HYDROMODIFICATION ASSESSMENT AND MANAGEMENT IN CALIFORNIA

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Southern California Coastal Water Research Project

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Executive Summary

Most jurisdictions in California are now required to address the effects of *hydromodification* through either a municipal stormwater permit or the statewide construction general permit. Hydromodification is generally defined as changes in channel form associated with alterations in flow and sediment due to past or proposed future land-use alteration. Hydromodification management has emerged as a prominent issue because degradation of the physical structure of a channel is often indicative of and associated with broader impacts to many beneficial uses, including water supply, water quality, habitat, and public safety. Conversely, reducing hydromodification and its effects has the potential to protect and restore those same beneficial uses. Although hydromodification has the potential to affect all water body types, this document focuses on assessing and managing effects to streams because they are the most prevalent, widely studied, and arguably most responsive type of receiving water.

Hydromodification by definition results from alteration of watershed processes; therefore, correcting the root causes of hydromodification ought to be most effective if based on integrated watershed-scale solutions. To date, such a watershed approach has not been adopted in California; most hydromodification management plans simply consist of site-based runoff control with narrow, local objectives and little coordination between projects within a watershed. Furthermore, each municipality is required to develop its own approach to meeting hydromodification management requirements rather than drawing from standard or recommended approaches that facilitate regional or watershed-scale integration. Long-term reversal of hydromodification effects, however, will require movement away from reliance on such site-based approaches to more integrated watershed-based strategies.

This document has two goals, and hence two audiences. The first goal is to describe the elements of effective hydromodification assessment, management and monitoring. The audience for this goal is primarily the State and Regional Water Boards, since meeting this goal will require integration of watershed and site-scale activities that are likely beyond the responsibility or control of any individual municipality. Success will require fundamental changes in the regulatory and management approach to hydromodification that will likely advance only iteratively and potentially require one or more NPDES permit cycles to fully implement. The second goal of this document is to provide near-term technical assistance for implementing current and pending hydromodification management requirements. This goal can be achieved by municipalities within the construct of existing programs and therefore the primary audience for this aspect of the document is local jurisdictions. Achieving this goal will facilitate greater consistency and effectiveness between hydromodification management strategies, giving them a stronger basis in current scientific understanding.

Watershed analysis should be the foundation of all hydromodification management plans (Figure ES-1). This analysis should begin with a documentation of watershed characteristics and processes, and past, current, and expected future land uses. The analysis should lead to identification of existing opportunities and constraints that can be used to help prioritize areas of greater concern, areas of restoration potential, infrastructure constraints, and pathways for potential cumulative effects. The combination of watershed and site-based analyses should be used to establish clear objectives to guide management actions. These objectives should articulate desired and reasonable physical and biological conditions for various reaches or portions of the watershed and should prioritize areas for protection, restoration, or management. Strategies to achieve these objectives should be customized based on consideration of current and expected future channel and watershed conditions. A one-size-fits-all approach should be avoided. Even where site-based control measures, such as flow-control basins, are judged appropriate, their location and design standards should be determined in the context of the watershed analysis.

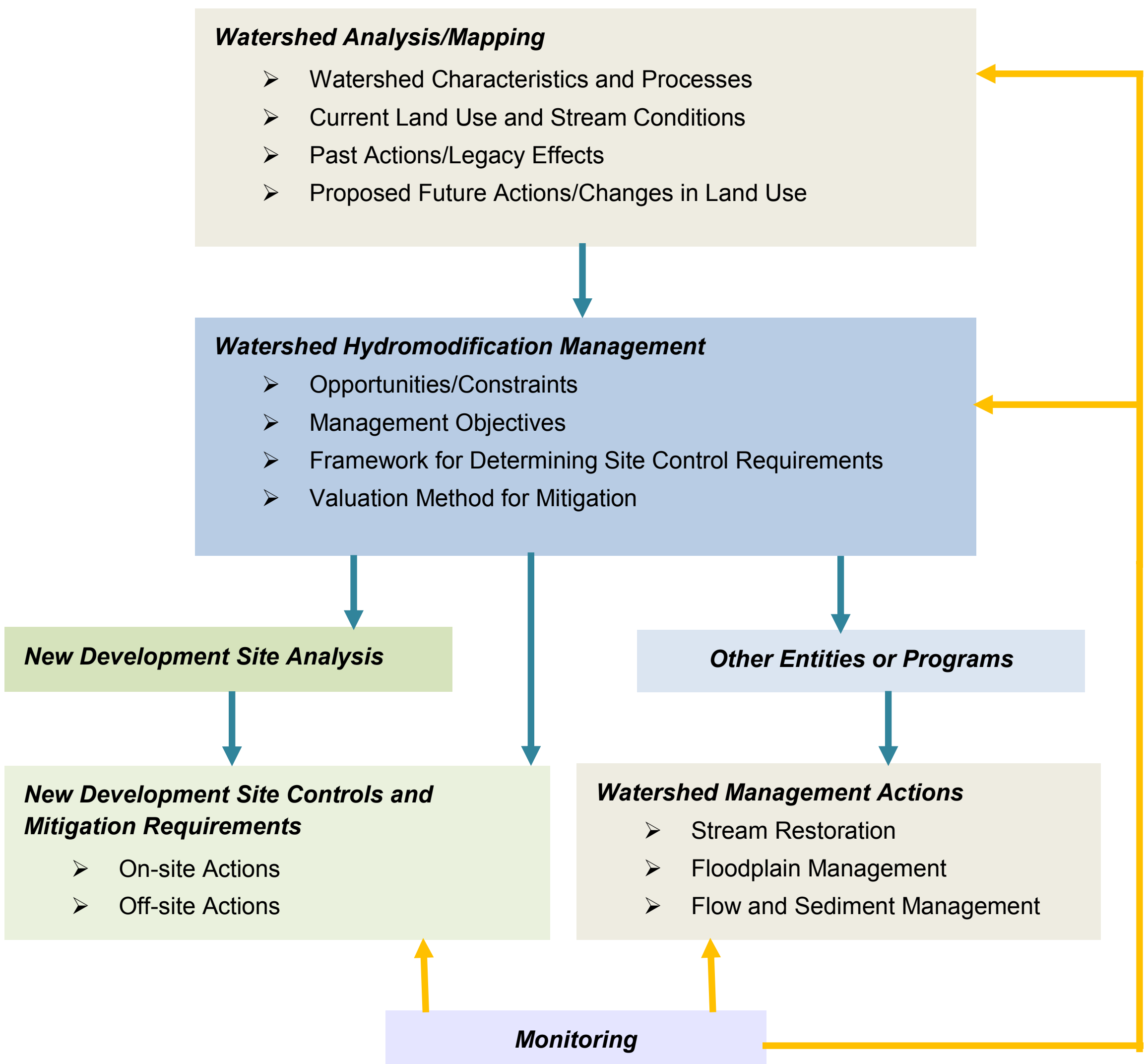


Figure ES-1. Framework for Integrated Hydromodification Management.

An effective management program will likely include combinations of on-site measures (e.g., low-impact development techniques, flow-control basins), in-stream measures (e.g., stream habitat restoration), floodplain and riparian zone actions, and off-site measures. Off-site measures may include compensatory mitigation measures at upstream locations that are designed to help restore and manage flow and sediment yield in the watershed.

Project-specific analysis and design requirements should vary depending on location, discharge point, and size. The range of efforts may include:

- Application of scalable, standardized designs for flow control based on site-specific soil type and drainage design. The assumptions used to develop these scalable designs should be conservative, to account for loss of sediment and uncertainties in the analysis and our understanding of stream impacts.
- Use of an erosion potential metric, based on long-term flow duration analysis and in-stream hydraulic calculations. Guidelines should specify stream reaches where in-stream controls would and would not be allowed to augment on-site flow control.
- Implementation of more detailed hydraulic modeling for projects of significant size or that discharge to reaches of special concern to understand the interaction of sediment supply and flow changes.
- Analysis of the water-balance for projects discharging into streams with sensitive habitat. This may include establishment of requirements for matching metrics such as number of days with flow based on the needs of species present.

Achieving these goals will require that hydromodification management strategies operate across programs beyond those typically regulated by NPDES/MS4 requirements. Successful strategies will need to be developed, coordinated, and implemented through land-use planning, habitat management and restoration, and regulatory programs. Regulatory coordination should include programs administered by the Water Boards, such as non-point source runoff control, Section 401 Water Quality Certifications and Waste Discharge Requirement programs, and traditional stormwater management programs. It should also include other agency programs, such as the Department of Fish and Game Streambed Alteration Program and the Corps of Engineers Section 404 Wetland Regulatory Program. Thus, all levels of the regulatory framework—federal, state, and local—will need to participate in developing and implementing such a program. The integrated watershed-based approach will likely take one or more permit cycles (i.e., at least ten years) to fully implement.

