INNOVATIVE APPROACHES TO URBAN RIVERSCAPE PLANNING AND DESIGN

by

HOLLY RUTH YARYAN HALL

(Under the Direction of Brian P. Bledsoe)

ABSTRACT

An urban riverscape holistically integrates natural forms and processes with human benefits and influences, and management can benefit from specialized tools for transdisciplinary communication, strategic planning, and technical design. I present three integrative approaches that support balancing social and ecological influences, equitable provisioning of benefits, and dynamic equilibrium of urban riverscapes. First, I create a framework for urban stream engineering that integrates the natural system with human dimensions, supporting discussion about key questions: What *is* an urban stream? What are the components, interactions, and potential functions and services of an urban riverscape? This communication tool is a conceptual model that serves as a natural infrastructure (NI) management and decision-support framework, reflecting a broad spectrum of urban riverscape benefits like flood protection, water quality, and ecosystem support, plus social influences and values including sustainable development, human connectivity, and environmental equity. Then, I develop a spatial prioritization approach for responding to additional questions: What do we want? How do we choose among the many environmental challenges and potential NI opportunities? The urban riverscape multi-criteria decision analysis (MCDA) is a planning tool that supports shared decision-making for equitable NI by incorporating multiple spatial scales and integrating management objectives. Through a collaborative, real-world case study, I implement novel variations of equity metrics, in the process discovering how watershed and sub-basin scales influence the identification of environmental inequity hotspots. I demonstrate how to tailor the urban riverscape MCDA for one particular social-ecological context and thereby illustrate its practical transferability for broader applications elsewhere. Finally, I expand and enhance a technical tool that supports stable channel design and aquatic ecosystems by balancing water and sediment, integrated over time. Sediment transport is one of the most misunderstood natural components of an urban stream system, despite being the fundamental link between channel forms and processes, key to physical equilibrium. While potential applications are not limited to urban riverscapes, this design and assessment tool is especially useful for areas constrained by built infrastructure and impacted streams that lack sufficient time and space for self-recovery.

INDEX WORDS: Ecological engineering, Environmental equity, Fluvial geomorphology, Natural infrastructure, Nature-based solutions, Stream restoration

INNOVATIVE APPROACHES TO URBAN RIVERSCAPE PLANNING AND DESIGN

by

HOLLY RUTH YARYAN HALL

BSE, Princeton University, 1997

MAT, University of South Carolina, 1999

ME, Colorado State University, 2013

A Dissertation Submitted to the Graduate Faculty of The University of Georgia in Partial Fulfillment of the Requirements for the Degree

DOCTOR OF PHILOSOPHY

ATHENS, GEORGIA

2022

INNOVATIVE APPROACHES TO URBAN RIVERSCAPE PLANNING AND DESIGN

by

HOLLY RUTH YARYAN HALL

Major Professor: Brian P. Bledsoe
Committee: C. Rhett Jackson
Amy Daum Rosemond
Ernest W. Tollner

Electronic Version Approved:

Ron Walcott
Vice Provost for Graduate Education and Dean of the Graduate School
The University of Georgia
December 2022

DEDICATION

To practitioners of healing arts
and unnamed tributaries

The first half of life
you were gathering
the waters
that will run
in the rivers of
your second half of life

ACKNOWLEDGEMENTS

This work is a multithreaded river, flowing from headwaters and tributaries, branching out, interconnecting, and exploring new channels. I am deeply grateful to Brian Bledsoe, who has inspired, guided, and encouraged me as a professor and advisor – a rock embedded in the river, steady in turbulent times, creating new opportunities downstream. My appreciation and admiration extend to my other committee members who have also motivated, challenged, and supported me: Amy Daum Rosemond, Rhett Jackson, and Bill Tollner. I am thankful for so many engineers, ecologists, hydrologists, geomorphologists, and others who have taught me how the water flows and what the river knows.

I gratefully acknowledge Janine Castro, Colin Thorne, Jacqueline Echols, Haley Selsor, and the Riverscapes group for improving this work through constructive feedback. The spatial prioritization case study was enriched by the participation of Charlotte-Mecklenburg Storm Water Services. I thank Rod Lammers for CSR tool collaboration and advice, Kevin Samples for GIS support, Brad Gilleland for 3D graphics, and many individuals for data collection. A river network of classmates, coworkers, colleagues, clients, collaborators, and conversation partners have given me friendship, encouragement, help, time, inspiration, and wisdom over the years.

From beginning to end to new beginnings, I am immensely grateful for the love and support of family both near and far. My parents showed me how to follow the call of gravity, jump into the deep end of a pool, and find my way back home. My sister is the ocean to my river, giving and receiving, always. The adventure partner of a lifetime, my husband has been by my side all the way from mountain stream to crashing sea.

TABLE OF CONTENTS

LIST OF ABBREVIATIONS

Abbreviation	**Definition**
ACD	Analytical Channel Design
CMSWS	Charlotte-Mecklenburg Storm Water Services
CSR	Capacity/Supply Ratio
FDC	Flow Duration Curve
FUSE	Framework for Urban Stream Engineering
MCDA	Multi-Criteria Decision Analysis
NCD	Natural Channel Design
NI	Natural Infrastructure
RARR	Risk Assessment / Risk Reduction
SRC	Sediment Rating Curve
SRRS	Stream Restoration Ranking System
SVI	Social Vulnerability Index

CHAPTER 1

INTRODUCTION AND LITERATURE REVIEW

Figure 1.1. Degraded (left) and revitalized (right) urban riverscapes (Bledsoe 2022). Reproduced with permission of Frank Ippolito.

"This stream needs help!" concludes a water resources manager visiting an urban channel reach with degraded bed, eroded banks, and marginal ecological habitat. In addition to feeling that the stream "could be better" and needs to be "cleaned up", a neighborhood resident describes property damage following a big storm event last year. At the same location, an environmental scientist observes low biodiversity, classifies water quality impairments, and ponders the multitude of possible influences and ecosystem responses. In my urban stream research, I ask how we can bridge the gap between restoration science and management practice for the sake of healthier streams and people, moving from left to right in Figure 1.1. Although a trashy waterway suffering from the urban stream syndrome (Walsh et al. 2005, Wenger et al. 2009, Booth et al. 2015) is far from fragrant, appetizing, or inviting, the metaphor of a potluck dinner can be helpful for imagining the interlocking relationships between people, context, and strategies for stream revitalization in a complex social-ecological-technical system.

To prepare for a potluck gathering with a wide variety of people, in addition to balancing main dishes, sides, and desserts, a thoughtful planner considers the invitees, event space, and dietary details. The guest list includes numerous stakeholders, practitioners, and experts: watershed and floodplain managers, regulatory agencies, neighborhood residents, politicians, technical specialists (engineers, fluvial geomorphologists, landscape architects, etc.), and researchers (biophysical and social sciences, public policy, etc.). Therefore, the potluck reflects a diversity of friendly faces, mutual strangers, and even some personality conflicts (e.g., perceived competing mandates). The well-coordinated meal to be shared by many represents a feast of potential benefits, services, and values, and the gathering space is where we want to experience a vibrant urban riverscape as a potential "third place" (Dolley and Bosman 2019). Expecting dinner conversations ranging from flooding and natural infrastructure (NI) to biodiversity and environmental justice, how can we encourage constructive discourse and collaborative learning that lead to positive outcomes? This dissertation details three different tools and techniques that support thoughtful, integrative planning and design in urban riverscapes. One of the following chapters decides where to have the potluck and who's invited, one is a healthy recipe, and one is a framework for holistically planning the entire menu.

In Chapter 2, I respond to a key research question that supports numerous additional research objectives: "How can we improve communications between scientists, managers, planners, engineers, and stakeholders?" (Wenger et al. 2009). Here, I introduce a conceptual model that supports balanced management approaches and transdisciplinarity in urban stream social-ecological systems. My framework for urban stream engineering (FUSE) synthesizes various stream restoration frameworks, objectives, and perspectives, and is therefore a novel approach to integrating fragments and otherwise missing pieces into a cohesive whole. FUSE is a three-dimensional mind-map that helps with organizing our thinking, communicating across disciplinary lines, and developing NI strategies that span multiple spatial scales. As a framework for NI, this conceptual model reflects a full spectrum of benefits like water quality, flood protection, and vital ecosystems, plus social influences and values like sustainable development and human connectivity. Incorporating complex themes like the urban stream syndrome (Walsh et al. 2005, Booth et al. 2015, Hawley and Vietz 2016) and environmental equity (Moran 2010, Davis et al. 2022, Díaz-Pascacio et al. 2022), FUSE reminds us that underlying tension can be resolved by balancing both social and ecological components (e.g., Smith et al. 2016, Murphy et al. 2022). While it can serve as an urban stream restoration party icebreaker, FUSE is intended to promote overall balance, holistic thinking, and knowledge co-production to help solve wicked social-ecological problems (Scoggins et al. 2022, Fork et al. 2022).

Demonstrating venue selection and a practical FUSE application, I present a spatial multi-criteria decision analysis (MCDA) case study in Chapter 3. The underlying research objective is to address a knowledge gap about management strategies in urban riverscapes that provide societal benefits in addition to improved stream ecosystems (Wenger et al. 2009). Spanning a broad array of both nested and independent scales (watershed, floodplain, reach, neighborhood, etc.) in the City of Charlotte and Mecklenburg County, North Carolina, this GIS-based prioritization integrates management objectives for multifunctional NI as part of a collaborative project with Charlotte-Mecklenburg Storm Water Services. While the ecological value of reach-scale stream restoration is debatable (Bernhardt et al. 2007, Smith et al. 2016, Doll et al. 2016), we can expect multiple potential benefits from restoration strategies at various spatial scales (Palmer et al. 2014a, 2014b, Hawley 2018, Polvi et al. 2020) when they are aligned with the corresponding stakeholder objectives (flood mitigation, water quality, habitat, outdoor recreation, etc.). The urban riverscape MCDA case study is also an innovative application for social equity (Smardon et al. 2018, Debbage 2019, Díaz-Pascacio et al. 2022), a critical NI component that helps address who gets a seat at the table, to make sure everyone is getting a good meal and enjoying the smorgasbord of environmental benefits.

In Chapter 4, digging into part of a healthy stream diet that is frequently neglected or misunderstood, I evaluate two sediment transport capacity methodologies: natural channel design (NCD) and analytical channel design (ACD). The research objective is to synthesize compatible approaches to the physical template and quasi-equilibrium needed to maintain roughness and habitat complexity, making them prerequisites for many aquatic ecosystems as well as protecting built infrastructure. Given the frequent space constraints of urban environments (Hess and Johnson 2001), management objectives typically coincide with a high priority for stream stability, often in response to erosion and sedimentation problems triggered by watershed development and channel disturbances (e.g., Doll et al. 2003, Booth and Fischenich 2015, Hawley and Vietz 2016). Physical equilibrium can be evaluated by comparing sediment transport and continuity for a pair of hydrologically connected stream reaches using the capacity/supply ratio (CSR). Here, I expand and enhance a CSR spreadsheet tool that is useful for reach-scale design and assessment (Bledsoe et al. 2016, 2017, Stroth et al. 2017). The CSR tool balances water and sediment, supporting dynamic equilibrium through integration over time, especially useful for urban streams with limited space and time for self-healing (Kondolf 2011). By incorporating NCD sediment transport and stream stability predictions, the modified CSR tool highlights convergent evolutionary pathways while integrating two competing stable channel design approaches. At the potluck meal, this chapter also corresponds to a civil conversation between a pair of guests about a potentially volatile topic. Therefore, in addition to being user-friendly for urban stream practitioners, the improved tool is a new contribution towards helping resolve part of a long-standing NCD controversy (Lave 2008, 2009) while rejecting the false dichotomy between stream form and processes (Niezgoda and Johnson 2005).

This dissertation spans multiple nested spatial scales, from the meta-scale conceptual model to watershed scale prioritization and reach scale analysis, with a focus on connectivity: lateral, longitudinal, temporal, and social-ecological. The progression of chapters reflects a transition from larger to smaller scales and logical order of tool implementation. First, comprehend the urban riverscape as a social-ecological-technical system and communicate with other people from diverse backgrounds. Then, decide where to intervene in the system partly through a process of multi-objective spatial prioritization. Finally, apply technical approaches (e.g., computational models) for detailed study and design. A common theme throughout this work is flexibility and applicability to the broad range of researchers, practitioners, and stakeholders found in an urban stream restoration context. To help bridge the gap between science and practice while supporting equitable delivery of the full spectrum of benefits, I also emphasize harmony with shared goals of healthier and more sustainable riverscapes. Welcome to the urban stream party – I'm thankful for what you bring to the table, too!

CHAPTER 2

A FRAMEWORK FOR URBAN STREAM ENGINEERING THAT INTEGRATES NATURAL INFRASTRUCTURE AND SOCIAL-ECOLOGICAL DIMENSIONS[1]

[1] Yaryan Hall, H.R. and B.P. Bledsoe. To be submitted to *River Research and Applications*.

Abstract

The interdisciplinary fields related to stream restoration science and practice draw from a variety of established conceptual models and frameworks, but these frameworks are highly constrained in urban contexts, especially with the typical division of people and goals among groups including managers (water quality, floodplain, stormwater), watershed organizations, landowners, and recreational interests. I developed a Framework for Urban Stream Engineering (FUSE) that supports multifunctional management approaches to ecological engineering and urban riverscape revitalization. FUSE combines a three-dimensional stream ecosystem pyramid and social sphere of influence in a visually memorable and flexible conceptual model. Previous models have poorly integrated water quality with hydrologic, geomorphic, and ecological structures, forms, and processes, and they have not explicitly incorporated key social dynamics. The stream ecosystem pyramid includes water chemistry along with geology, hydrology and biology as drivers of aquatic life and ecological functions, while the social sphere of influence captures human dimensions, such as land use, watershed management, social benefits, and environmental equity. A novel aspect of FUSE is the integration of established components from the natural and social sciences specific to urban streams with an ecological engineering focus on benefits and natural infrastructure (NI). FUSE is intended to serve as a tool that aids systems thinking (social-ecological-technical), asking essential questions, recognizing interactions, identifying root causes of impairment, and communicating to solve problems. By introducing FUSE as a transdisciplinary mental map for urban riverscapes, I hope to enhance multi-objective decision-making, guide NI innovations, support environmental equity, and encourage collaborative learning between specialists, stakeholders, and the general public.

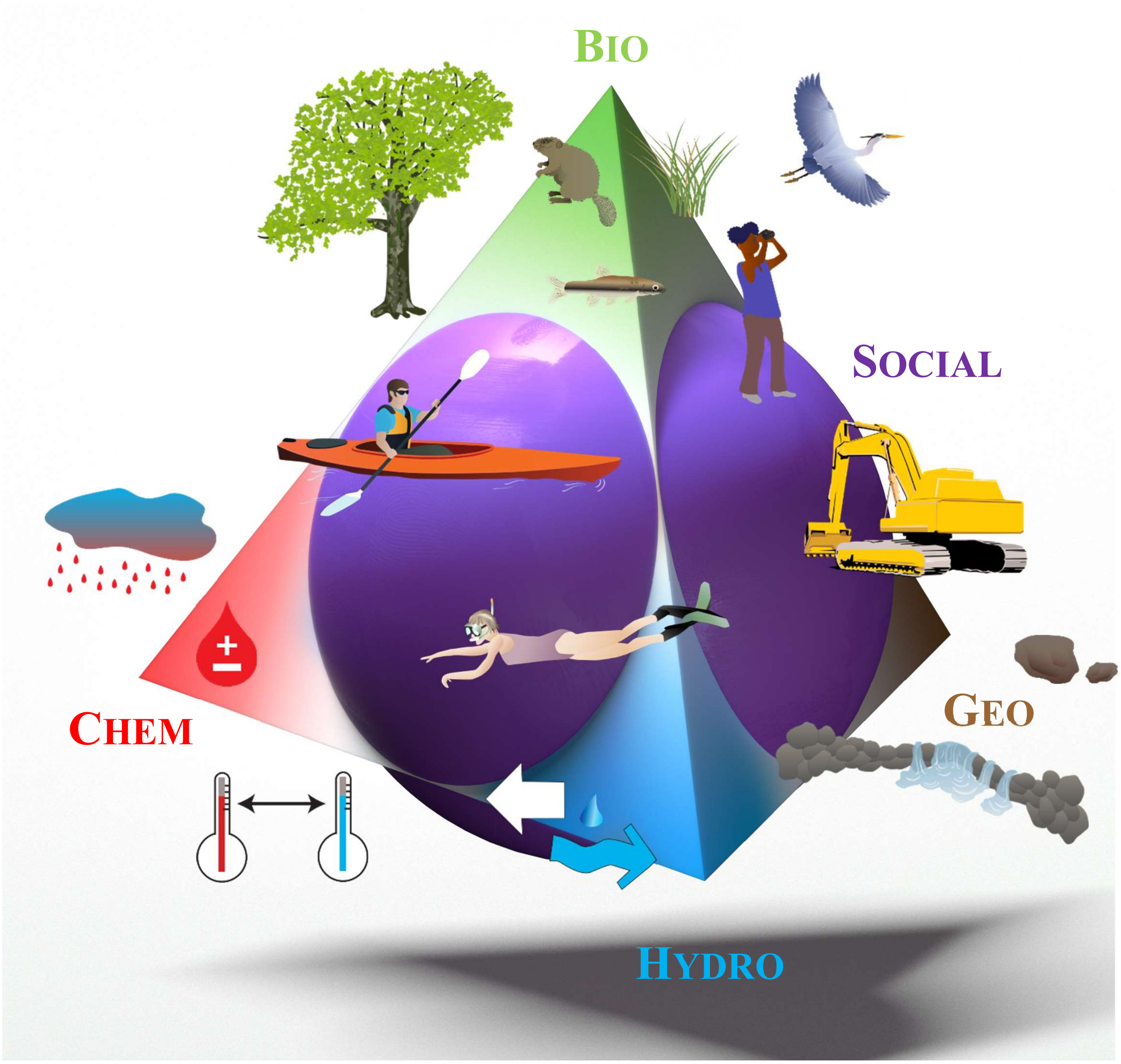


Figure 2.1. The framework for urban stream engineering (FUSE) represents a coupled human and natural system comprised of the stream ecosystem pyramid and social sphere of influence. The pyramid vertices (HYDRO, GEO, BIO, CHEM) are drivers of the natural system state and processes, including overall water quality and ecosystem structures and functions. In the social-ecological context of FUSE, the pyramid vertices also correspond to Anthropocene influences and environmental risks and benefits to humans. The social sphere encompasses people and policies integrated with the stream ecosystem, human values interacting with the natural system (management, development, equity, etc.), and social-ecological benefits emerging from the coupled system. The balanced form of FUSE shown here (pyramid edges tangent to the sphere) emphasizes benefits to both human society and the natural environment.

Introduction

Urban stream practitioners and researchers use a variety of conceptual models to characterize and communicate key social-ecological system components, interactions, and complexities. However, most existing models are either narrowly focused on specific facets, like geomorphology or stream ecology, or else broadly generic to all social-ecological systems, and therefore missing specific, critical elements of problems and solutions common to urban stream systems. The stream ecosystem pyramid and social sphere of influence form a conceptual framework for urban stream engineering (FUSE), shown in Figure 2.1, that integrates natural infrastructure (NI) and ecological engineering principles and practices, supporting riverscape management that provides both human benefits and aquatic ecosystem improvements. Building on the stream evolution triangle conceptual model (Castro and Thorne 2019) with geology, hydrology, and biology vertices, the stream ecosystem pyramid incorporates a chemistry vertex, while the combined FUSE integrates the social sphere of influence, expanding the scope from fluvial geomorphology to overall stream functions and human purposes (e.g., Baron et al. 2002). The three-dimensional model uses graphically efficient but flexible components to encompass complex social-ecological system interactions and concepts, serving as a mental map and communication tool to advance the science and practice of urban stream restoration and enhance shared decision-making. FUSE supports multi-objective management approaches, suggesting potential actions and strategies to achieve social benefits, services, and values in urban riverscapes, including environmental equity.

The current state of urban stream science and management has three distinct influences: stream restoration practices based on applied fluvial geomorphology concepts, an environmental science emphasis on the urban stream syndrome, and a growing social science inclusion of human dimensions. First, mainstream restoration practices are often dominated by channel stability, aquatic habitat, and design approaches based on geomorphological channel evolution models and stream classification systems (Rosgen 1994, NRCS 2007, Booth and Fischenich 2015). As such, common reach-scale restoration efforts are unlikely to effectively address underlying water quality problems (Bernhardt et al. 2007, Rubin et al. 2017, Polvi et al. 2020). However, because physicochemical factors (e.g., turbidity, dissolved oxygen, temperature, etc.) limit biology and therefore potential functional uplift (Palmer et al. 2010, Harman et al. 2012), decision-makers may avoid restoration investments in the most heavily impacted urban environments. Second, environmental science emphasizes biogeochemical variables, system disturbances, and ecological responses, characterizing human influences as a suite of detrimental driving forces outside the ecosystem boundaries, such as land use change (e.g., Walsh et al. 2005). Urban streams are often considered poor candidates for restoration projects due to links between land use and ecological functions (e.g., Roy et al. 2003, Sterling et al. 2016), so mitigation efforts strongly favor settings with relatively low contemporary human development. Third, there is a growing trend to view urban streams in the context of the larger social-ecological-technical system, with an emphasis on complex interactions manifesting as “wicked problems” as well as a wider range of potential project objectives (benefits, services, values) and measures of success (Clifford 2007, O’Donnell et al. 2020, Fork et al. 2022). Rather than conventional restoration projects, stream improvements may be characterized as renovation or enhancement efforts (e.g., Smith et al. 2016). Environmental equity is increasingly recognized as a leading edge of urban riverscape research, with evidence showing inequitable greenspace access, flood impacts, and surface water quality (Smardon et al. 2018, Debbage 2019, Davis et al. 2022, Díaz-Pascacio et al. 2022), in addition to concerns about neighborhood displacement and gentrification (e.g., Jelks et al. 2021).

Existing conceptual models and diagrams for urbans streams tend to fall into categories corresponding to disciplinary dominance. Two types of conceptual models are most common among stream restoration practitioners but lack water chemistry and/or social dimensions. Fluvial geomorphology models emphasize physical forms and processes, with common threads of channel evolution (e.g., Schumm 1981, Cluer and Thorne 2014, Booth and Fischenich 2015) and stream classification (e.g., Rosgen 1994, Kondolf et al. 2016). Recent variations include urban stream forms (Hawley et al. 2012, Cluer and Thorne 2014, Booth and Fischenich 2015), biological processes (Castro and Thorne 2019), and "self-healing" potential with bivariate axes for floodplain development/encroachment and stream power/sediment supply (Kondolf 2011). Broader function-based frameworks (e.g., Harman et al. 2012, Nadeau et al. 2018) are commonly used by regulators and practitioners to implement stream restoration planning and assessment. The popular stream functions pyramid (Harman et al. 2012), for example, includes physicochemistry, but the hierarchical structure masks multi-level cause-effect relationships and feedbacks among the layers (Johnson et al., 2016). Moreover, the corresponding stream quantification tools (SQTs) increasingly used by regulators (e.g., Harman and Jones 2017, TDEC 2017, U.S. Army Corps of Engineers 2018a, 2018b, Minnesota Stream Quantification Tool Steering Committee 2019) explicitly deprioritize streams with urbanized catchments. In general, existing conceptual models and frameworks developed and used by fluvial geomorphologists and stream practitioners lack key social components of urban stream systems, insufficiently accounting for human interactions and dynamics that perpetuate the urban stream syndrome.

Stream ecologists and environmental scientists, on the other hand, often generate complex system diagrams with boxes and arrows representing numerous interactions among stressors, ecosystem responses, and management actions, combining detailed aspects of flow, water chemistry, biological structures and functions, and geomorphology (e.g., Walsh et al. 2005, Wenger et al. 2009). Scientific models for urban streams often incorporate multiple spatial scales (e.g., Thoms et al. 2017, Polvi et al. 2020), but human components may be limited to external stressors or management actions, lacking the interconnections comprising environmental benefits as well as the potential for social feedback loops.

Scientists generally recognize that the urban stream syndrome is a coupled human and natural system problem (e.g., Walsh et al. 2015, Chien and Saito 2021), and social-ecological system literature includes frameworks with ecosystem services and multi-directional influences (feedback loops), but not necessarily specific to urban streams (e.g., Díaz et al. 2014, Grimm et al. 2017, Colding and Barthel 2019). A recent urban stream renovation framework (Smith et al. 2016) features a feedback loop with improvement actions, social and ecological outcomes, and public support for future actions, potentially leading to long-term gains in ecosystem structures and functions. However, the urban stream renovation framework does not differentiate between various biogeochemical drivers or specific social-ecological benefits and system constraints. Unfortunately, stream restoration (or renovation) efforts may be disconnected from social realities, with key planning and design decisions made by technical experts like engineers and environmental scientists interacting with a handful of government stakeholders, and public outreach limited to later project stages (e.g., Eden and Tunstall 2006). Moreover, I am not aware of any frameworks for urban streams that explicitly incorporate environmental equity, despite known patterns of inequity in riverscapes.

My objective in developing FUSE is to create a tool that encompasses multiple disciplinary themes, cutting across management boundaries to reflect the complexity of the urban stream syndrome, include common environmental benefits, support riverscape improvements, and contribute to productive discussions among stream specialists, decision-makers, and the public. In short, the broader goal of FUSE is to help bridge the gaps between science and practice, characterizing a coupled human and natural system with key commonalities of urban riverscapes. I aim to synthesize components from the natural and social sciences specific to urban streams with an ecological engineering focus on benefits and practices, holistically integrating water quality and human dimensions. By introducing a shared frame of reference, I hope to enhance multi-objective decision making and support ecological engineering and NI innovations like nature-based solutions (World Water Assessment Programme 2018, Cohen-Shacham et al. 2019, Sowińska-Świerkosz and García 2021), "blue-green cities" (Thorne 2020, O'Donnell et al. 2020) and Engineering With Nature® (Bridges et al. 2018, 2021, U.S. Army Corps of Engineers 2022). I anticipate that improved knowledge sharing among urban stream specialists and the general public, together with increased social cohesion, will help advance future restoration and renovation efforts, by promoting transdisciplinary planning approaches and leading to more successful project outcomes. As a practical tool, I created FUSE to guide balanced stream and watershed management, linking system states with actions and strategies to achieve social-ecological benefits and environmental equity in urban riverine corridors.

Framework for urban stream engineering (FUSE) components

The graphical simplicity of FUSE is meant to be memorable: the pyramid and sphere are fused together to depict a fully coupled social-ecological system. The color-coded vertices of the stream ecosystem pyramid help to recall the natural sciences underpinning stream ecology, which can be unpacked to include more detailed ecosystem structures, functions, and processes. The pyramid vertices further represent specific environmental benefits and risks to humans, as well as categorical human impacts to the stream ecosystem. At the same time, the sphere flexibly expands or contracts to indicate the degree of social influences. Moreover, the social sphere of influence can be pulled apart into multiple spheres representing key social dimensions. The pyramid volume corresponds to the ecosystem state, and changes to size and shape reflect system responses to specific system drivers. Finally, arrows can be used to depict directional interactions between individual stream ecosystem and social components, like more conventional stock-and-flow system diagrams (e.g., Meadows 2008).

FUSE integrates components from a variety of models and frameworks from fluvial geomorphology, environmental science, and social-ecological systems literature with the ecological links to human benefits, services, and values. The GEO, HYDRO, and BIO vertices of the stream ecosystem pyramid mirror the corners of the stream evolution triangle (Castro and Thorne 2019), with all three driving physical stream forms and processes (Table 2.1). For example, the GEO vertex can be interpreted as a suite of influences, such as channel confinement, dominant bed material, and sediment transport zones (Castro and Thorne 2019). In the context of benefits, GEO can represent erosion protection, for example, and urban streams with artificially stabilized channels resist geomorphological changes (Castro and Thorne 2019). Further, urban headwater streams may be completely confined to stormwater drainage systems, becoming artificial subterranean karst formations (Bonneau et al. 2017). Urban influences may likewise drive the stream through hydromodification (HYDRO) with increased flow flashiness due to imperviousness, and away from biological influences (BIO) by removal of vegetation, beaver, and instream wood (Castro and Thorne 2019). Going beyond drivers of fluvial geomorphology,

Table 2.1. Stream ecosystem pyramid components and intersections with social sphere of influence. Sciences underpinning stream ecology and water quality (biogeochemistry, hydrogeomorphology, etc.), with human interactions and motivations (environmental benefits, hazards, etc.).

	HYDRO	GEO	BIO	CHEM
Stream ecology and water quality processes and components	Hydrology Flow regime (low flow, baseflow, stormflow, snowmelt, floods) Groundwater Hyporheic zone Overbank flooding Connectivity	Geology Geomorphology Sediment (silt, sand, bed, bank) Erosion Sediment transport Floodplain connectivity Valley type, topography Turbidity	Biology (aquatic, riparian, terrestrial) Fish & wildlife (beaver, birds, insects, mussels) Vegetation, trees plants, algae, biofilm Wood Habitat Organic carbon	Chemistry Physical (dissolved oxygen, temperature) Nutrients (N, P) Metals (Pb, Mn, Hg, Cr, Zn) Bacteria (*E. coli*, fecal coliform) Turbidity
Ecosystem structures, functions, and characteristics	Environmental flows Hydraulics Chemical concentrations	Physical connectivity Habitat Refugia during floods, droughts, wildfires	Habitat Biodiversity Shade, temperature, available sunlight Food web Carbon, nutrient cycling	Aquatic species with pollution sensitivity or tolerance Metabolism Nutrient cycling, deposition
Human influences and alterations (direct or indirect)	Imperviousness Dams, reservoirs Road crossings Increased flashiness Lower baseflow Connections, disconnections	Bank stabilization Channelization Channel enclosure (storm sewers, culverts) Urban karst Fragmentation	Vegetation removal Beaver removal Large wood removal Habitat loss Less biodiversity	Sewer overflows Stormwater runoff from yards, streets, drains Pharmaceuticals Agriculture (N, P) Pesticides
Benefits, values	Water supply Boating	Stream, floodplain access	Greenspace Fishing, birdwatching	Clean water Swimming, wading
Risks, disbenefits	Flooding, drought	Erosion, sedimentation	Nuisance species	Pollution

however, the stream ecosystem pyramid encapsulates the variables contributing to ecological structures and functions (e.g., habitat, biotic species, etc.), human-centered benefits and services (e.g., flood mitigation, clean water, outdoor recreation), and potential motivations for management actions (Table 2.1). For example, fishable streams with species richness and biodiverse riparian zones that support bird watching are depicted where the social sphere of influence intersects the stream ecosystem pyramid near its BIO vertex.

The CHEM vertex of the stream ecosystem pyramid incorporates physicochemical variables such as available nutrients, dissolved oxygen, and temperature (Figure 2.2a). Water quality is a system property that emerges from aquatic chemistry variables and interactions with hydrology, biology, and geology (Karr and Yoder 2004, Booth and Bledsoe 2009), represented by the multidimensional space inside the pyramid. In turn, biologic metrics (e.g., species biodiversity) can be viewed as integrative indicators for water quality, as it's widely understood that ecological function is closely linked to physicochemistry and urban disturbances (e.g., Roy et al. 2003, O'Driscoll et al. 2010, Sterling et al. 2016). Removal of riparian vegetation, for example, can reduce instream shade and habitat, increasing the temperature and available sunlight, stimulating photosynthesis, and increasing dissolved oxygen, while simultaneously reducing carbon inputs that support the aquatic food web (e.g., Aldridge et al. 2009, Bernhardt et al. 2018). Human influences on urban stream aquatic chemistry and resulting water quality also include leaking sewers (e.g., combined sewer overflows, illicit discharges, failing pipes), lawn and street stormwater runoff (e.g., pesticides, road salt, vehicle pollution), groundwater contamination sources (e.g., old gas stations, underground storage tanks, industrial spills), and other toxins, pathogens, and nutrient pollution. Streams impacted by human development may thus contain complex mixtures of toxic metals, pharmaceuticals, and ions (e.g., Ca^{2+}, Na^{+}, Cl^{-}).

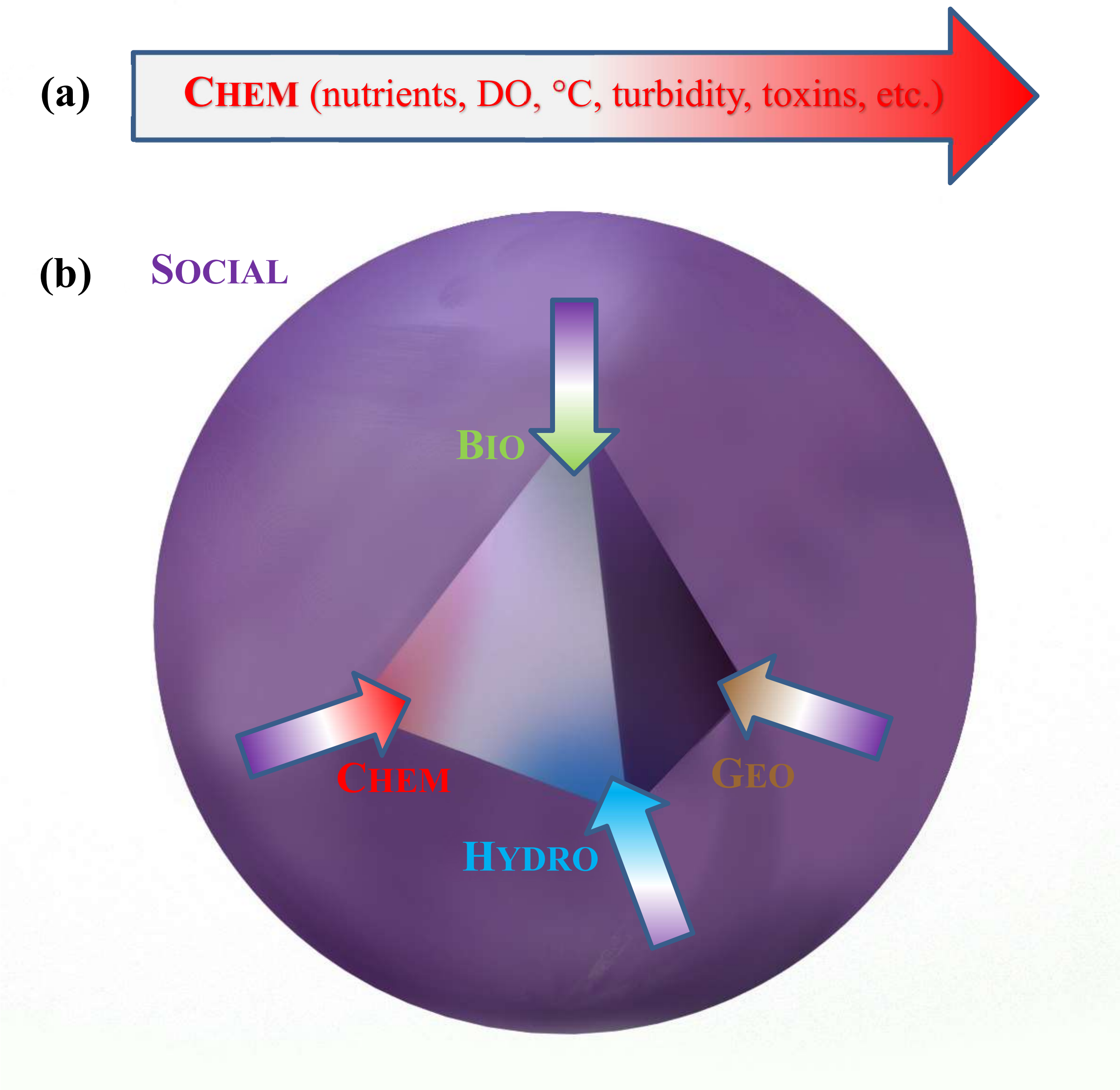


Figure 2.2. (a) The water chemistry spectrum and corresponding stream ecosystem pyramid vertex represent multiple physicochemical drivers of water quality and stream ecology: nutrients, dissolved oxygen (DO), temperature and light, turbidity, ions, toxins (e.g. fecal coliform), metals and plastics, and the resulting mixtures of chemical cocktails found in urban streams. (b) The urban stream syndrome evolves from the overwhelming interactions of the social sphere with the stream pyramid, and the arrows indicate system changes over time. Disturbances to hydrology, geology, and water chemistry, together with the removal of biological drivers (vegetation, beaver, wood), push the stream state in a negative direction with impairments to water quality and ecosystem structures and functions. The diminished stream pyramid also corresponds to reduced environmental benefits, such as greenspace (BIO), or increased risks, including pollution (CHEM), flooding (HYDRO), and erosion (GEO) impacts to human health and built infrastructure.

For example, winter road salt contributes to the freshwater salinization syndrome, which in turn combines with "urban karst" and other processors and mixers to create problematic "chemical cocktails" (Kaushal et al. 2018, 2020). Together with hydrology, geology, and biology alterations, the aquatic chemistry drivers of ecological structures and functions significantly contribute to the loss of water quality benefits and services and the associated urban stream syndrome (Walsh et al. 2005, Wenger et al. 2009). The diminished pyramid shown in Figure 2.2b thus represents a negative ecological scenario of an urban stream dominated by hydrology, geology, and chemistry stressors, devolved away from biology as both a positive driver and indicator of stream health, into a shrunken remnant of the former aquatic ecosystem, with significantly impaired water quality.