Short- and long-term recommendations for management are summarized in Table ES-1 below.

Table ES-1. Recommendations for implementing watershed-based hydromodification management.

Time Frame	Programmatic: State and Regional Water Boards	Local: City and County Jurisdictions
Short-term (<10 years)	• Establish consistent standards for HMPs • Promote use of watershed approaches in HMPs to move away from reliance on project-based management actions • Develop a valuation method to determine appropriate off-site mitigation • Transition to a broader set of monitoring endpoints including flow, geomorphology, and biology	• Implement watershed analysis of opportunities and constraints related to hydromodification • Implement a broader set of tools to improve on-site management actions • Develop institutional capacity to oversee and review modeling and assessment tools • Develop capacity for information/data management and dissemination
Long-term (1+ decades)	• Develop watershed-based regulatory programs and policies for hydromodification management • Integrate hydromodification management needs into other regulatory programs (e.g. TMDL, 401/WDR)	• Develop institution capacity to implement watershed-based hydromodification programs • Incorporate hydromodification and other water quality management into the land use planning process

To successfully accomplish these various recommendations for implementation, both agencies and private-sector practitioners will need to make use of a range of analytical tools. Such tools generally fall into three categories: descriptive tools, mechanistic models, and empirical/statistical models. Models may be used deterministically and/or in a probabilistic manner. These different types of tools can be selected or combined, depending on the specific objective, such as characterizing stream condition, predicting response, establishing criteria / requirements, or evaluating the effectiveness of management actions. Selection of tools should also consider the type of output, intensity of resource requirements (i.e., data, time, cost), and the extent to which uncertainty is explicitly addressed. It is important to note that deterministic modeling without accompanying probabilistic analysis may mask the uncertainties inherent in predicting hydromodification effects. Short-term and long-term recommendations for the application and improvement of tools to support the management framework are shown in Table ES-2.

Although there is sufficient scientific and engineering understanding of hydromodification causes and effects to begin implementing more effective management approaches now, improvements should be informed and adapted based on subsequent monitoring data. To be useful, monitoring programs should be designed to answer questions and test hypotheses that are implicit in the choice of management actions, such that practices that prove effective can be emphasized in the future (and those that prove ineffective can be abandoned). The focus of monitoring efforts, however, needs to be tailored to the time frame of the questions being addressed and the implementing agency (Table ES-3), reflecting the dual goals and audiences of this document.

Table ES-2. Recommendations for the application and improvement of tools in support of the proposed management framework.

Time Frame	Programmatic: State and Regional Water Boards	Local: City and County Jurisdictions
Short-term (<10 years)	• Develop quality control and standardization for continuous simulation modeling • Perform additional testing and demonstration of probabilistic modeling for geomorphic response • Pursue development of biologically- and physically-based compliance endpoints	• Work cooperatively with adjacent jurisdictions to implement hydromodification risk mapping at the watershed scale • Implement continuous simulation modeling for project impact analysis
Long-term (1+ decades)	• Improve tools for sediment analysis and develop tools for sediment mitigation design • Develop tools for biological response prediction • Improve tools for geomorphic response prediction	• Expand use of probabilistic and statistical modeling for geomorphic response • Apply biological tools for predicting and evaluating waterbody condition

Table ES–3. Recommendations for hydromodification monitoring.

Time Frame	Programmatic: State and Regional Water Boards	Local: City and County Jurisdictions
Short-term (<10 years)	• Define the watershed context for local monitoring (at coarse scale) • Evaluate whether permit requirements are making positive improvements	• Evaluate whether specific projects/ regulations are meeting objectives • Identify the highest priority action(s) to take
Long-term (1+ decades)	• Define watershed context and setting benchmarks for local-scale monitoring (i.e., greater precision, if/as needed) • Demonstrate how permit requirements can improve receiving-water "health," state-wide (and change those requirements, as needed)	• Evaluate and demonstrate whether actions (on-site, instream, and watershed scale) are improving receiving-water conditions • Assess program cost-effectiveness • Identify any critical areas for resource protection

Identifying and, ultimately, achieving the desired conditions in receiving waters requires multiple lines of evidence to characterize condition in an integrative fashion. At their most comprehensive, the chosen metrics should include measures of flow, geomorphic condition, chemistry, and biotic integrity. Biological criteria are key to integrative assessment: in general, biological criteria are more closely related to the designated uses of waterbodies than are physical or chemical measurements. This understanding is reflected in the State's proposed bio-objectives policy, which includes explicit links to hydromodification management.

In summary, transitioning from the current site-based to a more effective watershed-based approach to hydromodification management that addresses both legacy and future impacts will require cooperation between the State and Regional Water Boards and local jurisdictions. Both technical and regulatory/program approaches will need to be updated or revised altogether over the next several permit cycles to realize this long-term goal. Substantial resources will be necessary to realize these goals; therefore, opportunities for joint funding and leveraging of resource should be vigorously pursued from the onset. This cooperative approach should replace the current fragmented efforts among regions and jurisdictions.

1. OVERVIEW AND INTENDED USES OF THE DOCUMENT

1.1 Overall Objectives and Intended Audience

Regulation and management of hydromodification is in its infancy in California. As with any new endeavor, initial attempts to meet this need is unproven, inconsistent, and relatively narrow in focus. To improve on existing efforts, the State Water Resources Control Board (SWRCB) has engaged a team of experts to provide technical support to both regulators and permittees for development of Hydromodification Management Plans (HMPs) and their associated permit requirements. This resulting document has two goals and hence two audiences.

The first goal of this document is to provide broad perspectives on what would constitute effective hydromodification assessment, management and monitoring, based on our current best scientific understanding of the topic. The audience for this goal is primarily the State and Regional Water Boards, since meeting this goal will require integration of watershed and site-scale activities that are likely beyond the control or responsibility of any individual municipality. Success will require fundamental changes in the regulatory and management approach to hydromodification that will likely be possible only iteratively and potentially requiring one or more NPDES permit cycles to fully implement. The State and Regional Water Boards will need to provide leadership in implementing these changes, but they will also need to work cooperatively with permittees so that planning, management and monitoring programs can be adapted to operate in a more integrated manner over the broader spatial scales and longer time frames that are necessary to achieve genuine success. Furthermore, hydromodification management plans will need to address preexisting conditions from previous (i.e., legacy) land uses. Clearly, addressing such past effects will require approaches beyond regulation of new development.

This document provides broad perspectives on what would constitute effective hydromodification assessment, management and monitoring, based on our current best scientific understanding of the topic. The document also provides near-term technical assistance for implementing current and pending hydromodification management requirements.

The second goal of this document is to provide near-term technical assistance for implementing current and pending hydromodification management requirements. This goal can be achieved by municipalities within the construct of existing programs, and therefore the primary audience for this aspect of the document is MS4 permittees. Achieving this goal will facilitate greater consistency and effectiveness between HMPs, giving them a stronger basis in current scientific understanding, and will also serve as initial steps toward realizing the broader goal stated above.

1.2 Rationale and Justification

The process of urbanization has the potential to affect stream courses by altering watershed hydrology and geomorphic processes. Development and redevelopment can increase impervious surfaces on formerly undeveloped landscapes and reduce the capacity of remaining pervious surfaces to capture and infiltrate rainfall. The most immediate result is that as a watershed develops, a larger percentage of rainfall becomes surface runoff during any given storm. In addition, runoff reaches the stream channel much more efficiently, so that the peak discharge rates for floods are higher for an equivalent rainfall than they were prior to development. This process has been termed hydromodification. In some instances, direct channel alteration such as construction of dams and channel armoring has also been termed "hydromodification." Such direct alterations are not the focus of this document. Rather, this document focuses on the geomorphic and biological changes associated with changes in land use in the contributing watershed, which in turn alter patterns and rates of runoff and sediment yield. These changes can result in adverse impacts to channel form, stream habitat, surface water quality, and water supply that can alter habitat and threaten infrastructure, homes, and businesses.

The State and Regional Water Boards have recognized the need to manage and control the effects of hydromodification in order to protect beneficial uses in streams and other receiving water bodies. This recognition has led to the inclusion of requirements for development of "hydromodification management plans" (HMPs) in many Phase 1 and some Phase 2 Municipal Stormwater (MS4) permits. Most HMPs require the permitted municipalities to develop programs and policies to assess the potential effects of hydromodification associated with new development and redevelopment, to require the inclusion of management measures to control the impacts of hydromodification, and to develop monitoring programs to assess the effectiveness of HMP implementation at controlling and/or mitigating the impacts of hydromodification.