Just as each vertex of the stream ecosystem pyramid represents natural sciences, functions, system responses, and environmental hazards, the social sphere of influence captures key human components of an urban stream social-ecological-technical system: anthropogenic disturbances to aquatic and riparian ecosystems, stream and watershed management strategies, human-centered benefits and services, environmental equity, and public riverscape connectivity (Figure 2.3). For each sphere, the center is the desirable goal, not necessarily a starting point. Challenges at the outer edges need to be addressed, working from the outside in. However, it may also be possible to leverage interactions at the center, supporting changes through positive feedback loops, working from the inside out. For example, the management sphere (Figure 2.3a) has single factor interventions at the outside, in contrast with multi-objective approaches at the center. While it's necessary to assess specific concerns like flooding, erosion, and water quality, holistic riverscape management reaches across departmental boundaries. Supporting aquatic and riparian ecosystems at the center of the social sphere of influence likewise supports the broader

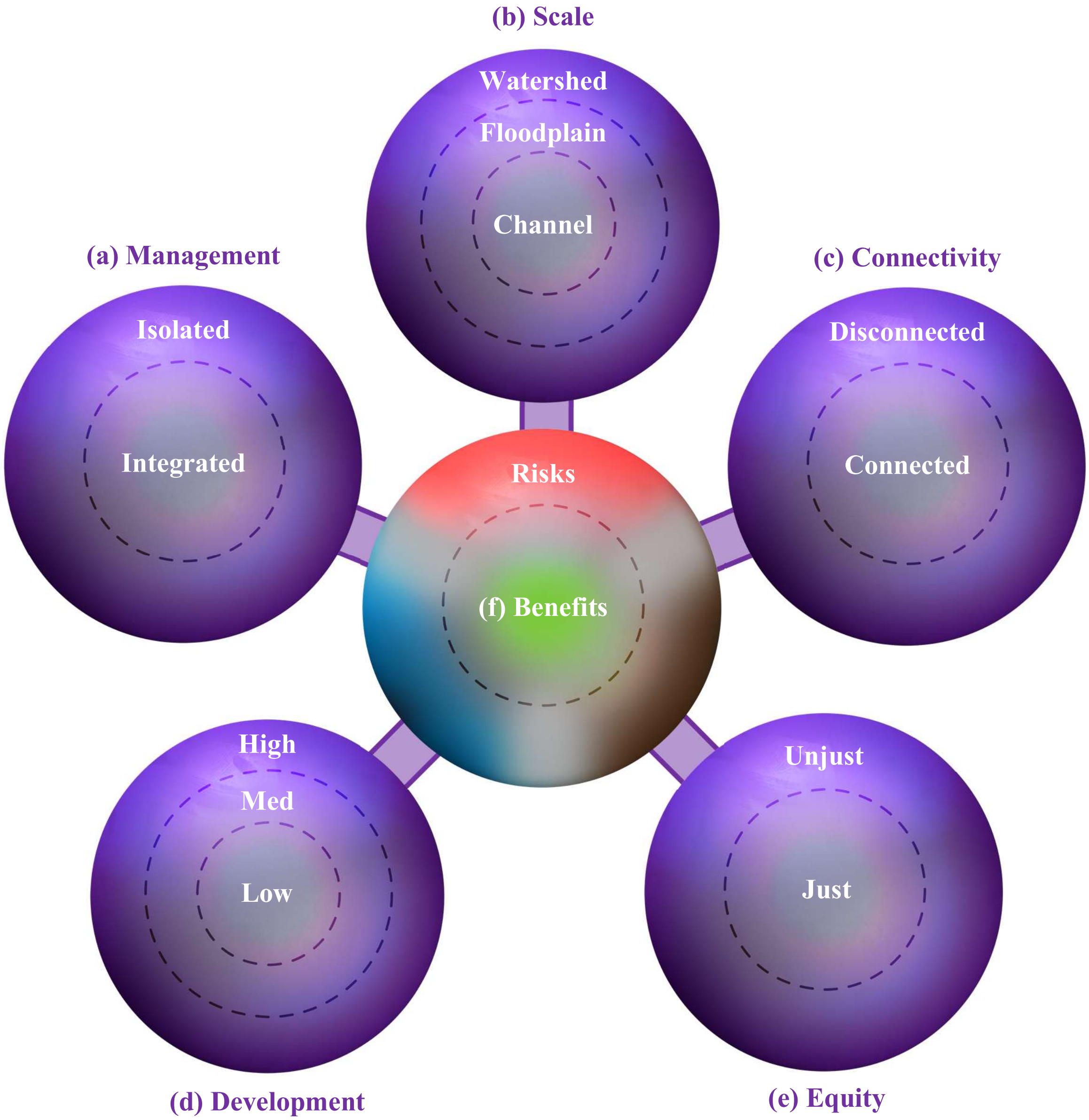


Figure 2.3. Social spheres of influence include (a) management policies and practices, (b) system scale, (c) social connectivity, (d) development intensity and land use, (e) social equity, and (f) environmental benefits and risks. Movement from the outside towards the center of each sphere corresponds to desirable goals, such as mitigating environmental risks (f), addressing injustice (e), or integrating riverscape management (a). The multicolored benefits sphere (f) reflects emergent properties of system interactions with the stream ecosystem pyramid (HYDRO, GEO, BIO, CHEM), and it is constrained by the other spheres as well as the pyramid (i.e., limited environmental benefits and ecosystem services). The social spheres shown here are especially critical to understanding urban stream systems and solving complex riverscape problems, not a comprehensive collection of social variables and processes.

range of positive benefits, and it tends to be more holistically interconnected than issues like flood risk, for example, with regard to the pyramid vertices. For spatial scale (Figure 2.3b), there can be value in channel-scale interventions, but it's also necessary to view the urban riverscape through the lens of nested spatial scales to recognize the full extent of disturbances and ecosystem responses. The development sphere (Figure 2.3d) ranges from low-impact (i.e., sustainable) at the center to high-impact at the external surface, and is not limited to land use change, but can also incorporate stormwater management practices, floodplain encroachment, and channel alterations as forms of human agency.

A sphere may depict social-ecological connectivity (Figure 2.3c), at its center closely overlapping the stream pyramid in a holistic fashion, with the external surface of the sphere representing fragmented aspects of the system, such as neighborhood residents who feel disconnected from their local stream or watershed. I use the term social-ecological connectivity in both concrete and abstract senses, meaning physical contact of communities and individuals with riverscapes (e.g., waterway access) as well as public engagement, knowledge co-production, decision-making processes, etc. In this way, the center of the connectivity sphere reflects closed social-ecological system feedback loops, such as public support for urban stream restoration and renovation leading to additional actions and greater long-term outcomes (Smith et al. 2016). The center of the connectivity sphere is not limited to transactional use and public participation but also includes the idea of relationship with the riverscape (e.g., sense of place) and "social geomorphology" (Mould et al. 2018, 2020). The environmental equity sphere (Figure 2.3e) emphasizes social-ecological connectivity in the context of race, ethnicity, income level, and other differences, and its center incorporates distributional equity, mutual respect, inclusive relationships, and collaborative goals. Social equity is a critical component of FUSE together with other aspects of human agency, just as scholars have begun to explore how environmental racism actively drives ecological outcomes and ecosystem services (Schell et al. 2020).

The multicolored benefits sphere (Figure 2.3f) reflects the overlap with the natural stream ecosystem pyramid (HYDRO, GEO, BIO, CHEM), and it includes a range of services and values, with positive benefits at the center and negative risks towards the outer edge. For example, flood mitigation is protection from an environmental hazard. Other desirable benefits in urban riverscapes include channel stability, water quality, greenspace, beauty, recreation, education, spirituality, human health, wellbeing, and economics. Positive benefits are constrained by the other social dimensions as well as the natural stream functions. In a positive sense, movement from the center outward could reflect the ecosystem services cascade and increasing degree of human agency, from recognition to mobilization and appropriation of benefits (Spangenberg et al. 2014, Fedele et al. 2017). In contrast, environmental risks are at the outer edge of the benefits sphere. Notably, the environmental equity sphere (Figure 2.3e) overlaps with benefits (Figure 2.3f) and connectivity (Figure 2.3c), such that inequity at the outer edge maps to negative risks and disconnection. Communities with higher Black or Hispanic populations may be disproportionately impacted by flooding (Smardon et al. 2018, Debbage 2019, Selsor et al. 2022), an example of environmental inequity near the stream pyramid HYDRO vertex. The social spheres shown in Figure 2.3 are critical FUSE components, but not comprehensive. The overall social sphere of influence may also incorporate economic costs, legal structures, insurance programs, social learning, etc.

Connecting multi-faceted viewpoints and transformative possibilities

Putting the FUSE pieces together, we can envision connections between the social sphere of influence and stream ecosystem pyramid vertices, such as the social-ecological benefits of

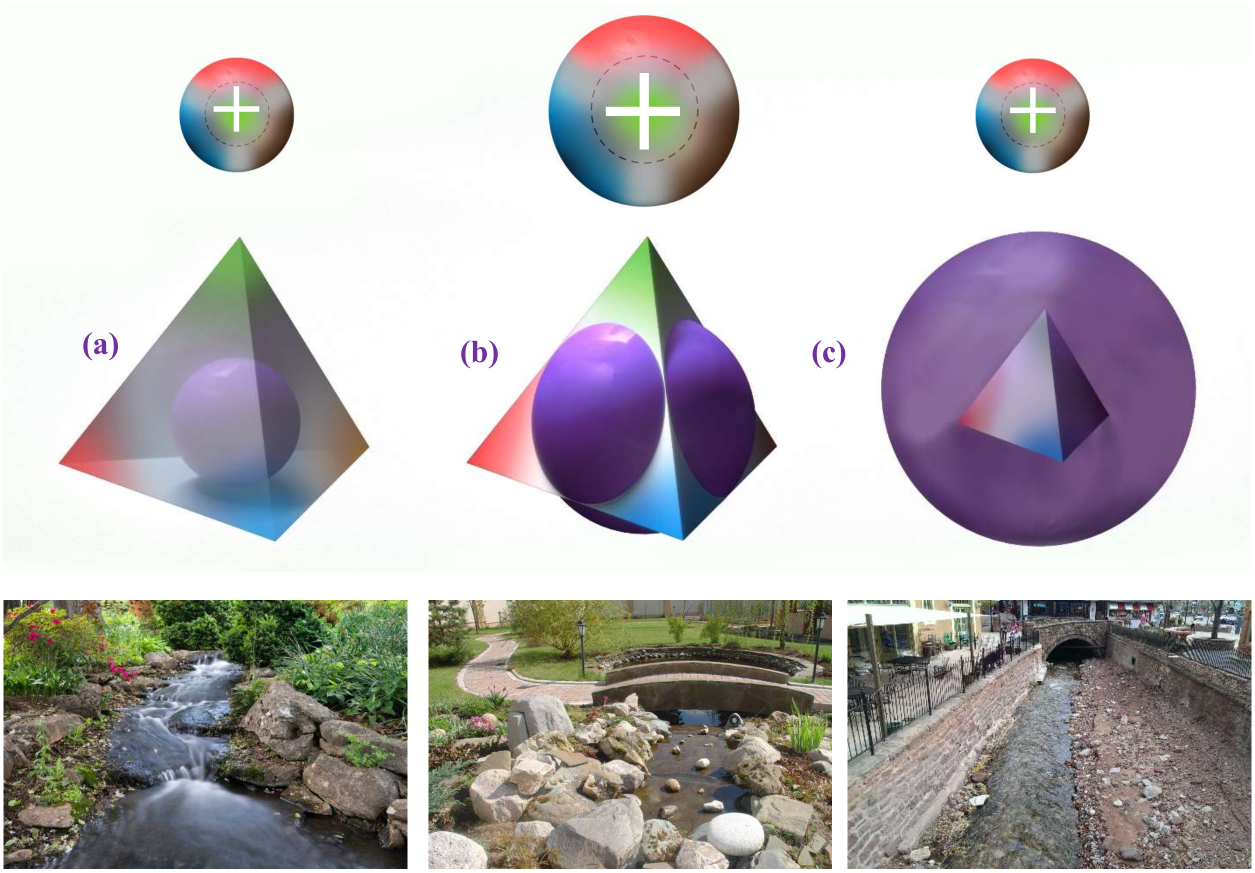


Figure 2.4. Variable size of the social sphere of influence relative to the stream ecosystem pyramid. (a) A small social sphere footprint corresponds to a predominantly natural system with few anthropogenic stressors; (b) a moderate sphere and pyramid together reflect sustainable development balancing human benefits and natural system functions; and (c) an oversized social sphere associated with the urban stream syndrome, characterized by negative environmental risks and undesirable economic consequences rather than positive benefits. The pyramid and sphere sizes both constrain the benefits sphere, because human benefits and services are supported by healthy ecosystems. A balanced approach to urban stream management emphasizes integrated riverscape design (b), rather than isolated ecological uplift (a) or risk reduction (c).

greenspace and species richness (Bio), land development impacts on sediment sizes (Geo), or snow and ice management (road salt) contributions to chemical cocktails (Chem). Moreover, the collective system can be depicted according to the lesser or greater role of the social sphere in relationship to the stream ecosystem, as shown in Figure 2.4. When the social sphere has a relatively small footprint compared to the stream pyramid (Figure 2.4a), the natural system retains the majority of its ecological structures, functions, and processes. Where a stream reach is closer to high quality reference conditions, logical management might include targeted conservation or stream restoration to support relatively sensitive species (BIO). While human benefits could be further increased through sustainable development (expanding the sphere within the pyramid), the system warrants monitoring for early signs of water quality and stream ecology problems. In contrast, where social influences (Figure 2.3) overwhelm the stream ecosystem (Figure 2.4c), we expect to find the urban stream syndrome intertwined with watershed-scale impacts to water quality, inequitable distributions of environmental risks (e.g., flooding, erosion, pollution), and lack of positive social connectivity (e.g., greenspace amenities, waterway access, stream adoption). Complex problems such as these are exacerbated by fractured stream and watershed management approaches, and stakeholders need to aim for a more balanced, integrated system (Figure 2.4b) that best supports a full range of benefits and services. The geometric elements of FUSE (e.g., pyramid faces, sphere overlaps) align with some key challenges and gaps in practice. With the following examples, I will explain how FUSE can help us organize thinking, recognize interactions that are root causes of the current state, and identify pathways to improvement.

Human benefits, ecosystem services, and social values emerge from the intersection of the stream ecosystem pyramid and social sphere of influence, reflecting variable social-ecological drivers, exchanges, and system feedbacks. Identifying and leveraging connections between the FUSE components involve forming perceptions and priorities to create the context that defines an interactive relationship. Some benefits and risks correspond most strongly with one specific vertex of the pyramid, for example water supply and flooding (HYDRO), while others, such as supporting aquatic life and ecological functions, more clearly integrate multiple system components. Similarly, the overlapping internal action spaces of the pyramid and sphere encompass multi-objective ecological engineering decisions and implementation approaches that benefit both people and nature, though not necessarily with equal emphasis, much like the urban stream renovation framework (Smith et al. 2016). Indeed, FUSE is intended to complement existing and future social-ecological system frameworks, much as the stream evolution triangle (Castro and Thorne 2019) accommodates various channel evolution and stream classification systems. The basic structure of FUSE can be adapted and combined with other conceptual models and frameworks to create more contextual social-ecological system models, such as those envisioned for the urban stream syndrome (e.g., Walsh et al. 2005, Wenger et al. 2009), or otherwise specific to particular applications (e.g., case studies), to help bridge disciplines and the gaps between science and practice.

The ideas driving FUSE lend themselves to communication via the geometric structures. For example, each two-dimensional, triangular face of the pyramid is opposite from a fourth vertex, with distance as a push-pull tension that impacts potential ecosystem functions and human benefits and services. FUSE helps organize our understanding of various social-ecological system complexities, lending itself to multiple points of view and encompassing otherwise hidden components of urban riverscapes and the people therein. Any given volume inside the intersecting pyramid and sphere can visually represent the system state in terms of dominant social-ecological drivers, ecosystem functions, and human benefits, while shape distortion reflects change over time, similar to movement between two points in the stream evolution triangle (Castro and Thorne 2019). There is no ideal stream improvement approach, only selecting context-specific strategies and implementing tools aligned with the relative importance of desired outcomes. Accordingly, multi-objective planning and problem-solving becomes the art of the possible within existing social-ecological constraints.

Urban stream syndrome: Seeking green solutions

The urban stream syndrome represented by the diminished pyramid (Figures 2.2b, 2.4c) is an ecological state driven by system disturbances. Where the social sphere of influence impacts the stream ecosystem pyramid near the GEO corner, we find artificially channelized and stabilized streams, like the arrested incision of some urban channel evolution models (Cluer and Thorne 2014, Booth and Fischenich 2015), or otherwise complete enclosure in storm sewer systems. Alterations to the sediment regime are also characteristic of urban streams (e.g., Russell et al. 2019). The influence of the social sphere on the stream pyramid near the HYDRO corner indicates alterations to the natural flow regime (Poff et al. 1997), a flashy stream that may run completely dry between rainfall events, and corresponding changes to the benthic disturbance regime (Hawley and Vietz 2016, Anim et al. 2019). Riverine and urban flooding may also increase. Where the social sphere meets the CHEM corner is a stream impaired by pollutants, increased temperatures, and reduced dissolved oxygen. All three corners of the urban stream syndrome face (HYDRO, GEO, CHEM) serve as negative press or pulse stressors affecting ecosystem structures and functions. Missing are the naturally diverse biological communities of flora and fauna, frequently used as indicators of stream health, following the downward trajectory from the BIO vertex shown in Figure 2.2b. Management challenges of the urban stream syndrome facet reflect a common “trifecta” of concerns facing many municipalities: channel erosion (GEO), water quality (CHEM), and flooding (HYDRO).

Opposite the urban stream syndrome face, a balanced social sphere intersects the stream pyramid closer to the BIO vertex, an attractive greenspace and “third place” (Bosman and Dolley 2019) providing cultural services (e.g., aesthetics, recreation, education, etc.). The BIO vertex of the pyramid also represents the potential role of biologic components in actively pulling the ecosystem upward in a positive direction (Johnson et al., 2020). For example, "process-based restoration" practices often incorporate or mimic beaver activity, promote instream wood, and enable vegetation to self-organize (e.g., Castro et al. 2015, Wheaton et al. 2019, Powers et al. 2019), and the emergent properties of river-wetland corridors include a suite of potential environmental benefits (Wohl et al. 2021). As another alternative to conventional stream restoration in urban settings, ecological engineering can feature a series of cascading wetlands (i.e., bioreactors) that de-emphasize aquatic biology scores in favor of diverse, heterotrophic bacteria that mineralize pollutants and retain carbon. The central idea is a natural intervention that pulls the system upwards to the BIO vertex.

Integrated stream restoration: Reconnecting fluvial geomorphology to water quality

The stream evolution triangle (Castro and Thorne 2019) comprises the fluvial geomorphology facet (HYDRO, GEO, BIO) of this conceptual model, corresponding to common channel evolution models and classification schemes as well as to dominant stream restoration practices. Although the stream evolution triangle emphasizes the reach scale, the interplay of geology, hydrology, and biology is also evident at smaller and larger scales, such as the hyporheic zone or valley setting, with corresponding stream restoration strategies (Hester and Gooseff 2010, Powers et al. 2019). Regardless, the disconnection between the fluvial geomorphology facet and the opposing CHEM vertex reflect ongoing conflict about the values of reach-scale restoration, even as water quality limits potential stream functions and motivates many management actions. However, people intervene in stream systems for a variety of reasons (Bernhardt et al. 2007, Palmer et al. 2014b), for example to meet regulatory requirements and avoid lawsuits, obtain mitigation credit, support valued biota (e.g., trout), experience natural beauty, alleviate flooding, and care for creation.

Shifting from a fluvial geomorphology lens to FUSE can enable regulators, decision-makers, and practitioners to be more transparent and clearheaded about a full range of potential management objectives (benefits, services, values), social-ecological constraints, and realistic outcomes. Given that water quality is frequently a top priority for stream restoration efforts (e.g., Bernhardt et al. 2007), stakeholders in an urban stream social-ecological-technical system need to recognize the potential water quality benefits (or lack thereof) associated with a range of restoration practices. Bank stabilization, for example, can reduce phosphorus loading (e.g., Newcomer Johnson et al. 2016, Lammers and Bledsoe 2017), while floodplain areas are closely associated with sediment storage and nitrogen uptake and removal (e.g., Groffman et al. 2002, Craig et al. 2008, Helton et al. 2011). In fact, it is not possible to understand the dynamics of adsorbed nutrients without accounting for sediment transport.

Scientists propose that valley-scale stream forms and restoration practices can significantly contribute to water quality as well as habitat benefits (Cluer and Thorne 2014, Wohl et al. 2021). In many urban settings, however, various constraints may impede corridor-scale improvements (e.g., Hess and Johnson 2001), potentially limiting uplift to water quality and aquatic ecosystems. Accordingly, many stream specialists carefully distinguish between "restoration" and various forms of "enhancement" (e.g., renovation, renewal, revitalization, naturalization, etc.), because "restoration" suggests turning back time to a natural, undisturbed condition – impossible in many, if not all, urban streams (e.g., Smith et al. 2016). Nonetheless, transformation of a dying stream to a novel riparian ecosystem that supports terrestrial flora and fauna as well as pollution-tolerant aquatic biota could be experienced as a substantial success for the surrounding community (Moran 2010, Mant et al. 2020), with opportunities for a range of values and benefits. In this way, FUSE can help project goals and objectives to be less myopic and also more realistic.

Environmental equity: Reconciling people with healthy streams

Some interactions between the stream ecosystem pyramid and social sphere of influence have resulted in various environmental inequities associated with race, ethnicity, and income level. Researchers have emphasized flood risk (Debbage 2019, Gourevitch et al. 2021, Wing et al. 2022), water quality (Scarlett et al. 2021, Davis et al. 2022), and greenspace access (e.g., Meerow and Newell 2017), which closely correlate to the HYDRO, CHEM, and BIO vertices of the pyramid, respectively. I suggest the connectivity sphere (Figure 2.3c) at the GEO vertex of the pyramid can represent physical connectivity to floodplains and riparian corridors as well as waterway conveyance, channel stability, etc. In urban watersheds, engineered drainage systems, channel alterations, and landscape elements form a physical infrastructure. By incorporating the additional overlapping environmental equity sphere (Figure 2.3e), FUSE can represent how combined natural and built infrastructure systems underserving Black and Brown communities are physical forms of structural racism. Some communities have been cut off from their natural waterways through storm sewer enclosures, such as the "urban-origin" South River with piped headwaters in Fulton County, Georgia (South River Watershed Alliance 2020). Elsewhere, people are discouraged from riverscape recreation, creating a "culture of avoidance" (Echols 2022a, 2022b) with fear of toxicity. At the other end of the connectivity spectrum we find vulnerable communities disproportionately impacted by too much encroachment of natural floodplain areas (e.g., Debbage 2019, Díaz-Pascacio et al. 2022), an interaction between the HYDRO and GEO vertices. Reflecting upon intersections of inequity near the GEO vertex of the pyramid, we might wonder about management actions linked to property owner complaints of channel erosion (homes and yards) as possible expressions of privilege (race, class, etc.), especially if underserved residents are less likely to reach out for government assistance.

Searching for equitable solutions, pathways towards environmental equity are located in FUSE where the social sphere of influence and stream ecosystem pyramid are closely integrated at the center of the model, helping us recognize the streams and water resources available to us. Environmental equity advocates promote solutions to water quality problems by reconnecting communities to their waterways, especially with recreation like water trails (Echols 2022a) for kayaking and canoeing. Riverscapes can provide increased greenspace access and mobility, settings in which a cultural fearfulness of nature can be transformed into excitement about the natural environment. In this way, cultural benefits (recreation, education, aesthetics, spirituality) can go beyond isolated outcomes to become a driving force for system change through closed feedback loops. Watershed managers should examine the natural drainage network from headwaters to floodplains and open waterways to identify patterns of inequity, for example with risk ratios (Debbage 2019, Selsor et al. 2022). However, shared decision-making is needed to overcome the complex problems of urban streams through bottom-up community involvement (Figure 2.3c) and a culture of inclusion (Figure 2.3e), a centered approach with FUSE.

Flow is not the only master variable: Shifting paradigms and priorities

Just as the stream evolution triangle incorporates biology and geology alongside hydrology as drivers of reach-scale morphology (Castro and Thorne 2019), the stream ecosystem pyramid conceptually reflects linkages between biogeochemistry, hydrogeomorphology, and stream ecology which can be absent or obscured in some conceptual models. FUSE further represents human elements embedded in urban stream systems and a spectrum of benefits, services, and values, not just flood regulation or water supply, as well as the centrality of multi-objective decision-making.

Unlike lost biology, hidden chemistry, and disconnected geology, the HYDRO vertex of the stream ecosystem pyramid has traditionally dominated scientific discourse about and engineering approaches to urban streams. The position of hydrology at the base of the stream functions pyramid (Harman et al. 2012), for example, reflects an established consensus about the importance of flow regimes and streamflow patterns (e.g., Poff et al. 1997), the most effective stream restoration strategies (e.g., Palmer and Bernhardt 2006), and general constraints to potential improvements (e.g., Roni and Beechie 2012). Indeed, hydrologic alterations are known to impact other key stream ecology variables, such as channel morphology and stability (e.g., Annable et al. 2012), sediment regime (e.g., Russell et al. 2018), concentrations of nutrients and other pollutants (e.g., O'Driscoll et al. 2010, Jefferson et al. 2017), and aquatic biota (e.g., Walsh et al. 2016). Moreover, hydrology remains a moving target for urban stream designers on account of ongoing land use and climate changes.

The selected descriptions of common disconnects, management gaps, and fractured systems may be familiar to stream practitioners and scientists, and other practical examples could be similarly explained as multi-faceted interactions between the social sphere of influence and stream ecosystem pyramid components. FUSE facilitates recognition of system vulnerabilities and shortcomings by including the missing pieces. Instead of dividing the world along the default lines of stormwater, floodplain, conservation, recreation, or development expertise, for example, FUSE provides a starting point to explore interconnections with potential management synergies.

Imagining natural infrastructure (NI) innovations with FUSE

The key components and connections of FUSE support a social-ecological framework for successful urban stream restoration and renovation. With NI as the backdrop and social elements in the foreground, FUSE keeps all of these dimensions present in the mind's eye, helping us to articulate questions, study problems, explore alternative solutions, and advance science and practice through transdisciplinary learning. While the balanced version of FUSE (Figure 2.4b) provides a useful mental scaffolding for NI that provides a full range of positive processes and outcomes, other configurations (e.g., Figure 2.4c) help us think about how imbalance between the social sphere of influence and stream ecosystem pyramid are manifested in the stream and in the community. For example, scientists can imagine how the FUSE imbalance could result in different states in various geomorphic or water quality classifications in a stream restoration context. Researchers have broadly described three axes of restoration success: ecological functions, human benefits, and learning processes (Palmer et al. 2005). Given existing social-ecological system constraints on aquatic ecosystems and potential uplift in urban settings, a shift from ecological restoration to urban stream renovation emphasizes social and educational successes. FUSE represents the holistic urban riverscape context needed for learning how to achieve realistic social-ecological successes across the axes: visioning, implementing, and evaluating outcomes.

Benefits, services, and values are emergent system properties where the stream ecosystem pyramid intersects the social sphere of influence. At the outermost shell of the benefits sphere (Figure 2.3f), environmental risks are manifestations of dominant themes of an urban stream social-ecological-technical system, namely flooding and water quality, and top-down management approaches to problems resulting from disbenefits. Closer to the center of the social sphere, connections to cultural benefits (recreation, aesthetics, education, spirituality) can serve as bottom-up, grassroots entry points to stream improvements through holistic social integration and community inclusion in planning and problem-solving. Conventional stream restoration projects for ecological restoration illustrate a similar spectrum of motivational differences, with EPA water quality regulation leading to mitigation banking and associated controversies (Lave et al. 2008, Lave and Doyle 2021), contrasted with outdoor recreational and environmental conservation organizations (Ducks Unlimited, Trout Unlimited, etc.) providing alternative funding sources. Leaning into urban stream riverscape services and benefits can facilitate positive system changes by encouraging greater alignment between popular conceptions of healthy, beautiful streams and functionally vibrant ecosystems, shifting away from “virtual” surface appearances (Wohl 2001) towards more ecologically resilient riparian corridors providing habitat and water quality benefits (Cluer and Thorne 2014). The balanced FUSE (Figure 2.4b) represents sustainability evidenced by social-ecological harmony and reciprocal states of wellbeing.

Socio-environmental learning opportunities

As a tool to help us organize our thinking, FUSE can support teaching, learning, and knowledge sharing in contexts ranging from formal education and professional development to natural science and social collaboration. Serving as a shared frame of reference, FUSE provides an educational scaffolding for connecting what we know to what we would like to learn, and a lens for developing and testing hypotheses about the relationships between the model components. For example, we may seek improved understandings of social-ecological constraints, change and response mechanisms, innovation opportunities, and shared decision-making. An educational framework like Bloom’s taxonomy of learning (e.g., Forehand 2010) can be understood as a social sphere with a progression from remembering facts and basic concepts to a central focus on integrative evaluations and creative actions such as urban stream renovation projects. Social learning for resilience (de Kraker 2017) begins with the urban riverscape setting, including the biophysical aspects of the stream ecosystem and the actors within the social spheres, with potential processes and outcomes that can lead to a more resilient social-ecological-technical system.

FUSE can provide a touchstone for developing inquiries, such as questions about spatial scale, transdisciplinary thinking, and environmental equity. The relative influence of a social sphere upon the stream ecosystem can be characterized with specific aspects of the pyramid vertices (e.g., geomorphic classification) as well as more holistic system properties (e.g., water quality, ecological functions). For example, researchers frequently emphasize watershed-scale stream restoration (e.g., Polvi et al. 2020), with strategies corresponding to the pyramid vertices, including stormwater control structures (HYDRO), nonpoint source pollution prevention (CHEM), and green infrastructure (BIO). The GEO vertex is associated with less common landscape approaches that may warrant additional investigation, such as permeable soils contrasted with subgrade compaction, or sediment budgets and subsidies.

As a boundary object, FUSE can cut across siloes common to watershed management and regulatory agencies. For example, municipal stormwater management and flood risk mitigation departments could begin communicating about how to integrate missions incorporating a broad array of benefits, values, and services, and all stakeholders can reflect upon how inter-jurisdictional hydrology and water quality issues affect mental maps of urban stream management. The holistic FUSE supports planners and designers from different departments who engage in self-inquiry about the overlap between healthy streams and healthy people, because single factor management results in dysfunctional systems that are far from their potential capacity for a full range of benefits. A shared communication tool enhances the ability to conduct integrative urban stream management and identify societal, technical, and ecological barriers to improving urban stream functions and services. Furthermore, FUSE helps decision makers to overcome barriers by linking action strategies to environmental stewardship. Considering the environmental equity sphere can help stakeholders communicate key questions and challenges in the context of urban stream management, engaging in collaborative learning about sustainable riverscape spaces for historically marginalized groups.

NI and new fusions of ideas

FUSE is compatible with frameworks for NI, Engineering With Nature®, nature-based solutions, and ecological engineering. In the context of urban streams, NI includes natural systems such as floodplains, riparian corridors, streams, and riverine wetlands supporting human wellbeing, health, and happiness in addition to flood risk reduction, mobility, ecological habitat, and water quality. NI performs engineering functions and services, and it can be strategically combined with conventional infrastructure such as reservoirs and levees to produce an array of social, environmental, and economic benefits. The FUSE representation of NI systems are the portions of the stream ecosystem supporting society and providing the benefits and services shown in Figure 2.3f. For example, Engineering With Nature® encompasses processes and actions taken to conserve, create, enhance, and restore NI, like solutions to riverine flooding including levee setbacks, watershed reforestation, and hydrologic reconnection between channel and floodplain (World Bank 2017, Whelchel et al. 2018). In fact, NI can be more resilient and cost-effective than conventional infrastructure or otherwise enhance the performance of traditional engineering works through hybrid applications (Browder et al. 2019).

In contrast with many stream restoration methods primarily focused on channel design, NI includes the entire riparian corridor as well as the contributing watershed, so it is inherently tied to spatial scale (Figure 2.3b). Similar to proponents of watershed-scale stream restoration for water quality or ecological improvement, some have advised that landscape-scale NI is required for effective flood risk mitigation (e.g., Dadson et al. 2017). However, I would suggest that "big things can come in small packages", particularly when NI is able to provide a spectrum of benefits (Figure 2.3f) incorporating multiple vertices of the stream ecosystem pyramid. If a multi-criteria decision analysis embodies the spirit of FUSE, then it should help point us to those opportunities in which small actions may have a disproportionately large effect. For example, a community disproportionately impacted by riverine flooding (Figure 2.3e) could benefit from localized flood risk reduction (HYDRO) with a floodplain scale NI that provides greenspace access (BIO) and improves surface water quality (CHEM) by disconnecting residential areas from the natural floodplain (GEO). In this context, "building back better" could mean turning flood-prone areas into natural recreation spaces that support health and wellbeing, while proactively working to decrease gentrification risk through policy applications (e.g., progressive property tax mechanisms). Thus, FUSE can support NI planning and prioritization as stakeholders consider what benefits, services and functions (e.g., aesthetics, tree cover, wildlife habitat, urban heat island reduction) they value from their urban streams as well as environmental risks that need to be addressed (e.g., flooding, erosion / land loss, wastewater leaks or overflows, trash).

Dynamic by design: Urban river-wetland corridors

To illustrate how FUSE can be applied as an NI learning tool for co-production and co-creation, I propose exploring innovative approaches to urban stream restoration. Currently, natural channel design and bank stabilization dominate restoration approaches, both of which can be characterized as static, quasi-equilibrium, single-thread channels with somewhat limited water quality and ecological benefits (Cluer and Thorne 2014), in contrast with dynamic guiding images (Palmer et al. 2005). However, researchers and practitioners have begun advocating more dynamic practices, such as self-forming streams (Mecklenburg 2008), self-healing (Kondolf 2011), beaver (Pollock et al. 2014, Castro et al. 2015), Stage 0/8 (Cluer and Thorne 2014, Powers et al. 2019), low-tech process-based restoration (Wheaton et al. 2019), and biomic river restoration (Johnson et al. 2020). Many assume that the physical constraints of urban settings limit possible interventions to conventional single-thread channels, locked into place through artificial hardening or bioengineered bank stabilization (e.g., Kondolf 2011, Johnson et al. 2020), and indeed this is true in many instances. However, I would argue that novel exceptions to the general rule of thumb are possible, potential riverscape "pearls on a string" that can provide a wealth of benefits and drive positive social-ecological system changes through feedback loops.

I would suggest that a term like "freedom space" (Biron et al. 2014) is well-suited to convey key biochemical and hydrogeomorphic aspects of dynamic riverscape restoration practices, improve stakeholder understandings of what constitutes a "healthy" stream, and enhance public perceptions of novel techniques. As one of the drivers of the urban stream syndrome (Figure 2.2b), GEO corresponds to channel alterations, artificial hardening, and sediment regime (Table 2.1). A geology dominated urban stream can also reflect geomorphic evolution stages associated with incision, floodplain disconnection, arrested degradation, channel entrenchment, and valley confinement (Cluer and Thorne 2014, Booth and Fischenich 2015). In contrast, decreasing the artificial geology influence corresponds to increased geomorphic freedom, reduced bank erosion, and either improved sediment balance or potential deposition and storage (Kondolf 2011, Dust and Wohl 2012). Where enclosed storm sewer systems define existing stream geology and geomorphology (urban karst), creation of a dynamic riverscape features stream daylighting with ample floodplain connectivity.

Figure 2.5 could be contextualized to show how creating freedom space improves geomorphic functions that can lead to improved floodplain and groundwater connectivity (GEO/HYDRO), biotic influences (GEO/BIO), and water quality benefits (GEO/CHEM) associated with river-wetland corridors. With respect to the HYDRO vertex, for example, dynamic riverscapes can potentially provide localized flood regulation with lower water surface elevations similar to levee setbacks, while the improved floodplain connectivity promotes infiltration, hydrologic storage time, and groundwater recharge, similar to natural wetlands. Reflecting upon the benefits sphere (Figure 2.3f), we can consider additional co-benefits associated with dynamic riverscapes in which both people and streams are free to move and interact, although potential cultural services may differ from traditional single-thread channels in urban environments. Both approaches can be compatible with greenway trails for recreational use, but freedom space aesthetics may seem more natural, a "rambunctious" floodplain (Marris 2013) or pocket of wilderness where beaver would be welcome, for example. Urban riverscapes that are dynamic by design have tremendous potential as outdoor classrooms, too, by shifting popular conceptions away from single-thread, "virtual" streams and towards more holistic intuitions about healthy waterways. Freedom spaces with multi-thread channels can capture the human imagination, enhancing sense of place while reflecting the interconnectedness of life.

While researchers and practitioners have already started to explore the physical and ecological mechanics of dynamic stream restoration approaches, we still have much to learn about how to create freedom spaces in urban riverscapes from a human standpoint, including social justice and management implications. FUSE can help stakeholders consider questions

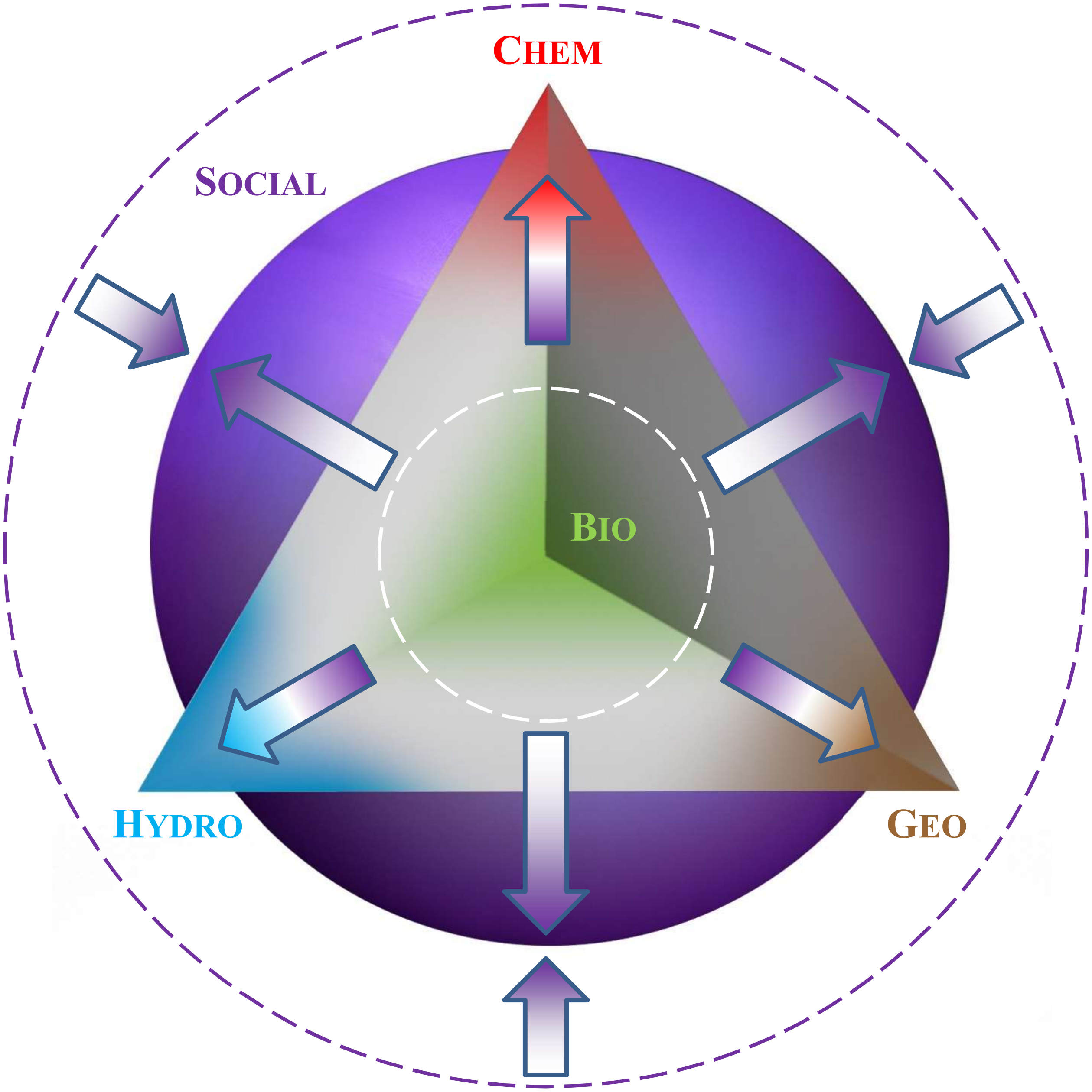


Figure 2.5. FUSE helps managers think holistically about where interventions are needed. Looking at the social sphere of influence relative to the stream ecosystem pyramid can help with prioritizing challenges and actions. For a sustainable urban riverscape system, all vertices of the pyramid need to be functioning, and the sphere needs to be balanced. FUSE is a communication tool about working on all of the key pieces of the social-ecological-technical system. The challenges at the outer edge of the social sphere need to be addressed, working from the outside in. At the same time, it is possible to support positive social changes, improve stream ecosystem functions, and increase benefits and services by working from the inside out. Not shown is the arrow corresponding to BIO improvements. The following scenarios begin with an imbalanced system (Figures 2.2b, 2.4c).

Balanced Floodplain Management: A city explores causes, effects, and potential solutions to localized flooding. Entry points to FUSE are the HYDRO vertex and floodplain management (Figures 2.3a,b). Initial evidence of system imbalance includes the emphasis on environmental risk (Figure 2.3f) and a pattern of high-impact development (Figure 2.3d) at the watershed scale (Figure 2.3b). FUSE suggests lines of related inquiry: causal physical links include loss of watershed vegetation (BIO/HYDRO) and floodplain encroachment (GEO/HYDRO), while altered hydrology affects fecal coliform levels (HYDRO/CHEM). More balanced flood risk assessment needs to include environmental equity analyses (Figure 2.3e) and incorporate public exposure to water pollution (CHEM). NI solutions like floodplain reconnection (GEO/HYDRO) also need to address social issues. Efforts to disconnect neighborhoods from flood risk could be strengthened by increasing social connectivity to potential riverine corridor benefits (Figures 2.3c,f).

Dynamic River-Wetland Corridors: Imbalances between the social sphere of influence and stream ecosystem pyramid are linked to unhealthy social-ecological-technical system states. For example, geomorphological (GEO) manifestations of disharmony include ad hoc bank armoring and waterway entombment. FUSE serves as a shared mental map to consider NI innovations and social co-production that can help with overall system alignment and sustainability. Shifting towards the GEO vertex corresponds to a more dynamic river-wetland corridor. Stream daylighting with access to the floodplain valley improves groundwater recharge (GEO/HYDRO), nutrient processing (GEO/CHEM), and ecological habitat (GEO/BIO). Stakeholders seek to create an urban riverscape at the floodplain scale (Figure 2.3b) that is dynamic by design, where both the stream and people are free to move and interact. Increased social connectivity (Figure 2.3c) with naturally “messy” spaces needs additional landscape design elements to create safe boundaries. New outdoor education and recreation opportunities support positive social-ecological system feedback through public experience with “healthy” streams.

Water Quality Equity: In FUSE, one barrier to equity (Figure 2.3e) exists where the urban stream syndrome is exacerbated by watershed management (Figures 2.3a,b). As regulatory leverage, when a waterbody gets on the 303d list, it is assigned total maximum daily loads (TMDLs) associated with the CHEM vertex, like fecal coliform, nutrients, and dissolved oxygen. However, most impaired waterbodies do not get listed, because we use standards and criteria to assess whether waterbodies are supporting their designated uses like recreation (Figure 2.3f), aquatic life (BIO), etc. Urban streams are partially supporting or in most cases not supporting their designated uses but in purgatory in terms of 303d listing. In the absence of fish and other biota (BIO), an effective change mechanism is positive connectivity with benefits (Figures 2.3c,f) like water trails for canoeing and kayaking. Supporting underserved communities with outdoor recreation programs can increase political capital for water quality monitoring as well as stream and watershed improvements, pushing the social-ecological-technical system in a positive direction.

about how social equity interacts with stormwater and floodplain management, and other urban stream issues. For example, the potential relationship between environmental equity and stream reconnection described earlier parallels a FUSE shift towards the GEO vertex (Figure 2.5) that characterizes riverscape freedom spaces. Although lack of greenspace access is one expression of environmental inequity, urban stream restoration projects in park settings or parallel to greenway trails carry the risks of gentrification and neighborhood displacement (e.g., Jelks et al. 2021). We could consider how dynamic riverscapes might be perceived by various communities, and if there might be differences compared to more traditional stream restoration approaches. In reality, healthy streams are often messy streams, and messy streams can be scary, so it will be critical to provide visual examples to help people understand freedom space as a potential stream improvement alternative, either with or without parks and recreational trails. A landscape architecture concept based upon "messy ecosystems" with "orderly frames" (Nassauer 1995) could be the critical link to connect riverscape freedom spaces with positive human experiences and perceptions.

Conclusions

FUSE improves understanding of system constraints, mechanisms of change and response, and opportunities for innovations, while incorporating social dimensions for goal-setting and decision-making. The vertices of the stream ecosystem pyramid represent biogeochemical factors in an NI construct, including stream functions and human benefits, while the social sphere of influence depicts the embedded nature of human interactions and intertwined aspects of the social-ecological-technical system. The graphical efficiency of FUSE combined with its flexibility can be leveraged in a variety of ways to capture the complexities underlying multi-objective decision making, regulatory policies, and management strategies. A novel aspect of the model is the integration of established components from the natural and social sciences specific to urban streams with an ecological engineering focus on benefits and practices. The chemistry axis of the stream pyramid expands an emphasis on fluvial geomorphology to a wider range of biogeochemical variables, supporting feedback loops that influence ecological structures and functions. The social sphere portrays human aspects of urban stream systems, and the combined FUSE reflects the presence of people as active participants rather than passive external drivers, using NI for environmental risk reduction and positive benefits like recreation, mobility, wellbeing, and sense of place.

As a shared thinking space, FUSE enhances communication between managers, organizations, and diverse stakeholders to holistically approach problems and possibilities in urban riverscapes. The strength of this conceptual model is the multidisciplinary inclusion of both social and ecological drivers of benefits, services, and values associated with urban streams. FUSE integrates dominant themes from science and practice, serving as a framework for collaborative management and knowledge co-production, while encouraging deeper thinking about the social-ecological system context, components, and interactions. By incorporating system states, spatial scales, and potential interventions, FUSE supports balanced and equitable approaches to stream and watershed management, with practical applications such as multi-objective decision making, NI innovations, and social learning.

While FUSE was developed specifically for urban stream social-ecological complexities, the fused stream ecosystem pyramid and social sphere of influence can also be applied in non-urban settings to support a full range of stream restoration and mitigation efforts and avenues for interdisciplinary research. Moreover, the qualitative conceptual model presented here could be adapted for quantitative purposes. For example, the HYDRO, GEO, BIO, and CHEM vertices of the stream ecosystem pyramid can align with generalized components of physical, biological, and chemical conditions (e.g., Jackson and Pringle 2010), restoration constraints and limiting factors, (e.g., Suding et al. 2004, Kondolf 2011), multivariable stream function assessments (e.g., Harman et al. 2012, Murphy et al. 2022), and environmental benefits (e.g., Hoang et al. 2016). While the pyramid could be geometrically simplified by combining HYDRO and GEO together as a single physical driver of ecosystem structures and processes, the proposed configuration best complements established frameworks and conceptual models from stream ecology and fluvial geomorphology (e.g., Wenger et al. 2009, Harman et al. 2012, Castro and Thorne 2019), captures geomorphic boundary conditions (e.g., lithotopo unit, physiographic setting), and emphasizes both water and sediment regimes for ecosystem management (e.g., Wohl et al. 2015, Hawley and Vietz 2016). FUSE is a simplified way to represent an extremely complicated system, but also flexible and inclusive like the stream evolution triangle (Castro and Thorne 2019), and thus helpful for thinking and communicating about urban riverscapes.

CHAPTER 3

INTEGRATED URBAN RIVERSCAPE PLANNING: SPATIAL PRIORITIZATION FOR EQUITABLE NATURAL INFRASTRUCTURE[2]

[2] Yaryan Hall, H.R. and B.P. Bledsoe. To be submitted to *Journal of Water Resources Planning and Management (ASCE)*.

Abstract

Natural infrastructure (NI) and nature-based solutions in urban riverscapes can provide a spectrum of environmental, societal, and economic benefits, but widespread implementation of NI remains limited due to their context-dependent nature. Windows of opportunity have opened through legislation and funding to expand NI solutions that address flooding, water quality, air pollution, extreme heat, and environmental equity. System-level approaches may offer these projects a framework that is flexible yet holistic enough to streamline implementation. In fact, a systems approach is essential to realize the potential of NI for equitably achieving these goals. The purpose of this study was to support decision-makers and managers in prioritizing their conservation and capital investments in urban riverscapes for flood risk reduction, water quality improvement, and ecosystem restoration in addition to identifying and leveraging social co-benefits. A spatial multi-criteria decision analysis (MCDA) is well suited to landscape-scale environmental risk management and scenario comparisons, as it provides a logical and transparent way to incorporate multiple, competing goals and priorities from a variety of stakeholder groups. We conducted an urban stream spatial MCDA case study with Charlotte-Mecklenburg Storm Water Services to guide equitable and efficient stream reach, floodplain, and watershed interventions. Our study assessed social and ecological characteristics of the system and prioritized watersheds and sub-basins for potential NI using a spatial MCDA. We developed an urban stream prioritization framework that could be tailored to complement existing management strategies and also more broadly implemented in other social-ecological systems.

Introduction

Surface water managers and organizations face multiple complex challenges in urban riverscapes, particularly flooding, water quality, and associated environmental equity concerns.

Natural infrastructure (NI) and nature-based solutions can improve all of these problems while providing greenspace with wildlife habitat and ample social benefits, like recreation, education, and other cultural values (Whelchel et al. 2018, O'Donnell et al. 2020, Wheaton and Skidmore 2022). While the ecological potential of urban streams may be limited compared to non-urban settings (e.g., Roy et al. 2003, O'Driscoll et al. 2010, Sterling et al. 2016), there nonetheless may be opportunities to improve riparian and aquatic ecosystems to better support a full range of benefits (Wenger et al. 2009). In addition to targeting flood risk reduction, water quality and ecosystem enhancement, combining NI with conventional systems can achieve a broad array of co-benefits like better health outcomes through outdoor recreation, improved air quality, and regulation of extreme heat (Jackson et al. 2014, Meerow and Newell 2017, Simperler et al. 2020). Moreover, implementation of the Infrastructure Investment and Jobs Act and similar legislation presents a window of opportunity for expanding NI to address environmental hazards and social justice. Significant funding is increasingly available for generational transformation in the context of greater awareness of environmental inequities and the untapped potential of hybrid natural and conventional infrastructure to achieve multiple objectives. However, creating resilient urban riverscapes through resources like the Infrastructure Investment and Jobs Act requires a strategic approach to doing "the right projects, the right way" (ASCE 2022).

The overwhelming scope of urban riverscape problems, possibilities, and priorities is exacerbated by multiple spatial scales (watershed, floodplain, channel). Unfortunately, spatial mismatches commonly manifest as fragmented stream management approaches and departmental "silos," even in municipalities and utilities with relatively strong programs and planning capacity. At the same time, social injustice (e.g., environmental racism) is often visible through spatial relationships between neighborhood demographics and environmental risks (e.g., Pulido 2000, Debbage 2019). Thus, planning and implementing equitable, multifunctional NI solutions requires practical, straightforward spatial analysis tools that can help a range of stakeholder groups to better understand, prioritize, and address the intertwined complexities of urban streams and watersheds.

Flood risk reduction arguably has been the dominant focus in urban stream systems, likely due to the direct, negative economic impacts of flood events. Floodplain managers and researchers often employ economic cost benefit analyses to identify priorities and guide projects (e.g., ten Veldhuis 2011), and much of applied research has emphasized decision support tools (e.g., Habersack et al. 2014, Hammond et al. 2015, Whelchel et al. 2018). Floodplain hydrology and hydraulics encompass multiple spatial scales: the primary focus is a medium valley or floodplain scale, and localized hotspots are handled at the smaller channel or reach scale, but land-use factors and various nature-based solutions to flood risks for many storm events are best understood at the larger landscape or watershed scale (Cohen-Shacham et al. 2019).

Water quality regulation is typically completely separated from floodplain management by organizational structures and missions, despite close coupling via riparian management, and it is more likely to be driven by policies stemming from the Clean Water Act than the direct economics of environmental risk reduction. Aquatic insects and fish are often used as holistic indicators of overall water quality, and ecological uplift is a frequent objective for stream restoration efforts (e.g., Palmer et al. 2014, Smith et al. 2016). Many water quality problems are generally tied to nonpoint source pollution at the largest watershed-scale in urban and agricultural settings (e.g., Kaushal et al. 2018, Stets et al. 2020), while medium-scale riparian buffers may be implemented to pre-treat stormwater runoff, and monitoring is performed at the reach scale. Conversely, ecological restoration frequently focuses on reach-scale channel geomorphology (e.g., Bernhardt et al. 2007), though growing attention has also been given to potential floodplain and watershed scale interventions to improve water quality and ultimately aquatic biology (e.g., Polvi et al. 2020).

The consequences of flood damages and poor water quality (i.e., pollution) are further compounded by long-standing social inequities. For example, recent studies found increased flood risk among Black, Hispanic, and low-income populations in the southeastern US (Debbage 2019, Selsor et al. 2022), while others have documented greater exposure to water pollution threats (Davis et al. 2022). At the same time, underserved communities are less likely to have access to greenspace amenities and the positive benefits associated with riverine corridors (Smardon et al. 2018), with neighborhoods at greater risk for gentrification and displacement (Jelks et al. 2021).

In addition to environmental risk assessment and reduction, stream management practices and research have begun to incorporate several key themes: nature-based solutions (including NI), additional co-benefits, shared decision making, and social equity. For example, "blue-green cities," Engineering With Nature®, and other similar visions emphasize multiple social-ecological benefits in addition to sustainable flood risk mitigation, such as water quality, ecosystem support, and outdoor recreation opportunities (Bridges et al. 2018, 2021, Mant et al. 2020, Sowińska-Świerkosz and García 2021, U.S. Army Corps of Engineers 2022). In this paper, we describe a spatial multi-criteria decision analysis (MCDA) that we developed for urban riverscapes. Such an approach is well-suited to addressing complex problems like watershed and stream prioritization, because it offers a flexible, transparent way for stakeholders to select projects and evaluate potential alternatives while combining a range of variable inputs across multiple spatial scales, thereby allocating capital resources and/or seeking funding opportunities.

Under the umbrella of shared decision making, conventional MCDA tools are broadly applied and documented for environmental management (e.g., Kiker et al. 2005, Linkov et al. 2011), although they may alternately be called multi-criteria evaluation approaches or decision support systems (e.g., Renaud et al. 2016, Meerow and Newell 2017). The participatory aspects of MCDAs further include collaborative modeling (Evers et al. 2018) and citizen perceptions (Hong and Chang 2020). At the cutting edge of applied research, however, GIS-based prioritization and spatial MCDAs are especially relevant for stream and watershed improvements that provide a wide range of benefits, services, and values. Recent urban case studies of green infrastructure (Meerow and Newell 2017) and flood management (Vercruysse et al. 2019) have focused on benefit evaluation (Hoang et al. 2016), spatial planning (Meerow and Newell 2017), site suitability (Vercruysse et al. 2019), and project alternatives (Lim and Lee 2009). To our knowledge, however, past efforts have not targeted urban riverscapes with a multi-scale integration of stream restoration (renovation, revitalization, naturalization, etc.), flood mitigation, watershed management, and social objectives. While social vulnerability and resilience continue to be popular themes (e.g., Meerow and Newell 2017, Evers et al. 2018), we have seen no examples of riverscape spatial prioritization that explicitly tackles measurable environmental inequities. Overlooking social equity when planning for improvements to natural and built infrastructure ultimately perpetuates systemic racism and other forms of environmental injustice.

Our primary objective for the current study was to develop a spatial MCDA that includes flood risk reduction, water quality improvements, social-ecological benefits, and environmental equity, thereby facilitating system-level prioritization for the "right projects" in urban riverscapes. In addition to identifying system hotspots, we wanted to investigate potential synergies and tradeoffs among the various criteria. We intended to create a practical and transferable framework that could eventually be used to support shared decision-making, evaluate local project alternatives, and design sustainable NI solutions. Using adaptable social metrics, it was our intent to apply an urban stream management strategy that could be flexibly implemented in a variety of environmental equity contexts. Our overarching goal is to provide water managers and urban riverscape communities with useful tools to incorporate multiple planning objectives, prioritize opportunities, and leverage NI to achieve efficient and equitable benefits and services.

Methods

Spatial prioritization case study

For our spatial MCDA application, we collaborated with the City of Charlotte and Mecklenburg County in North Carolina (Figure 3.1), located in the southeastern piedmont region along the Charlanta megaregion. Similar to other municipalities, Charlotte-Mecklenburg Storm Water Services (CMSWS) divides surface water management responsibilities among several main groups, including watershed planning, engineering and flood mitigation, and water quality. Some of the existing CMSWS approaches to prioritization include a watershed-scale water quality matrix (Hunt 2022), building-level flood risk assessment / risk reduction (RARR) tool (Charlotte-Mecklenburg Storm Water Services 2020), and reach-scale stream restoration ranking system (SRRS) (Mecklenburg County Storm Water Services 2021), all of which are supported by extensive spatial data. When planning stream improvements, CMSWS often partners with the Mecklenburg County Park and Recreation Department to incorporate greenway trails and other outdoor amenities. Mecklenburg County recently developed an Equity Action Plan, and CMSWS wanted to better understand how they could include social components with their surface water

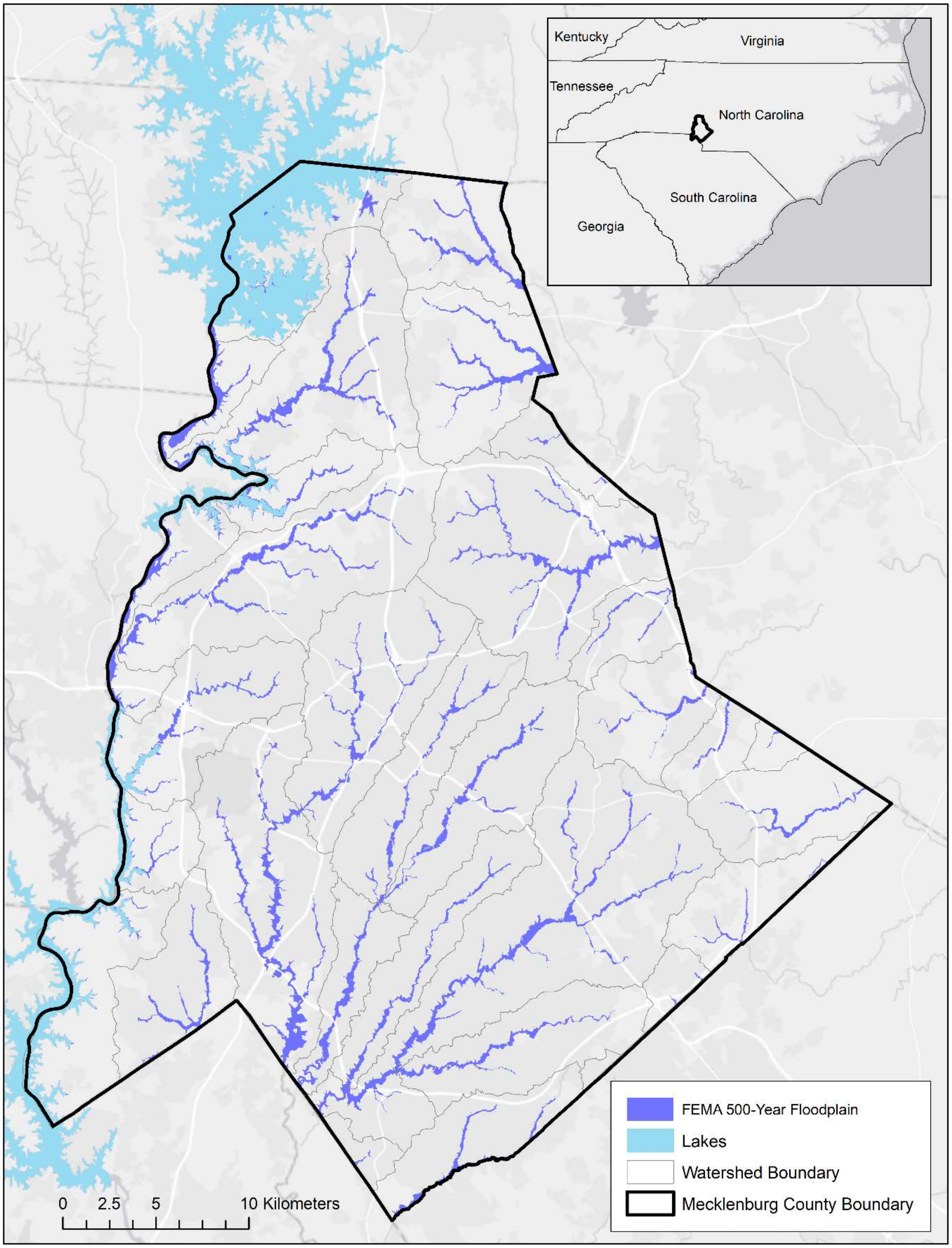


Figure 3.1. Mecklenburg County watersheds. The western and southern portions of the county drains to the Catawba River Basin, and the eastern streams are tributaries to the Yadkin River. The 500-year floodplain excludes the reservoirs along the western boundary (Lake Norman, Lake Wylie). The study focused on the portions of 33 watersheds and their sub-basins within the county boundaries.

projects and initiatives. In general, our goal was to integrate the established CMSWS components (RARR, SRRS, etc.) while taking steps to identify environmental equity objectives. At the same time, our intent was to develop a generalized approach that could be broadly applied in other urban stream systems.

With a structured decision-making process, stakeholder groups typically work collaboratively to identify objectives and possible metrics (Bridges et al. 2015). This research was intended to support CMSWS with early planning and outreach initiatives by identifying areas for follow-up with neighborhoods and communities. Through conversations with CMSWS, we elicited baseline criteria and sub-criteria for the spatial MCDA, which can be modified later as part of collaborative conversations. For analysis at the watershed and sub-basin scales, the main criteria included three "riverscape" criteria (flood regulation, water quality regulation, ecosystem support) as well as amenity access and environmental justice (Table 3.1). We used a two-part approach to environmental justice by including measures of social vulnerability and historic injustice in a general "landscape" category, complemented by sub-criteria in riverscape categories targeting specific aspects of social inequity, such as disproportionate exposure to flood risk or lack of greenspace access. Following prioritization at the large (watershed) and medium (sub-basin) spatial scales, the same objectives may be combined with additional feasibility criteria to evaluate local project alternatives. Preferences from multiple stakeholder groups can be separately elicited and then weighted together, and a tradeoff matrix can be used to show the highest rated alternative for each group (Bridges et al. 2015).

Data acquisition

As part of the flood regulation objective, we prioritized areas based on flood damage risks and hazards to human life – locations with high flood impact and probability, and

Table 3.1. Spatial multi-criteria decision analysis (MCDA) priority criteria, sub-criteria, and data descriptions.

Criteria	Sub-criteria	Data Description
Riverscape Criteria		
Flood Regulation	- Flood risk score - Flood risk equity	- Risk Assessment / Risk Reduction (RARR) building polygons[1] - RARR 500-year flood polygons[1]
Water Quality Regulation	- Fecal coliform bacteria - Turbidity - Regulatory status - Pollution risk equity	- Water quality matrix[1] - 303d list[1] - Water quality buffer polygons[2]
Ecosystem Support	- Channel stability and habitat - Riparian buffers	- Stream Restoration Ranking System (SRRS) scores (polylines)[1]
Landscape Criteria		
Amenity Access	- Near outdoor recreation - Amenity benefit equity	- Neighborhoods polygons/table[1]
Environmental Justice	- Social vulnerability index (SVI) & housing change score - Historic redline areas - Population density	- SVI polygons[1] - Home Owners' Loan Corporation (HOLC) Redlining polygons[3] - Neighborhoods polygons/table[1]
General Data		
Various	- Watersheds, sub-basins - Parcels - Developed areas - Census blocks, block groups, tracts	- Watershed, sub-basin polygons[1] - Parcel polygons[2] - 2019 landcover raster[4] - Population demographics polygons[2] and tables[5]

Data Sources: (1) Charlotte-Mecklenburg Storm Water Services (personal communication), (2) Mecklenburg County Open Mapping, (3) ArcGIS Online, (4) NLCD, (5) US Decennial Census and American Community Survey. Combinations of various datasets were used to calculate risk and benefit equity layers.

sensitivity to depth and velocity. We extracted parcels overlapping the 500-year flood hazard zone, and those having buildings with RARR scores, which were based on numerous variables, such as water surface and building elevations, flow depth-velocity zones, accessibility and parking, and residential building types (e.g., single- or multi-family). The RARR total risk score is a single value that incorporates multiple flood events (2, 5, 10, 25, 50, 100, and 500 year) to account for major risks that are relatively rare plus lesser risks that occur more often and have potential damages that accumulate over time (ten Veldhuis 2011). All of these GIS layers were from the CMSWS RARR dataset.

Under the water quality category, the MCDA structure emphasizes streams with watershed impairments based on monitored levels of fecal coliform and turbidity as well as regulatory status. Our data sources included the water quality matrix and 303(d) list, water quality buffers (35 to 100 feet), and 500-year flood hazard zone shapefiles provided by CMSWS. We used the flood zone and water quality buffers to define potential exposure of homes and properties to surface water pollution. In determining fecal coliform and turbidity levels, we used supplementary monitoring data for 6 watersheds (Goose, Rocky River, Clear, McDowell, Clarke, Gar): we interpreted the average fecal coliform levels as non-compliant or severe in all 6 watersheds, and the turbidity levels were compliant only in Goose and Car Creek Watersheds.

We divided ecosystem support into aquatic and riparian subcategories, assigning the highest priorities to streams needing improvements in channel stability and habitat conditions (aquatic ecosystem) as well as buffer vegetation (riparian ecosystem). While the SRRS program includes both desktop and field components, we used only existing desktop data and scores provided by CMSWS. For aquatic ecosystem support, we limited the potential priorities to the SRRS group of reaches recommended for restoration on the basis of stream functions and constructability, which comprised 58 percent of all stream miles (Mecklenburg County Storm Water Services 2021). However, we did not make the same distinction for the riparian corridor, because there might be room for improvement with a vegetated buffer, even if the channel itself was deemed unsuitable. While we opted to prioritize areas with poor riparian buffers that need improvement, this particular metric could also be a predictor of potential aquatic ecological uplift, so corridors with high buffer scores alternatively could be used to prioritize opportunities, not just deficiencies.

The amenity access criteria were identified as areas in the landscape with relatively fewer benefits based on an existing countywide layer for neighborhoods near public outdoor recreation. CMSWS indicated this data layer was a proximity analysis, with the percentage of housing units within ½-mile of an outdoor public recreation area. Elsewhere, a similar spatial layer could be created by starting with an entire dataset of housing units, determining how many are in the ½-mile proximity, and then overlaying the watershed geographies to get the percentages within the watersheds and sub-basins. While CMSWS doesn't have a housing unit layer, the proximity analysis could perhaps be based on a zoning layer, or otherwise use a previously studied approach to greenspace access (e.g., Meerow and Newell 2017). The watersheds and sub-basins we used were delineated by CMSWS.

Under the general environmental justice objective, higher priority areas included those with high social vulnerability as well as historically redlined neighborhoods. As a metric for social vulnerability, the Charlotte Housing Authority uses a neighborhood-level "change score", and this was supplemented by the standard CDC social vulnerability index (SVI) in portions of Mecklenburg County outside of the city limits. The neighborhood change score identifies areas most vulnerable to gentrification and displacement based on income level and housing changes (sales prices, permit volumes). SVI, on the other hand, incorporates multiple variables from the American Community Survey: socioeconomic and minority status, household type and composition, disability, language, and transportation. While we included population-density here, it is also possible to calculate population densities just within flood-prone areas or surface water quality exposure.

Additional data used for various sub-criteria included the 2019 National Land Cover Dataset (NLCD), stormwater watershed and sub-basin polygons, and the 2020 US Decennial Census and American Community Survey (census blocks, block groups, and tracts). Demographic data for race and ethnicity were available at the smallest census block scale, while income (above or below poverty level) was only available from the American Community Survey at the larger census block group scale. The studied populations included 45% White (non-Hispanic), 29% Black (non-Hispanic), 15% Hispanic, 85% non-Hispanic, 10% below poverty level, and 90% above poverty level in Mecklenburg County. Table 3.1 summarizes the data descriptions and sources for the spatial MCDA criteria and sub-criteria.

Environmental equity metrics

Spatial prioritization approaches that incorporate social objectives typically use some type of demographic-based SVI (e.g., Meerow and Newell 2017), and environmental justice is especially important for the vulnerable communities identified by SVI. Environmental equity, on the other hand, relates to the distribution of risks and benefits, and metrics involve both social vulnerability (e.g., race, ethnicity, income) and either exposure or access, respectively. This is because environmental hazards to human health and wellbeing (e.g., flood damage, water pollution) also involve exposure, just as environmental benefits (e.g., greenway trails, outdoor education) require access opportunities. For example, a poor neighborhood is socially vulnerable in a general sense, whereas a poor neighborhood in a low-lying area floodplain area is both socially vulnerable and at risk of exposure to a flooding.

Our approach to environmental equity built upon prior methods of analyzing inequitable flooding in the Charlanta megaregion (Debbage 2019). Starting with Mecklenburg County parcels and the 2019 NLCD, we selected parcels only where all or the majority of the parcel was in the developed range between open space and high-density areas (NLCD classes 21-24). While the landcover raster was used to identify developed parcels (and can be easily accessed for other locations), there might be more precise ways to filter out parcels that are undeveloped open spaces or otherwise vacant.

For a baseline flood scenario, we then identified developed parcels overlapping the 500-year flood hazard zone. We opted to use a parcel-based approach to area calculations following Selsor et al. (2022), rather than raster coverage within each census block used in an earlier study (Debbage 2019), because parcels provide more accurate spatial resolution. In the GIS attribute table for parcels, we added fields to include census block group number as well as selection categories (developed, 500-year flood, water quality exposure, etc.), with 1 (yes) or 0 (no). Using the Summary Statistics tool (ArcMap v. 10.5), we calculated area sums and exposure risk factor for each census block group (N=624) as shown here:

$$Risk\ Factor = \frac{\sum At - Risk\ Developed\ Parcel\ Area}{\sum Developed\ Parcel\ Area} \qquad (3.1)$$

We then estimated the number of exposed individuals in each category by multiplying the risk factor with the group populations from the census block group demographic data (Non-Hispanic Black, Non-Hispanic White, Hispanic, Non-Hispanic, Below Poverty, Above Poverty). The total number of individuals for each category are summed together for a larger spatial unit, typically the census tract. However, we also assigned each census block group to a sub-basin and watershed based on the centroid of the census block group, thereby creating “demographic watersheds” as an alternative to conventional topographic delineation. The overall topographic and demographic watersheds are similar but not identical, and we would not recommend using the latter for hydrologic calculations. However, these demographic boundaries enabled us to directly compute categorical populations and risks without needing to apply weighting based on spatial areas and variable population densities. As with prior flood inequity studies (Debbage 2019, Selsor et al. 2022), we calculated risk ratios as follows:

$$Race\ Risk\ Ratio = \frac{\left(\frac{At - Risk\ Non - Hispanic\ Black}{Total\ Non - Hispanic\ Black}\right)}{\left(\frac{At - Risk\ Non - Hispanic\ White}{Total\ Non - Hispanic\ White}\right)} \qquad (3.2)$$

$$Ethincity\ Risk\ Ratio = \frac{\left(\frac{At - Risk\ Hispanic}{Total\ Hispanic}\right)}{\left(\frac{At - Risk\ Non - Hispanic}{Total\ Non - Hispanic}\right)} \qquad (3.3)$$

$$Poverty\ Risk\ Ratio = \frac{\left(\frac{At - Risk\ Below\ Poverty}{Total\ Below\ Poverty}\right)}{\left(\frac{At - Risk\ Above\ Poverty}{Total\ Above\ Poverty}\right)} \qquad (3.4)$$

A risk ratio greater than 1 indicates environmental inequity, such as a predominantly Black community with disproportionately high exposure to flood hazards. Approximately one third of the developed and flooded parcels also had RARR scores greater than zero, which we used for a separate flood scenario based on an established CMSWS flood mitigation strategy. We used the RARR scenario for the MCDA, but the baseline 500-year scenario could easily be implemented elsewhere. We used a similar risk calculation method for water quality equity. To approximate exposure to surface water pollution (e.g., fecal coliform bacteria), we combined the 500-year flood zone with the stormwater buffers, which ranged in width between 35 and 100 feet, thereby including the smaller streams that also convey polluted water. Figure 3.2 illustrates the different delineation methods used for environmental risk exposure.

We calculated benefit ratios for amenity access in a similar fashion, except that the data for population near public outdoor recreation had its own neighborhood spatial units (N=464),

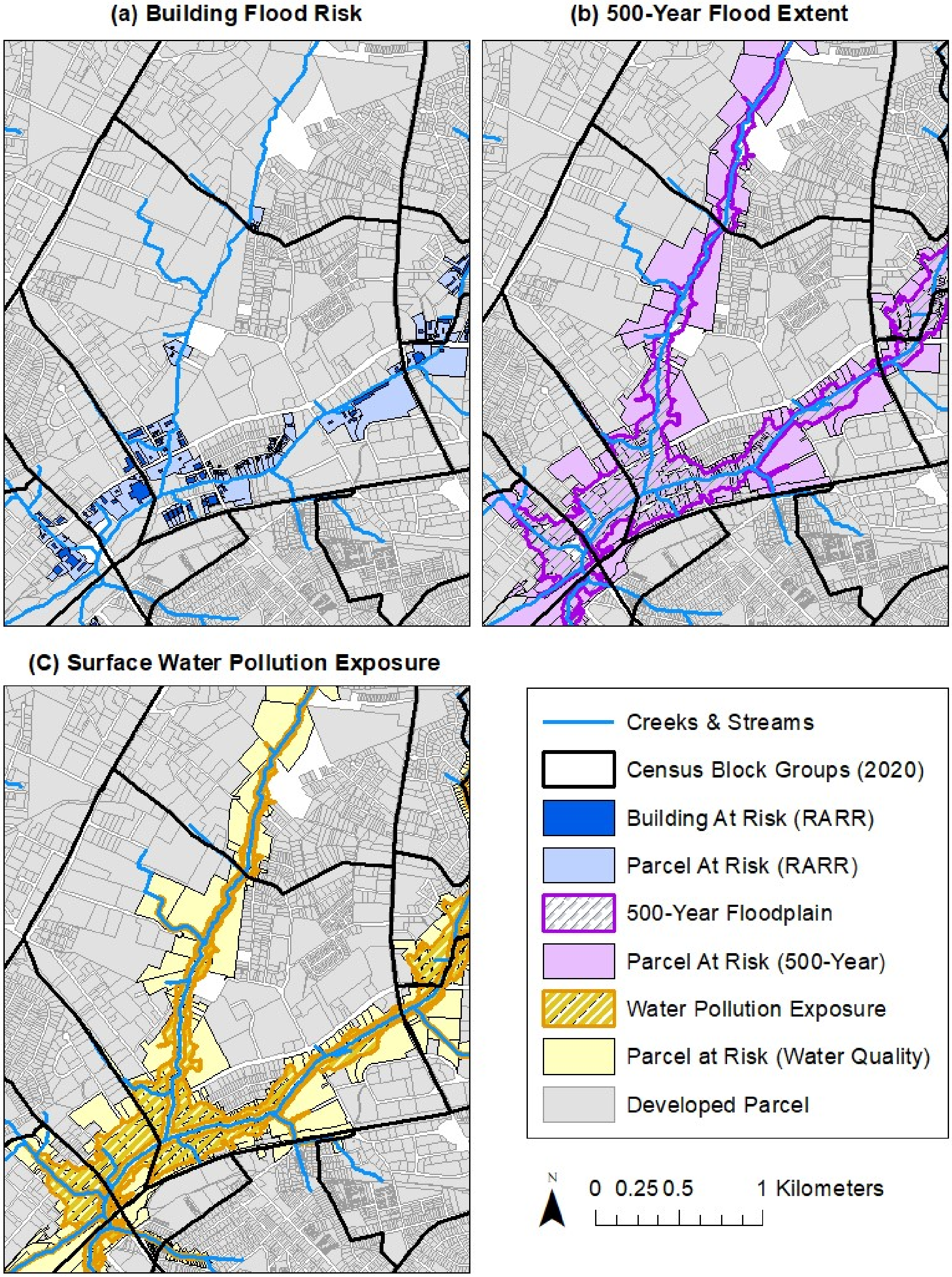


Figure 3.2. Examples of three different environmental risk exposure areas used for calculating risk ratios and social equity priorities. Risk factors were based on developed parcels (a) with building flood risk assessment / risk reduction (RARR) scores greater than one, (b) overlapping the 500-year flood zone, and (c) overlapping the water quality zone (combined stormwater quality buffers and 500-year flood zone).

and we did not use parcel areas or the NCLD raster. The equation for a benefit ratio looks nearly identical to a risk ratio, like the following example:

$$Poverty\ Benefit\ Ratio = \frac{\left(\frac{Below\ Poverty\ with\ Access}{Total\ Below\ Poverty}\right)}{\left(\frac{Above\ Poverty\ with\ Access}{Total\ Above\ Poverty}\right)} \qquad (3.5)$$

However, opposite from risk ratios, a benefit ratio less than 1 indicates social inequity, such as a low-income neighborhood with less access to public parks. For each scenario, after summing up the at-risk and total populations for the demographic groups at the tract, sub-basin, and watershed scales, we obtained risk ratios using R statistical software. While the ratios can be calculated easily using a spreadsheet or GIS attribute table, the fmsb package in R also provided p-values to describe statistical significance (< 0.05). However, the p-values were not used for spatial MCDA prioritization purposes.

Following the method of Selsor et al. (2022), to create a single combined equity metric for each spatial unit (census tract, watershed, sub-basin), we first re-assigned values of 1 for all risk ratios less than 1 and benefit ratios greater than 1 (i.e., no inequity) and then added together the ratios for race, ethnicity, and income level. In addition to calculating risk ratios for the Mecklenburg County census tracts (N=305), we assigned each census block group to both a watershed (N=33) and sub-basin (N=113) and calculated equity at the larger scales to better support spatial prioritization. Nine sub-basins, all located at the county boundaries, did not contain centroids of any census block groups, and therefore received the lowest priority for the various equity sub-criteria.