Development of HMPs is challenging for several reasons. First, there are few accepted approaches for assessing the impacts of hydromodification. Traditional modeling tools are generally untested and may be difficult to apply or inappropriate for use in some California watersheds and streams. Responses of streams to hydromodification are difficult to assess, given inherent climatic variability and the highly stochastic nature of rainfall and the resulting response of streams to runoff events. There are few local examples or case studies from which to draw experiences or conclusions.

As a result of these challenges, individual HMPs to date have utilized a variety of approaches with little coordination or consistency between them. Little information is available on the relative efficacy of any of these approaches. Furthermore, where approaches and tools developed for HMPs in one region of the State (or even from a different region of the country altogether) have been used in subsequent HMPs elsewhere, there has been little or no consideration of the effect of regional climatological or physiographical differences on the transferability of analytical techniques and tools.

1.3 Need for an Expanded Approach

Current site-based hydromodification management approaches are limited in their ability to address the underlying processes that are responsible for most deleterious impacts of hydromodification. Hydromodification effects, by definition, are watershed-dependent processes that are influenced by water and sediment discharge, movement, and storage patterns that may be occurring up- or downstream of a specific project site. Ideally, then, the first step of any hydromodification management plan (HMP) should be a watershed analysis; management of processes at the site or project scale should be done only in the context of such a watershed analysis. Understanding larger-scale processes facilitates prioritization of activities in areas of greatest need and allows for management measures to be located where they have the largest potential benefit, even if that is not on or adjacent to the project site where the current impact is occurring. It also allows for expansion of site based management beyond simple flow control and/or channel stabilization toward strategies that consider flow, sediment, and biological conditions as an integrated set of desired endpoints.

Because watershed boundaries are often not the same as geopolitical boundaries of cities or counties, incorporation of watershed analysis will require leadership from the State and Regional Water Boards. Changes to the current regulatory structure may be necessary to accommodate inter-jurisdictional cooperation and regional information sharing. Similarly, program implementation by both large and small municipalities must include mechanisms that allow site-specific decisions to be informed by watershed-scale analysis.

This document is intended to help address some of these challenges and needs by providing technical recommendations, both to state and regional program developers and to local implementing agencies, for assessment, modeling, development of management strategies, and monitoring. This document can support current HMP development and, at the same time, serve as a first step toward achieving the longer term goals of more integrated, watershed-based hydromodification management.

Current site-based approaches are limited in their ability to address the underlying processes that are responsible for hydromodification impacts.

Effective management of hydromodification will require coordinated approaches across programs at the watershed scale that address all aspects of runoff, sediment generation and storage, instream habitat, and floodplain management.

Adopting this broader approach means that managing the effects of hydromodification cannot be the purview of the stormwater (MS4) program alone. Effective management of hydromodification will require coordinated approaches across programs at the watershed scale that address all aspects of runoff, sediment generation and storage, instream habitat, and floodplain management. Various SWRCB programs have the opportunity and ability to contribute to the goals of comprehensive hydromodification management, including the non-point source control program, water quality certifications, waste discharge requirements, basin planning, SWAMP, and the emerging State Wetland Policy and Freshwater Bio-objectives program. Each of these programs can take advantage of the tools and approaches outlined in this paper to contribute to coordinated management of hydromodification in order to protect beneficial uses and meet basin plan objectives. Furthermore, successful control and mitigation of hydromodification effects will support other programs by improving water quality, enhancing groundwater recharge, and protecting habitat. Therefore, hydromodification management can be a unifying element of many programs and support integrated regional watershed planning.

It is important to note that hydromodification has the potential to affect all water body types; therefore, HMPs should address potential effects to all streams and receiving waters. Because streams are most directly affected by hydromodification, they have been the focus of current regulatory requirements and, therefore, most HMPs. Consequently, this document emphasizes tools and approaches applicable to fluvial systems, which are broadly defined to include wadeable streams, large rivers, headwater streams, intermittent and ephemeral drainages, and alluvial fans (although new specific tools may be necessary for assessment and management of alluvial fans). We recognize, however, that hydromodification can also affect nearshore and coastal environments, including bays, harbors, and estuaries, by altering estuary channel structure, water quality, sand delivery, siltation, and salinity. These effects have been less extensively studied or documented and have received substantially less attention in current hydromodification requirements. Future efforts should more directly address hydromodification effects to all receiving waters, but the information is not presently available to provide equally comprehensive guidance here.

1.4 Scope and Organization

This document is not intended to be prescriptive or to serve as a "cookbook" for development of hydromodification management strategies. Rather, it is a resource to evaluate the utility of existing tools and approaches, and it proposes a framework for integrating multiple approaches for more comprehensive assessment and management. This framework should be used to aid in the development of HMPs that are appropriate for specific regions and settings and take advantage of the best available science. It can also be used to improve consistency in assessment and monitoring approaches so that information collected across regions and programs can be compiled and leveraged to provide more comprehensive assessments of the effectiveness of management actions. Ultimately, such consistency should improve the effectiveness of all programs.

The authors, a team of technical experts, developed the content for this document in consultation with agency staff and regulated entities. The document begins with a brief general discussion of the effects of hydromodification and stream response mechanisms, providing the best available science to support subsequent recommendations. The main body of the document focuses on presenting a proposed new management paradigm where site-based management is nested within an overall watershed assessment that accounts for past, current, and proposed future land use. The body of the document also includes a discussion of existing tools and how they can be used more effectively and appropriately to evaluate potential impacts and guide decisions on selection and design of management practices. The third major section of the document focuses on monitoring that includes evaluation of hydrologic, geomorphic, and biologic conditions with an overriding goal of adaptive management. The document concludes with several technical appendices that offer specific guidance on the appropriate application of tools and models within the existing HMP approaches, and a bibliography of resources.

2. Hydromodification Science

2.1 Introduction

Land-use changes can alter a wide variety of watershed processes, including site water balance, surface and near-surface runoff, groundwater recharge, and sediment delivery and transport. Although alteration to these watershed processes (referred to collectively as hydromodification) can affect many elements of a landscape, the focus of this document is on impacts to stream systems. Furthermore, while this paper will often refer to urbanization, it is recognized that other types of land-use changes (grazing, agricultural, forestry, etc.) can have similar impacts. This section reviews relevant hydrologic processes and summarizes the impact of urbanization on hydrologic, biologic, and geomorphic systems, and it describes our current understanding of the physical mechanisms underlying these impacts. This provides a foundation for establishing assessment tools and predictive models, as well as for developing management and monitoring programs.

Although not addressed by this report, urbanization also has a range of effects on water quality (*Heaney and Huber* 1984, *Brabec et al.* 2002) by increasing pollutant loads (*Owe et al.* 1982), increasing nutrient loads (*Wanielista and Yousef* 1993, *Hubertz and Cahoon* 1999), and diluting dissolved minerals through increased runoff and decreased infiltration and soil contact (*Loucaides et al.* 2007). As a result of both its physical and chemical effects, urbanization also affects the integrity of biota (*Heaney and Huber* 1984) including fishes (*Klein* 1979, *Weaver and Garman* 1994, *Wang et al.* 2000) and invertebrates (*Sonneman et al.* 2001, *Wang and Kanehl* 2003). These impacts are acknowledged and evaluated in the discussion of monitoring Section 4, but the details of their interactions and effects are not otherwise addressed here.

Land-use changes can alter a wide variety of watershed processes, including site water balance, surface and near-surface runoff, groundwater recharge, and sediment delivery and transport. Alteration to these watershed processes are referred to collectively as hydromodification.

2.2 Hydrology Overview

To understand the effects of urbanization, the basic processes of the hydrologic system must be highlighted. A watershed's drainage system consists of all the features of the landscape that water flows over or through (*Booth* 1991). These features include vegetation, soil, underlying bedrock, and stream channels. Urban elements such as roofs, gutters, storm sewers, culverts, pipes, impervious surfaces such as parking lots and roads, and cleared and compacted surfaces fundamentally change the rate and character of hydrologic processes. Generally, the hydrologic changes associated with development and urbanization increases the speed and efficiency with which water enters and moves through the drainage system. In undeveloped watersheds, only a portion of the precipitation that falls ever enters the stream channel. Instead, precipitation may be: 1) evaporated off the ground surface or intercepted by vegetation and evaporated; 2) transpired from the soil; or 3) infiltrated deeply into regional aquifers. For the portion of precipitation that ultimately enters the stream, the rate and processes of delivery vary between watersheds, with important implications for how urbanization will affect runoff.

Flow can be classified as stormflow (or "quickflow") if it enters the stream channel within a day or two of rainfall (*Dunne and Leopold* 1978). Quickflow occurs through 1) infiltration excess (also called "Horton") overland flow, wherever rainfall intensity exceeds the infiltration capacity of the soil and water flows over the ground surface; 2) saturation excess overland flow, where overland flow occurs following filling of all pore space in surface soils; 3) shallow subsurface flow, where water flows relatively quickly through permeable shallow soils (but still more slowly than either Horton or saturation overland flow); and 4) precipitation directly into stream channels. Conversely, water that infiltrates more deeply is classified as delayed flow, because it travels slowly as deep groundwater and emerges into a stream slowly over time.

As a storm progresses, runoff patterns and rates can change, even within the same catchment. For example, surficial soils may become saturated during the course of a storm (or a storm season) as the water table rises, and this can induce a shift in runoff from shallow (or even deep) subsurface flow to the quickflow process of saturation excess overland flow (*Booth* 1991). Even under scenarios in which rainfall intensity exceeds infiltration capacity, Horton overland flow will not be connected to stream channels until surface depressions are filled.

2.3 Impacts of Urbanization

The archetypal model of development involves clearing vegetation; grading, removing, and compacting soils; building roads and stormwater sewers; constructing buildings; and re-landscaping. The specific ways in which these activities alter runoff processes are discussed below. Development may also directly alter stream, such as through channel straightening, levee construction, and flood control reservoirs; however, discussion of the impacts of these alterations is beyond the scope of this document.

2.3.1 Decreased Interception

When rainfall occurs in a watershed, some of the precipitation will be intercepted by vegetation and leaf litter and prevented from entering the stream channel network (Figure 2-1). The percentage of precipitation that can be intercepted varies according to cover type and the character of rainfall (rainfall intensity, storm duration, storm frequency, evaporation conditions) (*Dunne and Leopold* 1978). The effectiveness of interception decreases as a storm progresses because once the surface area of a tree is completely wetted, water will drip off leaves and run down the vegetation as stem flow. Typically, 10-35% of precipitation is intercepted by trees and 5-20% by crops, though these amounts vary widely (*Dunne and Leopold* 1978, *Xiao and McPherson* 2002, *Reid and Lewis* 2009, *Miralles et al.* 2010). In urban environments where vegetative cover is greatly reduced, landscape-scale interception may be lower by an order of magnitude (*Xiao and McPherson* 2002). Precipitation that is not intercepted enters the drainage system. Thus, the mere reduction in interception in urban areas may produce the hydrologic equivalent of a storm that is 10-30% larger.

Figure 2-1. Vegetation reduces runoff by intercepting a portion of the total rainfall and preventing water from entering the drainage system. (Illustration by Jennifer Natali).

The influence of urbanization on climate is complex and varied. For example, urbanization has been shown to increase temperature (*Kalnay and Cai* 2003), increase or decrease wind speeds (*Oke* 1978, *Balling and Brazel* 1987, *Grimmond* 2007), increase pan-evaporation rates (*Balling and Brazel* 1987), and increase shading of the ground surface (*Kalnay and Cai* 2003). In most studies of urban hydrology, the dynamics of evapotranspiration (ET) are typically, explicitly or implicitly, ignored (*Grimmond and Oke* 1999). This exclusion exists because of the widespread assumption that urban ET is negligible compared to rural areas with higher proportions of vegetation-covered soils (*Chandler* 1976, *Oke* 1979). In cases such as urban deforestation in the temperate Eastern United States, it is appropriate to assume a net loss of ET due to urbanization (*Bosch and Hewlett* 1982, *Sun et al.* 2005, *Roy et al.* 2009). However, spatial variability and the site-specific dynamics of climate, vegetation, and land-use should be considered carefully in arid and semi-arid regions where vegetation is limited prior to development. In drier climates (including much of southern California), primary productivity (and ET) may be substantially increased through the irrigation of urban landscaping (*Buyantuyev and Wu* 2008).

2.3.2 Decreased Infiltration

Infiltration in urban areas is decreased due to several factors: impermeable surfaces such as roads, parking lots, and roofs prevent infiltration by blocking water from reaching soils; heavy-equipment construction operations cause soil compaction and degrade soil structures; construction projects may remove surface soils and expose subsurface soils with poorer infiltration capacity; vegetation-clearing and bare-earth construction increase erosion and loss of topsoil (*Pitt et al.* 2008). The effect of impervious surfaces is intuitive, visible, and dramatic (*Booth and Jackson* 1997), but not all impervious areas affect runoff processes equally. For example, if an impervious surface is built over clayey soils with poor infiltration, the overall runoff rates will be less affected than if built over sandy soils with high natural infiltration rates. While the loss of pervious area has received substantial attention within scientific and policy communities, until recent years considerably less attention has been paid to the effects of compaction and the reductions in infiltration capacity of soils (*Pitt et al.* 2008). Commonly, an area of green is assumed to be permeable, but playing fields and even ornamental lawns may have very low infiltration capacities (*Pitt et al.* 2008). A study of urban runoff in Washington found that impervious areas generated only 20% more runoff than what appeared to be green, pervious areas of lawns (*Wigmosta et al.* 1994). Factors such as excavation and lawn-establishment methods appear to be more significant for infiltration than any other factor including grain size of the original sediments (*Hamilton and Waddington* 1999). Tillage may increase infiltration slightly, while compost or peat soil amendments can increase infiltration by 29 to 50 percent (*Kolsti et al.* 1995).

2.3.3 Increased Connectivity and Efficiency of the Drainage System

Rainfall in urban areas moves quickly as overland flow into storm sewers and the stream channel network (Figure 2-2). The delivery of precipitation into urban stream channels is extremely efficient, transforming essentially all precipitation into stormflow and creating nearly instantaneous runoff. Under natural conditions, in contrast, most runoff to streams is via groundwater paths that typically flow at least one or two orders of magnitude slower than surface water. Thus converting subsurface flow into surface stormflow has dramatic consequences. Furthermore, artificial surfaces such as roofs, pavement, and storm sewers are 1) straight, which shortens the travel distance required for delivery into the channel network; and 2) smooth, which decreases friction and allows flow to travel more quickly than in natural channels (*Hollis* 1975). Storm sewer systems increase the density of “channels,” which further shortens runoff travel distances (Figure 2-3). In particular, upland regions that may not have had any surface channels prior to urbanization are frequently fitted with storm sewers, which dramatically increase delivery efficiency into the channel network (*Roy et al.* 2009). In sum, urbanization transforms watershed processes and flow paths that were once slow, circuitous, and disconnected into engineered and non-engineered systems that are highly efficient, direct, and connected.

In contrast to the slow measured runoff to natural streams by surface and subsurface pathways, the delivery of precipitation into urban stream channels is extremely efficient, transforming essentially all precipitation into stormflow and creating nearly instantaneous runoff.

2.3.4 Decreased Infiltration into Stream Beds

Concreting of bed and banks, channel narrowing, and channel straightening limit infiltration from a stream into the ground. Concrete channel margins create infiltration barriers, while channel narrowing and straightening limit the surface area accessible for infiltration and also create a less complex channel. Channel complexity such as pools, riffles, steps, and debris dams create hydraulics that slow flow velocities and also divert water into the subsurface (*Lautz et al.* 2005). In arid and semi-arid watersheds where streams may flow only occasionally, infiltration through bed, banks, and floodplain areas may significantly lower peak flows and may sustain aquifers vital to regional water supplies and natural habitats (*Kresan* 1988, *Dahan et al.* 2008). Increasing recognition is being paid in the scientific literature to the infiltration services provided by natural channels and floodplains (*Macheleidt et al.* 2006, *Schubert* 2006).