Weighting and prioritization

The basic procedure was to develop scores for all of the sub-criteria at the watershed and sub-basin scales, convert to a common priority scale (e.g., 0 to 1), and then apply weighted

MCDA Priority Weights	Min	Max		
	1	10		

Criteria & Sub-Criteria	Criteria Importance	Criteria Weights	Sub-Criteria Importance	Sub-Criteria Weights
Flood Risk Regulation	10	0.2		
Flood Risk			10	0.50
Flood Risk Equity			10	0.50
Water Quality Regulation	10	0.2		
Fecal Coliform Bacteria			10	0.25
Turbidity			10	0.25
303d			10	0.25
Water Pollution Risk Equity			10	0.25
Ecosystem Support	10	0.2		
Aquatic Ecosystem			10	0.50
Riparian Corridor			10	0.50
Amenity Access	10	0.2		
Outdoor Amenity Access			10	0.50
Greenspace Benefit Equity			10	0.50
Environmental Justice	10	0.2		
Historic Inequity			10	0.33
Social Vulnerability			10	0.33
Population Density			10	0.33

Flood Regulation (RARR)			Water Quality Regulation					Ecosystem Support (SRRS)			Amenity Access			Environmental Justice			
RARR_Overall	RARR_Scores_Priority	RARR_Equity_Priority	WQ_Overall	Fecal_Priority	Turbidity_Priority	Listed_Priority	WQ_Equity_Priority	Eco_Overall	SRRS_Priority	Buffer_Priority	AA_Overall	Amenity_Priority	AA_Equity_Priority	EJ_Overall	Redlined_Priority	RARR_SVI_Priority	Pop_Priority
1.1	0.02	0.00	5.5	0.50	0.00	1.00	0.03	3.1	0.12	0.19	5.7	0.52	0.28	6.0	0.00	0.86	0.53
1.0	0.00	0.00	4.8	0.25	0.50	0.00	0.54	5.9	0.48	0.24	10.0	0.84	0.71	3.2	0.00	0.44	0.18
4.3	0.60	0.02	5.6	0.25	0.25	1.00	0.05	6.7	0.20	0.63	3.3	0.39	0.00	6.4	0.00	0.73	0.76
1.0	0.00	0.00	4.0	0.00	0.00	1.00	0.01	1.7	0.00	0.11	1.8	0.13	0.00	2.9	0.00	0.43	0.11
1.0	0.00	0.00	1.0	0.00	0.00	0.00	0.00	2.7	0.00	0.24	1.0	0.00	0.00	4.1	0.00	0.78	0.07
1.9	0.04	0.12	7.1	0.50	0.50	1.00	0.05	10.0	1.00	0.31	6.5	0.66	0.29	2.4	0.00	0.22	0.16
3.4	0.06	0.39	4.1	0.50	0.25	0.00	0.28	5.6	0.42	0.25	1.5	0.08	0.01	2.1	0.00	0.20	0.12
4.7	0.01	0.67	1.0	0.00	0.00	0.00	0.01	2.3	0.00	0.19	8.1	0.50	0.73	6.6	0.00	0.55	1.00
1.3	0.06	0.00	4.0	0.00	0.00	1.00	0.02	7.1	0.42	0.48	5.2	0.73	0.00	2.2	0.00	0.10	0.23
3.2	0.11	0.30	3.6	0.50	0.25	0.00	0.13	7.7	0.77	0.21	2.9	0.32	0.02	3.9	0.00	0.36	0.44
1.3	0.06	0.00	2.5	0.50	0.00	0.00	0.00	3.5	0.31	0.06	1.9	0.15	0.00	3.0	0.00	0.50	0.05
1.7	0.06	0.07	2.5	0.50	0.00	0.00	0.00	3.8	0.07	0.34	5.0	0.33	0.37	1.6	0.00	0.05	0.12
3.2	0.40	0.00	9.4	1.00	0.75	1.00	0.09	9.8	0.81	0.47	3.6	0.42	0.02	8.9	0.68	1.00	0.53
1.0	0.00	0.00	1.0	0.00	0.00	0.00	0.00	1.0	0.00	0.00	3.2	0.29	0.09	1.9	0.00	0.07	0.18
1.0	0.00	0.00	3.7	0.00	0.00	0.00	0.90	1.8	0.00	0.12	1.2	0.03	0.00	2.6	0.00	0.26	0.20
1.8	0.16	0.00	4.7	0.50	0.75	0.00	0.00	5.9	0.48	0.24	2.8	0.31	0.00	4.8	0.00	0.75	0.30
1.3	0.06	0.00	1.0	0.00	0.00	0.00	0.00	6.6	0.00	0.82	6.8	1.00	0.00	4.7	0.00	0.45	0.58
4.7	0.29	0.39	10.0	1.00	0.75	1.00	0.28	9.6	0.25	1.00	4.8	0.31	0.35	6.0	0.00	0.72	0.68
10.0	0.66	1.00	4.0	0.00	0.00	0.00	1.00	3.1	0.00	0.31	6.7	0.44	0.55	3.3	0.00	0.33	0.32
2.0	0.18	0.00	7.8	0.50	0.75	1.00	0.02	7.7	0.69	0.29	3.9	0.39	0.11	5.7	0.00	0.74	0.57
1.7	0.11	0.02	9.4	1.00	0.75	1.00	0.06	5.1	0.31	0.28	3.1	0.33	0.04	5.2	0.00	0.59	0.57
1.9	0.17	0.00	2.6	0.25	0.25	0.00	0.03	6.2	0.36	0.40	3.8	0.37	0.12	3.0	0.00	0.19	0.35
2.0	0.04	0.14	8.1	1.00	0.25	1.00	0.12	5.8	0.50	0.20	2.3	0.21	0.01	4.9	0.00	0.70	0.38
3.0	0.20	0.16	6.9	0.50	0.25	1.00	0.23	5.7	0.48	0.21	6.7	0.49	0.49	5.5	0.00	0.64	0.61
6.1	0.68	0.26	3.5	0.50	0.25	0.00	0.10	7.6	0.79	0.17	2.9	0.29	0.03	4.6	0.00	0.75	0.25
2.0	0.06	0.11	6.6	0.50	0.25	1.00	0.13	7.4	0.82	0.11	5.0	0.67	0.02	5.3	0.00	0.88	0.31
1.2	0.03	0.01	4.4	0.50	0.50	0.00	0.14	8.8	0.91	0.23	9.5	0.46	1.00	2.1	0.00	0.11	0.19
1.3	0.06	0.00	4.7	0.25	0.00	1.00	0.00	9.6	0.52	0.73	2.1	0.19	0.00	4.5	0.00	0.39	0.58
4.7	0.51	0.18	5.2	1.00	0.25	0.00	0.15	7.8	0.80	0.19	4.4	0.49	0.10	4.9	0.00	0.65	0.44
6.6	1.00	0.04	9.2	1.00	0.75	1.00	0.02	7.5	0.64	0.31	3.9	0.46	0.04	4.7	0.00	0.76	0.27
1.0	0.00	0.00	1.0	0.00	0.00	0.00	0.01	1.0	0.00	0.00	1.0	0.01	0.00	7.0	0.00	0.91	0.75
3.6	0.47	0.00	7.8	0.50	0.75	1.00	0.02	8.5	0.35	0.75	3.6	0.45	0.00	10.0	1.00	0.71	0.80
1.1	0.02	0.00	1.0	0.00	0.00	0.00	0.01	3.0	0.29	0.00	3.8	0.48	0.00	1.5	0.00	0.07	0.06

Figure 3.3. The urban stream multi-criteria decision analysis (MCDA) spreadsheet user interface enables variable weights for criteria and sub-criteria. The combined criteria and overall MCDA priority scores are automatically calculated for watersheds and sub-basins.

averages to calculate combined criteria and overall scores. For purposes of this study, we assigned equal weights to all sub-criteria to calculate the overall priority scores for the criteria, and then we assigned equal weights to all of the criteria to calculate the combined priority score. It is possible to use a Weighted Sum tool in ArcMap, but compiling the data in a spreadsheet with inputs for variable weights is more user-friendly for stakeholders (Figure 3.3).

For the flood regulation objective, we used the Summary Statistics tool to find the mean RARR score for each watershed and sub-basin and then normalized based on the highest average value to assign a score between 0 and 1, so that 1 was the highest priority. We calculated overall flood risk equities for both spatial scales using the procedures described above and then reclassified to the common priority scale. Other possible variations could include a different RARR cutoff score, filter method for "developed" parcels, or simplification based on 500-year overlap. Although we computed watershed and sub-basin risk ratios for the baseline 500-year flood scenario, they were not used in the spatial MCDA calculations. Figure 3.4 shows the numerical distributions of watershed scores, and Figure 3.5a depicts the spatial distributions of sub-basin flood regulation priorities.

In the water quality category, the fecal coliform conditions (and categorical priority scores) were either non-compliant (=0.5) or very bad (=1.0) for all 23 monitored watersheds, although only 6 watersheds were also listed with a fecal TMDL, so the highest combined priority (=1.0) was a listed watershed with very bad conditions. None of the watersheds with the worst turbidity levels also had a regulatory status, so the highest combined priority scores (=0.75) for turbidity were found in watersheds with non-compliant conditions (=0.5) and also a turbidity TMDL or 303(d) listed (=1.0). The water quality regulatory sub-criteria also included a standalone priority score for 303(d) listed watersheds for any reason. To calculate pollution risk

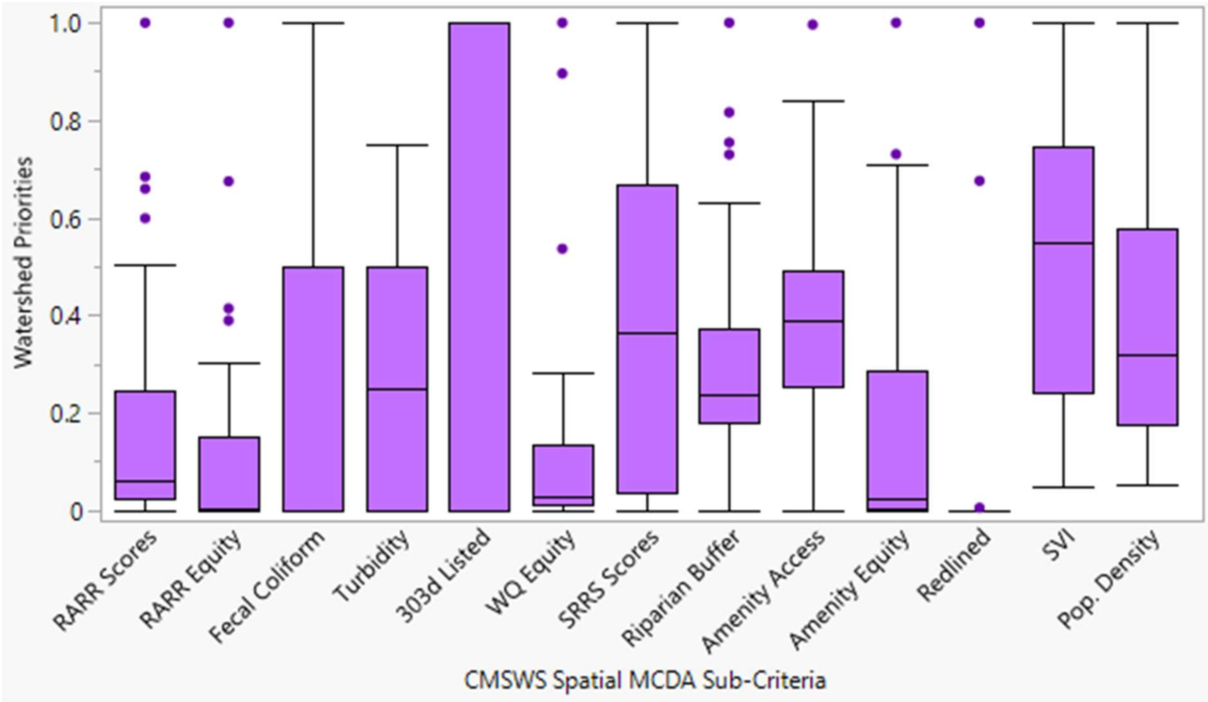


Figure 3.4. Distributions of watershed sub-criteria priority scores in all five categories. Priorities were calculated based on MCDA spreadsheet input data, and then weighted to calculate combined criteria and overall total scores. Average flood risk (RARR Scores), flood risk equity (RARR Equity), water quality equity (WQ Equity), and amenity access equity (Amenity Equity) were all heavily skewed towards the low end of the priority range with outliers in the upper tails, while riparian vegetation (Riparian Buffer) was skewed towards higher priorities with only a couple low-scoring outliers. No watersheds had a turbidity priority equal to one, which would have required a severe level plus a turbidity TMDL or 303d listed status. The 303d listed priorities for any reason were either 0 (not listed) or 1 (listed). Ecosystem restoration priority scores based on channel stability and habitat (SRRS Scores) were normally distributed, as were access to public outdoor recreation (Amenity Access), social vulnerability (SVI), and population density priorities. Only two watersheds had significant historic redlining with corresponding high priorities (Redlined). The minimum SVI scores and population densities were greater than zero.

equity, we used exposure to non-compliant or very bad fecal conditions for all developed parcels overlapping the 500-year flood zone or streamside water quality buffers. The ranges of water quality sub-criteria scores are shown in Figure 3.4. Because the original water quality data was at the larger watershed scale, equity accounted for the only differences between sub-basins in any single watershed (Figure 3.5b). Although pollution exposure may also be linked to subsurface utilities (e.g., Alves 2022), our spatial MCDA focused on surface water environmental hazards.

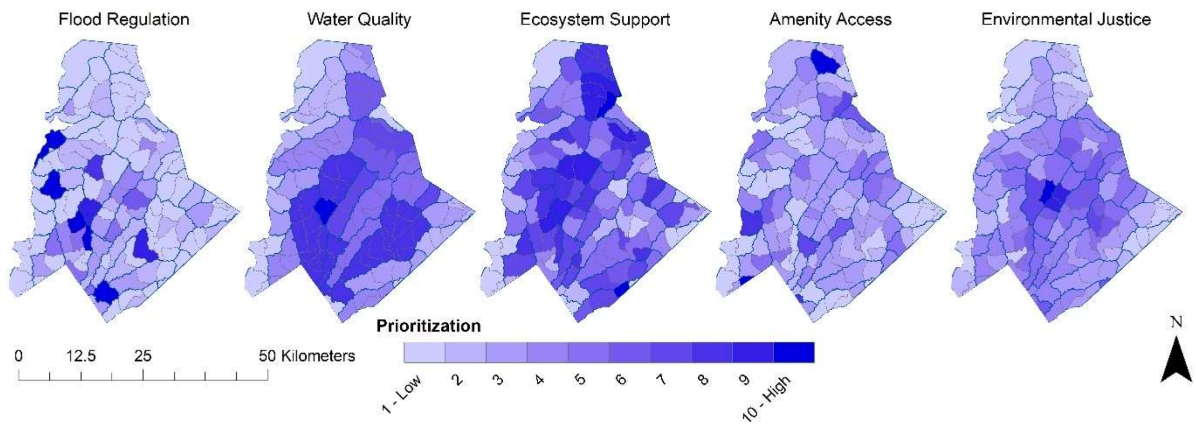


Figure 3.5. Sub-basin combined criteria scores for (a) flood regulation, (b) water quality regulation, (c) ecosystem support, (d) amenity access, and (e) environmental justice. Most of the water quality sub-criteria (fecal coliform, turbidity, 303d listed) were characteristics inherited from the parent watershed, so water pollution exposure equity accounted for the only combined differences at the sub-basin scale within a given watershed (b).

Under the ecosystem support category, we converted the polyline shapefile with SRRS scores to raster format and used the Zonal Statistics as Table tool to find mean scores for each watershed and sub-basin. Because higher SARR condition scores in the existing channel and riparian buffer corresponded to lower restoration priority, we inverted the scores as well as normalizing from 0 to 1. The ecological watershed sub-criteria score distributions are provided in Figure 3.4, and combined sub-basin priorities are shown in Figure 3.5c. Alternative ecological protocols like the Stream Function Assessment Method for Oregon (Nadeau et al. 2018) may include similar desktop analyses (e.g., aerial imagery).

For access to amenities, we converted the neighborhood polygons to raster format, used the Zonal Statistics as Table tool for watersheds and sub-basins to find the mean proportion near public outdoor education, inverted the values, and then reclassified to the common priority scale. The benefit equity for amenity access was calculated and prioritized similar to the flood and pollution risk ratios, except that inequity was characterized by disproportionately lower access to benefits rather than greater exposure to risk. The numerical and spatial distributions of watershed sub-criteria and overall sub-basin priorities related to amenity access are shown in Figures 3.4 and 3.5d, respectively.

Under the general environmental justice category, we converted the population densities, SVI polygons, and redline areas to raster format, used the Zonal Statistics as Table tool, and then normalized the sub-criteria priorities to a maximum score of 1. As indicated in Figure 3.4, no watersheds had an average SVI score or population density of 0. While the SVI and population density exhibited normally distributed values and corresponding priorities across the sub-basins and watersheds (Figure 3.4), redlining was much more isolated, being limited mostly to the Upper Little Sugar and Irwin watersheds. The overall sub-basin environmental justice priorities are depicted in Figure 3.5e.

The watershed distributions of sub-criteria scores are shown in Figure 3.4, in many cases exhibiting highly skewed trends, also evident at the sub-basin scale. For example, the median flood risk assessment (RARR score) priority was quite low compared to the water quality, ecosystem support, amenity access, and general environmental justice metrics. Likewise, all of the equity priority scores had skewed distributions with low median scores and a handful of outliers at the upper end of the range, the hotspots for environmental inequity. We organized the sub-criteria scores for both watersheds and sub-basins in a spreadsheet (Figure 3.4) as the primary spatial MCDA user interface with variable weights. Eliciting weights from CMSWS would typically be the next step for full implementation of the urban riverscape MCDA. For purposes of this study, however, we assigned equal weights to all sub-criteria to calculate the overall priority scores for the various criteria (Figure 3.5), and we used the spreadsheet to explore two spatial MCDA scenarios. In the first scenario (Combined MCDA), we assigned equal weights to all five of the criteria to calculate the combined priority score. For the alternative scenario (Riverscape MCDA), we used only flood and water quality regulation and ecosystem support – the main criteria most physically linked to arterial waterways and their contributing watersheds, as well as the associated environmental hazards and benefits. In addition to reviewing the watershed and sub-basin priorities resulting from the two MCDA scenarios, we investigated tradeoffs and synergies by performing Pearson's bivariate correlations to test the criteria and sub-criteria priority scores for possible relationships.

Results

Spatial prioritization

We found that inclusion of the landscape criteria (amenity and environmental justice) in the Combined MCDA substantially altered the spatial prioritization compared to the Riverscape MCDA based only on flood regulation, water quality, and ecosystem support, but the differences were evident only at the sub-basin scale (Figure 3.6c-d). The subset of criteria used for the Riverscape MCDA scenarios most closely align with the primary management goals elicited from our CMSWS collaborators as well as potential landscape-scale interventions and natural infrastructure strategies. For the watershed scenario including both riverscape and landscape criteria, Lower Little Sugar and Irwin had the highest combinations of priority scores (Figures 3.6a, 3.7), with Upper Little Sugar and Sugar closely tied for third place – spatially, these watersheds are directly adjacent to one another. The water quality, ecosystem support, and amenity access overall criteria scores were comparable among the top few watersheds, but Upper Little Sugar had the highest environmental justice metrics (social vulnerability, population density, historic redlining), whereas Sugar exhibited the highest average flood risk (RARR scores). The highest priorities for the Riverscape MCDA scenario (Figures 3.6b, 3.7) again included Lower Little Sugar, Sugar, and Irwin watersheds. Not surprisingly, Upper Little Sugar

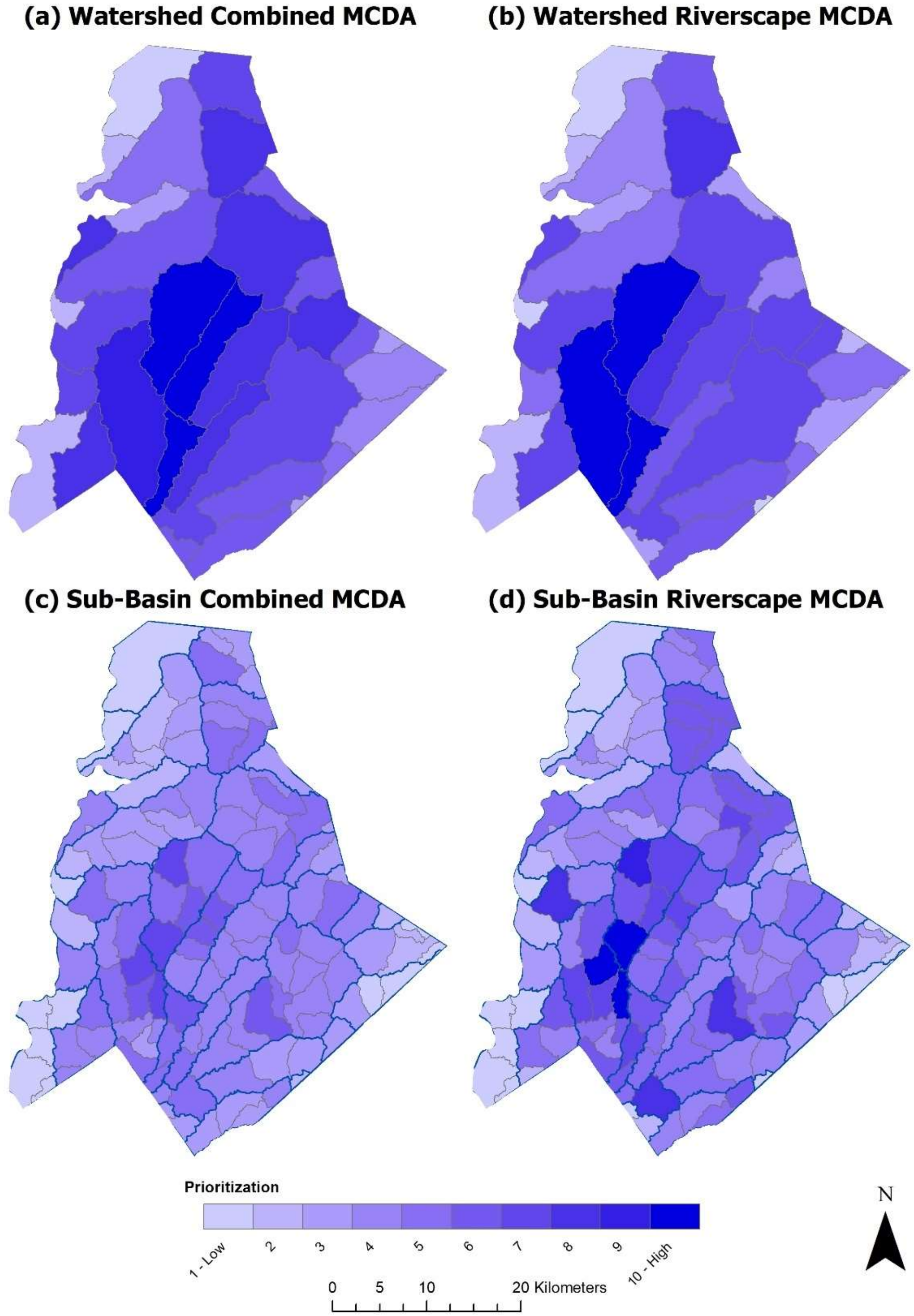


Figure 3.6. Spatial MCDA results for watershed and sub-basin scenarios that combined (a,c) all five criteria and (b,d) only flooding (Figure 3.5a), water quality (Figure 3.5b), and ecosystem support (Figure 3.5c), which are most directly linked to the arterial waterways. Sub-basin hotspots are evident with both scenarios, but (c) shows more spatial similarity within any given watershed, likely due to the inclusion of landscape-based amenity access (Figure 3.5d) and environmental justice (Figure 3.5e). The splotchy appearance of the riverscape scenario (d) corresponds to just a few sub-basin outliers with the highest priority scores.

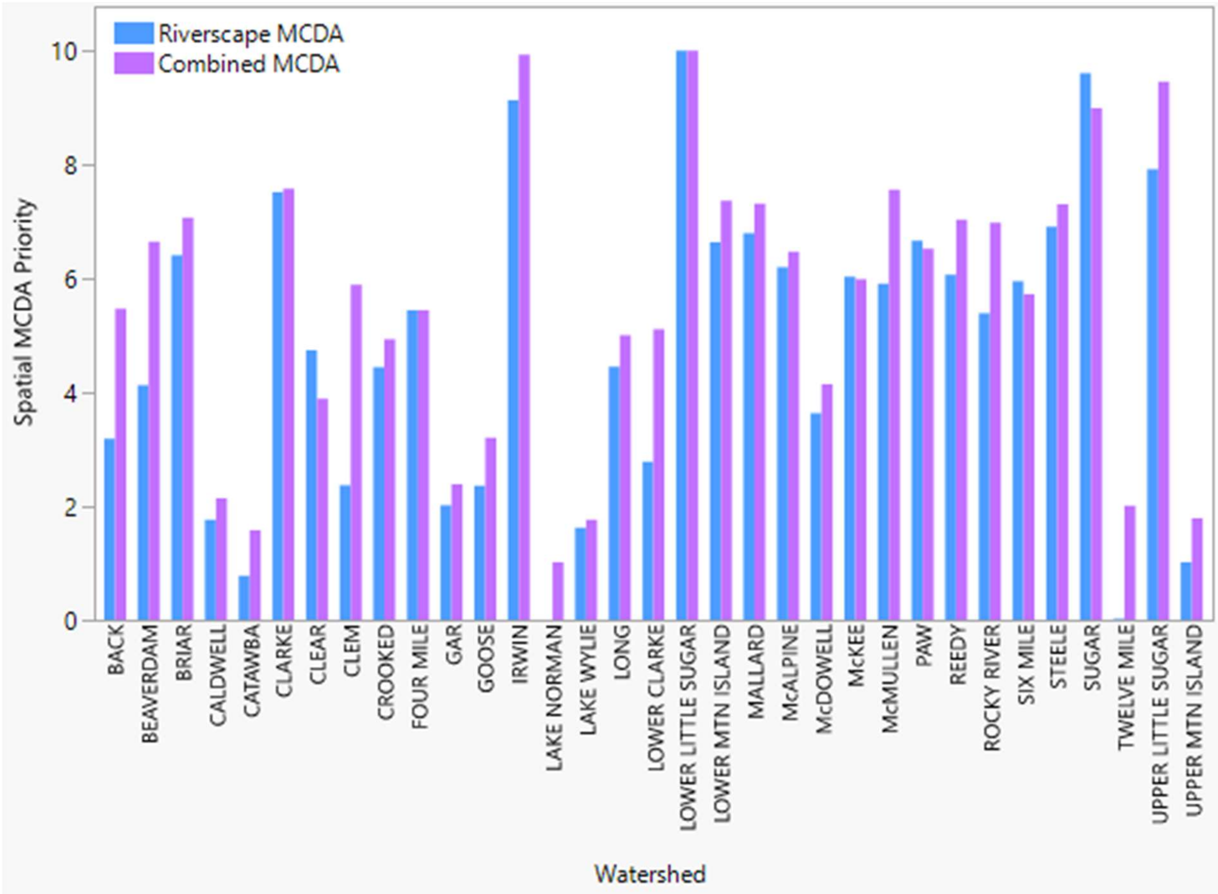


Figure 3.7. Watershed priority scores for two MCDA scenarios show similar overall totals. Lower Little Sugar scored highest for both scenarios, and the other watersheds with the highest priorities include Irwin, Sugar, and Upper Little Sugar. The Riverscape MCDA watershed priorities for Lake Norman and Twelve Mile were zero, because they had no “developed” parcels with environmental risk exposure (Figure 3.2) or stream reaches with Stream Restoration Ranking System (SRRS) scores.

dropped in priority without the general environmental justice criteria. However, the Riverscape MCDA scenario still incorporated context-specific environmental risk equity. With side-by-side comparisons of the two priority results, we found similar watershed priorities between the two scenarios (Figure 3.7).

The sub-basin prioritization highlighted the spatial hotspots combining social-ecological system vulnerabilities and deficiencies across multiple dimensions. We found a distinct contrast between the sub-basin priorities for the two MCDA scenarios (Figures 3.6c-d, 3.8). For the combined scenario based on all five criteria, sub-basins within the same watershed showed a

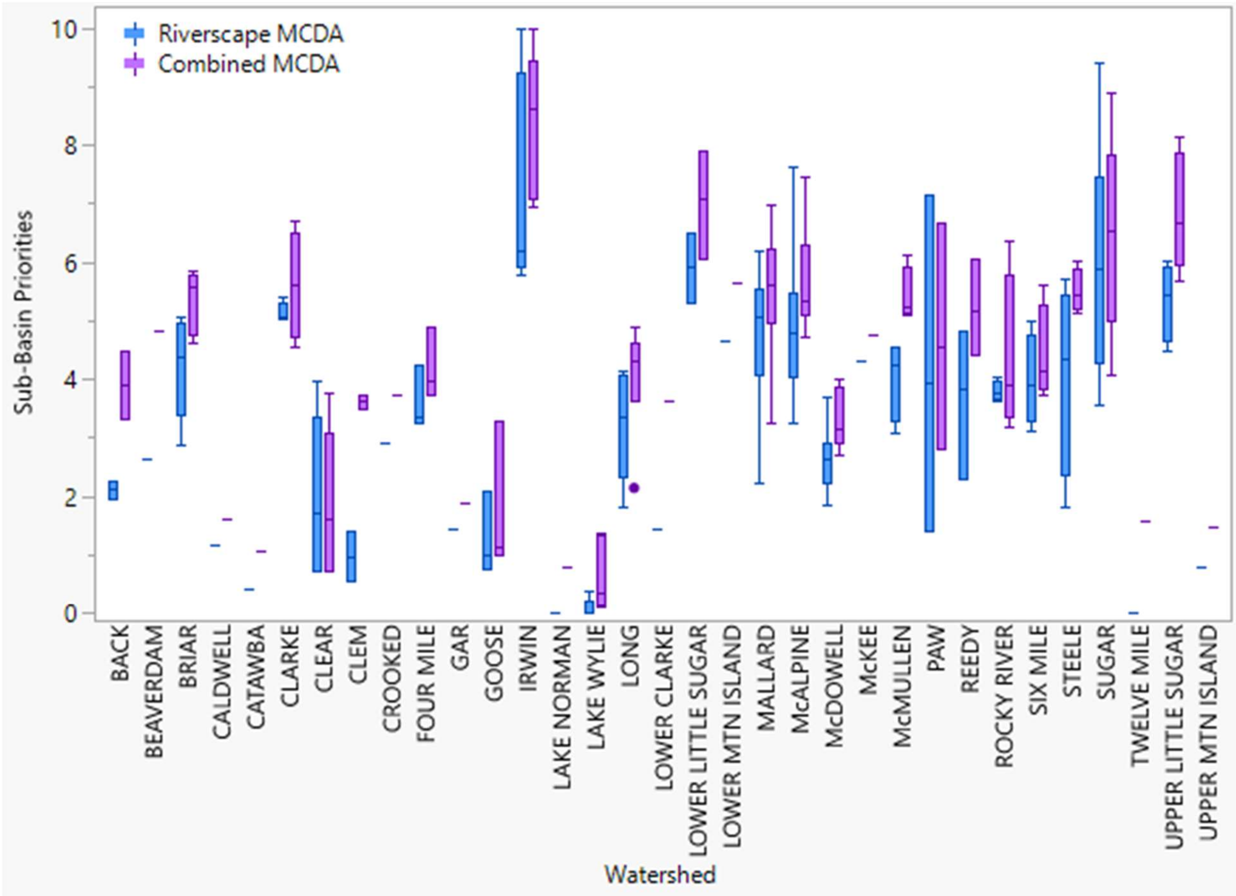


Figure 3.8. Distributions of sub-basin priorities for two MCDA scenarios within each watershed. In many cases, the Combined MCDA scenario including both riverscape and landscape criteria generated higher mean values of basin priorities compared to the Riverscape MCDA. The corresponding spatial distributions are shown in Figure 3.6.

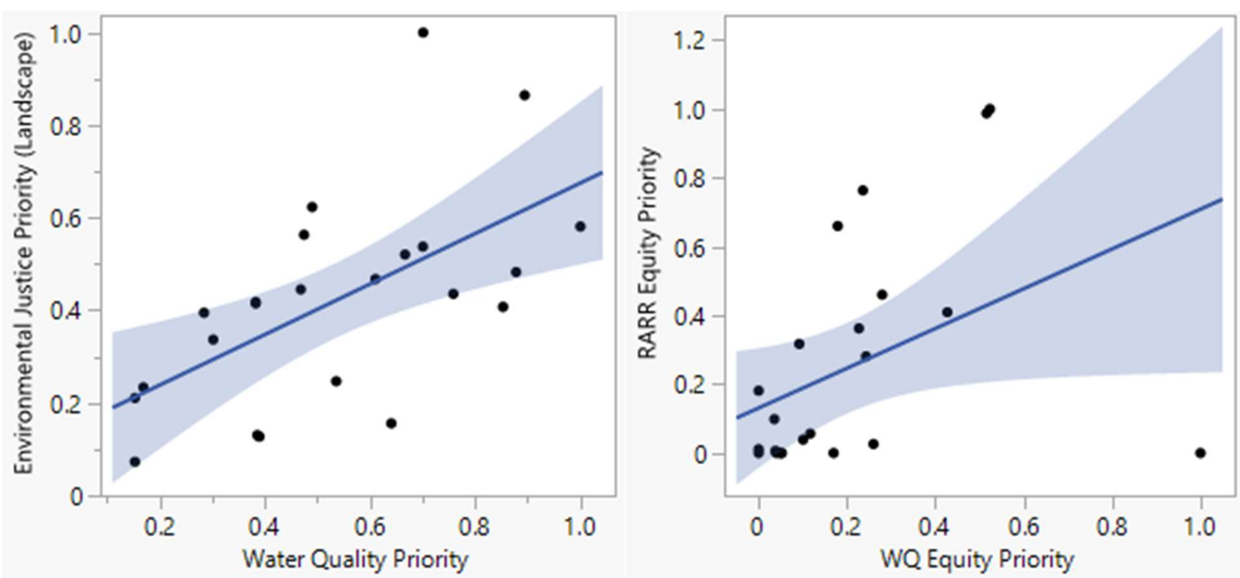
degree of spatial similarity (Figure 3.6c), but the riverscape scenario generated a distinct spatial pattern of isolated hotspots (Figure 3.6d). These results make sense, given that the additional criteria (amenity access, environmental justice) used in the first scenario are distributed across the entire landscape rather, not just the arterial waterways.

System level synergies

Part of the purpose of a spatial MCDA is to help stakeholders leverage potential synergies among multiple objectives. An individual watershed or sub-basin with a much higher overall priority score compared to others has greater combined effects from its constituent criteria. For example, the Lower Little Sugar watershed rose to the top of both MCDA scenarios

(Figure 3.7) with upper quartile scores for flood risk, water quality, ecosystem support, and environmental justice criteria (Figure 3.4).

When we looked for positive or negative correlations between criteria across all watersheds, we found little statistical power to support conclusions about system-wide synergies or tradeoffs. However, for watersheds with monitoring data, we did find evidence of a positive correlation (0.60, p=0.0027) between the overall water quality and general environmental justice criteria (Figure 3.9a). This result intrigued us, especially because none of the water quality equity sub-criteria scores (race, ethnicity, income, combination) were correlated with any of the environmental justice sub-criteria. However, the lack of relationship between the equity scores and general environmental justice criteria underscores the value of including both types of metrics, and not just social vulnerability. The broader metrics are still valuable for capturing



(a) (b)

Figure 3.9. Statistical relationships among criteria and sub-criteria at the watershed scale. (a) The subset of watersheds with water quality monitoring data, showed a positive correlation (0.60, p=0.0027) between the combined water quality and environmental justice (SVI, population density, redlined) priorities. (b) A positive relationship (0.51, p=0.0022) between the flood and water pollution risk equity priorities could result from the similar methods used to calculate risk ratios for both, with the only difference being exposure (Figure 3.2).

neighborhood characteristics beyond environmental risk exposure because neighborhoods may not spatially correspond to topographically delineated watersheds and sub-basins.

When we analyzed the sub-criteria, we also found a positive correlation between the equity scores for flood risk (RARR) and water pollution exposure at both sub-basin (0.63, $p<.0001$) and watershed scales (0.51, $p=0.0022$), as shown in Figure 3.9b. The environmental risk equity relationship makes sense given the similar methods we used to develop the two metrics, with the only difference being the extent of exposure. This correlation might also have contributed to the pattern of sub-basin priority outliers visible in Figures 3.6b and 3.8 if there was an amplifying effect by including separate flood and water pollution risk ratios in the waterway-focused MCDA scenario.

Environmental equity

The environmental equity aspect of our study produced several important results and applications. First, we found spatial trends in flood inequity in agreement with earlier findings about the areas in Charlotte with socio-economic disparities, and the magnitudes of risk ratios above one, despite the differences in how we defined flood exposure and aggregated risk areas using parcels. For example, Debbage (2019) found that the Landsowne neighborhood (Census Tract 37119002004) had some of the highest risk ratios that were statistically significant, the worst being 3.28 for the Non-Hispanic Black population, and we found the corresponding RARR and 500-year flood risk ratios were both 3.40 ($p<.0001$) for the same tract. However, the spatial resolution of the analysis matters, because Debbage found only one significant risk ratio (below poverty vs. above poverty) for Mecklenburg County as a unified whole (Debbage 2018). Like Debbage, however, we found a different story when zooming into the census tract level for equity comparisons by demographic groups (Figure 3.10). While the presence of any risk ratio

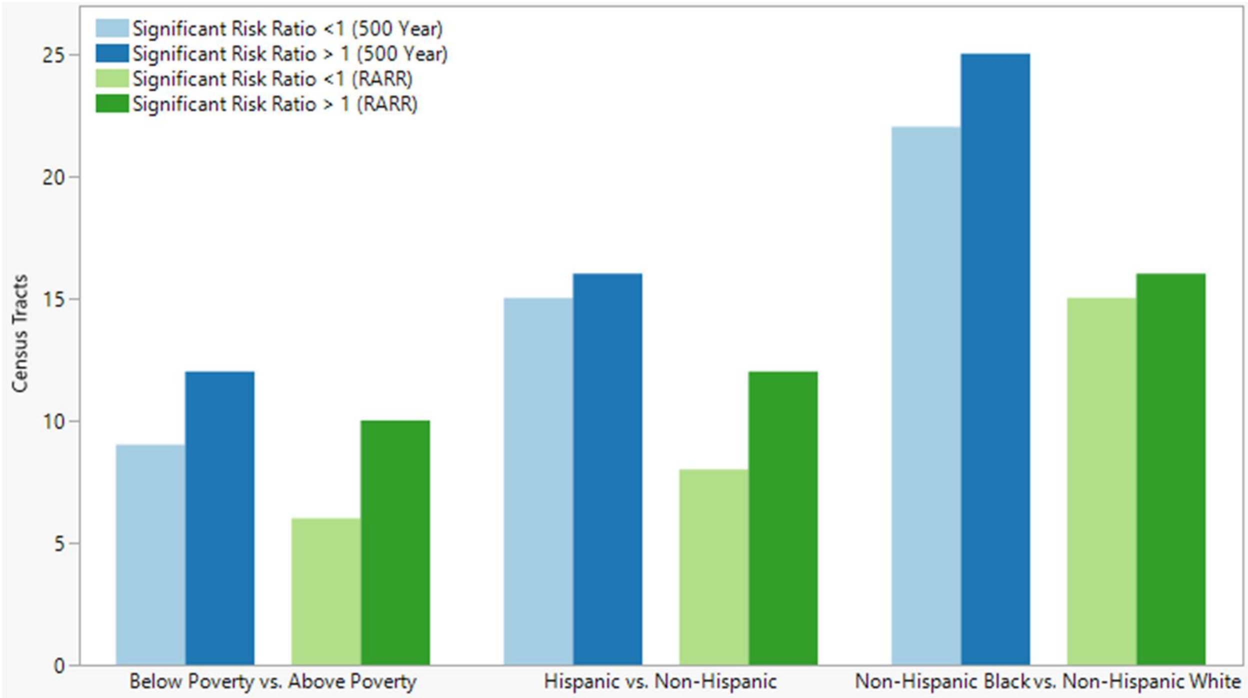


Figure 3.10. Frequencies of census tract risk ratios with significant p-values greater and less than one for studied socio-economic groups and two different metrics of flood risk exposure: developed parcels with non-zero RARR scores (building level) and overlapping the 500-year flood hazard zone. The Hispanic, Non-Hispanic Black, and Below Poverty demographic categories all had more census tracts with inequitable risk ratios (>1) for both flood scenarios.

greater than one shows that there are equity concerns that need to be addressed, the numbers of tracts with statistically significant risk scores greater than one exceeded those with risk scores less than one, suggesting overall inequities across race, ethnicity, and income level. In contrast, among sub-basins with statistically significant risk ratios, only the poverty category continued to show a negative disparity, and the larger watershed scale altogether erased this tendency through overall aggregation.

In addition to confirming overall trends in flood risk, we were able to take a step towards operationalizing flood risk equity through spatial prioritization. For starters, we combined multiple minority and income categories (Hispanic, Non-Hispanic Black, Below Poverty) to create a single equity score for each sub-basin and watershed, thereby including intersectionality of these socially vulnerable classes (e.g., low-income Black and Hispanic people). Leveraging multiple spatial scales was also useful when characterizing the distributions of environmental risks and benefits. Even if risk ratios at the large county scale failed to demonstrate overall inequities, the medium and small spatial scales enabled us to determine which watersheds, sub-basins, and tracts were the areas of greatest concern. Thus, we applied the environmental equity scores as part of the criteria for a spatial MCDA.

Moreover, we found that the environmental equity techniques were both relatively simple and flexible enough for a range of practical applications. For example, we were able to use the 500-year flood zone as a proxy for flood risk similar to earlier work (Debbage 2019), and the same basic technique worked with alternative definitions of environmental risk (e.g., surface water pollution exposure) and a specialized approach that incorporated risk frequency and probability (i.e., RARR scores). Access to environmental benefits (i.e., public outdoor recreation) was also readily incorporated in a comparable fashion through the use of benefit ratios. Furthermore, it was relatively straightforward to estimate risk and benefit factors (i.e., exposure and access), combine them with census data, and then perform risk and benefit ratio calculations with tools widely used by municipalities and utilities (GIS, spreadsheets).

Discussion

The urban stream spatial MCDA that we developed through collaboration with CMSWS provides a flexible approach to identifying social-ecological system vulnerabilities, prioritizing streams and watersheds for future interventions, and working across departments to meet multiple management objectives. As decision-makers prepare to make significant NI investments, advanced effort is needed to do the "right" projects that will best support sustainable cities for generations to come. Rather than responding to isolated environmental hazards and challenges in a piecemeal fashion, our spatial MCDA facilitates more holistic system-level planning, laying the groundwork for strategic NI innovations.

Facing challenges

Properties and people suffering from flood damages, under-functioning stream reaches that fail to support aquatic life, and impaired watersheds that fall short of regulatory goals are just the tip of the urban riverscape iceberg encountered by organizations and stakeholders. Our spatial MCDA incorporates all of these specific concerns in an approach that further addresses some of the underlying challenges and system drivers. For example, these problems involve multiple spatial scales with variable degrees of interaction, and our MCDA approach includes both the large (watershed) and medium (sub-basin) landscape context, with further potential to modify metrics for evaluating local project alternatives while sharing the same overarching criteria. The sheer scope of responsibilities held by stream and watershed managers has led to a natural division of labor, which can function as a barrier. Our spatial MCDA bridges departmental boundaries and includes input from CMSWS individuals tasked with multiple missions, reflected in the range of criteria and incorporation of various established management tools (i.e., RARR, SRRS, Water Quality Matrix). Engineers and scientific specialists might also struggle with questions related to human dimensions outside of their technical expertise, such as how to incorporate social equity in a meaningful way. For example, this study agreed with earlier findings of flood inequity in the Charlotte metropolitan area (Debbage 2019), with regard to locations and magnitudes of socio-economic disparities, most evident at smaller spatial scales. Our dual approach to environmental equity in the urban stream spatial MCDA uses widely recognized metrics of social vulnerability in tandem with equitable distributions of specific environmental risks and benefits, similar to prior spatial planning for green infrastructure

(Meerow and Newell 2017). Finally, the spreadsheet serving as the user interface for the spatial MCDA helps address the inherent challenge of multi-objective prioritization in a transparent and flexible fashion through weighted criteria and sub-criteria that can be easily modified by multiple stakeholder groups to explore alternative riverscape scenarios.

Finding opportunities

With our urban stream spatial MCDA, we wanted to help CMSWS identify areas in Mecklenburg County with opportunities to realize multiple potential benefits. In contrast with a green infrastructure study in Detroit (Meerow and Newell 2017), we found little evidence for synergies or tradeoffs across criteria, perhaps because we were focused on riverscapes with different environmental risks and benefits. However, our prioritization did highlight specific watersheds and sub-basins as promising locations for addressing multiple objectives while finding alignment with ecosystem restoration – ideal settings for potential NI investments.

Identifying hotspots on the basis of more environmental risks coupled with fewer existing benefits was the primary task of our MCDA approach, but we envision a complementary spatial prioritization with benefits-related sub-criteria like future greenway trails, adopted streams, outdoor education, and other social connectivity. Furthermore, specific natural infrastructure solutions, such as those for flood mitigation, involve different types of spatial criteria (e.g., Hovis et al. 2021). If stream renovation (restoration, revitalization, naturalization, etc.) priorities feature ecological uplift, then upgrading urban aquatic and riparian ecosystems from poor to fair may involve different success indicators than improvement from fair to good. For example, a poorly vegetated stream buffer zone presents a potential opportunity to enhance riparian conditions and support terrestrial wildlife, but connectivity to a high quality forested stream reach may improve the likelihood of restoration success for fish and aquatic insects. In short, what makes for a "good opportunity" for environmental benefits and co-benefits is highly context specific, so it may be most appropriate to conduct separate analyses at the sub-basin, floodplain, and/or stream reach scales rather than lumping opportunities together with social-ecological system vulnerabilities and deficiencies.

The inclusion of environmental equity in our case study corresponds to particularly valuable opportunities, providing a potential starting point for water managers struggling to incorporate social components. We demonstrated how to include social vulnerability and advance environmental equity with our spatial MCDA. Moreover, our approach is intended to be a collaborative decision support tool for community inclusion and diverse stakeholder groups. That being said, when adding new sub-criteria with priority scores based on potential opportunities (e.g., specific NI solutions, alignment with other planned improvements, etc.), it will be especially important to review potential tradeoffs with environmental equity criteria, such as risks associated with neighborhood displacement and gentrification (neighborhood change score). We are concerned about the potential for "win-win-win" scenarios based on some combination of "good opportunities" or a resilience narrative that inadvertently reinforce existing systemic injustices due to unacknowledged social tradeoffs (e.g., Eakin et al. 2017, Béné et al. 2018). However, the potential alignment we found between the overall water quality and environmental justice criteria is a promising avenue for further study and action.

Flexible applications

Working together with CMSWS on the urban stream MCDA case study demonstrated one practical application of our approach. However, our larger intent was to create a multifunctional spatial prioritization framework to support the larger body of managers, practitioners, and communities tackling urban riverscape challenges and planning for NI solutions. Integrating existing management tools and strategies like those used by CMSWS (RARR, SRRS) showed how the general approach could be tailored for specific organizations and departments. However, the spatial MCDA could easily transfer to other municipalities with alternative methods for characterizing flood risk, channel stability, water quality, etc. For example, floodplain managers could apply our spatial MCDA in tandem with probabilistic mapping (Stephens and Bledsoe 2020) or frequency-based risk equity (Selsor et al. 2022), and use different methods for determining which parcels are developed and at risk. Environmental scientists could evaluate and prioritize stream restoration using ecological potential based on the ratio of existing to predicted biotic scores (Paul 2022). Sub-criteria related to fecal coliform and water quality could incorporate spatial data about basement backups (Alves et al. 2021, Alves 2022). Census data, the CDC version of SVI, and historic redlining maps are readily available in the US to support environmental equity objectives, and our methods for delineating "demographic watersheds" and calculating environmental risk and benefit ratios are highly adaptable. The spatial MCDA used software tools that are already familiar to technical specialists in our target audience, and GIS can be coupled with story maps to present information to the wider group of stakeholders (e.g., Meerow and Newell 2017, Environmental Protection Agency 2020). Finally, our urban stream MCDA can support tradeoff analyses when evaluating NI criteria, priorities, and alternatives with multiple stakeholder groups (e.g., Bridges et al. 2015, Meerow and Newell 2017).

Conclusions

The spatial MCDA we developed for urban watersheds and streams was used in collaboration with CMSWS for preliminary planning to meet multiple priorities: flood risk, water quality, aquatic ecosystems, amenity access, and environmental equity. With an eye on multifunctional NI investment, the urban stream MCDA was an initial step towards prioritizing the "right projects", and our social-ecological-technical approach supports further steps towards doing projects the "right way". Our environmental equity methods and findings can help the City of Charlotte and Mecklenburg county to advance social justice by expanding the scope from vulnerability to the distributions of environmental risks and benefits, keeping in mind that inclusion of landscape criteria like access to amenities and general environmental justice (SVI, historic redlining, population density) can substantially shift the perspective. The spatial analysis revealed synergies and identified hotspots to begin conversations with neighborhoods and communities as part of structured decision making: problem definitions, knowledge co-production, and opportunity identification leading to community-based solutions (NI, conventional systems, hybrid approaches, and other potential innovations). At the same time, the practical transferability of our spatial MCDA supports broader NI applications in other social-ecological systems and urban riverscapes, and can be used to operationalize equity in infrastructure decisions.

CHAPTER 4

INTEGRATING CHANNEL DESIGN AND ASSESSMENT METHODS BASED ON SEDIMENT TRANSPORT CAPACITY IN GRAVEL BED STREAMS[3]

[3] Yaryan Hall, H.R. and B.P. Bledsoe. Submitted to *Journal of American Water Resources Association,* 9/7/22.

Abstract

Natural channel design (NCD) and analytical channel design (ACD) are two competing approaches to stable channel design that share fundamental similarities in accounting for sediment transport processes with designs based on hybrid fluvial geomorphology and hydraulic engineering methods. In this paper we highlight the linkage between ACD's capacity/supply ratio (CSR) and NCD's sediment capacity models (FLOWSED/POWERSED), illustrating how ACD and NCD have reached a point of convergent evolution within the stream restoration toolbox. We modified an existing CSR analytical spreadsheet tool which enabled us to predict relative channel stability using both conventional bed-load transport equations and regional sediment regression curves. The stable channel design solutions based on measured data most closely matched the Parker (ACD) and/or Pagosa Good/Fair (NCD) relationships, which also showed the greatest CSR sensitivity in response to channel alterations. We found that CSR differences among the transport relationships became more extreme the further the design width deviated from the supply reach, suggesting that a stable upstream supply reach may serve as the best design analog. With this paper we take a step towards resolving lingering controversy in the field of stream restoration, advancing the science and practice by reconciling key differences between ACD and NCD in the context of reach scale morphodynamics.

Introduction

Stable channel design has historically dominated the fields of hydraulic engineering and fluvial geomorphology, guiding practitioners and policy makers alike, and underpinning stream interventions from roadway crossings (Richardson et al. 2001, Lagasse et al. 2012) to restoration efforts (e.g., Shields et al. 2003, Johnson et al. 2020). Here, we define a geomorphologically stable river as "one that has adjusted its width, depth, and slope such that there is no significant aggradation or degradation of the stream bed or significant planform changes (meandering to braided, etc.) within the engineering time frame (generally less than about 50 years)" (Biedenharn et al. 1997). Proponents of stable channel design – typically those grappling with problematic erosion and sedimentation processes (e.g., Niezgoda and Johnson 2005) – share objectives like protecting human infrastructure, improving water quality, and supporting aquatic ecosystems, and most aim for long-term predictability and reliability.

Within the stable channel design framework, we have witnessed a gradual evolution from armored trapezoidal conveyance systems (e.g., historical engineering works for flood protection) to fully alluvial channel designs (e.g., Shields et al. 2003, NRCS 2007). The fundamental relationship between channel forms and processes has been broadly studied, described, and applied (e.g., Niezgoda and Johnson 2005, Church 2006). However, lingering misperceptions and myths about sediment transport continue to characterize many stream channel designs and policies. First is confusion about the necessary level of sediment transport analysis (Bledsoe et al. 2016, 2017). Second is a common failure to distinguish between sediment competency and capacity (Rosgen 2006), potentially leading to a false sense of design security. Third is a conclusion that sediment transport analysis is too cumbersome, based on the perceived difficulty of obtaining measured sediment loads and/or understanding conventional transport equations. Finally, is a tendency to discount analytical results, based on values that could plausibly be inaccurate by an order of magnitude (e.g., Hinton et al. 2018). Consequently, stream designers may fail to fully account for sediment transport, with real-world consequences: potential channel instability leading to vulnerabilities of infrastructure, water quality, and stream ecosystems, for example. Furthermore, practitioners need to understand the watershed conditions and design implications of upstream sediment supply (e.g., legacy material) and active channel response to ongoing disturbance (e.g., Reid and Dunne 1996).

Stable channel design further faces a persistent obstacle of infighting between different stream restoration schools of thought (Lave 2009, 2012). Natural channel design (NCD) has been widely propagated by practitioners and regulators (Lave 2012), while analytical channel design (ACD) has been more commonly endorsed by academia and the U.S. Army Corps of Engineers (e.g., Copeland et al. 2001, Soar and Thorne 2001, Thomas et al. 2002, Stroth et al. 2017). Although NCD incorporates multiple levels of sediment assessment, practitioners may stop short of performing capacity-based design due to lack of familiarity with or access to the proprietary FLOWSED/POWERSED software models (Rosgen 2006), which are also not fully understood by many in the research community. ACD, on the other hand, depends upon structurally complex equations for sediment transport analysis (e.g., Brownlie 1981, Parker 1990, Wilcock and Curran 2003), which are not fully understood by many of those outside the research community. Additionally, a relative paucity of real-life ACD case studies (e.g., Dierks et al. 2003) may impede broader acceptance by those looking for reassurance about construction and monitoring outcomes. Wading into these waters, we intend to avoid the false dichotomy of form versus process (Lave 2009) by recognizing the most evolved approaches to both NCD and ACD manipulate channel geometry and profile while accounting for natural sediment transport processes.

The objective of this paper is to compare channel stability predictions using the analytical sediment transport approaches of ACD and NCD, thereby illuminating key similarities, differences, and sensitivities to design parameters for both methods. We used a modified version of the publicly available capacity/supply ratio (CSR) spreadsheet tool (Bledsoe et al. 2017) to overcome sediment transport myths and misperceptions. Previous research has already provided recommendations about when capacity-based design is warranted, and the original CSR tool eliminated the need for measured sediment loads while leveraging the power of relative comparison when dealing with analytical uncertainty (Bledsoe et al. 2016, 2017, Stroth et al. 2017). To equitably compare NCD to ACD using the CSR tool, we incorporated sediment rating curves and a bed-load prediction model based upon specific (unit width) stream power (ω), both central features of FLOWSED/POWERSED (Rosgen 2006, 2009), and we tested our modified tool on four gravel/cobble bed streams with measured sediment data. . Despite substantial inaccuracies in analytical estimates of sediment transport magnitude for a single reach, we show how time-integrated, capacity-supply comparisons between two reaches can be quite accurate and useful for practitioners. We demonstrate the importance of sediment transport in stable channel design and take an initial step towards reconciling capacity-based analytical differences between ACD and NCD. Through constructive discourse we intend to help bridge the long-standing divide and ultimately advance the science and practice of stream restoration.

Background

Stable Channel Design Evolution

Stable channel design comprises a long history of our collective attempts to impose static, non-erosive designs on dynamic systems, while inconsistently building upon the critical linkage between channel form and sediment transport processes. The earliest channel interventions, in fact, were typically designed for hydraulic conveyance alone, as needed for water supply (e.g., irrigation, trans-basin diversions), flood “control” (e.g., stormwater conveyance, notorious river channelization), or infrastructure protection (Lagasse et al. 2012). Constructed trapezoidal channels can be highly efficient at moving water downstream, although sediment balance is another story. Tractive force design limits water velocity (*V*) and shear stress (τ) relative to particle size (gravel, riprap, etc.), underpinning criteria for channel lining materials (vegetation, concrete, etc.). Alternatively, alluvial channel design endeavors to account for the continuity of both water and sediment loads over time.

Through convergent evolution, science and practice have gradually shifted from simplistically estimating sediment competency for a single design discharge to more holistically evaluating sediment transport capacity across the entire hydrologic regime. Our understanding of sediment transport began with the threshold of motion and predicting individual particle movement. Shields pioneered the incipient motion concept, as predicted by critical shear stress (τ_c), and the Shields parameter is a dimensionless form relative to diameter grain size (Shields 1936). Stream restoration applications of incipient motion include determining the depth of (bankfull) flow necessary to move a particle of a given size (e.g., D_{84}). As such, sediment competency is a cornerstone of threshold channel design (Shields et al. 2003). With mobile gravel bed riffles, sediment competency might be a desirable target value, whereas rock for constructed grade control would be sized large enough to preclude movement. Hawley and Vietz (2016) used the competency concept to determine the critical discharge necessary to reduce the frequency of benthic disturbance in urban streams. Fundamentally, sediment competency relates a single flow (bankfull, critical discharge, etc.) to a single particle size.

Sediment capacity, in contrast, more broadly balances flows of water and sediment (NRCS 2007). Whereas sediment *competency* underpins threshold channel design, sediment *capacity* supports active-bed approaches (Shields et al. 2003). We view capacity-based stream restoration design as a key point of evolutionary convergence for the stable channel spectrum. Participants in Rosgen's Level III short course learn about channel stability, and

FLOWSED/POWERSED (Rosgen 2006) are linked models used to evaluate sediment capacity by NCD practitioners. The HEC-RAS stable channel design module includes a Copeland analytical design option, an ACD tool that creates a family of stable width-slope solutions for a specific channel depth and bankfull flow (Copeland 1994, Brunner 2010). Soar and Thorne (2001) further developed the CSR approach by incorporating full-spectrum design across an entire flow regime. Most recently, the CSR tool developed for the National Cooperative Highway Research Program (NCHRP) (Bledsoe et al. 2016, 2017, Stroth et al. 2017) broadly embodies ACD concepts while being straightforward enough for widespread use in practice.

As sediment transport analytical tools, the conceptual similarities between the CSR spreadsheet tool and FLOWSED/POWERSED are threefold: the use of flow duration curves (FDCs) to represent a broad range of flows, reliance upon hydraulic relations (e.g., Manning's equation), and leveraging relative comparison to predict channel stability, aggradation, or degradation. The most significant differences are the actual relationships powering the transport calculations (physics-based equations versus empirical relationships). In general, though, both ACD and NCD support stream restoration designs using hybrid approaches of hydraulic engineering combined with fluvial geomorphology.

Capacity/Supply Ratio

The CSR spreadsheet tool used in this study compares the cumulative sediment transport capacity of a design reach and its upstream supply reach across a broad range of flows (Bledsoe et al. 2016, 2017):

$$CSR = \frac{\int_{time} transport\ capacity\ of\ Design\ Reach}{\int_{time} transport\ capacity\ of\ Supply\ Reach} \tag{4.1}$$

In this context, a stable channel design is characterized by a CSR value close to 1, with higher or lower values corresponding to predicted degradation or aggradation, respectively. It has been suggested that CSR values within 10% of 1 may suffice for quasi-stable equilibrium (Soar and Thorne 2001).

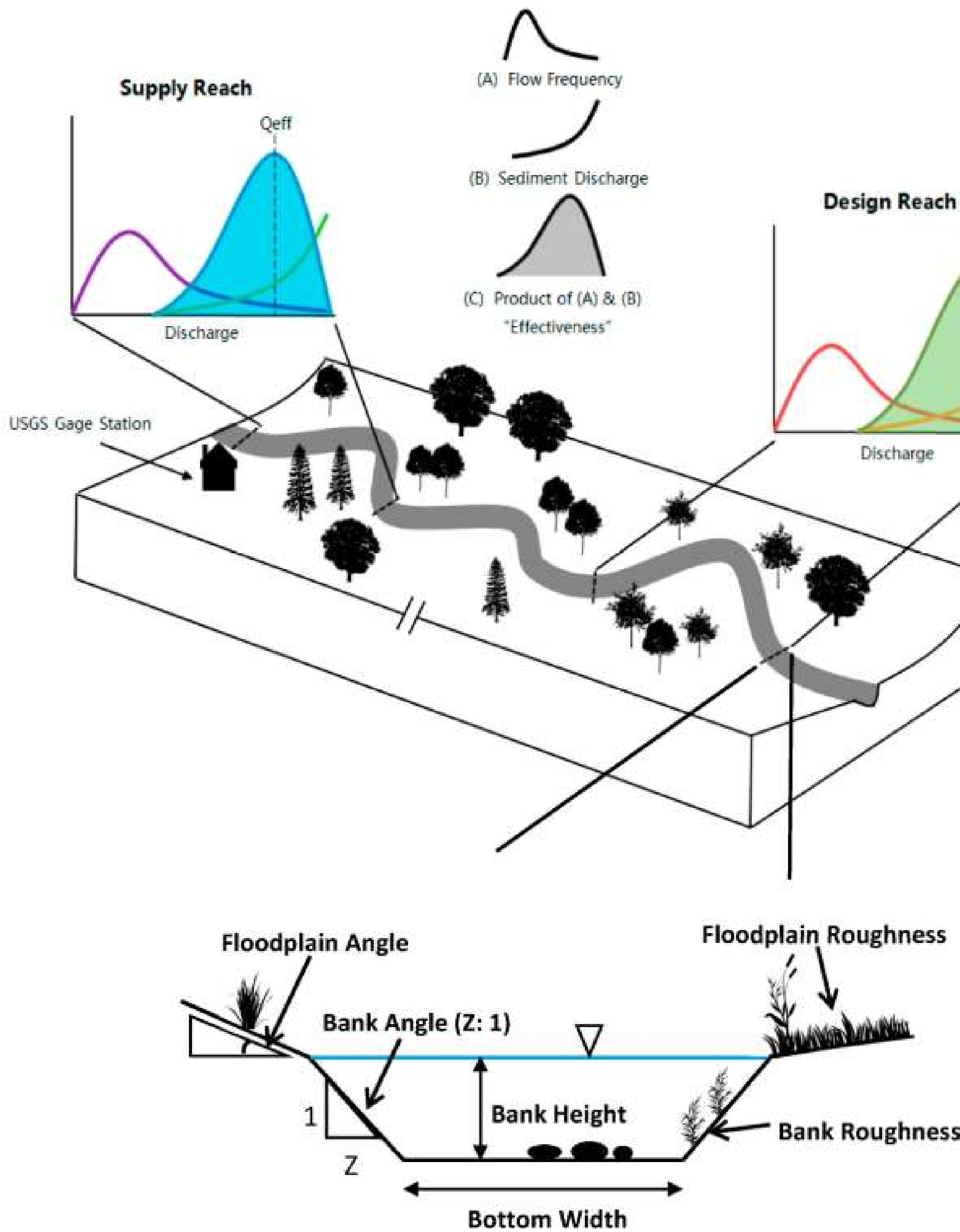


Figure 4.1. Graphical Representation of the Capacity/Supply Ratio (CSR) Tool (Bledsoe et al. 2016).

Integration across time involves a flow duration curve (FDC) of exceedance probability plotted against discharge, and sediment discharge is calculated for each defined hydrology range (bin). For any given discharge, the effectiveness is the product of probability (likelihood of discharge) and sediment discharge. Total effectiveness integrated over time can be strongly influenced by flows at the upper end, underscoring the importance of accurate flow records or otherwise good estimates. Figure 4.1 conceptually illustrates the effective discharge, Q_{eff}, as the bin that generates the largest effectiveness (Biedenharn et al. 2000), while the CSR is the green area under the design effectiveness curve divided by the blue area under the supply curve. Whereas Q_{eff} is calculated mathematically and broadly referenced in ACD, bankfull discharge (Q_{bkf}) is determined through field observation and more commonly associated with NCD. Q_{eff} and Q_{bkf} are both linked to the broader idea of dominant discharge, a theoretical channel forming discharge delineated by the threshold between in-channel and floodplain flows in naturally stable alluvial systems. While bankfull may be a point of contention (Lave 2009) or practical difficulty (e.g., incised channels, urban streams), floodplain connectivity is universally valued for energy dissipation and numerous ecological functions among stream restoration specialists, regardless of ACD or NCD background.

While the CSR is an ACD tool, NCD employs a parallel approach in FLOWSED/POWERSED, such that these sediment capacity models belong together in a single stable channel design toolbox. NCD practitioners are advised to calculate sediment transport capacity for “comparative” and “evaluation” conditions (Rosgen 2006, NRCS 2007). A decision point in the analysis is reached by comparing the total average annual transport rates. If the capacity of the evaluation reach is more or less than the comparative condition, degradation or aggradation is predicted, respectively. Thus, the CSR equation could be rewritten as:

$$CSR = \frac{\int_{time} transport\ capacity\ of\ Evaluation\ Reach}{\int_{time} transport\ capacity\ of\ Comparative\ Reach} \qquad (4.2)$$

In this way, NCD and ACD are more similar than different in terms of accounting for sediment transport. In an absolute sense, FLOWSED/POWERSED can evaluate a downstream reach by comparison to a stable upstream supply reach, conceptually comparable to the CSR tool. If the total transport rate in the design (evaluation) reach does not match the supply (comparative) reach, the practitioner is advised to adjust the design depth and/or slope (NRCS 2007). In addition to the spatial or design evaluation of downstream/upstream reaches, FLOWSED/POWERSED has been recommended for a range of scenario types: departure (existing/reference), temporal (after/before), or floodplain connectivity (floodplain/no floodplain) (Rosgen 2006). Bridge or culvert analysis can also be performed as a type of FLOWSED/POWERSED design evaluation (Athanasakes and Rosgen 2010). While the current CSR spreadsheet tool uses supply and design reach terminology, it can similarly be applied to a wider range of evaluation scenarios.

As with the CSR tool, the hydrologic cornerstone of FLOWSED/POWERED is an FDC. The primary difference between NCD and ACD with regard to FDCs is that FLOWSED/POWERSED users may truncate analysis, using flows only up to the bankfull discharge and disregarding flood flows (Rosgen 2009). The main exception is when the comparative condition and evaluation reaches have notably different floodplain geometry, such as an existing incised channel versus proposed floodplain reconnection (Wildland Hydrology 2016). The CSR tool, on the other hand, typically uses the full spectrum of flows, including the most infrequent but extreme events. However, less is understood about widely varying degrees of uncertainty in the influence of flow and transport extremes as well as floodplain hydraulic processes.

While a stable upstream supply reach serves as an ideal comparison for downstream sediment transport capacity, the reality may be more complex. Regional legacy sediment, for example, may strongly influence sediment supply and channel stability (e.g., Jackson et al. 2005, Walter and Merritts 2008), and the proponents of both ACD and NCD readily acknowledge the importance of watershed context. While documentation for the CSR tool describes how to identify a stable supply reach, it also includes guidance for projecting future streamflow changes due to land use (Bledsoe et al. 2017). Furthermore, the CSR approach can provide useful information about the relative magnitude of design consequences where the upstream sediment supply is unstable. Similarly, FLOWSED/POWERSED is explicitly intended for use in performing watershed assessments of river stability in the context of sediment supply (Rosgen 2009), and NCD assumes that a stable channel design closely depends upon stable watershed conditions.

Sediment Transport Relationships

Before tabulating sediment transport rates and effectiveness, the CSR spreadsheet tool first calculates hydraulic characteristics of both the supply and design reaches in a stage-discharge fashion. For each hydrologic bin, the code embedded in the tool iteratively calculates depth based on the Manning roughness coefficient (n). Floodplain and channel bank roughness values are input by the user, while the channel bed roughness is estimated using a relative roughness (R/D_{84}) approach (Limerinos 1970), where R is the hydraulic radius, and D_{84} is bed material size. R/D_{84} increases with depth, so the calculated bed roughness gradually decreases with additional discharge. While most practitioners may be more accustomed to entering a single Manning n value for the channel, the CSR tool uses a shear stress partitioning method to incorporate bank roughness.

The CSR spreadsheet tool estimates sediment transport using conventional equations for sand bed and gravel/cobble bed streams. The sand bed calculations use the Brownlie (1981) total load equation (bed load plus suspended load), combined with Brownlie's bedform and depth predictor. The gravel stream calculations use two different surface-based bed-load equations, depending on the sand fraction. Parker (1990) is used for channels with a low sand fraction (<3-5%), and Wilcock and Crowe may be preferred if the channel has an appreciable sand fraction (>5%) (Wilcock et al. 2001, Wilcock and Curran 2003). These three conventional transport relationships share similarities. First, all three require grain size distributions: D_{16}, D_{50}, *and* D_{84} for Brownlie, and full grain size distributions for the gravel equations. Second, they each incorporate incipient motion based on some type of hydraulic excess compared to a reference value: grain Froude number for Brownlie, and τ for Parker as well as Wilcock and Crowe. For bed-load prediction in gravel streams, inclusion of subsurface material gradation could help reduce one known source error (e.g., Parker and Klingeman 1982, Andrews 1984), but the CSR spreadsheet tool is not intended to ascribe a high degree of precision to sediment transport rates or total effectiveness. The predicted sediment transport rate in either the supply or design reach is subject to error in an absolute sense, but the CSR relative comparison provides useful information about the magnitude of difference between the two reaches, and thus the ability to predict aggradation or degradation despite simplifying assumptions.

FLOWSED/POWERSED combines two types of sediment transport relationships. First, FLOWSED uses a sediment rating curve (SRC) to predict transport rates for a given comparative reach (Rosgen 2006). SRCs are not unique to NCD, simply being a method to create individualized regression equations based on measured sediment loading to predict sediment transport as a function of discharge (e.g., Asselman 2000):

$$Q_s = aQ^b \qquad (4.3)$$

where Q_s is sediment transport rate, and Q is discharge. As a fully empirical relationship, the SRC readily sidesteps the mechanistic physics underpinning sediment transport (e.g., V, τ), which are not fully understood (e.g., Vanoni 2006, García 2008). However, apart from their inability to incorporate theoretical nuances, SRCs have two basic drawbacks. First, the necessary data are not easy to collect, being based on measured sediment data across a broad range of flows, requiring time and effort. Researchers recently compiled and continue to build a publicly available database which should help future research overcome the relative data scarcity (Hinton et al. 2017), and the Rosgen River Database is currently under development. Second, SRCs are sensitive to the data collection methodology (bed-load traps, samplers, etc.). Helley-Smith samplers are commonly used (Williams and Rosgen 1989, Andrews 1994), though not without concerns (Bunte and Abt 2009, Bunte et al. 2010). Finally, the lognormal transformation used to create SRCs may bias regressions to underpredict transport, so there is a recommended adjustment factor (Ferguson 1986).

Researchers advanced the use of SRCs by developing dimensionless sediment rating curves (DSRCs), taking the general form (Troendle et al. 2001, Rosgen 2010):

$$Q_s/Q_{sbkf} = a\,(Q/Q_{bkf})^b + c \qquad (4.4)$$

where Q_s is sediment transport rate, Q_{sbkf} is bankfull sediment transport rate, Q is discharge, and Q_{bkf} is bankfull discharge. Instead of sediment measurements across a range of flows, applying this type of DSRC to create a local SRC requires a single pair of bankfull values (Q_{bkf}, Q_{sbkf}). The literature focuses on four regional DSRCs, the Pagosa bed-load and suspended load regression curves for channels with Good/Fair or Poor stability (Rosgen 2010). Though less onerous than creating an SRC from sediment data collected over time, it may still be difficult to obtain the necessary field measurements (Hinton et al. 2018). While Q_{bkf} can be estimated using widely available regional regression equations, DSRC proponents also advocate the development and use of regional sediment curves where localized data are unavailable (Rosgen 2006, 2010). The Pagosa curves have been poorly received in the larger scientific community (Lave 2008), although researchers have partly addressed the controversial nature of these empirical relationships by finding underlying similarities to the theoretically based Parker equation (Hinton et al. 2012). Comparing the FLOWSED model to the CSR framework, sediment transport in the supply (comparative) reach is predicted using an SRC, independent of hydraulic relations.

The POWERSED model, on the other hand, combines the stage-discharge hydraulic relations in the comparative reach with the corresponding transport rates (from FLOWSED) to predict sediment transport as a function of ω (Rosgen 2009), defined here as (Bagnold 1960):

$$\omega = \tau V \qquad (4.5)$$

where ω is specific stream power, τ is shear stress, and V is velocity. Bagnold was the first to develop an sediment transport relationship based on ω (1980), an approach recommended by Gomez and Church (1989) and subsequently modified by others (Martin and Church 2000, Ferguson 2005, Lammers and Bledsoe 2018). The POWERSED sediment reference curve (Q_s versus ω) is then applied to the evaluation reach. The general idea is that an increase or loss of ω corresponds to increased or reduced sediment transport, respectively, and the designer is advised to consider contributing factors, namely width, depth, and slope. While the CSR spreadsheet tool first calculates specific sediment discharge (q_s) and then multiplies by bottom width to find Q_s, POWERSED directly predicts Q_s. This distinction matters most when the design reach width significantly departs from the supply reach.

Table 4.1. Selected Sediment Transport Relationships for Gravel and Cobble Bed Streams.

Relationship	Description	Sediment Type	References
Parker (CSR Tool)	Incipient motion based on grain size, shear stress (hydraulic relations)	Bedload with sand fraction <3-5%	(Parker 1990, Bledsoe et al. 2017)
Wilcock & Crowe (CSR Tool)	Incipient motion based on grain size, shear stress (hydraulic relations)	Bedload with sand fraction >5%	(Wilcock et al. 2001, Wilcock and Curran 2003, Bledsoe et al. 2017)
FLOWSED	Empirical relationship based on *Q* (e.g., measured SRC, Pagosa DSRC)	Bedload, suspended load	(Troendle et al. 2001, Rosgen 2006, 2009, NRCS 2007)
POWERSED	Empirical relationship based on unit stream power (hydraulic relations)	Bedload, suspended load	(Rosgen 2006, 2009, NRCS 2007)

Methods

We compared ACD and NCD sediment transport analyses of four gravel and cobble bed streams in Colorado to investigate capacity-based stable channel design using multiple bed-load relationships. Modifying an existing CSR spreadsheet tool enabled us to generate channel stability predictions for both approaches and assess their similarities, differences, and relative sensitivities to channel geometry and slope alterations. It can be difficult to follow both ACD and NCD through the various stages of capacity-based sediment transport analyses, and helpful overviews of the CSR spreadsheet tool and FLOWSED/POWERSED, respectively, can be found in Figure 2-7 of NCHRP Report 853 (Bledsoe et al. 2017), and Figures 11-19 and 11-25 of the Stream Restoration Design National Engineering Handbook (NRCS 2007).

Study Reaches

The study sites were selected based on availability of the input data needed to implement both ACD and NCD in the modified CSR tool, drawing from existing literature (Williams and Rosgen 1989, Hinton et al. 2017, 2018) and supplemented with additional data from the Rosgen River Database. While the CSR tool accommodates both flow records and pre-existing FDCs

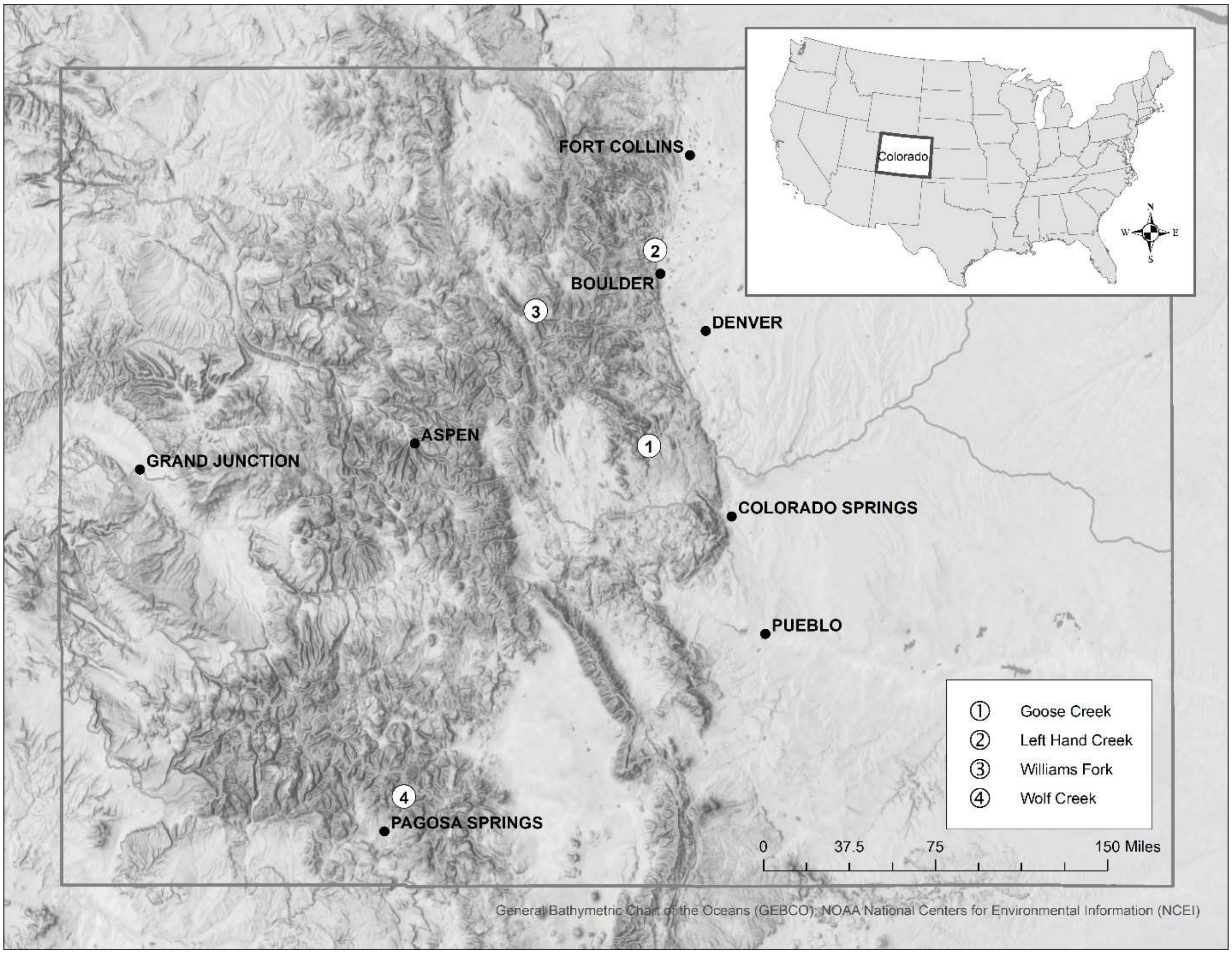


Figure 4.2. Colorado Site Locations.