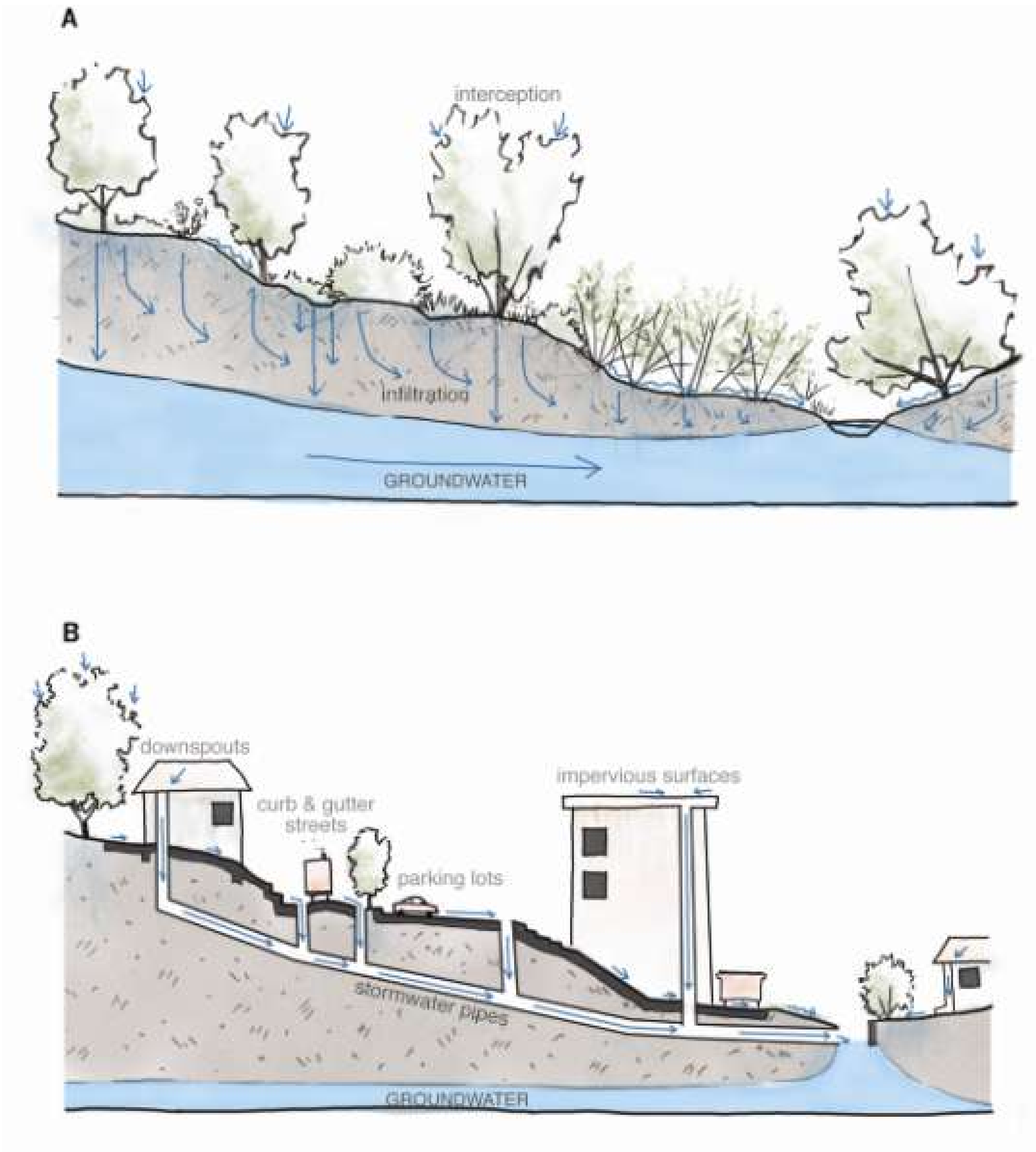


Figure 2-2. Stormwater flowpaths are shortened and quickened through paving, building, soil compaction, and sewer infrastructure. The rapid concentration of streamflow increases storm peaks. Rapid runoff and reduced infiltration prevent groundwater recharge. (Illustration by Jennifer Natali).

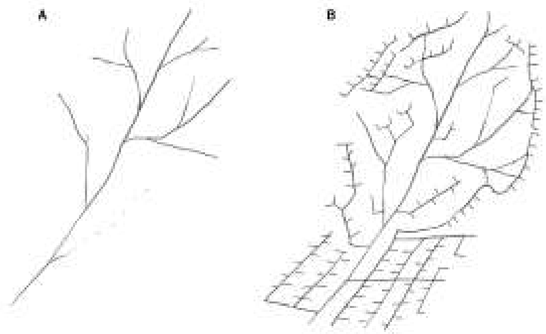


Figure 2-3. Increased surface runoff causes an extension of the channel network. This occurs through increased channel erosion or through constructed networks (to manage increased surface flow). The expanded channel network delivers runoff to downstream reaches much more efficiently. (Illustration by Jennifer Natali).

2.4 Changes in Instream Flow

The instream flow changes resulting from urbanization depend upon site-specific watershed and development characteristics, but typically they include modification of the timing, frequency, magnitude, and duration of both stormflows and baseflow. Urbanization has been shown to increase the magnitude of stormflows, increase the frequency of flood events, decrease the lag time to peak flow, and quicken the flow recession (Figure 2-4; *Hollis* 1975, *Konrad and Booth* 2005, *Walsh et al.* 2005). Because the effects of urbanization manifest differently for different components of the hydrograph, the hydrologic alterations of moderate storms, large storms, and baseflow are discussed individually below.

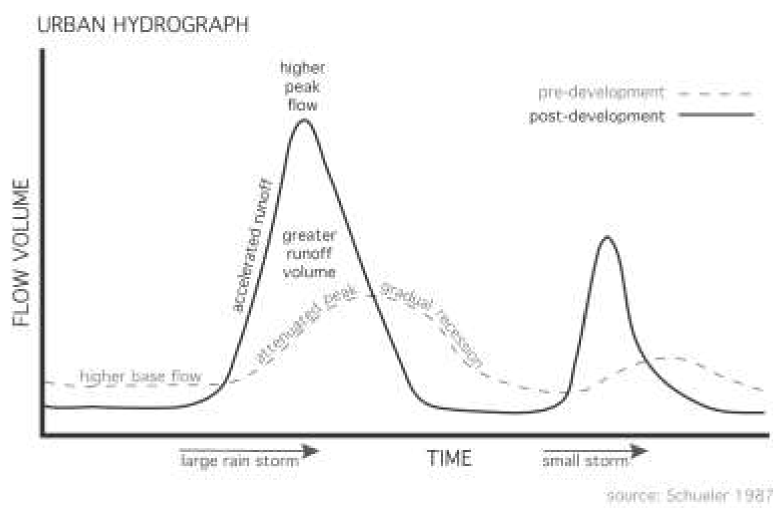


Figure 2-4. Increased runoff efficiency causes higher magnitude peak flows, shorter duration runoff events, decreased baseflow, and dramatic increases in small storms that may have generated little or no runoff under pre-development conditions. (Illustration by Jennifer Natali).

2.4.1 Moderate Stormflow

Urbanization of a watershed can drastically increase the frequency and magnitude of small and moderate flow events (Hawley and Bledsoe 2011). The magnitude of flow amplification increases generally in proportion to the amount of impervious area (*Leopold* 1968, *Hollis* 1975). For example, flows with a return period of one year or longer were shown to be unaffected by paving 5% of the watershed, yet the magnitude of a one-year flow could be more than ten times higher when 20% of a watershed is paved (*Hollis* 1975). In undeveloped watersheds, small storms may not generate any overland flow or streamflow increase at all, because interception, infiltration, soil absorption, and evapotranspiration contain all the precipitation.

Urbanization of a watershed can drastically increase the frequency, duration, and magnitude of small and moderate flow events by factors of 10 or more.

The change to a flashier regime with larger magnitude streamflow generated from small and moderate storms has two primary consequences. First, the stream power and sediment-transport capacity of the stream increase significantly, potentially creating channel erosion and/or stressing instream biota. Second, the season of stormflow is likely to be extended. In undeveloped watersheds, early or late-season storms typically do not generate significant runoff because soils are dry, can effectively absorb most precipitation, and therefore do not generate overland flow or streamflow. Antecedent moisture conditions are less important in urban watersheds where overland flow is generated regardless, and streamflow is generated by even a small storm in a dry watershed. Through magnifying small and moderate storms, urbanization may increase the duration of sediment-transporting and habitat-disturbing flows by factors of 10 or more (*Booth* 1991, *Booth and Jackson* 1997).

2.4.2 Large, Infrequent Storms

In large storms with return intervals of 10 or more years, the influence of urbanization is less pronounced though still present. Whereas a 1-year stormflow may be increased by ten times by paving 20% of the watershed, historical data from humid-region watersheds suggest that the peak magnitude of a 100-year flood would not even be doubled (*Hollis* 1975). The diminishing influence of urbanization on floods of higher recurrence intervals is understood by recognizing that the hydrologic processes of large storms resemble the processes of urban runoff. Essentially, a 100-yr flood is an event that is long in duration, severe in intensity, and likely occurs when soils are already wet. Even in an undeveloped watershed, a storm of this magnitude can typically generate (saturation) overland flow and transport water efficiently into the channel network in a manner more generally comparable to an urban setting.