(for example, estimated by hydrologic models), for purposes of this study, we limited potential sites to streams with U.S. Geological Survey (USGS) gage records. Measured bed-load rates were necessary to generate SRCs, while the DSRCs required Q_{bkf}, Q_{sbkf}, and channel stability (good/fair, poor). Full grain size distributions were necessary for the Parker (1990) and Wilcock and Crowe (2003) equations, and we looked for sites with a range of sand fractions. D_{84} values were used for hydraulic calculations (relative roughness). Bankfull channel slope, width, and depth as well as floodplain width were needed for the supply reaches.

From a surprisingly small number of studied reaches meeting all of the selection criteria, we identified four gravel and cobble bed streams, all located in the Rocky Mountain region of

Table 4.2. Reach Characteristics of Study Sites.

	Goose	Left Hand	Williams	Wolf
Hydrology				
Elevation (m)	2298	1752	2682	2375
Drainage Area (km^2)	211	131	232	46
Q_{bkf} (cms)	7.59	5.55	22.65	7.93
Sediment				
Bed material	Gravel	Cobble	Gravel	Gravel
D_{84} (mm)	50	102	79	87
Sand fraction	0.00	0.06	0.05	0.10
Channel Stability	Poor	Poor	Poor	Good/Fair
Q_{sbkf} (kg/s)	5.21	14.01	0.07	0.10
Channel				
W_{bkf} (m)	16.56	9.39	17.50	13.53
D_{bkf} (m)	0.37	0.46	0.55	0.49
W_{bkf}/D_{bkf}	45	20	32	28
Slope (m/m)	0.0037	0.0229	0.0058	0.0163
Floodplain				
W_{fp} (m)	32.6	46.6	520	25.0
W_{fp}/W_{bkf}	2.0	5.0	29.7	1.8
Data source	a, b, c	a, b, c	b, d	a, b, e

Data sources: (a) Brigham Young University Bedload Transport Database (Hinton et al. 2017), (b) Rosgen River Database, (c) US Forest Service Water Division 1 Fluvial Study Site, (d) Williams and Rosgen (1989), (e) Pagosa Dataset.

Colorado (Figure 4.2). Goose Creek and Left Hand Creek are located in the South Platte River Basin, while Williams Fork and Wolf Creek belong to the Upper Colorado region. Table 4.2 summarizes relevant site characteristics and inputs to the CSR spreadsheet tool. The site elevations are between 1,700 and 2,700 m, with channel slopes ranging between 0.004 and 0.023 m/m. The streams at higher elevations are characterized by predominantly snowmelt-driven hydrologic regimes. Left Hand Creek at the base of the Front Range is also influenced by summer monsoons.

CSR Tool Modifications

<u>SRCs</u>

We started with the existing CSR tool (Bledsoe et al. 2016, 2017) as a practical ACD application based on conventional sediment transport relationships (Brownlie 1981, Parker 1990, Wilcock and Curran 2003). For NCD practitioners, FLOWSED/POWERSED are the recommended models for predicting sediment transport (Rosgen 2006, 2009). Calibrated using Q_{bkf} and Q_{sbkf}, FLOWSED uses Equation 4.4 to predict Q_s as function of Q. To create an equitable basis for NCD comparison, we added SRC options to the CSR tool based on either measured sediment data or a user-defined relationship. Similar to FLOWSED, the modified CSR tool estimates supply reach bed-load transport rates as a function of discharge (Equation 4.3). For field measured sediment data, the SRC feature adjusts the coefficient of the power function per Ferguson (1986).

Unlike FLOWSED, the CSR tool code was not modified to directly calculate Q_s from a DSRC (e.g., Pagosa), but requires the intermediate step of a user-defined SRC in the form of Equation 4.3. Figure 4.3a shows the difference between the values predicted by the Pagosa formulas (Equation 4.4) and the corresponding user-defined SRCs (Equation 4.3). The Pagosa good/fair equation has a much closer power regression fit than the poor equation as a result of the constants of -0.0113 and 0.07176 used in Equation 4.4 for Pagosa Good/Fair and Poor, respectively. Calibrated using the same Q_{sbkf}, a channel with poor stability is predicted to move much more sediment at the lowest flows compared to one with better stability. The reader is also advised that the FLOWSED/POWERSED models are capable of simultaneously calculating both bed-load and suspended sediment transport rates using the corresponding Pagosa formulas, including the sand fraction component of the suspended load. The suspended sand differs from the sand fraction (Table 4.2) which is in the gravel matrix and influences gravel mobility (Wilcock et al. 2001). For this study, however, we focused on bed load alone for reasons mentioned in the Sediment Transport Relationships section above.

Supply Reach

With the SRC mode in the modified CSR tool, the supply reach functions like a POWERSED comparative condition. The spreadsheet tool combines the estimated Q_s and calculated ω at each stage, generating a reference curve to predict Q_s as a function of ω. Figure 4.3 shows an example of the discharge-based prediction (Figure 4.3b) and supply reach ω calculation (Figure 4.3c) used to create the final power-based prediction (Figure 3d). Our

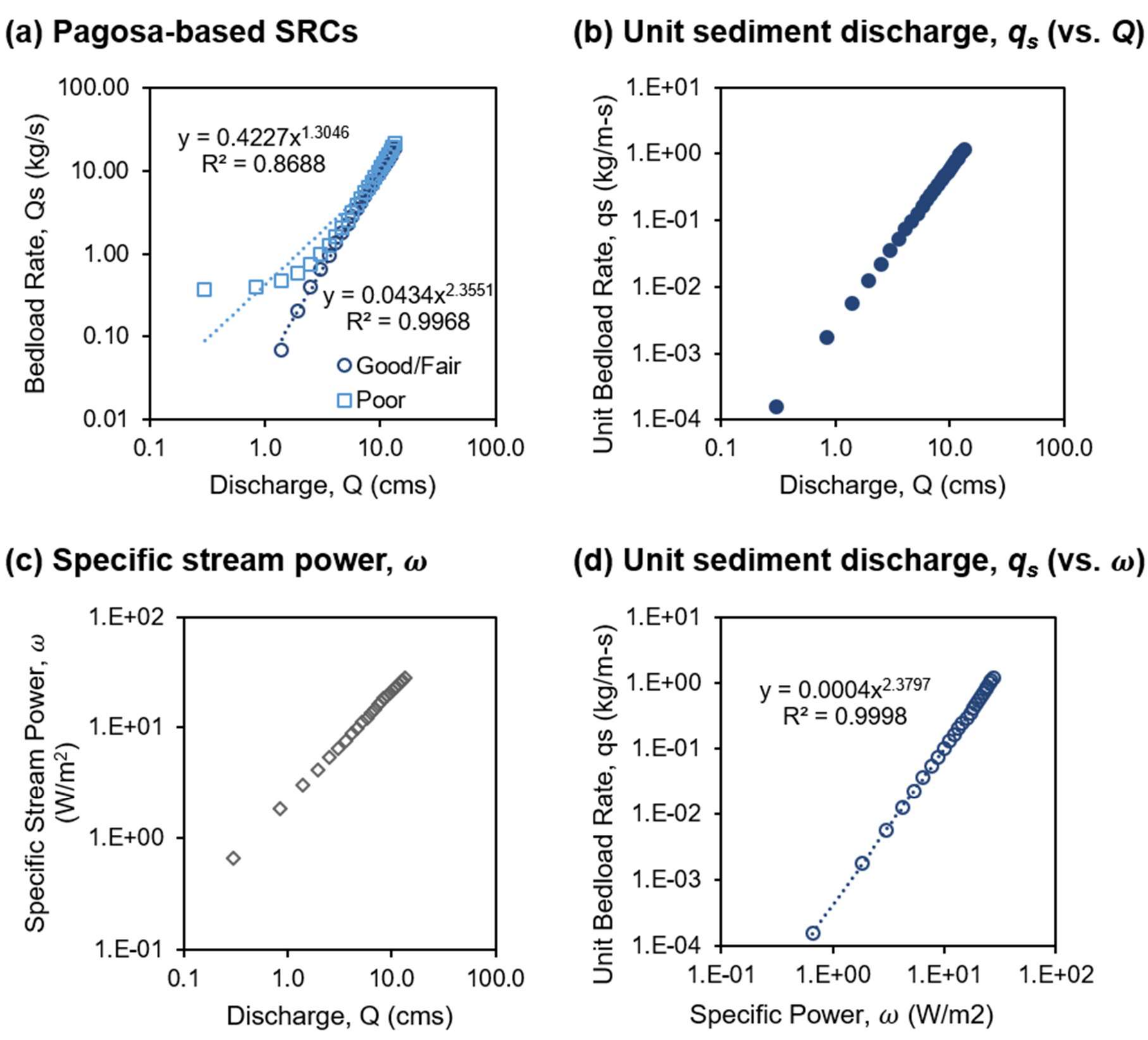


FIGURE 4.3. Goose Creek Supply Reach. (a) Predicted bed-load transport rates based on the Pagosa Good/Fair and Poor formulas and their corresponding sediment rating curves (SRCs); (b) SRC per unit width (based on Pagosa Good/Fair); (c) specific stream power from hydraulic relations; (d) reference curve used to predict design reach q_s.

application potentially departed from POWERSED in three primary ways. First, the CSR tool calculates bed roughness for gravel streams with the Limerinos (1970) based on the D_{84} value from the grain size distribution rather than user-defined values. As a result, the resulting calculated discharge corresponding to bank height may not be equal to a field-measured Q_{bkf} value. Second, bed-load transport is limited to the active channel bed, so for overbank flows, ω is based on the channel τ and V rather a total average including the floodplains. In nature, when flow spreads out onto the floodplain, the channel flow remains fast and deep, and ω continues to increase with depth rather than abruptly dropping off. Finally, we created reference curves (e.g., Figure 4.3d) to predict unit sediment discharge (q_s) rather than total sediment discharge (Q_s) as a function of ω, a distinction first made by Bagnold (1960) and that especially matters the further the design width deviates from the supply reach:

$$q_s = a\omega^b \qquad (4.6)$$

where q_s is sediment discharge per unit width and ω is specific stream power.

Design Reach

The modified CSR spreadsheet tool includes two separate design routines, the "design reach" and "design analysis" tabs. For both of the CSR design routines, the SRC mode calculates hydraulics at each stage and predicts q_s using the supply reach reference curve (Equation 4.6, Figure 4.3d), similar to the POWERSED approach (Rosgen 2009). If the regression (Equation 4.6) closely fits the supply reach (e.g., Figure 4.3d), a design with channel and floodplain input values equal to the supply reach should likewise predict the same bed-load transport at each stage as well as overall effectiveness, with CSR = 1.

The CSR tool's original design reach routine uses the total effectiveness of the supply reach and generates a width-slope curve or "swoosh" comprising a family of stable channel design solutions with CSR =1, similar to the Copeland analytical design in HEC-RAS and SAM (Copeland 1994, Brunner 2010). Values above or below the curve (CSR > 1, CSR < 1) predict design reach degradation or aggradation, respectively. The user can manually adjust the target total effectiveness (tons/day) to explore a broader range of design solutions for CSR values of 0.9 or 1.1, for example. The swoosh generated by the CSR tool does not have a POWERSED equivalent.

In contrast to the width-slope curve, the design analysis routine in the modified CSR tool enables the user to tweak any of the input values in the design reach, as would correspond to a modified slope (S) or width/depth ratio (W_{bkf}/D_{bkf}), for example. The tool tabulates the total sediment capacity of the design reach and calculates a CSR value using Equation 4.1. If CSR ≈ 1, the design is estimated to be in quasi-equilibrium.

Floodplain

We added floodplain width to the tool inputs for the supply and design reaches, as the original version had only a floodplain slope and assumed an infinite width. The modified tool adds vertical walls to the edge of the floodplain, which supported our investigation of the effects of valley confinement and channel entrenchment. To the extent that NCD relies heavily upon bankfull concepts, there is a characteristic distinction between bankfull and flood flows, and FLOWSED/POWERSED practitioners may opt to truncate an FDC at a value close to Q_{bkf} (Rosgen 2009). Accordingly, we modified the CSR tool to calculate both bankfull and total effectiveness and their corresponding CSR values for the design analysis routine. The modified CSR tool's "bankfull" discharge is the highest supply reach discharge bin that does not result in overbank flow. As noted in the Design Reach section above, this may not match a field-measured Q_{bkf}.

Analyses

We analyzed each study site using both standard transport relations and SRCs. We ran design scenarios using the Parker relationship for all sites as well as the Wilcock-Crowe relationship for sites with applicable sand fractions (Left Hand Creek and Wolf Creek). For the SRCs based on measured sediment data, the tool fit a regression curve with bias-corrected coefficient (Ferguson 1986). For the DSRCs (Pagosa formulas), we first predicted the supply reach Q_s for each hydrology bin, and then fit a regression curve (e.g., Figure 4.3a). Table 4.3 provides a summary of the final power regression values for each of these SRC scenarios.

Table 4.3. Sources Used to Generate SRCs and Parameters Used to Predict Supply Reach Bed-Load Transport. Includes Pagosa dimensionless sediment rating curves (DSRCs).

Stream	**Measured**		**Pagosa Good/Fair DSRC**		**Pagosa Poor DSRC**	
	a	**b**	**a**	**b**	**a**	**b**
Goose	0.0052	3.59	0.043	2.36	0.42	1.30
Left Hand	0.00052	6.57	0.27	2.28	1.3	1.63
Williams	0.016	0.51	0.000046	2.34	0.0011	1.37
Wolf	0.0025	1.98	0.00058	2.52	0.0089	1.15

To explore the influence of channel stability, we ran scenarios for each site using both the Pagosa Good/Fair and Poor bed-load equations. The Pagosa equations were empirically derived for specific channel types, and we should expect dissatisfactory results using the Poor parameters with a Good/Fair channel (and vice versa). Nonetheless, we wanted to simulate scenarios with uncertainty about the actual channel stability and assess the potential design consequences of applying inappropriate equations. To test the viability of our SRC approach for sites with no measured sediment loads, we created a new Pagosa scenario for each site with a unit value for Q_{sbkf} (1 kg/s) used in Equation 4.4. While altering the Q_{sbkf} value impacted the absolute value of sediment discharge rates, the difference was captured by the magnitude of the coefficient in Equation 4.3, and the exponent remained unchanged. This behavior was mathematically similar to the Ferguson adjustment. Additionally, to explore sensitivity to Q_{bkf}, we ran two separate scenarios with Q_{bkf} equal to one half and twice the field observations. These last analyses were meant to reflect real-life scenarios for ungaged streams lacking field bankfull indicators, where otherwise a $Q_{1.5}$ or Q_2 might be estimated using widely available regional regression equations (e.g., StreamStats).

For supply and design reach inputs, the CSR tool uses simplified channel sections, in contrast with detailed cross section survey inputs to POWERSED. We assumed rectangular channel geometry (W_{bkf} x D_{bkf}), a common approximation for relatively wide channels. In general, the CSR tool showed extremely low sensitivity to floodplain input values, and site specific data were unavailable, so we assumed floodplain slopes of 100:1. Similarly, we used Manning roughness values of 0.04 and 0.06 for the channel banks and floodplains, respectively, realistic values (Chow 1959, Barnes 1967) in lieu of site specific data. We were concerned that introducing additional complexity and variation (e.g., floodplain and roughness characteristics) between sites could have confounded interpretation of results, but regardless, analyses with the CSR tool were ultimately insensitive to our simplifying assumptions, as the vast majority of sediment transport occurs along the bottom width of the channel. For each transport relationship, we generated standard width-slope curves (CSR = 1) for design reaches with a bank height equal to the supply reach, as well as curves corresponding to CSR values of 0.9 and 1.1. To explore scenarios that might depart from CSR ≈ 1, we investigated sensitivity to three basic variables: channel *W*/*D* ratio, slope, and floodplain width. First, we computed CSR values for design reaches with *W*/*D* values equal to 15, 25, and 40. The first two are representative values for single thread streams in the Colorado region, while the higher value is approaching the threshold for braiding. Second, we calculated CSR values for design reaches with slopes +/- 10 percent compared to the supply reach. Finally, to investigate the effects of floodplain confinement, we varied the design reach ratio of W_{fp}/W_{bkf} (2, 3, 5, 10). For each design analysis, we calculated CSR values for both total and "bankfull" effectiveness, with the latter value corresponding to the supply reach hydraulics (but not necessarily field observations).

Results

Our modified CSR spreadsheet tool enabled us to predict relative channel stability using both conventional bed-load transport equations (ACD) and empirical sediment regression curves (NCD), and neither approach required measured sediment loads. Using a generic Pagosa DSRC with either actual or arbitrary Q_{sbkf} values resulted in identical stable channel designs with the NCD approach, although field data could be used to generate site-specific SRCs. For most sites, the stable channel design solutions based on measured data most closely matched Parker and/or Pagosa Good/Fair, both of which also showed the greatest CSR sensitivity in response to altered W/D ratio and slope. We found that design swoosh and CSR differences among the various transport relationships became more extreme the further the design width deviated from the supply reach, such as channel or floodplain constrictions at roadway crossings.

Supply Reaches

Figure 4.4 shows predicted Q_s for each supply reach based on multiple sediment transport relationships. For Parker and Wilcock & Crowe, Q_s is a function of hydraulic calculations, not just Q. For example, the Parker curves for Williams Fork and Left Hand Creek (Figures 4.4b,c) show a characteristic shift from lower to upper phase as it passes a shear stress inflection point. However, we found unexpected discontinuities at the calculated Q_{bkf}, most visible in Goose Creek (Figure 4.4a), caused by the CSR tool shift from in-channel to overbank flow calculations which results in different flow partitioning methods. Briefly, the tool accounts for bank

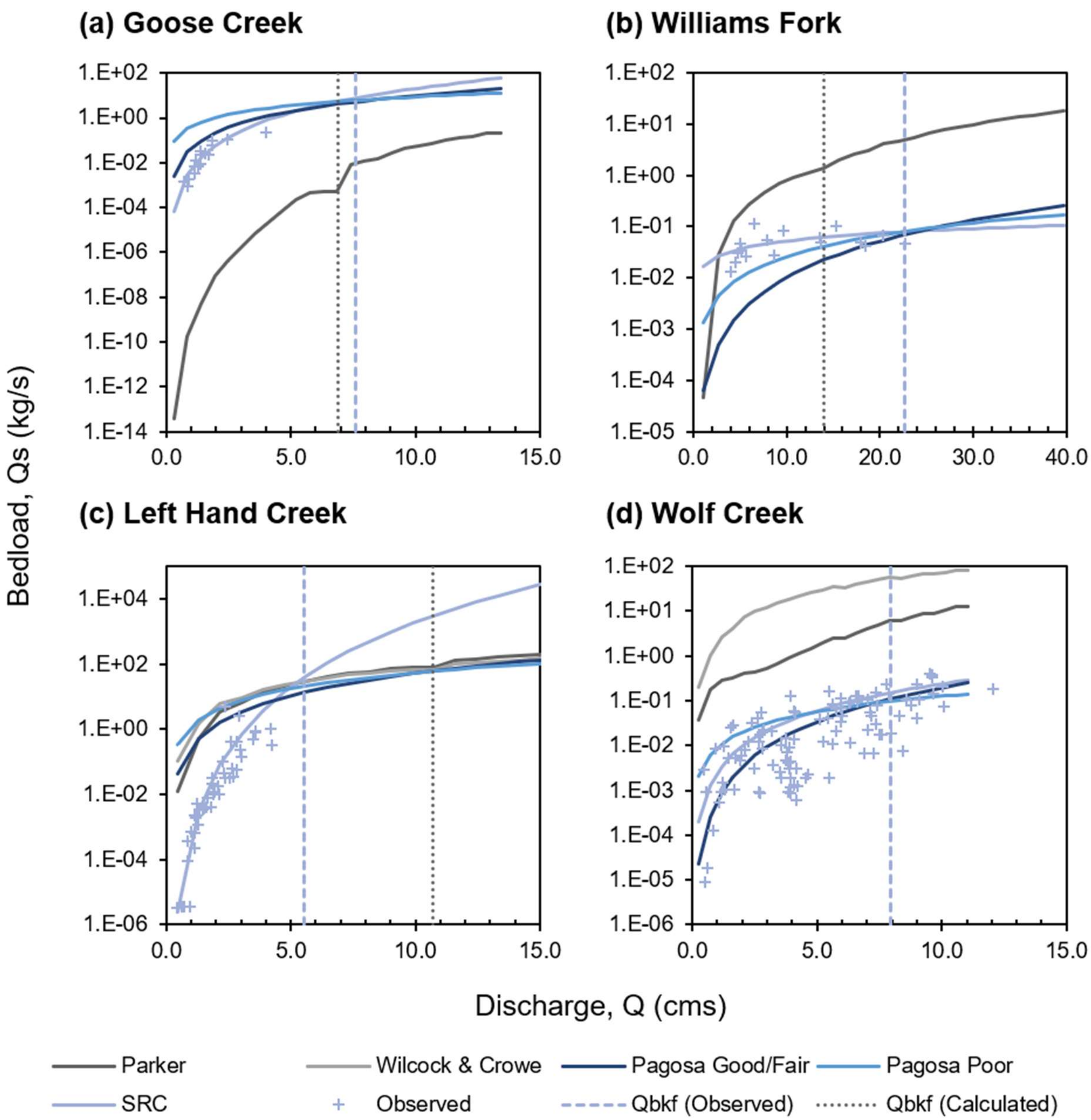


Figure 4. 4. Supply Reach Predicted and Observed Q_s. Wilcock & Crowe was only applicable for Left Hand Creek (c) and Wolf Creek (d) based on appreciable sand fractions. For Wolf Creek (d), all flow calculations remained in-channel, so Q_{bkf} (Calculated) could not be determined and only Q_{bkf} (Observed) is shown. Helley-Smith bed-load sampling method.

roughness effects on in-channel flows, but not overbank flows. As a result, the tool overestimates the bed hydraulic radius and corresponding transport capacity when the flow goes over bank. In contrast, the SRCs and Pagosa variations are directly estimated as a function of Q for the supply reach, regardless of hydraulic relations, so the overbank discontinuity only impacts predicted Qs in the design reach.

For most sites, the Q_s predicted by standard transport relationships exceeded the sampled sediment data. This may not be surprising since Parker was developed using a bed-load trap, and Wilcock & Crowe was based on flume data, whereas the observed values were obtained with a Helley-Smith sampler (which can underestimate total sediment transport rates). Discrepancies between predictions from the physics-based equations and measured data may also have been related to variables underlying the hydraulic calculations, namely roughness. The roughness values for the active channel bed were calculated by the CSR tool as a ratio of hydraulic radius to grain size (D_{84}), such that the relative roughness decreased with discharge. Although the CSR tool also incorporated roughness values for floodplain and banks, our analyses showed relative insensitivity to those particular inputs. However, the overall roughness of a natural channel includes additional factors such as sinuosity and bedforms. For all sites, the Pagosa Good/Fair curve intersected the Poor curve at Q_{bkf}, because both were estimated as a function of Q/Q_{bkf}, and the same Q_{sbkf} was assumed for both equations. In reality, we would reasonably expect Q_{sbkf} for a supply reach with poor stability to exceed that of the same channel in better condition.

Figure 4.5 shows the supply reach effectiveness curves for each site, in which the predicted Q_s for each hydrology bin was multiplied by probability of that discharge occurring. The Wolf Creek curves (Figure 4.5d) all predicted maximum effectiveness at the same discharge (Q_{eff} = 7.5 cms), while the other sites showed more variation between different sediment transport relationships. For Goose Creek (Figure 5a), the Q_{eff} for Parker and the SRC were closest, while Left Hand Creek had the same Q_{eff} for all curves except the SRC. Notable curves include sites with Q_{eff} in the first hydrology bin, such as SRC and Pagosa Poor in Figure 4.5b, or in the last hydrology bin, as with SRC in Figure 4.5c. Total effectiveness corresponds to the integrated area under each curve. Closely tied to the sediment discharge in Figure 4.4, we found differences in supply reach effectiveness spanning multiple orders of magnitude. In most cases, the greatest differences occurred at the lowest flow ranges.

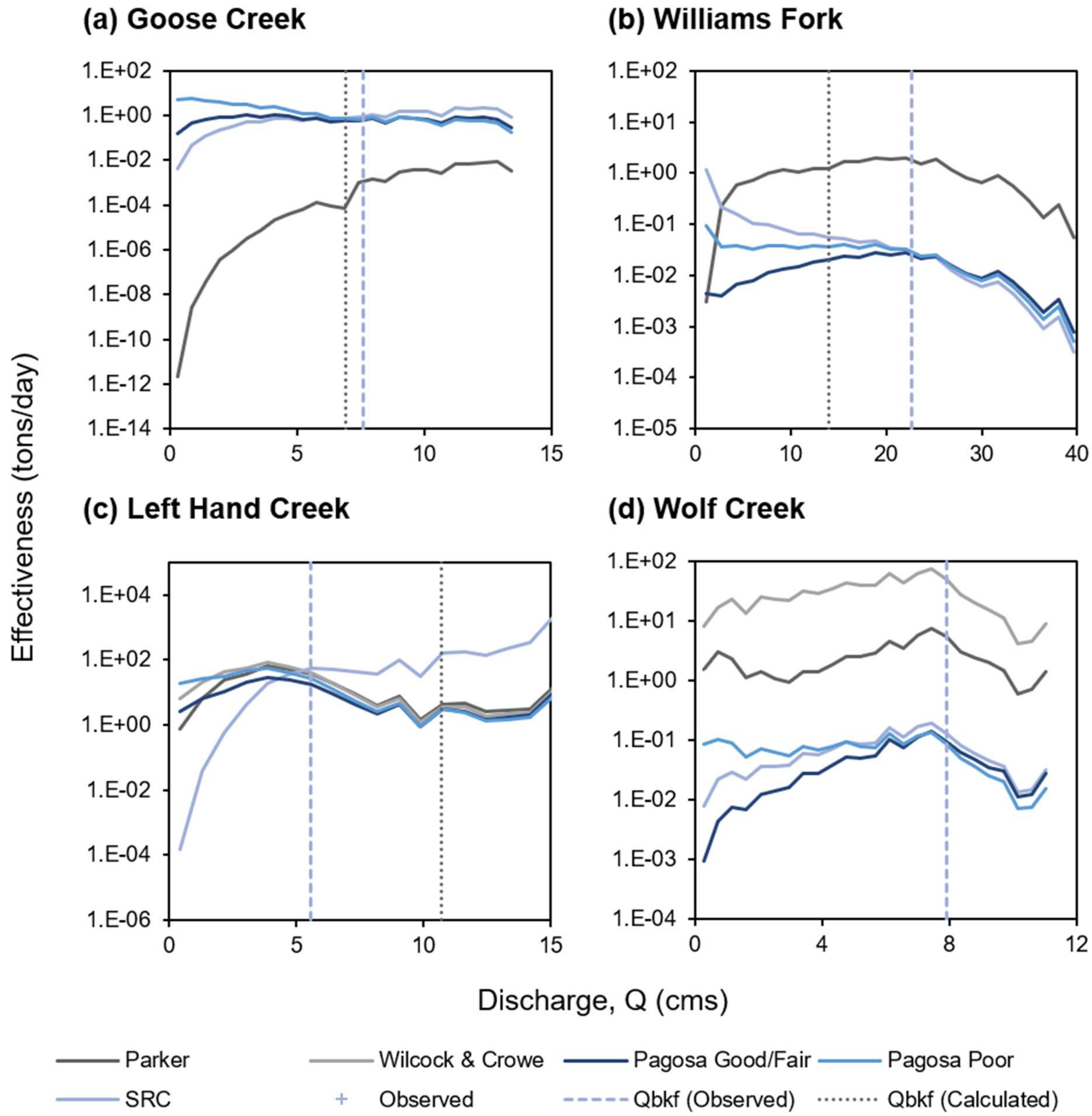


Figure 4.5. Supply Reach Effectiveness Curves. For Wolf Creek (d), all flow calculations remained in-channel, so Q_{bkf} (Calculated) could not be determined and only Q_{bkf} (Observed) is shown. Wilcock & Crowe was only applicable for Left Hand Creek (c) and Wolf Creek (d) based on appreciable sand fractions.

Design Curves

Figure 4.6 shows slope-width solutions for design reaches with a constant bank height equal to the supply reach. The points along each curve predict CSR = 1, and the areas above and below each curve predict degradation and aggradation, respectively. The intersection of curves at the supply reach width (W_{bkf}) corresponds to a design channel width and slope equal to the supply reach. In general, the greatest divergences occurred where the design width significantly deviated from the supply reach. Even so, some conventional transport relationships and SRCs (including Pagosa variations) generated similar designs.

For Goose Creek (Figure 4.6a), Parker was nearly identical to the Pagosa Good/Fair (+/- 0.0003), while Pagosa Poor differed the most (+/- 0.002). In Williams Fork (Figure 4.6b), Parker closely matched Pagosa Good/Fair (+/- 0.0003), while the SRC was a distinct outlier (+/- 0.02), and a backwards swoosh by comparison. Considering possible underlying reasons, we noted that the R^2 value of the Williams Fork SRC was only 0.29, with an unusually low exponent (0.51) and relatively high bed-load rates at the lower flows (Figure 4.4b), such that Q_{eff} was the lowest hydrology bin (Figure 4.5b). In Left Hand Creek (Figure 4.6c), however, both Parker and Wilcock & Crowe were closest to Pagosa Poor (+/- 0.002). For Wolf Creek (Figure 4.6d), Parker was closest to the SRC (+/- 0.001) and Pagosa Good/Fair (+/- 0.002), while Wilcock & Crowe was more similar to Pagosa Poor (+/- 0.006).

Setting aside differences compared to conventional transport relationships, we were somewhat surprised by the similarities and differences between Pagosa variations, with Wolf Creek (Figures 4.7a,b) as an example of good/fair channel stability. First, altering Q_{sbkf} affected only the supply reach absolute magnitudes (Figure 4.7a), while the slope-width solution remained identical (Figure 4.7b). In other words, focusing on CSR means we did not need any

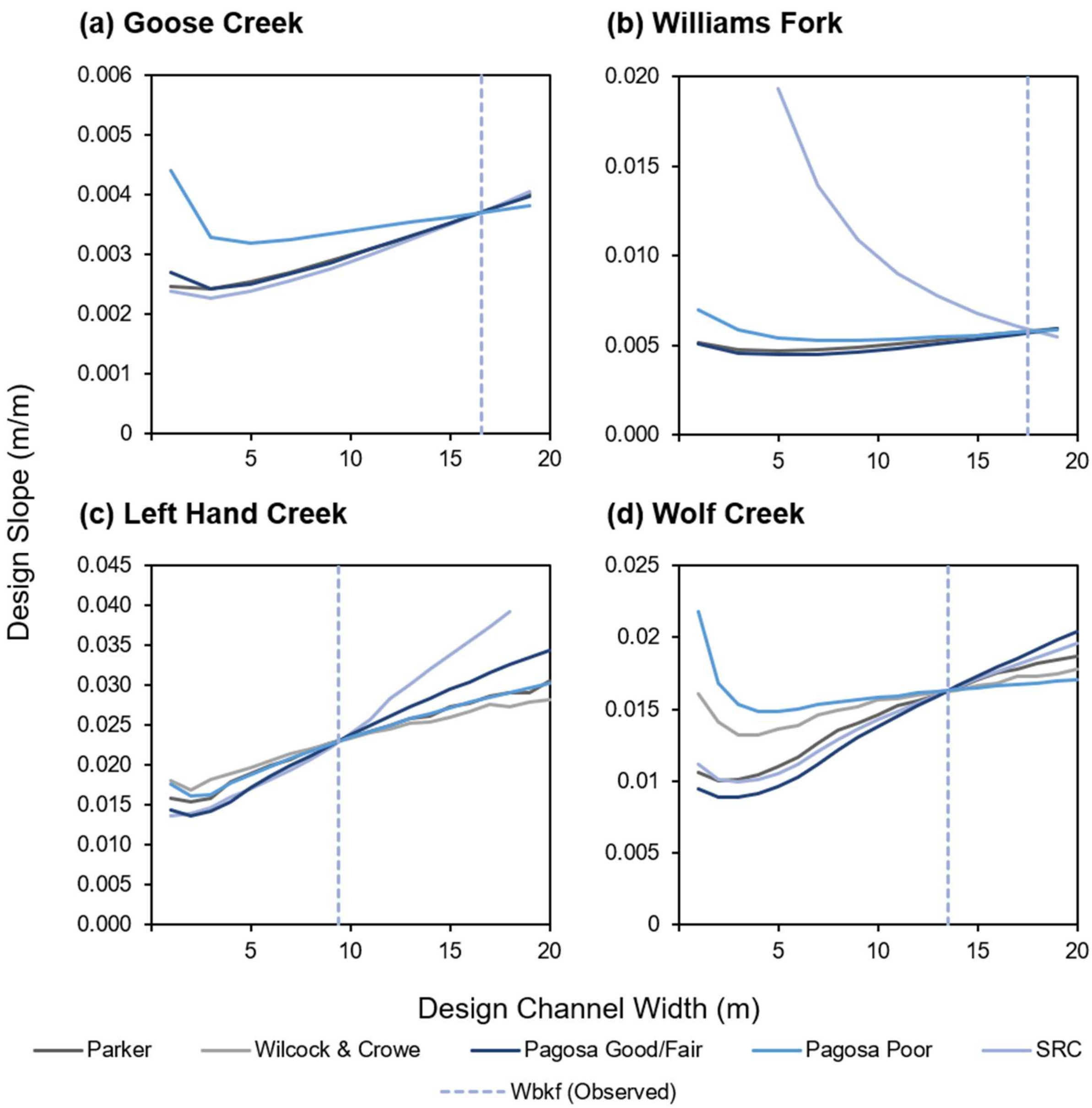


Figure 4.6. Width-Slope Curves Representing Families of Stable Channel Design Solutions. Values above or below each curve (CSR > 1, CSR < 1) predict degradation or aggradation, respectively. The supply reach bankfull width is indicated by the dashed vertical line. Wilcock & Crowe was only applicable for Left Hand Creek (c) and Wolf Creek (d) based on appreciable sand fractions.

measured sediment transport data, a distinct benefit shared with Parker and Wilcock & Crowe. Similarly, Good/Fair 0.5Q_{bkf} and 2 Q_{bkf} differed primarily when it came to absolute magnitudes, otherwise having very similar curves (Figure 4.7b). Of the two variations, 0.5 Q_{bkf} was a bit closer to the original Pagosa Good/Fair curve (+/- 0.0005). At Wolf Creek, Pagosa Poor was an

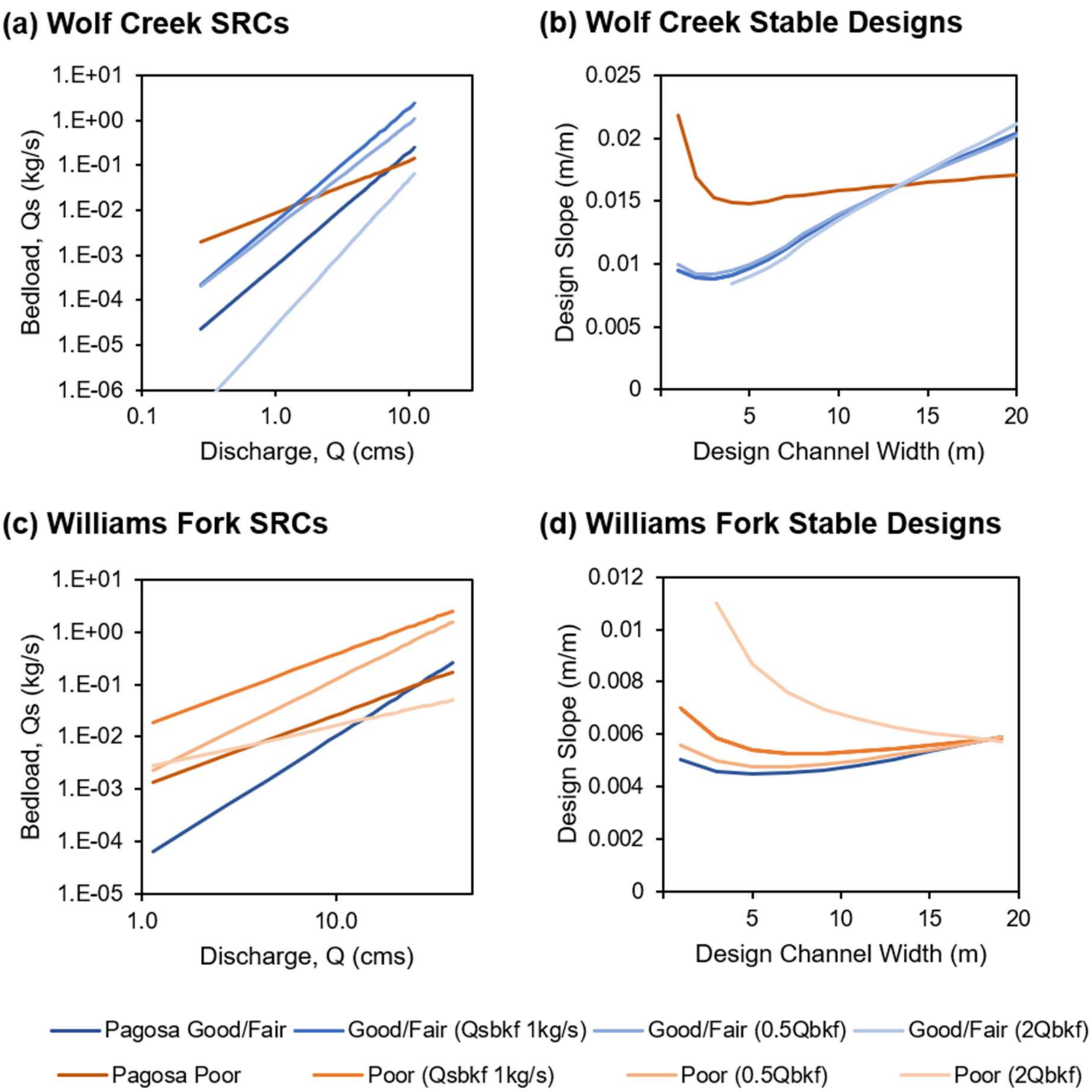


Figure 4.7. Pagosa Relationships with Q_{sbkf} and Q_{bkf} Scenarios. (a,b) Wolf Creek variations based on Pagosa Good/Fair. (c,d) Williams Fork Creek variations based on Pagosa Poor. The SRCs for (a) Pagosa Good/Fair is parallel to Good/Fair (Q_{sbkf} 1kg/s), but separated by an order of magnitude, while their stable channel solutions (b) exactly coincide, so only 4 swooshes are visible. Williams Fork (c,d) shows the same relationships between Pagosa Poor and Poor (Q_{sbkf} 1kg/s).

outlier compared to all of the Good/Fair variations (+/- 0.01), perhaps not surprising given it was a channel with good/fair stability. Williams Fork (Figures 4.7c,d), with poor channel stability, showed identical patterns between Pagosa Poor and Poor (Q_{sbkf} 1 kg/s), but otherwise different trends for the Poor Q_{bkf} variations. Interestingly, the Poor 0.5Q_{bkf} curve was closest to the Pagosa

Good/Fair curve (+/- 0.002), while the Poor 2 Q_{bkf} was the far outlier (+/- 0.007). Other channels with poor stability had results similar to Williams Fork. To summarize, we found the most critical distinction between Good/Fair or Poor, and a secondary distinction between the Poor Q_{bkf} variations. If using the Pagosa formula with the CSR tool and Q_{bkf} is uncertain, it may be better to use the lowest value of an estimated range. Importantly, Q_{sbkf} did not affect the stable channel design curves.

CSRs

When we tested floodplain confinement, we discovered that altering floodplain width had little to no effect, regardless of transport relationship, resulting in an unchanging CSR value of 1. The only possible exception we found was by severely reducing the Williams Fork W_{fp}/W_{bkf} for the Pagosa Good/Fair analyses, which slightly increased the CSR value to 1.06. Furthermore, when comparing CSR values based on total versus bankfull effectiveness, preliminary results were counterintuitive in some cases close to the floodplain threshold as discussed below; therefore, we only present the results of *W/D* and slope modifications.

Figure 4.8 shows CSR values for design scenarios with variable *W/D* ratios and slopes. In general, Parker and Pagosa Good/Fair showed the greatest sensitivity with highest CSR values greater than 1 (degradation) as well as lowest values below 1 (aggradation), with the exception of Left Hand Creek's SRC (Figure 4.8c). In Williams Fork, the SRC had unexpected values, moving in the opposite direction from 1 compared to the other sediment transport relationships, and similar in nature to the backwards swoosh for CSR =1 (Figure 4.8b). Pagosa Poor tended to be least sensitive to change in both *W/D* ratio and slope, with values remaining closer to 1 compared to the other relationships.

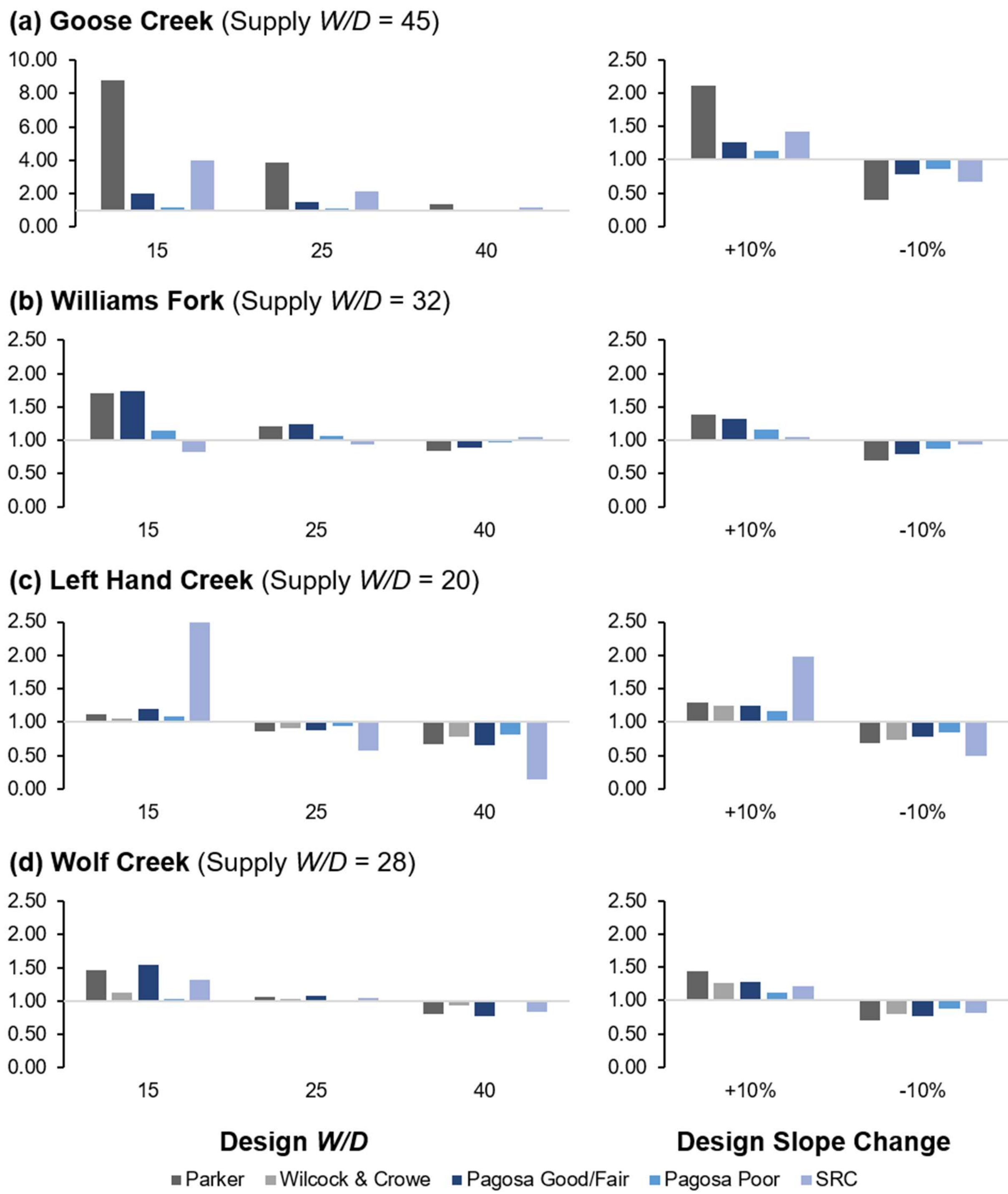


Figure 4.8. CSR Values for *W/D* and Slope Alterations. Shown relative to CSR = 1, with higher ratios predicting degradation, and lower ratios predicting aggradation. Note different scales for (a) *W/D* scenarios.

Discussion

The modified CSR spreadsheet tool offers a straightforward yet flexible approach to stable channel design, and the incorporation of SRCs expands its utility for both NCD and ACD. The analysis results and inherent commonalities underscore shared compatibilities within integrated tools like this one. Most importantly, both methods rely upon capacity-based design and FDCs combined with relative comparison of two reaches. The predictive relationships are the primary differences, with NCD leaning more heavily on empirical relationships, and ACD incorporating more physical variables. Furthermore, ACD uses identical models for both reaches, while NCD uses a coupled model with a discharge-based prediction (SRC) for the first reach and a stream power-based prediction for the second reach.

For most sites, certain ACD and NCD bed-load relationships produced similar stable channel designs. Results of this study showed that stable design solutions based on measured sediment loads were most likely to align with both the Parker (1990) bed-load equation and/or Pagosa Good/Fair DSRC, confirming an association between those two sediment transport relationships (Hinton et al. 2012). Moreover, relieving practitioners of the need for measured sediment loads greatly increases the practicality of capacity-based design implementation, and the power of relative comparison using the CSR method minimizes a noted Pagosa formula disadvantage (Hinton et al. 2018).

Nonetheless, the results of this study found design curve differences and variable CSR sensitivities across sediment transport relationships, which could potentially be explained by some of the predominant underlying mechanisms: grain size distribution, supply reach stability, and hydrologic regime. Wolf Creek was the only gravel bed stream with appreciable sand, and the stable channel design curves (Figure 4.6d) showed a greater degree of variability compared to the other sites. Additionally, while any given width-slope swoosh represents an entire family of stable channel designs, a simultaneous comparison across multiple transport relationships (Figure 4.6) highlights a hinge point where the design and supply reach widths and slopes are equal, perhaps unsurprising in hindsight. To the extent that differences between curves reflect design uncertainty, the preferred stable channel design may necessarily have a channel cross section and slope very similar to the supply reach, unless the supply reach is inherently problematic. This conclusion is consistent with the idea that a stable upstream supply reach may serve as a preferred reference reach (Bledsoe et al. 2017).

Influences of Driving Mechanisms

The results of this study revisit the significance of grain size distribution to sediment transport as well as the underlying assumptions of different models, whether the relation uses the full distribution or no grain size or something in between (e.g., Brownlie 1981, Wilcock and Kenworthy 2002). Whereas this study's conventional equations (Parker 1990, Wilcock and Curran 2003) explicitly incorporate bed material gradation, the NCD empirical regressions predict only total volumetric sediment transport capacity. At Left Hand Creek, the only cobble bed site, many of the design curves remained relatively close, even far below the supply reach W_{bkf} (Figure 4.6c). Furthermore, the various sediment transport relationships showed much less CSR sensitivity compared to the Left Hand SRC (Figure 4.8c), such that CSR values were relatively close to each other for the conventional equations as well as both Pagosa relationships.

At the same time, for both sites where the Wilcock and Crowe relationship were tested (Left Hand, Wolf), we found consistent differences compared to Parker in terms of both the design curve and CSR values, with Parker being more sensitive compared to Wilcock and Crowe. Both of these standard transport relationships incorporate incipient motion using grain size and shear stress, while Wilcock & Crowe additionally accounts for the sand fraction. The higher CSR sensitivity of the Parker relationship is expressed through steeper design swooshes having a narrower range of permissible solutions for CSR values ranging from 0.9 to 1.1. For design widths below the supply reach W_{bkf}, the design slope that satisfied Wilcock and Crowe was actually steeper than Parker, and vice versa for design widths above the supply reach W_{bkf}. It may seem counterintuitive that the Wilcock and Crowe relationship would have CSR values less sensitive to design alterations, given its premise that higher sand fractions in the bed increase gravel transport rates. The reason for this is not immediately clear, but may reflect differences in power scaling of transport relations (Wilcock and Curran 2003). Additionally, the Wilcock and Crowe stable design solutions for Wolf Creek align poorly with most of the other bed-load relationships, being closest to Pagosa Poor, and yet Wolf Creek was used to develop the Pagosa Good/Fair DSRCs. These results suggest that Wilcock and Crowe may poorly model Wolf Creek, despite its appreciable sand, as the predicted transport is super sensitive to high sand content, and it may be difficult to estimate the sand fraction very accurately. Otherwise, perhaps accounting for suspended sand (e.g., Pagosa suspended sediment formula) might be necessary for this site.

There are still unanswered questions about appropriate use of a single formula to characterize all channels with poor stability. Our selection criteria for study sites resulted in one channel with good/fair channel stability (Wolf Creek) and the remaining three with poor stability, although we tested both Pagosa bed-load formulas for all sites. This is a source of concern to the extent that the CSR tool is ideally suited to sites with stable sediment supply reaches. Furthermore, the Pagosa Poor curve consistently resulted in CSR values that were least sensitive to design alterations compared to either Pagosa Good/Fair or the conventional equations. Even more than Wilcock and Crowe, Pagosa Poor had relatively flat design swooshes with the widest ranges of permissible solutions for CSR values ranging from 0.9 to 1.1. Nonetheless, designers may have little control over the upstream sediment supply reach, such that a stable channel design may not be feasible.

These results must be interpreted with caution for at least one other reason. For this study, our selection criteria included USGS gaged sites, but the CSR tool is also applicable to ungaged sites by using a regional FDC. Whereas the tool creates a frequency distribution using arithmetic bins (equal flow ranges) when provided with a flow record, a method favored by research (Biedenharn et al. 2000), the pre-existing FDC option currently uses a logarithmic approach. Regardless, any sensitivity to binning and the corresponding probabilities may be reflected in the total effectiveness for both the supply and design reaches, potentially affecting the CSR values and design swooshes. Questions also remain about the signal versus noise in the tails, and the relative influences of flows above bankfull. In addition to addressing FDC sensitivity for ungaged streams, recommended future research would include sites with a wider range of hydrologic characteristics. All four study sites were either snowmelt dominated or snowmelt combined with monsoon influences, and none of the sites were flashy. However, relatively high degrees of stream sensitivity to disturbance have been linked to flashy hydrology and fine grained bed material (Bledsoe et al. 2016, 2017).

We were most surprised by study results that failed to show significant floodplain influences, which is quite inconsistent with the traditional significance placed on floodplain connectivity by most stream restoration specialists (e.g., Rosgen 1994, Palmer and Febria 2012, Fryirs et al. 2016). We expected to see greater floodplain connectivity reducing CSR values and vice versa, but it became apparent that altering floodplain width had little to no effect, regardless of transport relationship. There are several possible explanations for this finding. First, the relatively low frequency of overbank flows mathematically reduces their contributions to total effectiveness, because the sediment discharge rates are multiplied by very low probabilities. Second, we know the CSR tool's shear stress partitioning differential between in-channel and overbank flows creates discontinuities in hydraulic relations (velocity, depth, hydraulic radius, etc.) that causes a sharp increase in sediment rates immediately above the transition point. Therefore, we have concluded that further investigation of channel entrenchment and floodplain effects may require an alternative approach to shear stress partitioning. We expect the tool to be sensitive to floodplain connectivity based on physical understanding, so this behavior provides a useful test of fidelity to the actual physical processes and can inform selection of appropriate portioning schemes. In particular, floodplain connection would be expected to contribute significantly to sediment capacity in labile, sand bed streams. In general, floodplain connectivity is an important design consideration as these systems provide a wealth of hydrologic and ecological benefits independent of sediment transport processes (e.g., Helton et al. 2011, Cluer and Thorne 2014, Lammers and Bledsoe 2017).

Uncertainty and Management Implications

The CSR spreadsheet tool helps stream restoration practitioners predict relative channel stability on projects without measured flow or sediment data. For both ACD and NCD, a flow duration curve is required input, but these may be readily estimated for ungaged locations (Bledsoe et al. 2016, 2017). With regard to sediment, those applying conventional ACD relationships need only a grain size distribution. Alternately, users with NCD experience may create a reasonable SRC using a Pagosa relationship with a regional estimate for Q_{bkf} (e.g., lower end of a plausible range) and an arbitrary value for Q_{sbkf} (e.g., 1 kg/s). In either case, the key is to focus on the CSR value and width-slope design curve rather than estimated sediment transport rates. Although we used simplifying assumptions for cross section shape, floodplain characteristics, and bank roughness to focus on comparative analyses of channel geometries for our case studies, we recommend that designers tailor input for site specific conditions to the extent practical.

Differences in sediment transport relationships, as illustrated by our results, is one factor contributing to overall design uncertainty. Hydrologic data are a second source of potential uncertainty, whether due to record lengths, measurement accuracy, evolving watershed conditions or climate change. Supply reach stability may also be temporary, partly due to changing streamflows, and partly due to physical channel alterations or other factors. Nonetheless, the CSR tool enables users to overcome lingering myths and misperceptions about sediment transport. In particular, the ratio (CSR value) of sediment transport capacities for a pair of upstream and downstream reaches provides useful information about relative channel stability (e.g., Figure 4.6) despite absolute estimates for a single reach that could be inaccurate by an order of magnitude (e.g., Figure 4.4). In other words, relative comparisons of time-integrated sediment transport capacity can be quite accurate despite individual inaccuracies in estimates of absolute sediment transport magnitude. Using the CSR tool to generate design solutions with a CSR value equal to one was the original intent of the software designers, but a singular swoosh could be an unintentional source of overconfidence. Prior research suggested that a CSR value of 1 +/- 10 percent may be good enough to support "dynamic stability", assuming that the channel may naturally adjust (Soar and Thorne 2001). However, the study results showed that Wilcock and Crowe and Pagosa Poor were relatively less sensitive to design alterations within this specified range. This warrants caution because those particular relationships are intended for channels with appreciable sand or poor stability, respectively, both of which suggest greater care may be needed to create a stable channel design. Furthermore, the paired curves we generated with CSR values of 0.9 and 1.1 were not statistical confidence intervals, per se. There is abundant room for further progress in determining appropriate ranges for CSR values, although designing a stream with a CSR of 1 +/- 10 percent may still be a reasonable goal when using the Parker or Pagosa Good/Fair relationships. Regardless, changing upstream conditions and site constraints may result in anticipated aggradation or degradation. This doesn't mean the CSR tool isn't applicable to such situations. Rather, it provides a way to quantitatively assess the potential degree of relative instability, including sensitivity analysis. Where a stable channel design may not be possible, predicting aggradation or degradation in advance gives stakeholders realistic expectations and points the way toward adaptive management.

While this paper endeavors to compare stable channel design approaches based on sediment transport capacity, a more holistic validation of any stream restoration method will involve significant time and resources to build and monitor projects. A robust monitoring program seeks to address the question of whether or not the design tool performed as intended, and then closes the feedback loop with the design team so future efforts can benefit from the collective knowledge gained. Our goal here is neither to validate nor choose one approach over the other, but rather understand similarities and differences.

Conclusions

Although we assert that alluvial channel design based on relative sediment transport capacity is the most evolved approach to stable channel design, we do not claim that our modified CSR tool should be the default approach to stream restoration. Even in the context of stable channel design, nuanced views exist about enforcing stability through construction techniques (e.g., toe wood, instream structures) or simply letting the channel naturally self-adjust (Lave 2009, 2012). Natural scientists have been historically careful to draw a distinction between engineering and ecological equilibrium, arguing against "command and control" approaches (e.g., Holling and Meffe 1996). Evolving thought structures and practices among stream specialists have recently trended in the direction of intentionally supporting relatively dynamic systems, and a growing number of alternatives to stable channel design are gaining traction (e.g., Beardsley and Boyd 2019). Self-forming channels (Mecklenburg 2008), Stage 0 restoration (e.g., Powers et al., 2019), and low-tech process-based restoration (Wheaton et al. 2019) are examples of dynamic approaches that leave design decisions to the system rather than dictating natural processes through constructed channels, and they may generate greater benefits in terms of water quality and ecosystem resilience (e.g., Cluer and Thorne 2014). However, the most naturally dynamic approaches to stream restoration may not be suitable where constraints generally dictate the use of stable channel design, particularly sites with corridor encroachments common to transportation projects and highly urban settings (Kondolf 2011). The degree to which social-ecological systems are characterized by multiple objectives and embedded constraints suggests that stable channel design will continue to be the preferred alternative in many settings. Despite the challenges of uncertainty and variable upstream conditions, relative comparison of supply and design reach sediment transport capacities can still be a useful tool for channel design and assessment. Our analysis showed the compatibility of NCD and ACD capacity-based approaches and that it is logical to integrate them in a common toolbox for stable channel design.

CHAPTER 5

CONCLUSIONS

The complexities of urban riverscapes are revealed through nested spatial scales and various forms of physical and social connectivity, including multidisciplinary stakeholder groups with potentially competing objectives. Therefore, planning and design in urban riverscapes require situational awareness and strategies for integrating the missing or hidden facets of the coupled human and natural system. Ongoing struggles with the urban stream syndrome and wicked social-ecological problems indicate no shortage of research questions and opportunities for improvement in our strategies for multifunctional urban riverscapes (Wenger et al. 2009, Booth et al. 2015, Fork et al. 2022, Díaz-Pascacio et al. 2022). This dissertation responded to a broad question: How can we advance the science and practice of urban stream restoration? More specifically, I was interested in how communication could be improved between everyone at the table, management approaches could provide both societal and ecosystem benefits, and ideological differences could be reconciled. Integrative approaches like the framework for urban stream engineering (FUSE), spatial multi-criteria decision analysis (MCDA), and capacity/supply ratio (CSR) tool can support holistic planning and design in urban riverscapes. As such, this work presented an original conceptual model, communication tool, and implementation framework for bridging a knowledge gap that has been limiting management strategies for multifunctional natural infrastructure (NI) and nature-based solutions, plus an expanded and enhanced technical tool that supports stable channel design, ecosystem support, and built infrastructure protection. Collectively, these contributions advance multiple stages and spatial scales of urban riverscape management. Following a logical project sequence, the improvements to preliminary conceptualization, planning-level decision-making, and engineering design phases complement one another and integrate urban stream restoration as a social-ecological-technical learning process over time.

I created a new conceptual model to support holistic ecological engineering in urban riverscapes. FUSE is a tool that was designed to improve communication among various groups and disciplines, achieving one research objective. Oriented towards benefits and services, FUSE synthesized interdisciplinary perspectives of riverscape forms and process in urban environments with an NI framework that included water quality and social dimensions. FUSE both incorporated and transcended stream classification systems (e.g., Castro and Thorne 2019) and existing frameworks (e.g., Wenger et al. 2009, Harman et al. 2012) to better support urban stream contexts and constraints. By providing a relatively simple and flexible way to conceptualize a coupled human and natural stream system, FUSE visually captures the urban stream syndrome concept and other interconnected social-ecological problems, whilst representing the idea of balanced urban riverscape management. Fully implementing FUSE for its intended purpose as a shared thinking space will require real-world applications in a variety of settings: management policies and practices, formal and informal education, community focus groups, and transdisciplinary research.

The urban riverscape MCDA I developed collaboratively with Charlotte-Mecklenburg Storm Water Services (CMSWS) was a planning tool that addressed the social-ecological context of NI across multiple spatial scales. Like FUSE, the spatial MCDA integrated water quality and social dimensions with flooding and aquatic ecosystems (i.e., risk assessments), emphasizing system vulnerabilities using both landscape and riverscape criteria. The urban riverscape MCDA identified the highest priority watersheds and sub-basins for a synergistic combination of potential NI benefits and co-benefits (flood regulation, water quality, ecosystem support, amenity access, environmental justice). As part the environmental justice emphasis, I incorporated NI equity using environmental risk and benefit ratios, and I found that a riverscape emphasis combined with a sub-basin spatial scale was most helpful for identifying system hotspots, comparable to earlier flood risk investigation at the census tract scale (Debbage 2019). Aside from the positive relationship between water quality and environmental justice, there was little statistical evidence for overall synergies or tradeoffs, in contrast with a green infrastructure study in Detroit (Meerow and Newell 2017). While this spatial MCDA approach focused on system vulnerabilities, additional spatial analyses can be used in the future to search for specific NI opportunities (floodplain reconnection, natural channel design, process-based riverscape restoration, etc.). However, community inclusion should be the next step, rather than jumping ahead to potential NI solutions. The spreadsheet I created allows for modification of criteria and priority weights, enabling potential application by CMSWS as a shared decision-making tool, as well as transferability to other social-ecological systems.

To manage urban riverine corridors for built infrastructure protection, aquatic ecosystem stability, and physical equilibrium, I synthesized analytical approaches to sediment transport capacity by drawing together two compatible channel design methodologies. I integrated the natural channel design (NCD) approach to channel stability prediction (NRCS 2007, Rosgen 2009, 2013) into an analytical channel design (ACD) tool (Bledsoe et al. 2016, 2017, Stroth et al. 2017), demonstrating how this was possible due to the fundamental similarities between the approaches. The original CSR spreadsheet tool predicted relative channel stability by comparing sediment transport capacity of two reaches (upstream supply and downstream design) across a flow duration curve, and I expanded the tool to accommodate relative comparisons based on sediment rating curves and unit stream power. I subsequently evaluated ACD and NCD approaches using the enhanced CSR tool, finding that the sediment transport relationships produced comparable results and stable channel design solutions, similar to a previous study (Hinton et al. 2018). However, caution is still advised for any channel design approach in a sensitive setting with changing flow and sediment regimes (e.g., unstable supply reach, fine grained bed material). I used measured sediment for validation but found that NCD, like ACD, did not require measured sediment loads when using the updated CSR tool. Thus, despite long-standing differences, two popular channel design approaches made similar capacity-based stability predictions and were compatible in a common toolbox. A step towards helping resolve the sediment controversy is especially relevant for practitioners seeking to apply appropriate stable channel designs in urban riverscapes.

I ultimately conclude that we need to start from a holistic view of urban riverscapes as multifunctional NI, and the concepts and tools presented above can mainstream this perspective and facilitate implementation with context-sensitive tools for planning, design, and communication. Collectively, the urban stream framework, spatial MCDA, and modified CSR spreadsheet tool fill knowledge gaps about how to communicate among disciplines and stakeholders, apply multi-objective strategies for equitable NI, and support physical equilibrium with practical and flexible approaches to urban riverscape planning and design. FUSE and the spatial prioritization tool are both straightforward enough to be meaningful to a wide range of stakeholders for communication and collaboration, and the technical tool has utility for both researchers and practitioners. Urban riverscape management nonetheless remains a significant area for present opportunities and future research. With the 50th anniversary of the Clean Water

Act, urban streams are still severely degraded, just as windows of opportunity are opening through funded legislation for local governments to expand equitable and multifunctional NI solutions for generational transformation. However, depending on different stream conditions and system stressors, questions remain about which management strategies are cost-effective and likely to improve aquatic ecology (Wenger et al. 2009). Going forward, FUSE can certainly be used to thematically organize a broad spectrum of urban riverscape solutions that helps practitioners align the most appropriate planning, design, and implementation approaches for meeting various objectives. Future research is needed to continue refinement of these tools through application, and to further build out a toolbox that incorporates multiple spatial scales (e.g., watershed, floodplain, channel), driving variables (e.g., hydrology, geology, biology, water chemistry), and connectivity types (e.g., floodplain, stormwater, hyporheic zone). At the same time, an urban riverscape MCDA could be leveraged to spatially identify appropriate NI options and improvement strategies, such as locations where an intentionally dynamic system is possible or stable channel design is most critical.

REFERENCES

Aldridge, K. T., J. D. Brookes, and G. G. Ganf. 2009. Rehabilitation of stream ecosystem functions through the reintroduction of coarse particulate organic matter. Restoration Ecology 17:97–106.

Alves, P. B. R. 2022. Effective engagement with diverse communities, policy makers, and academics in watershed sustainability and resiliency. American Ecological Engineering Society. Baltimore, MD.

Alves, P. B. R., S. Djordjević, and A. A. Javadi. 2021. An integrated socio-environmental framework for mapping hazard-specific vulnerability and exposure in urban areas. Urban Water Journal 18:530–543.

Andrews, E. D. 1984. Bed-material entrainment and hydraulic geometry of gravel-bed rivers in Colorado. Geological Society of America Bulletin 95:371–378.

Andrews, E. D. 1994. Marginal bed load transport in a gravel bed stream, Sagehen Creek, California. Water Resources Research 30:2241–2250.

Anim, D. O., T. D. Fletcher, G. J. Vietz, G. B. Pasternack, and M. J. Burns. 2019. Restoring in-stream habitat in urban catchments: Modify flow or the channel? Ecohydrology 12:1–16.

Annable, W. K., C. C. Watson, and P. J. Thompson. 2012. Quasi-equilibrium conditions of urban gravel-bed stream channels in southern Ontario, Canada. River Research and Applications 28:302–325.

ASCE. 2022. ASCE’s Roadmap to Sustainable Development: Four Priorities for Change. (Available from: https://www.asce.org/communities/institutes-and-technical-groups/sustainability/sustainability-roadmap)

Asselman, N. E. M. 2000. Fitting and interpretation of sediment rating curves. Journal of Hydrology 234:228–248.

Athanasakes, G., and D. L. Rosgen. 2010. Application of the FLOWSED and POWERSED Models in River Stability, Bridge Design and River Restoration. 2nd Joint Federal Interagency Conference. Las Vegas, NV.

Bagnold, R. A. 1960. Sediment discharge and stream power: A preliminary announcement, Geological Survey Circular 421. Washington, D.C.

Bagnold, R. A. 1980. An Empirical Correlation of Bedload Transport Rates in Flumes and Natural Rivers. Pages 453–473 Proceedings of the Royal Society of London. Series A, Mathematical and Physical Sciences, Vol. 372, No. 1751.

Barnes, H. H. 1967. Roughness Characteristics of Natural Channels (No. 1849). US Government Printing Office.

Baron, J. S., N. L. Poff, P. L. Angermeier, C. N. Dahm, P. H. Gleick, N. G. Hairston, Jr., R. B. Jackson, C. A. Johnston, B. D. Richter, and A. D. Steinman. 2002. Meeting Ecological and Societal Needs for Freshwater. Ecological Applications 12:1247–1260.

Beardsley, M., and K. Boyd. 2019. Breaking up with Lane: Rethinking equilibrium and stability in stream restoration. River Restoration Northwest, Stevenson, WA.

Béné, C., L. Mehta, G. McGranahan, T. Cannon, J. Gupte, and T. Tanner. 2018. Resilience as a policy narrative: potentials and limits in the context of urban planning. Climate and Development 10:116–133.

Bernhardt, E. S., J. B. Heffernan, N. B. Grimm, E. H. Stanley, J. W. Harvey, M. Arroita, A. P. Appling, M. J. Cohen, W. H. McDowell, R. O. Hall, J. S. Read, B. J. Roberts, E. G. Stets, and C. B. Yackulic. 2018. The metabolic regimes of flowing waters. Limnology and Oceanography 63:S99–S118.

Bernhardt, E. S., E. B. Sudduth, M. A. Palmer, J. D. Allan, J. L. Meyer, G. Alexander, J. Follastad-Shah, B. A. Hassett, R. Jenkinson, R. Lave, J. Rumps, and L. Pagano. 2007. Restoring Rivers One Reach at a Time: Results from a Survey of U.S . River Restoration Practitioners. Restoration Ecology 15:482–493.

Biedenharn, D. S., R. R. Copeland, C. R. Thorne, P. J. Soar, R. D. Hey, and C. C. Watson. 2000. Effective Discharge Calculation: A Practical Guide. ERDC/CHL TR-00-15. Coastal and Hydraulics Laboratory, Vicksburg, MS.

Biedenharn, D. S., C. M. Elliott, and C. C. Watson. 1997. The WES Stream Investigation and Streambank Stabilization Handbook. U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

Biron, P. M., T. Buffin-Bélanger, M. Larocque, G. Choné, C.-A. Cloutier, M.-A. Ouellet, S. Demers, T. Olsen, C. Desjarlais, and J. Eyquem. 2014. Freedom Space for Rivers: A Sustainable Management Approach to Enhance River Resilience. Environmental Management 54:1056–1073.

Bledsoe, B. P. 2022. Bledsoe Lab. (Available from: http://bledsoe.engr.uga.edu/)

Bledsoe, B. P., D. W. Baker, P. Nelson, T. Rosburg, J. S. Sholtes, and T. Stroth. 2016. National Cooperative Highway Research Program (NCHRP) Project 24-40 Final Report: Design Hydrology for Stream Restoration and Channel Stability at Stream Crossings.

Bledsoe, B. P., D. W. Baker, P. Nelson, T. Rosburg, J. S. Sholtes, and T. Stroth. 2017. National Cooperative Highway Research Program (NCHRP) Research Report 853: Guidance for Design Hydrology for Stream Restoration and Channel Stability.

Bonneau, J., T. D. Fletcher, J. F. Costelloe, and M. J. Burns. 2017. Stormwater infiltration and the ‘urban karst’ – A review. Journal of Hydrology 552:141–150.

Booth, D. B., and B. P. Bledsoe. 2009. Streams and Urbanization. Pages 93–123 *in* L. A. Baker (editor). The Water Environment of Cities. Springer, New York, NY.

Booth, D. B., and C. J. Fischenich. 2015. A channel evolution model to guide sustainable urban stream restoration. Area 47:408–421.

Booth, D. B., A. H. Roy, B. Smith, and K. A. Capps. 2015. Global perspectives on the urban stream syndrome. Freshwater Biology 35:412–420.

Bosman, C., and J. Dolley. 2019. Rethinking third places and community building. Pages 1–19 Rethinking Third Places: Informal Public Spaces and Community Building. Edward Elgar Publishing.

Bridges, T. S., E. M. Bourne, J. K. King, H. K. Kuzmitski, E. B. Moynihan, and B. C. Suedel. 2018. Engineering With Nature: An Atlas. Vicksburg, MS.

Bridges, T. S., E. Bourne, B. Suedel, E. Moynihan, and J. King. 2021. Engineering With Nature: An Atlas, Volume 2. Vicksburg, MS.

Bridges, T. S., P. W. Wagner, K. A. Burks-Copes, M. E. Bates, Z. A. Collier, C. J. Fischenich, J. Z. Gailani, L. D. Leuck, C. D. Piercy, J. D. Rosati, E. J. Russo, D. J. Shafer, B. C. Suedel, E. A. Vuxton, and T. V Wamsley. 2015. Use of Natural and Nature-Based Features (NNBF) for Coastal Resilience. US Army Corps of Engineers - Engineer Research and Development Center.

Browder, G., S. Ozment, I. Rehberger Bescos, and T. Gartner. 2019. Integrating Green and Gray: Creating Next Generation Infrastructure. Washington, DC.

Brownlie, W. R. 1981. Prediction of flow depth and sediment discharge in open channels, Report No. KH-R-43A. California Institute of Technology, Pasadena, CA.

Brunner, G. W. 2010. HEC-RAS River Analysis System, User’s Manual; Report No. CPD-68, Version 4.1; Davis, CA.

Bunte, K., and S. R. Abt. 2009. Transport relationships between bedload traps and a 3-inch Helley-Smith sampler in coarse gravel-bed streams and development of adjustment functions.

Bunte, K., S. R. Abt, K. W. Swingle, and J. P. Potyondy. 2010. Functions to Adjust Transport Rates from a Helley-Smith Sampler to Bedload Traps in Coarse Gravel-Bed Streams (Rating Curve Approach). 2nd Joint Federal Interagency Conference.

Castro, J. M., M. M. Pollock, C. E. Jordan, G. Lewallen, and K. Woodruff. 2015. The Beaver Restoration Guidebook: Working with Beaver to Restore Streams, Wetlands, and Floodplains. 1.02. United States Fish and Wildlife Service, Portland, Oregon.

Castro, J. M., and C. R. Thorne. 2019. The stream evolution triangle: Integrating geology, hydrology, and biology. River Research and Applications 35:315–326.

Charlotte-Mecklenburg Storm Water Services. 2020. RARR User Guide Update User Guide Update: Risk Assessment / Risk Reduction Tool. Charlotte, NC.

Chien, H., and O. Saito. 2021. Evaluating social–ecological fit in urban stream management: The role of governing institutions in sustainable urban ecosystem service provision. Ecosystem Services 49:101285.

Chow, V. Te. 1959. Open-Channel Hydraulics. McGraw-Hill, New York, NY.

Church, M. 2006. Bed Material Transport and the Morphology of Alluvial River Channels. Annual Review of Earth and Planetary Sciences 34:325–354.

Clifford, N. J. 2007. River restoration: paradigms, paradoxes and the urban dimension. Water Science & Technology: Water Supply 7:57–68.

Cluer, B. L., and C. R. Thorne. 2014. A Stream Evolution Model Integrating Habitat and Ecosystem Benefits. River Research and Applications 30:135–154.

Cohen-Shacham, E., A. Andrade, J. Dalton, N. Dudley, M. Jones, C. Kumar, S. Maginnis, S. Maynard, C. R. Nelson, F. G. Renaud, R. Welling, and G. Walters. 2019. Core principles for successfully implementing and upscaling Nature-based Solutions. Environmental Science and Policy 98:20–29.

Colding, J., and S. Barthel. 2019. Exploring the social-ecological systems discourse 20 years later. Ecology and Society 24:2.

Copeland, R. R. 1994. Application of Channel Stability Methods Case Studies. Technical report HL-94-11. US Army Corps of Engineers, Waterways Experiment Station, Vicksburg, MS.

Copeland, R. R., D. N. McComas, C. R. Thorne, P. J. Soar, and M. M. Jonas. 2001. Hydraulic design of stream restoration projects. Vicksburg, MS.

Craig, L. S., M. A. Palmer, D. C. Richardson, S. Filoso, E. S. Bernhardt, B. P. Bledsoe, M. W. Doyle, P. M. Groffman, B. A. Hassett, S. S. Kaushal, P. M. Mayer, S. M. Smith, and P. R. Wilcock. 2008. Stream restoration strategies for reducing river nitrogen loads. Frontiers in Ecology and the Environment 6:529–538.

Dadson, S. J., J. W. Hall, A. Murgatroyd, M. Acreman, P. Bates, K. Beven, L. Heathwaite, J. Holden, I. P. Holman, S. N. Lane, E. O’Connell, E. Penning-Rowsell, N. Reynard, D. Sear, C. R. Thorne, and R. Wilby. 2017. A restatement of the natural science evidence concerning catchment-based ‘natural’ flood management in the UK. Proceedings of the Royal Society A: Mathematical, Physical and Engineering Sciences 473.

Davis, L. J., R. Milligan, C. E. Stauber, N. O. Jelks, L. Casanova, and S. H. Ledford. 2022. Environmental injustice and Escherichia coli in urban streams: Potential for community-led response. Wiley Interdisciplinary Reviews: Water e1583:1–19.

Debbage, N. A. 2018. Urban Flooding Vulnerability: A Multifaceted Comparative Assessment of the Charlanta Megaregion. University of Georgia.

Debbage, N. A. 2019. Multiscalar spatial analysis of urban flood risk and environmental justice in the Charlanta megaregion, USA. Anthropocene 28:100226.

Díaz-Pascacio, E., M. M. Castillo, and N. O. Jelks. 2022. Including equity in urban stream restoration: From historical wrongs to new paradigms. Freshwater Science 41:539–547.