2.4.3 Baseflow

Urbanization does not affect instream baseflows consistently. Many studies have documented baseflow reductions and/or lowered groundwater levels that have been attributed to decreased infiltration (*Simmons and Reynolds* 1982, *Ferguson and Suckling* 1990) and groundwater extraction (*Postel* 2000). In extreme cases, baseflow in urban watersheds can disappear completely during drought years, dry seasons, or even between storm events during the wet season. The effect of reducing infiltration may be counteracted in urban and suburban landscapes, however, through irrigation of lawns, parks, golf courses, and other water inputs such as septic systems, leaky pipes, and sewage treatment outflow which typically import water from outside the watershed and contribute to both streamflow and groundwater recharge (*Konrad and Booth* 2005, *Walsh et al.* 2005, *Roy et al.* 2009). Indeed, imported water volumes in very dense cities may be an order of magnitude greater than precipitation. Lerner (2002) judged that leakage in water importation and delivery infrastructure typically ranges from 20-50%, and in general this leakage will increase groundwater recharge in urban areas. Similarly, other studies have found municipal irrigation capable of raising groundwater levels and causing surface flooding (*Rushton and Al-Othman* 1994) and changing ephemeral streams into perennial streams (*Rubin and Hecht* 2006, *Roy et al.* 2009). In summary, the magnitude and direction baseflow and groundwater recharge alteration depends on climate, land use, water use, and the infrastructure system of the watershed. There are no simple “rules.”

2.5 Changes in Sediment Yield

The role of watershed sediment yield in the behavior of watersheds was first characterized systematically by Wolman (1967) in a three-part conceptual framework of how rivers respond to urban development, in which 1) pre-development quasi-equilibrium conditions are followed by 2) a period of active construction involving grading, vegetation removal, and bare earth exposed to erosion; and 3) the establishment of an urban landscape consisting of pavement, houses, gutters and sewers etc. The construction period is marked by an increase in sediment (typically 2-10 times pre-development rates) produced from bare surfaces and the disturbances associated with construction (*Chin* 2006). The sediment produced during construction is often deposited within stream channels, initiating aggradation and/or channel widening. Following the construction period, sediment production decreases (Figure 2-5) and runoff increases, resulting in increased transport capacity and the potential for severe channel erosion that can result in channel enlargement of commonly 2-3 (and as much as 15) times the original channel cross-section (*Chin* 2006). Changes in post-construction sediment production rates are not well studied, though case studies have found sediment yields in post-construction watersheds to be somewhat higher than rural, undeveloped basins.

The combination of increased runoff and decreased sediment production can result in channel enlargement of commonly 2-3 (and as much as 15) times the original channel cross-section.

Post-construction sediment loads are typically derived from channel enlargement as a result of increased peak flows and the legacy of construction-phase disturbance (*Trimble* 1997, *Nelson and Booth* 2002). The rate of decline in post-construction sediment yields is therefore predominantly controlled by the degree of channel instability caused by the construction phase and the effect of increased peak flows. If the channel margins are armored, densely vegetated, or otherwise erosion resistant, sediment yields may decline quickly following urbanization. If channel instability ensues, elevated sediment yields may persist for decades or more.

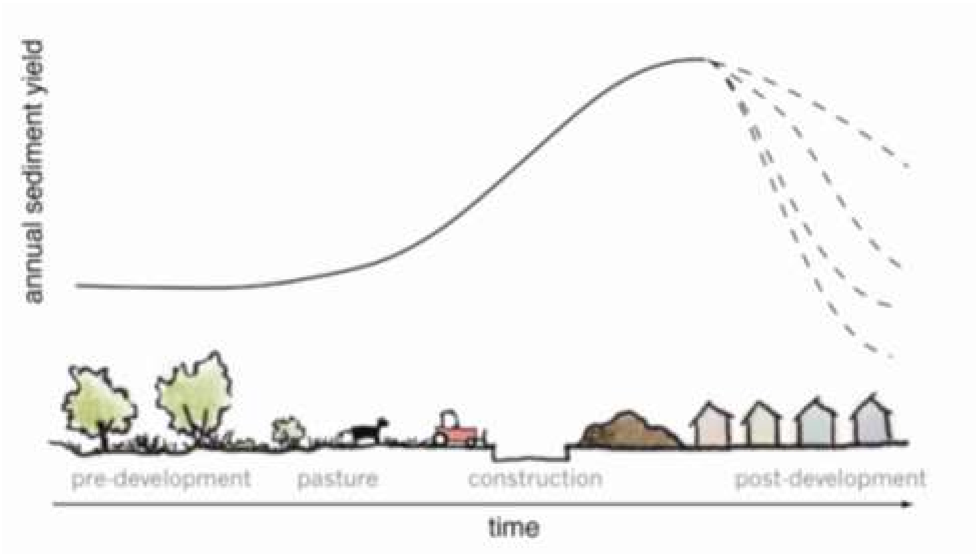


Figure 2-5. Increased sediment yields occur during the land-clearing and construction phases of development. Post-construction sediment yields decrease, though the rate of decrease varies considerably depending on the degree of channel instability caused by the construction phase and by increased runoff. (Illustration by Jennifer Natali).

2.6 Impacts on Channel Form and Stability

Channel form and stability reflect both hydrologic and geomorphic processes. Changes to runoff characteristics and sediment supply can affect all aspects of stream morphology, including planform, cross-sectional geometry, longitudinal profile, bed topography (e.g., pools, riffles), and bed sediment size and mobility. While many factors influence the type and degree of impacts (discussed below), a suite of commonly observed morphological changes due to hydromodification include channel enlargement (incision and widening), decreased bank stability, increased local sediment yield from eroding reaches, overall simplification of stream habitat features such as pools and riffles, changes in bed substrate conditions, loss of connectivity between channel and floodplain (*Segura and Booth* 2010), and changes in sediment delivery to coastal waters (*Jacobson et al.* 2001). Impacts may also propagate upstream as headcuts resulting from reductions in base level due to excess erosion. Likewise, tributaries entering downstream of a developed area may also experience the upstream propagation of headcuts due to base level reductions of the mainstem.

In addition to Jacobson *et al.* (2001), two well-researched literature reviews of morphological impacts (as well impacts to riparian habitat and biota) can be found in: “Impacts of Impervious Cover on Aquatic Systems” by The Center for Watershed Protection (2003) and “Physical Effects of Wet Weather Flows on Aquatic Habitats: Present Knowledge and Research Needs” published by Water Environment Research Foundation (*Roesner and Bledsoe* 2003). Note that these two studies differ significantly in how they synthesize and interpret the reviewed literature, and the CWP publication acknowledges that it does not necessarily apply to streams in the arid west.

2.6.1 Physical Principles Underlying Channel Impacts

A convenient conceptual framework for the physical impacts of hydromodification on stream morphology is “Lane’s Balance” (*Lane* 1955; Figure 2-6). This framework encapsulates a fundamental (albeit qualitative) relationship between the hydrologic and geomorphic processes that balance water flow and sediment in a channel. It expresses the condition of sediment transport capacity, as controlled by water discharge and slope, in broad balance with the supplied load and size of bed sediment for a channel in equilibrium. An increase in streamflow or a decrease in sediment supply (for example) will typically initiate a corresponding decrease in slope and/or increase in grain size in order to reestablish equilibrium. That decrease in slope is expressed by channel incision or degradation. In contrast, an increase in sediment supply or decrease in streamflow will typically result in aggradation and a corresponding increase in slope.