Díaz, S., S. Demissew, J. Carabias, C. Joly, M. Lonsdale, N. Ash, A. Larigauderie, J. R. Adhikari, S. Arico, A. Báldi, A. Bartuska, C. Mbizvo, A. Duraiappah, T. Koetz, E. Brondizio, G. Bartus, M. Roué, S. T. Asah, M. Fischer, H. Moller, D. Darnaedi, P. Smith, J. B. Mikissa, P. Lyver, R. Allahverdiyev, H. A. Mooney, D. Pacheco, S. Hashimoto, H. Tallis, A. M. M. Fouda, F. Homer, B. Martin-Lopez, J. Caillaux, G. Erpul, C. Nesshöver, G. M. Mace, M. Okumura, Z. Asfaw, R. J. Scholes, M. Mehrdadi, P. Leadley, M. Schultz, S. Thomas, D. Zlatanova, E. Amankwah, B. Reyers, R. Hill, J. P. Metzger, M. Verma, Y. Yeo-Chang, I. A. Baste, G. Dalle, V. E. Figueroa, E. S. Pérez, R. Sumaila, L. A. Brooks, E. Roth, T. Yahara, S. Lavorel, A. Bilgin, G. Lichtenstein, O. Saito, Y. Al-Hafedh, G. Pataki, P. Escobar-Eyzaguirre, B. Fu, R. Watson, W. Mandivenyi, R. Thaman, Z. A. Hamid, A. Driver, W. A. Mala, C. Akosim, K. Takeuchi, A. A. Oteng-Yeboah, U. Pascual, H. Nagendra, J. Rubis, P. Matczak, H. Gundimeda, N. Sharma, P. Mumby, P. Failler, and K. M. Chan. 2014. The IPBES Conceptual Framework — connecting nature and people. Current Opinion in Environmental Sustainability 14:1–16.

Dierks, S., K. Gustavson, J. Schneider, and S. Wilson. 2003. Stream restoration design using the Stable channel Analytical Model (SAM). Pages 155–164 Proceedings of the Symposium - Protection and Restoration of Urban and Rural Streams. American Society of Civil Engineers.

Doll, B. A., G. L. Grabow, K. R. Hall, J. Halley, W. A. Harman, G. D. Jennings, and D. E. Wise. 2003. Chapter 5: Priority Options for Restoring Incised Streams. Pages 50–61 Stream Restoration: A Natural Channel Design Handbook. NC State University, Raleigh, North Carolina.

Doll, B. A., G. D. Jennings, J. Spooner, D. L. Penrose, J. L. Usset, J. Blackwell, and M. Fernandez. 2016. Identifying Watershed, Landscape, and Engineering Design Factors that Influence the Biotic Condition of Restored Streams. Water 8:151.

Dolley, J., and C. Bosman (Eds.). 2019. Rethinking Third Places: Informal Public Spaces and Community Building. Edward Elgar Publishing.

Dust, D., and E. E. Wohl. 2012. Conceptual model for complex river responses using an expanded Lane's relation. Geomorphology 139–140:109–121.

Eakin, H. C., L. A. Bojórquez-Tapia, M. A. Janssen, M. Georgescu, D. Manuel-Navarrete, E. R. Vivoni, A. E. Escalante, A. Baeza-Castro, M. Mazari-Hiriart, and A. M. Lerner. 2017. Urban resilience efforts must consider social and political forces. Proceedings of the National Academy of Sciences 114:186–189.

Echols, J. 2022a. Influence of Watershed Policy on Socio-Environmental Systems and Impacts.

Echols, J. 2022b. Personal communication. Atanta, GA.

Eden, S., and S. Tunstall. 2006. Ecological versus Social Restoration? How Urban River Restoration Challenges but Also Fails to Challenge the Science – Policy Nexus in the United Kingdom. Environment and Planning C: Government and Policy 24:661–680.

Environmental Protection Agency. 2020. Proctor Creek Watershed Story Map: The Intersection of Green Infrastructure and Health. (Available from: https://www.epa.gov/sites/default/files/2020-09/documents/proctor_creek_story_map_fact_sheet_sep2020.pdf)

Evers, M., A. Almoradie, and M. M. de Brito. 2018. Enhancing Flood Resilience Through Collaborative Modelling and Multi-criteria Decision Analysis (MCDA). Pages 221–236 Urban Disaster Resilience and Security. Springer International Publishing.

Fedele, G., B. Locatelli, and H. Djoudi. 2017. Mechanisms mediating the contribution of ecosystem services to human well-being and resilience. Ecosystem Services 28:43–54.

Ferguson, R. I. 1986. River Loads Underestimated by Rating Curves. Water Resources Research 22:74–76.

Ferguson, R. I. 2005. Estimating critical stream power for bedload transport calculations in gravel-bed rivers. Geomorphology 70:33–41.

Forehand, M. 2010. Bloom's taxonomy. Emerging perspectives on learning, teaching, and technology 41:47–56.

Fork, M. L., K. G. Hopkins, J. Chappell, R. J. Hawley, S. S. Kaushal, B. Murphy, B. Ríos-Touma, and A. H. Roy. 2022. Urbanization and stream ecology: Moving the bar on multidisciplinary solutions to wicked urban stream problems. Freshwater Science 41:398–403.

Fryirs, K. A., J. M. Wheaton, and G. J. Brierley. 2016. An approach for measuring confinement and assessing the influence of valley setting on river forms and processes. Earth Surface Processes and Landforms 41:701–710.

García, M. H. (Ed.). 2008. Sedimentation engineering: Processes, measurements, modeling, and practice (Manual 110). American Society of Civil Engineers.

Gomez, B., and M. Church. 1989. An Assessment of Bed Load Sediment Transport Formulae for Gravel Bed Rivers. Water Resources Research 25:1161–1186.

Gourevitch, J. D., R. M. Diehl, B. C. Wemple, and T. H. Ricketts. 2021. Inequities in the distribution of flood risk under floodplain restoration and climate change scenarios. People and Nature 00:1–13.

Grimm, N. B., S. T. A. Pickett, R. L. Hale, and M. L. Cadenasso. 2017. Does the ecological concept of disturbance have utility in urban social–ecological–technological systems? Ecosystem Health and Sustainability 3:e01255.

Groffman, P. M., N. J. Boulware, W. C. Zipperer, R. V. Pouyat, L. E. Band, and M. F. Colosimo. 2002. Soil nitrogen cycle processes in urban riparian zones. Environmental Science and Technology 36:4547–4552.

Habersack, H., B. Schober, and C. Hauer. 2014. Floodplain evaluation matrix (FEM): An interdisciplinary method for evaluating river floodplains in the context of integrated flood risk management. Natural Hazards 75:5–32.

Hammond, M. J., A. S. Chen, S. Djordjević, D. Butler, and O. Mark. 2015. Urban flood impact assessment: A state-of-the-art review. Urban Water Journal 12:14–29.

Harman, W. A., and C. J. Jones. 2017. North Carolina Stream Quantification Tool Data Collection and Analysis Manual, NC SQT v3.0. Raleigh, NC.

Harman, W. A., R. Starr, M. Carter, K. Tweedy, M. Clemmons, K. Suggs, and C. Miller. 2012. A Function-Based Framework for Stream Assessment and Restoration Projects (EPA 843-K-12-006). US Environmental Protection Agency, Office of Wetlands, Oceans, and Watersheds, Washington, DC.

Hawley, R. J. 2018. Making Stream Restoration More Sustainable: A Geomorphically, Ecologically, and Socioeconomically Principled Approach to Bridge the Practice with the Science. BioScience 68:517–528.

Hawley, R. J., B. P. Bledsoe, E. D. Stein, and B. E. Haines. 2012. Channel Evolution Model of Semiarid Stream Response to Urban-Induced Hydromodification. Journal of the American Water Resources Association 48:722–744.

Hawley, R. J., and G. J. Vietz. 2016. Addressing the urban stream disturbance regime. Freshwater Science 35:278–292.

Helton, A. M., G. C. Poole, J. L. Meyer, W. M. Wollheim, B. J. Peterson, P. J. Mulholland, E. S. Bernhardt, J. A. Stanford, C. P. Arango, L. R. Ashkenas, L. W. Cooper, W. K. Dodds, S. V. Gregory, R. O. Hall, S. K. Hamilton, S. L. Johnson, W. H. McDowell, J. D. Potter, J. L. Tank, S. M. Thomas, H. M. Valett, J. R. Webster, and L. Zeglin. 2011. Thinking outside the channel: Modeling nitrogen cycling in networked river ecosystems. Frontiers in Ecology and the Environment 9:229–238.

Hess, A. J., and P. A. Johnson. 2001. A Systematic Analysis of the Constraints to Urban Stream Enhancements. Journal of the American Water Resources Association 37:213–221.

Hester, E. T., and M. N. Gooseff. 2010. Moving beyond the banks: Hyporheic restoration is fundamental to restoring ecological services and functions of streams. Environmental Science and Technology 44:1521–1525.

Hinton, D. D., R. H. Hotchkiss, and D. P. Ames. 2017. Comprehensive and Quality-Controlled Bedload Transport Database. Journal of Hydraulic Engineering 143:06016024.

Hinton, D. D., R. H. Hotchkiss, and T. P. Ballestero. 2012. Theoretical Basis for Rosgen's Pagosa Good/Fair Equation. International Conference on Fluvial Hydraulics - River Flow 2012. San Jose, Costa Rica.

Hinton, D. D., R. H. Hotchkiss, and M. Cope. 2018. Comparison of Calibrated Empirical and Semi-Empirical Methods for Bedload Transport Rate Prediction in Gravel Bed Streams. Journal of Hydraulic Engineering 144:04018038.

Hoang, L. N., R. A. Fenner, and M. Skenderian. 2016. A conceptual approach for evaluating the multiple benefits of urban flood management practices. Journal of Flood Risk Management 11:S943–S959.

Holling, C. S., and G. K. Meffe. 1996. Command and Control and the Pathology of Natural Resource Management. Conservation Biology 10:328–337.

Hong, C. Y., and H. Chang. 2020. Residents' perception of flood risk and urban stream restoration using multi-criteria decision analysis. River Research and Applications 36:2078–2088.

Hovis, M., J. C. Hollinger, F. Cubbage, T. Shear, B. A. Doll, J. J. Kurki-Fox, D. Line, A. Fox, M. Baldwin, T. Klondike, M. Lovejoy, B. Evans, J. West, and T. Potter. 2021. Natural Infrastructure Practices as Potential Flood Storage and Reduction for Farms and Rural Communities in the North Carolina Coastal Plain. Sustainability 13:9309.

Hunt, J. 2022. Personal communication. Charlotte, NC.

Jackson, C. R., J. Martin, and D. S. Leigh. 2005. A southeastern piedmont watershed sediment budget: Evidence for a multi-millennial agricultural legacy. Journal of Soil and Water Conservation 60:298–310.

Jackson, C. R., and C. M. Pringle. 2010. Ecological Benefits of Reduced Hydrologic Connectivity in Intensively Developed Landscapes. BioScience 60:37–46.

Jackson, R. J., T. D. Watson, A. Tsiu, B. Shulaker, S. Hopp, and M. Popovic. 2014. Urban River Parkways An Essential Tool for Public Health Urban River Parkways: An Essential Tool for Public Health. Los Angeles, CA.

Jefferson, A. J., A. S. Bhaskar, K. G. Hopkins, R. Fanelli, P. M. Avellaneda, and S. K. McMillan. 2017. Stormwater management network effectiveness and implications for urban watershed function: A critical review. Hydrological Processes 31:4056–4080.

Jelks, N. O., V. Jennings, and A. Rigolon. 2021. Green Gentrification and Health: A Scoping Review. International Journal of Environmental Research and Public Health 18:907.

Johnson, B., M. Beardsley, and J. Doran. 2016. FACStream 1.0: Functional Assessment of Colorado Streams. (Available from: http://www.ecometricscolorado.com/functional-assessment.html)

Johnson, M. F., C. R. Thorne, J. M. Castro, G. M. Kondolf, C. S. Mazzacano, S. B. Rood, and C. Westbrook. 2020. Biomic river restoration: A new focus for river management. River Research and Applications 36:3–12.

Karr, J. R., and C. O. Yoder. 2004. Biological Assessment and Criteria Improve Total Maximum Daily Load Decision Making. Journal of Environmental Engineering 130:594–604.

Kaushal, S. S., A. J. Gold, S. Bernal, T. A. Newsomer Johnson, K. Addy, A. J. Burgin, D. A. Burns, A. A. Coble, E. Hood, Y. H. Lu, P. M. Mayer, E. C. Minor, A. W. Schroth, P. G. F. Vidon, H. Wilson, M. A. Xenopoulos, T. R. Doody, J. G. Galella, P. J. Goodling, K. A. Haviland, S. Haq, B. Wessel, K. L. Wood, N. Jaworski, and K. T. Belt. 2018. Watershed 'chemical cocktails': forming novel elemental combinations in Anthropocene fresh waters. Biogeochemistry 141:281–305.

Kaushal, S. S., K. L. Wood, J. G. Galella, A. M. Gion, S. Haq, P. J. Goodling, K. A. Haviland, J. E. Reimer, C. J. Morel, B. Wessel, W. Nguyen, J. W. Hollingsworth, K. Mei, J. Leal, J. Widmer, R. Sharif, P. M. Mayer, T. A. Newcomer Johnson, K. D. Newcomb, E. Smith, and K. T. Belt. 2020. Making 'chemical cocktails' – Evolution of urban geochemical processes across the periodic table of elements. Applied Geochemistry 119:104632.

Kiker, G. A., T. S. Bridges, A. Varghese, P. T. P. Seager, and I. Linkov. 2005. Application of multicriteria decision analysis in environmental decision making. Integrated environmental assessment and management 1:95–108.

Kondolf, G. M. 2011. Setting Goals in River Restoration: When and Where Can the River "Heal Itself"? Pages 29–43 Stream Restoration in Dynamics Fluvial Systems: Scientific Approaches, Analyses and Tools. American Geophysical Union, Geophysical Monograph Series.

Kondolf, G. M., H. Piégay, L. Schmitt, and D. R. Montgomery. 2016. Geomorphic classification of rivers and streams. Pages 133–158 Tools in Fluvial Geomorphology. John Wiley & Sons, Ltd, Chichester, UK.

de Kraker, J. 2017. Social learning for resilience in social–ecological systems. Current Opinion in Environmental Sustainability 28:100–107.

Lagasse, P. F., L. W. Zevenbergen, W. J. Spitz, and L. A. Arneson. 2012. Stream Stability at Highway Structures, Hydraulic Engineering Circular No. 20. Fourth Edi.

Lammers, R. W., and B. P. Bledsoe. 2017. What role does stream restoration play in nutrient management? Critical Reviews in Environmental Science and Technology 47:335–371.

Lammers, R. W., and B. P. Bledsoe. 2018. Parsimonious sediment transport equations based on Bagnold's stream power approach. Earth Surface Processes and Landforms 43:242–258.

Lave, R. 2008. The Rosgen Wars and the Shifting Political Economy of Expertise. University of California, Berkeley.

Lave, R. 2009. The Controversy Over Natural Channel Design: Substantive Explanations and Potential Avenues for Resolution. Journal Of The American Water Resources Association 45:1519–1532.

Lave, R. 2012. Fields and Streams : Stream Restoration, Neoliberalism, and the Future of Environmental Science. University of Georgia Press.

Lave, R., and M. W. Doyle. 2021. Streams of Revenue: The Restoration Economy and the Ecosystems it Creates. MIT Press.

Lave, R., M. M. Robertson, and M. W. Doyle. 2008. Why you should pay attention to stream mitigation banking. Ecological Restoration 26:287–289.

Lim, K.-S., and D.-R. Lee. 2009. The spatial MCDA approach for evaluating flood damage reduction alternatives. KSCE Journal of Civil Engineering 13:359–369.

Limerinos, J. T. 1970. Determination of the Manning Coefficient From Measured Bed Roughness in Natural Channels. U.S. Geological Survey, Washington, D.C.

Linkov, I., P. Welle, D. Loney, A. Tkachuk, L. Canis, J. B. Kim, and T. S. Bridges. 2011. Use of Multicriteria Decision Analysis to Support Weight of Evidence Evaluation. Risk Analysis 31:1211–1225.

Mant, J., C. R. Thorne, J. Burch, and M. Naura. 2020. Restoration of urban streams to create blue-green infrastructure. Pages 77–97 *in* C. R. Thorne (editor). Blue-Green Cities. ICE Publishing, London, UK.

Marris, E. 2013. Rambunctious Garden: Saving Nature in a Post-Wild World. Bloomsbury Publishing.

Martin, Y., and M. Church. 2000. Re-examination of Bagnold's empirical bedload formulae. Earth Surface Processes and Landforms 25:1011–1024.

Meadows, D. H. 2008. Thinking in Systems : A Primer. (D. Wright, Ed.). Chelsea Green Publishing, White River Junction, VT.

Mecklenburg County Storm Water Services. 2021. Stream Restoration Ranking System (SRRS): SRRS Administrative and Desktop Score Procedure Manual. Charlotte, NC.

Mecklenburg, D. E. 2008. Tracking and monitoring self-forming streams and stream mitigation sites. Columbus, OH.

Meerow, S., and J. P. Newell. 2017. Spatial planning for multifunctional green infrastructure: Growing resilience in Detroit. Landscape and Urban Planning 159:62–75.

Minnesota Stream Quantification Tool Steering Committee. 2019. Minnesota Stream Quantification Tool and Debit Calculator (MNSQT) User Manual, Version 1.0. Washington, D.C.

Moran, S. 2010. Cities, creeks, and erasure: Stream restoration and environmental justice. Environmental Justice 3:61–69.

Mould, S. A., K. A. Fryirs, and R. Howitt. 2020. The importance of relational values in river management: Understanding enablers and barriers for effective participation. Ecology and Society 25:1–16.

Mould, S. A., K. Fryirs, and R. Howitt. 2018. Practicing Sociogeomorphology: Relationships and Dialog in River Research and Management. Society & Natural Resources 31:106–120.

Murphy, B. M., K. L. Russell, S. Mould, G. Vietz, and P. A. Nelson. 2022. Managing urban riverscapes: An assessment framework to integrate social-ecological values and physical processes. Journal of Environmental Management 322:115862.

Nadeau, T., D. M. Hicks, C. Trowbridge, N. Maness, R. Coulombe, and N. Czarnomski. 2018. Stream Function Assessment Method for Oregon. Salem, OR.

Nassauer, J. I. 1995. Messy Ecosystems, Orderly Frames. Landscape Journal 14:161–169.

Newcomer Johnson, T. A., S. S. Kaushal, P. M. Mayer, R. M. Smith, and G. M. Sivirichi. 2016. Nutrient retention in restored streams and rivers: A global review and synthesis. Water (Switzerland) 8:1–28.

Niezgoda, S. L., and P. A. Johnson. 2005. Improving the Urban Stream Restoration Effort: Identifying Critical Form and Processes Relationships. Environmental Management 35:579–592.

NRCS. 2007. Chapter 11 Rosgen Geomorphic Channel Design. Pages 11-1–76 Part 654 Stream Restoration Design National Engineering Handbook. U.S. Department of Agriculture, Washington, D.C.

O'Donnell, E. C., C. R. Thorne, S. Ahilan, S. Arthur, S. Birkinshaw, D. Butler, D. A. Dawson, G. Everett, R. A. Fenner, V. Glenis, L. Kapetas, C. Kilsby, V. Krivtsov, J. Lamond, S. Maskrey, G. O'Donnell, K. Potter, K. Vercruysse, T. Vilcan, and N. Wright. 2020. The Blue-Green Path to Urban Flood Resilience. Blue-Green Systems 2:28–45.

O'Driscoll, M., S. M. Clinton, A. J. Jefferson, A. Manda, and S. K. McMillan. 2010. Urbanization effects on watershed hydrology and in-stream processes in the southern United States. Water (Switzerland) 2:605–648.

Palmer, M. A., and E. S. Bernhardt. 2006. Hydroecology and river restoration: Ripe for research and synthesis. Water Resources Research 42:W03S07.

Palmer, M. A., E. S. Bernhardt, J. D. Allan, P. S. Lake, G. Alexander, S. Brooks, J. Carr, S. Clayton, C. N. Dahm, J. F. Shah, D. L. Galat, S. G. Loss, P. Goodwin, B. A. Hassett, R. Jenkinson, G. M. Kondolf, R. Lave, J. L. Meyer, T. K. O. Donnell, and E. B. Sudduth. 2005. Standards for Ecologically Successful River Restoration. Journal of Applied Ecology 42:208–217.

Palmer, M. A., and C. M. Febria. 2012. The heartbeat of ecosystems. Science 336:1393–1394.

Palmer, M. A., S. Filoso, and R. M. Fanelli. 2014a. From ecosystems to ecosystem services: Stream restoration as ecological engineering. Ecological Engineering 65:62–70.

Palmer, M. A., K. L. Hondula, and B. J. Koch. 2014b. Ecological Restoration of Streams and Rivers: Shifting Strategies and Shifting Goals. Annual Review of Ecology, Evolution, and Systematics 45:247–269.

Palmer, M. A., H. L. Menninger, and E. S. Bernhardt. 2010. River restoration, habitat heterogeneity and biodiversity: A failure of theory or practice? Freshwater Biology 55:205–222.

Parker, G. 1990. Surface-based bedload transport relation for gravel rivers. Journal of Hydraulic Research 28:417–436.

Parker, G., and P. C. Klingeman. 1982. On why gravel bed streams are paved. Water Resources Research 18:1409–1423.

Paul, M. 2022. Biological Potential of Urban Streams: Development and Application. Symposium on Urbanization and Stream Ecology (SUSE), Webinar. (Available from: https://www.urbanstreamecology.org/webinars.html)

Poff, N. L., J. D. Allan, M. B. Bain, J. R. Karr, K. L. Prestegaard, B. D. Richter, R. E. Sparks, and J. C. Stromberg. 1997. The natural flow regime: A paradigm for river conservation and restoration. BioScience 47:769–784.

Pollock, M. M., T. J. Beechie, J. M. Wheaton, C. E. Jordan, N. Bouwes, N. Weber, and C. Volk. 2014. Using beaver dams to restore incised stream ecosystems. BioScience 64:279–290.

Polvi, L. E., L. Lind, H. Persson, A. Miranda-Melo, F. Pilotto, X. Su, and C. Nilsson. 2020. Facets and scales in river restoration: Nestedness and interdependence of hydrological, geomorphic, ecological, and biogeochemical processes. Journal of Environmental Management 265:110288.

Powers, P. D., M. Helstab, and S. L. Niezgoda. 2019. A process-based approach to restoring depositional river valleys to Stage 0, an anastomosing channel network. River Research and Applications 35:3–13.

Pulido, L. 2000. Rethinking environmental racism: White privilege and urban development in southern California. Annals of the Association of American Geographers 90:12–40.

Reid, L. M., and T. Dunne. 1996. Rapid evaluation of sediment budgets. Catena, Reiskirchen, Germany.

Renaud, F. G., K. Sudmeier-Rieux, M. Estrella, and U. Nehren (Eds.). 2016. Ecosystem-Based Disaster Risk Reduction and Adaptation in Practice. Advances in Natural and Technological Hazards Research.

Richardson, E. V., D. B. Simons, and P. F. Lagasse. 2001. River Engineering for Highway Encroachments: Highways in the River Environment. Hydraulic Design Series No. 6.

Roni, P., and T. J. Beechie. 2012. Stream and Watershed Restoration: A Guide to Restoring Riverine Processes and Habitats. John Wiley and Sons.

Rosgen, D. L. 1994. A classification of natural rivers. Catena 22:169–199.

Rosgen, D. L. 2006. FLOWSED/POWERSED – Prediction models for suspended and bedload transport. Pages 761–769 Eighth Federal Interagency Sedimentation Conference (8th FISC).

Rosgen, D. L. 2009. Watershed Assessment of River Stability and Sediment Supply (WARSSS). Second Edi. Wildland Hydrology, Fort Collins, CO.

Rosgen, D. L. 2010. The application and validation of dimensionless sediment rating curves. 2nd Joint Federal Interagency Conference 1–11.

Rosgen, D. L. 2013. Natural Channel Design: Fundamental Concepts, Assumptions, and Methods. Pages 69–93 Stream Restoration in Dynamic Fluvial Systems: Scientific Approaches, Analyses, and Tools, Volume 194. American Geophysical Union (AGU).

Roy, A. H., A. D. Rosemond, M. J. Paul, D. S. Leigh, and J. B. Wallace. 2003. Stream macroinvertebrate response to catchment urbanisation (Georgia, U.S.A.). Freshwater Biology 48:329–346.

Rubin, Z., G. M. Kondolf, and B. Rios-Touma. 2017. Evaluating stream restoration projects: What do we learn from monitoring? Water (Switzerland) 9:1–16.

Russell, K. L., G. J. Vietz, and T. D. Fletcher. 2018. Urban catchment runoff increases bedload sediment yield and particle size in stream channels. Anthropocene 23:53–66.

Russell, K. L., G. J. Vietz, and T. D. Fletcher. 2019. Urban sediment supply to streams from hillslope sources. Science of the Total Environment 653:684–697.

Scarlett, R. D., M. Subramaniam, S. K. McMillan, A. T. Ingermann, and S. M. Clinton. 2021. Stormwater on the margins: Influence of race, gender, and education on willingness to participate in stormwater management. Journal of Environmental Management 290:112552.

Schell, C. J., K. Dyson, T. L. Fuentes, S. Des Roches, N. C. Harris, D. S. Miller, C. A. Woelfle-Erskine, and M. R. Lambert. 2020. The ecological and evolutionary consequences of systemic racism in urban environments. Science 4497.

Schumm, S. A. 1981. Evolution and Response of the Fluvial System, Sedimentologic Implications. The Society of Economic Paleontologists and Mineralogists.

Scoggins, M., D. B. Booth, T. Fletcher, M. L. Fork, A. Gonzalez, R. L. Hale, R. J. Hawley, A. H. Roy, E. E. Bilger, N. Bond, M. J. Burns, K. G. Hopkins, K. H. Macneale, E. Martí, S. K. Mckay, M. W. Neale, M. J. Paul, B. Rios-Touma, K. L. Russell, R. F. Smith, S. Wagner, and S. Wenger. 2022. Community-powered urban stream restoration: A vision for sustainable and resilient urban ecosystems. Freshwater Science 41:404–419.

Selsor, H., B. P. Bledsoe, and R. Lammers. 2022. Recognizing Flood Risk Inequities Across Flood Frequencies (In Review).

Shields, A. 1936. Anwendung der Aehnlichkeitsmechanik und der Turbulenzforschung auf die Geschiebebewegung. Preußischen Versuchsanstalt für Wasserbau und Schiffbau.

Shields, F. D., R. R. Copeland, P. C. Klingeman, M. W. Doyle, and A. Simon. 2003. Design for Stream Restoration. Journal of Hydraulic Engineering 129:575–584.

Simperler, L., T. Ertl, and A. Matzinger. 2020. Spatial Compatibility of Implementing Nature-Based Solutions for Reducing Urban Heat Islands and Stormwater Pollution. Sustainability 12:5967.

Smardon, R., S. Moran, and A. K. Baptiste. 2018. Revitalizing Urban Waterway Communities: Streams of Environmental Justice. Routledge.

Smith, R. F., R. J. Hawley, M. W. Neale, G. J. Vietz, E. Diaz-Pascacio, J. Herrmann, A. C. Lovell, C. Prescott, B. Rios-Touma, B. Smith, and R. M. Utz. 2016. Urban stream renovation: incorporating societal objectives to achieve ecological improvements. Freshwater Science 35:364–379.

Soar, P. J., and C. R. Thorne. 2001. Channel Restoration Design for Meandering Rivers. Coastal and Hydraulics Laboratory, Vicksburg, MS.

South River Watershed Alliance. 2020. Our River Overview. (Available from: https://www.southriverga.org/overview)

Sowińska-Świerkosz, B., and J. García. 2021. A new evaluation framework for nature-based solutions (NBS) projects based on the application of performance questions and indicators approach. Science of The Total Environment 787:147615.

Spangenberg, J. H., C. Görg, D. T. Truong, V. Tekken, J. V. Bustamante, and J. Settele. 2014. Provision of ecosystem services is determined by human agency, not ecosystem functions. Four case studies. International Journal of Biodiversity Science, Ecosystem Services and Management 10:40–53.

Stephens, T. A., and B. P. Bledsoe. 2020. Probabilistic mapping of flood hazards: Depicting uncertainty in streamflow, land use, and geomorphic adjustment. Anthropocene 29:100231.

Sterling, J. L., A. D. Rosemond, and S. J. Wenger. 2016. Watershed urbanization affects macroinvertebrate community structure and reduces biomass through similar pathways in Piedmont streams, Georgia, USA. Freshwater Science 35:676–688.

Stets, E. G., L. A. Sprague, G. P. Oelsner, H. M. Johnson, J. C. Murphy, K. R. Ryberg, A. V Vecchia, R. E. Zuellig, J. A. Falcone, and M. L. Riskin. 2020. Landscape Drivers of Dynamic Change in Water Quality of U.S. Rivers. Environmental Science and Technology 54:4336–4343.

Stroth, T., B. P. Bledsoe, and P. Nelson. 2017. Full Spectrum Analytical Channel Design with the Capacity/Supply Ratio (CSR). Water 9:271.

Suding, K. N., K. L. Gross, and G. R. Houseman. 2004. Alternative states and positive feedbacks in restoration ecology. Trends in Ecology & Evolution 19:46–53.

TDEC. 2017. Tennessee Stream Quantification Tool Spreadsheet User Manual TN SQT v1.0. Nashville, TN.

Thomas, W. A., R. R. Copeland, and D. N. McComas. 2002. SAM hydraulic design package for channels. Engineer Research & Development Center, US Army Corps of Engineers, Washington, DC.

Thoms, M. C., H. Piégay, and M. Parsons. 2018. What do you mean, "resilient geomorphic systems"? Geomorphology 305:8–19.

Thorne, C. R. (Ed.). 2020. Blue-Green Cities: Integrating urban flood risk management with green infrastructure. ICE Publishing, London, UK.

Troendle, C. A., D. L. Rosgen, S. E. Ryan, L. S. Porth, and J. M. Nankervis. 2001. Developing a "Reference" Sediment Transport Relationship. Pages II-73–80 Seventh Federal Interagency Sedimentation Conference.

U.S. Army Corps of Engineers. 2018a. Wyoming Stream Quantification Tool (WSQT) User Manual and Spreadsheet, Version 1.0. Cheyenne, WY.

U.S. Army Corps of Engineers. 2018b. Savannah District's 2018 Standard Operating Procedure for Compensatory Mitigation.

U.S. Army Corps of Engineers. 2022. Engineering With Nature. (Available from: https://ewn.erdc.dren.mil/)

Vanoni, V. A. (Ed.). 2006. Sedimentation Engineering (Manual 54). American Society of Civil Engineers.

ten Veldhuis, J. A. E. 2011. How the choice of flood damage metrics influences urban flood risk assessment. Journal of Flood Risk Management 4:281–287.

Vercruysse, K., D. A. Dawson, V. Glenis, R. Bertsch, N. Wright, and C. Kilsby. 2019. Developing spatial prioritization criteria for integrated urban flood management based on a source-to-impact flood analysis. Journal of Hydrology 578:124038.

Walsh, C. J., D. B. Booth, M. J. Burns, T. D. Fletcher, R. L. Hale, L. N. Hoang, G. Livingston, M. A. Rippy, A. H. Roy, M. Scoggins, and A. Wallace. 2016. Principles for urban stormwater management to protect stream ecosystems. Freshwater Science 35:398–411.

Walsh, C. J., T. D. Fletcher, D. G. Bos, and S. J. Imberger. 2015. Restoring a stream through retention of urban stormwater runoff: A catchment-scale experiment in a social-ecological system. Freshwater Science 34:1161–1168.

Walsh, C. J., A. H. Roy, J. W. Feminella, P. D. Cottingham, P. M. Groffman, and R. P. Morgan. 2005. The urban stream syndrome: current knowledge and the search for a cure. Journal of the North American Benthological Society 24:706–723.

Walter, R. C., and D. J. Merritts. 2008. Natural streams and the legacy of water-powered mills. Science 319:299–304.

Wenger, S. J., A. H. Roy, C. R. Jackson, E. S. Bernhardt, T. L. Carter, S. Filoso, C. A. Gibson, W. C. Hession, S. S. Kaushal, E. Martí, J. L. Meyer, M. A. Palmer, M. J. Paul, A. H. Purcell, A. Ramírez, A. D. Rosemond, K. A. Schofield, E. B. Sudduth, and C. J. Walsh. 2009. Twenty-six key research questions in urban stream ecology: an assessment of the state of the science. Journal of the North American Benthological Society 28:1080–1098.

Wheaton, J. M., S. N. Bennett, N. Bouwes, J. D. Maestas, and S. M. Shahverdian. 2019. Low-Tech Process-Based Restoration of Riverscapes: Design Manual. Version 1.0. Logan, UT.

Wheaton, J. M., and P. Skidmore. 2022. Riverscapes as natural infrastructure: Meeting challenges of climate adaptation and ecosystem restoration. Anthropocene 38:100334.

Whelchel, A. W., B. G. Reguero, B. van Wesenbeeck, and F. G. Renaud. 2018. Advancing disaster risk reduction through the integration of science, design, and policy into eco-engineering and several global resource management processes. International Journal of Disaster Risk Reduction 32:29–41.

Wilcock, P. R., and J. C. Curran. 2003. Surface-based Transport Model for Mixed-Size Sediment. Journal of Hydraulic Engineering 129:120–128.

Wilcock, P. R., and S. T. Kenworthy. 2002. A two-fraction model for the transport of sand/gravel mixtures. Water Resources Research 38:12-1–12.

Wilcock, P. R., S. T. Kenworthy, and J. C. Curran. 2001. Experimental study of the transport of mixed sand and gravel. Water Resources Research 37:3349–3358.

Wildland Hydrology. 2016. Sediment Transport and the FLOWSED and POWERSED Models. Wildland Hydrology, Fort Collins, CO.

Williams, G. P., and D. L. Rosgen. 1989. Measured Total Sediment Loads (Suspended Loads and Bedloads) for 93 United States Streams. U.S. Geological Survey, Denver, CO.

Wing, O. E. J., W. Lehman, P. D. Bates, C. C. Sampson, N. Quinn, A. M. Smith, J. C. Neal, J. R. Porter, and C. Kousky. 2022. Inequitable patterns of US flood risk in the Anthropocene. Nature Climate Change 12:156–162.

Wohl, E. E. 2001. Virtual rivers : lessons from the mountain rivers of the Colorado front range. Yale University Press.

Wohl, E. E., B. P. Bledsoe, R. B. Jacobson, N. L. Poff, S. L. Rathburn, D. M. Walters, and A. C. Wilcox. 2015. The natural sediment regime in rivers: Broadening the foundation for ecosystem management. BioScience 65:358–371.

Wohl, E. E., J. M. Castro, B. Cluer, D. Merritts, P. Powers, B. Staab, and C. R. Thorne. 2021. Rediscovering, Reevaluating, and Restoring Lost River-Wetland Corridors. Frontiers in Earth Science 9:511.

World Bank. 2017. Implementing nature-based flood protection: Principles and implementation guidance. Washington, DC.

World Water Assessment Programme. 2018. Nature-Based Solutions for Water. Paris, France.

APPENDIX A

URBAN STREAM PYRAMID

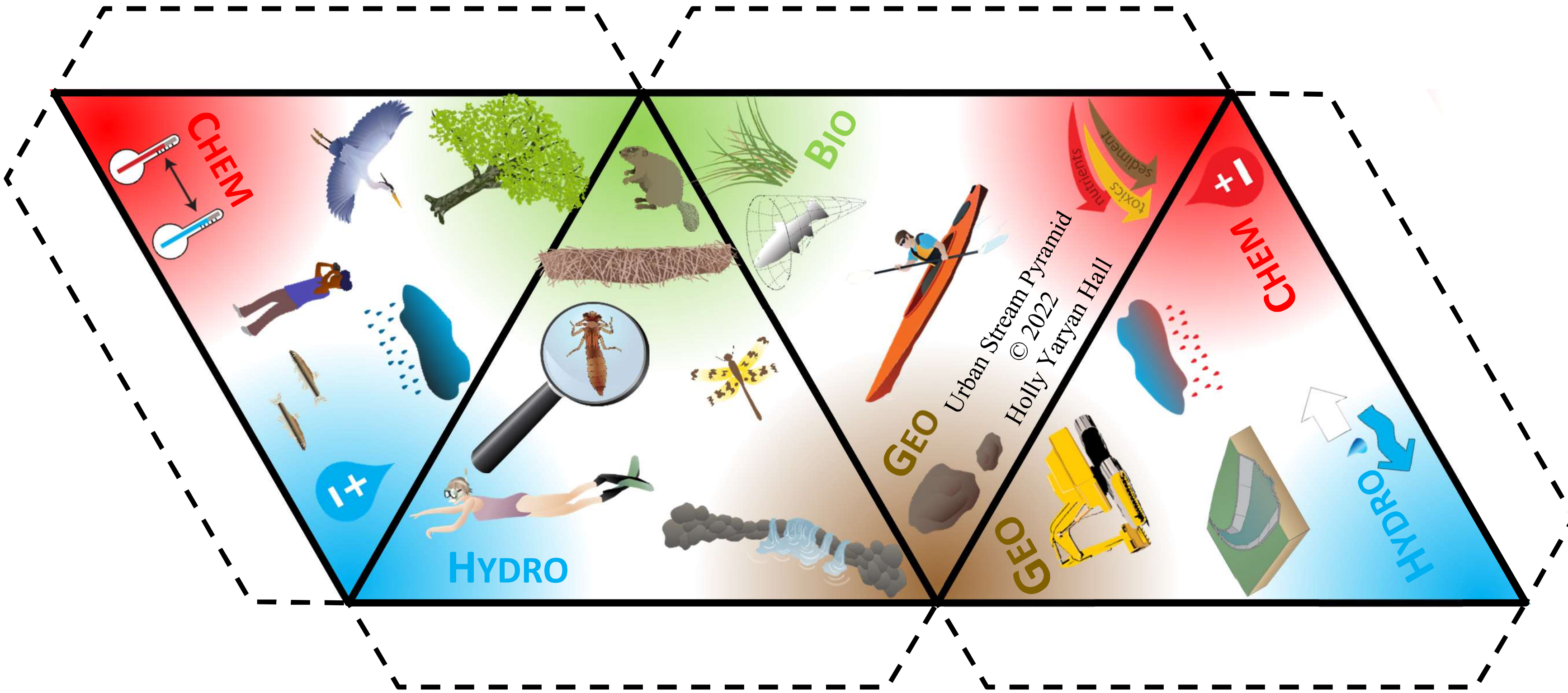


Figure A1. The urban stream pyramid (i.e., stream ecosystem pyramid) is half of the framework for urban stream engineering (FUSE). The pyramid vertices are drivers of natural system processes and functions, and they also correspond to environmental hazards, social influences, and human values (Table 2.1), with positive benefits inside the pyramid volume. This template can be printed to construct a three-dimensional object (cut along dashed lines, fold along solid lines, and then use tape or glue).