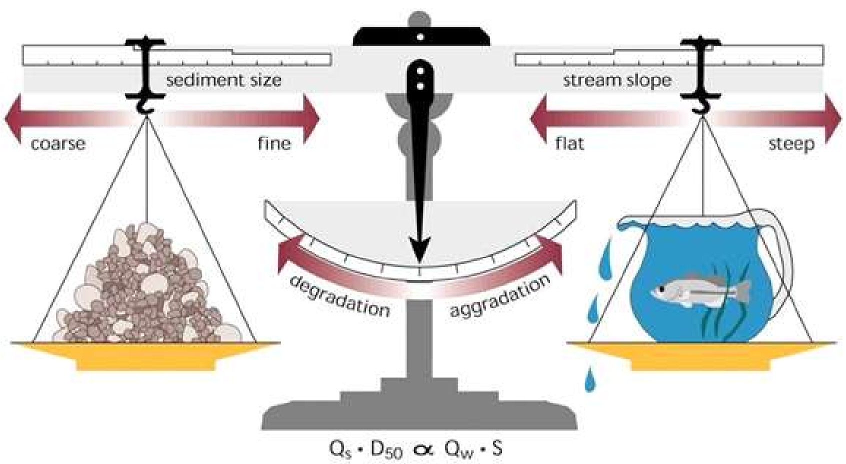


Figure 2-6. Lane’s Balance, showing the interrelationship between sediment discharge (Q_s), median bed sediment size (D_{50}), water discharge (Q_w), and channel slope (S).

Slope and grain size are not the only modes of adjustment, as stream channels have many more degrees of freedom in responding to changes in streamflow and sediment supply. For example, Schumm (1969) extended Lane’s Balance to include width, depth, sinuosity, and meander wavelength. More quantitatively (and more complexly), adjustments to channel form resulting from hydromodification are controlled by interactions among flow-generated shear stresses (described by hydraulic equations for open channel flow, as a function of channel geometry, roughness, and longitudinal slope), inflowing sediment load, and the shear strength of the bed and bank sediments (a function of their size distribution and cohesiveness).

2.6.2 Natural Variability in Stream Systems

Understanding natural variability in streams is critical to predicting and assessing anthropogenic impacts. A stream may be considered "stable" or "at equilibrium" when its overall planform, cross-section and profile are maintained with no net degradation or aggradation within a range of variance, over extended timeframes (*Mackin* 1948, *Schumm* 1977, *Leopold and Bull* 1979, *Biedenharn et al.* 1997). Such systems can often withstand short-term disturbances without significant change. Even without discrete disturbances, natural streams may be in a state of dynamic equilibrium (*Schumm* 1977), where the channel exhibits stability over the long term even while actively migrating laterally such that erosion of outer banks is accompanied by sediment deposition and bar building on inner banks. Streams may also be fluctuating between aggradation/ degradation/ stability, all within a limited range of conditions. A large-scale event, like a flood or landslide, can cause dramatic changes in channel form, but the channel will often re-established its pre-event planform, geometry and slope over time.

In contrast, a persistent alteration like hydromodification can cause the rate of change to increase. As a result, the channel may begin an evolutionary (or catastrophic) change in morphology, leading to enlargement and instability. A geomorphic threshold is the condition at which there is an abrupt and significant channel adjustment or failure because the channel has evolved to a critical situation. It is the condition at which the proverbial straw breaks the camel's back. Channels that are near a geomorphic threshold can exhibit significant adjustments in response to a relatively small degree of hydromodification. For example, a channel with banks that are near the height and angle for geotechnical failure may widen abruptly due to slight incision.

2.6.3 The Role of Sediment Transport and Flow Frequency in Channel Morphology

Extensive research has been devoted to establishing specific relationships between flow frequency and characteristics of channel morphology. The concept of "effective discharge" was introduced by Wolman and Miller (1960), using a magnitude-frequency analysis to assess the effectiveness of flow events to transport sediment. They concluded that, for the rivers in their analysis, relatively frequent events (occurring on average about 1 times/year) are most effective over the long term in transporting sediment. This concept has formed the basis for a large body of literature (and occasional controversy) over the subsequent five decades relating to the relationships between these flow frequencies and principal channel dimensions (e.g., bankfull stage, width-to-depth ratio), and the application of these relationships to stream design and restoration, as well as prediction and control of hydromodification impacts. Much of the controversy has related to the use of a single event ("dominant discharge" or "bankfull flow") as the basis for such applications, with the implicit assumption that control for that single discharge will result in commensurate channel changes regardless of the distribution of flow frequencies and flow durations over a wider range of discharges.

More recently, the concept of a *range* of moderately frequent, "geomorphically significant" flows that transport the majority of the sediment over the long term (King County 1990, *Bledsoe* 2002, *Roesner and Bledsoe* 2003) was proposed to replace the focus on a single event. The geomorphically significant flow range is considered to be the most influential in determining channel form, as this collective group of flows typically does the most "work" on the channel boundary over engineering time scales. Controlling changes to the frequency of flows within this range is therefore critical to reducing impacts to stream morphology, and is the scientific basis for the "flow-duration" control criteria discussed in the following sections. A flow-duration criterion aims to match the pre-development volumes, durations, and frequencies of this critical range of sediment transporting flows over a period of many decades. Even this concept, however, relies on the implicit assumption that infrequent large events, no matter how dramatic their effects, typically occur "too infrequently" to reset channel morphology and habitat over the timescales of concern in meeting regulatory requirements. These events are typically managed through traditional flood control practices as opposed to hydromodification management.

A flow-duration management approach aims to match the pre-development volumes, durations, and frequencies of this critical range of sediment transporting flows over a period of many decades.

2.6.4 Applicability to California Streams

The traditional concepts of dynamic equilibrium in streams and geomorphically significant flows, discussed above, derive largely from studies on perennial streams in humid areas. An important question is: to what extent do these concepts apply to managing hydromodification impacts to streams within arid and semi-arid areas (such as large portions of California, and particularly the southern and eastern regions)? In such climate regions, precipitation is highly variable, with low annual totals and episodic, large events. Many streams are ephemeral or intermittent and located in a setting of extremely high sediment production associated with erosive geology resulting from high rates of tectonic uplift, sparse vegetative cover and frequent fires (*Graf* 1988, *Stillwater Sciences* 2007). These streams are often characterized by multi-thread sand-bed channels that are inherently unstable and readily respond to changes in flow conditions. In the ephemeral streams described by Bull (1997), for example, the natural behavior is one of alternating periods and locations of aggradation and degradation, varying both temporally and spatially. In such "episodic" streams, the vast majority of sediment may be moved by extreme, highly infrequent events. The importance of understanding the role of episodic events has been emphasized for semi-arid and arid fluvial systems (e.g., *Wolman and Gerson* 1978, *Brunsden and Thornes* 1979, *Yu and Wolman* 1987). The latter authors reviewed concepts of frequency and magnitude in geomorphology research and noted that episodic behavior hinges on frequency of episodic events relative to the time required to return to an "equilibrium" channel form. Episodic behavior is more prevalent where the average long-term disturbance is low but the year-to-year variability is high, a characteristic of arid and semi-arid climates.

Although the morphology of arid and semi-arid streams may be more strongly influenced by extreme events under natural conditions, hydromodification has nevertheless been shown to cause rapid and significant physical changes in such California streams (*Trimble* 1997, *Coleman et al.* 2005, *Hawley and Bledsoe* 2011). Such dramatic responses to the effects of urbanization on relatively frequent flows, often over periods of a decade or less, have profound implications for aquatic life and physical habitat. Despite the flashy streamflow regimes, high sediment supplies, and steep gradients of many streams in the region, the responses of California streams are controlled by the same physical processes as those in other regions that have been studied more extensively. As such, the key controls of stream response can be identified and managed to mitigate the chronic effects of hydromodification between infrequent extreme events. However, it is always advisable to ensure that the application of tools and approaches for prediction and assessment should be based on reference data and empirical models (where applicable) drawn from stream types that are similar in both hydrologic and geomorphic characteristics.

2.6.5 Factors Determining Extent of Impacts

The extent and nature of impacts to stream morphology and habitat from a given change in runoff and sediment supply vary widely, depending on the channel geometry, longitudinal slope, channel material type(s) and size(s), and the type and density of channel vegetation (*Center for Watershed Protection* 2003, *Roesner and Bledsoe* 2003). For example, increased flows within a deep, narrow channel may result in significantly higher shear stresses at the bed; this same increase in a wide, shallow channel may become predominantly overbank flow, with less effect on bed shear stress. Where all other factors are equal, fewer impacts would be expected where flows have access to broad overbank areas (i.e., floodplains) during relatively common floods (*Segura and Booth* 2010), channel materials are more resistant, and stabilizing riparian vegetation is present. Conversely, where erosion and bank instability result in the loss of vegetation reinforcement, a positive feedback response may cause erosion to be accelerated. Furthermore, the relative erosive resistance of bed and bank materials will influence the extent of lateral versus vertical channel adjustments (*Simon and Rinaldi* 2006, *Simon et al.* 2007). For example, if bank resistance is lower than bed resistance, then the channel will tend to widen rather than deepen.

The extent of impacts will also depend on the stream's physiographic context and spatial and temporal patterns of urban development within the watershed (Konrad and Booth 2005). Large-scale studies of hydrologic responses to urbanization (Chin 2006, Poff et al. 2006) also highlighted the regional variation in these responses and reinforced the need to understand local watershed and channel characteristics when managing hydromodification impacts. The presence of road crossings and other infrastructure can provide local grade control and create sediment bottlenecks which often translate to exacerbated erosion in the immediately downstream areas.

The extent and nature of impacts to stream morphology and habitat from a given change in runoff and sediment supply vary widely, depending on the channel geometry, longitudinal slope, channel material type(s) and size(s), and the type and density of channel vegetation, and the spatial and temporal patterns of urban development

An additional consideration relates to the pre-development balance between sediment and streamflow, which is dependent on precipitation patterns, the location of a stream reach within the watershed, the associated sediment behavior of that reach (i.e., production, transport or deposition zone), and local rates of sediment production.

While many of these factors may be quantified for a given time and location, stream systems are enormously complex both spatially and temporally. The existence of physical thresholds and feedback systems can cause an incremental change to result in a disproportionately large response (*Schumm* 1977, 1991). Furthermore, there may be significant temporal lags between the point in time at which land use is altered and when channel impacts are observed (*Trimble* 1995, 1997). In recognition of these effects and the associated uncertainty, predictive models and management tools may present results in terms of probabilities or within the context of a risk-based approach, as discussed further in this document. Such effects also have substantial implications for the design of assessment and monitoring programs.

There may be significant temporal lags between the point in time at which land use is altered and when channel impacts are observed.

2.6.6 Impacts on Other Types of Receiving Waters

Although outside the scope of this document, hydromodification impacts to other water body types are recognizable and should be the subject of additional research and future consideration.

Wetlands, Estuaries, and Coastal Ecosystems. Urbanization can alter water quality, quantity and sediment delivery to wetlands and sensitive coastal ecosystems. Urbanization has led to loss or degradation of wetlands and estuaries as a result of 1) draining and conversion to agriculture (*Dahl*, 1997); 2) upstream alterations to flow and sediment regimes that can change the magnitude, frequency, timing, duration, and rate of change of estuarine salinity, turbidity, freshwater flooding, freshwater baseflow, and groundwater recharge dynamics (*Azous and Horner* 2001); and 3) contaminated runoff from urban areas (*Paul and Meyer* 2001, *J Brown et al.* 2010). Urbanization may also lead to coastal erosion in circumstances where reservoir sediment trapping or post-development decreases in sediment yield reduce the sediment supply to the coast (*Pasternack et al.* 2001, *Syvitski et al.* 2005).

Alluvial Fans. Alluvial fans are dynamic landforms that are under increased development pressure in recent decades, particularly in the expanding cities of the American West. Upstream urbanization, and the resultant flashier flow regime, shortens the time available for infiltration and groundwater recharge in alluvial fans. Furthermore, development on fans themselves results in channel straightening and/or construction of concrete flood conveyance channels that also reduce or eliminate infiltration. The reduction in infiltration amplifies the flood risk further downstream. Additionally, alluvial fans may be more vulnerable than other landscapes to channel instability resulting from hydromodification, because they lack intrinsic geologic controls on channel gradient, and commonly have little vegetation or bank cohesion to provide stability in the purely alluvial deposits (*Chin* 2006).

2.6.7 Influence of Scale

The ability to detect impacts from land-use changes depends upon the spatial and temporal scale at which they are measured. Issues of hydrograph timing and the relative size of the storm system with respect to the watershed area may confound relationships at larger spatial scales. Furthermore, a number of fluvial geomorphic features that are commonly used as metrics of geomorphic condition are scale-dependent. For example, width-depth ratio, tendency toward braiding, and channel depth relative to stable bank height all commonly increase downstream. Other factors, such as the influence of vegetation, depend on protrusion relative to width and rooting depth relative to bank height. The temporal scale over which channel changes occur will be influenced by precipitation variability, in addition to the many physical factors already discussed.

These scale considerations, as well as previous discussion of factors influencing stream response, are important when determining the choice of both management tools and monitoring approaches. It is generally much easier to predict the direction of response than the magnitude. Accurate, detailed predictions of response are difficult to make, and they are generally only possible when applied to specific locations, using extensive data input, to answer very specific questions; even then they are subject to uncertainty. Policies or assessment methods aimed to address a range of streams and geographic conditions are better suited to probabilistic approaches that explicitly acknowledge uncertainty, as described further in subsequent sections.

2.7 Impacts on Fluvial Riparian Vegetation

Stream channel form and stability is closely linked with the ecology of instream and floodplain habitats (Figure 2-7). Spatial and temporal distributions of plant communities are tied to moisture availability and seasonality. The ability of vegetation to stabilize soils, trap sediments, and reduce flow velocities (*Sandercock et al.* 2007) can create positive feedback that promotes further vegetation establishment and enhancement of these stabilizing features. This can result in a strong influence on channel geometric features, specifically channel narrowing (*Anderson et al.* 2004). The change in frequency of overbank flows resulting from channel incision will also affect riparian processes, including nutrient transfer and seed dispersal. For example, it is believed that *Tamarix* dominance over native species along Western US rivers would be less extensive if not for anthropogenic alteration of streamflow regimes (most recently supported by Merritt and Poff (2010)).

Impacts to stream biota may occur through the alteration of habitat structure and habitat dynamics caused by hydrologic and geomorphic changes, as well as directly from hydrologic alteration.

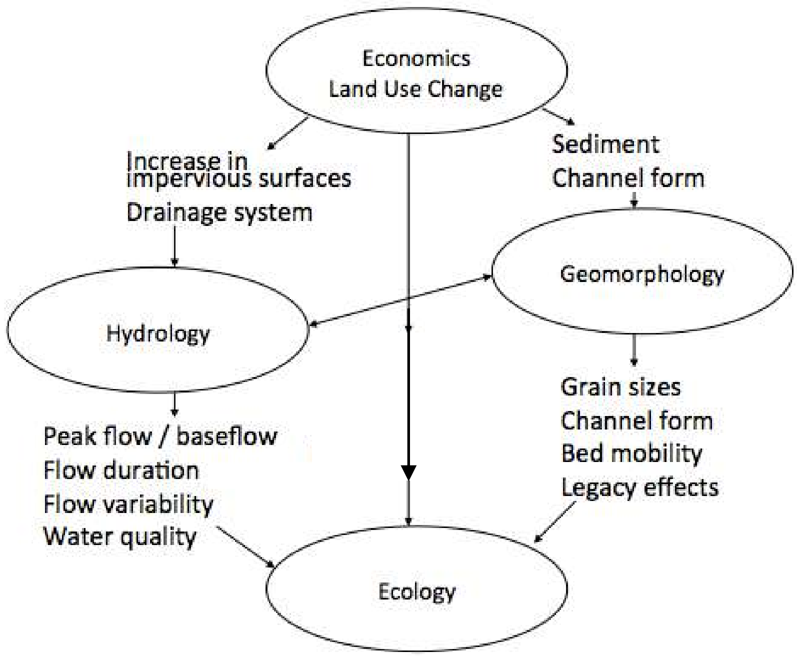


Figure 2-7. Land use changes, hydrology, geomorphology and ecology are closely and complexly interrelated. (Adapted from Palmer *et al.* 2004).

Vegetation changes not only are a result of morphological impacts but also can result directly from changes in streamflow. These findings continue to be supported by recent studies; for example, increases or decreases in baseflow or changes to the seasonal availability of water can determine the extent and type of riparian vegetation capable of thriving in that environment (*White and Greer* 2006). Vegetation changes can have cascading effects on indigenous fauna that require native plants for food or nesting (*Riley et al.* 2005). Channel incision can also result in phreatic draining of adjacent wetland and floodplain habitats and result in loss of key riparian species (*Scott et al.* 2000).

2.8 Impacts on In-Stream Biota

As shown in Figure 2-7, impacts to stream biota may occur through the alteration of habitat structure and habitat dynamics caused by hydrologic and geomorphic changes, as well as directly from hydrologic alteration. (The term biota is used here to refer to a range of non-plant species including algae, macroinvertebrates, amphibians, fishes, etc.) Because of these relationships, the condition of in-stream biota is considered to reflect the effects of all other impacts and has been recommended as an integrative measure of stream health (discussed further in Section 5).

Studies continue to build on Poff *et al.* (1997), who highlighted the importance of the “natural flow regime” and its variability as critical to ecosystem function and native biodiversity. Streamflow pattern or “regime” interacts with the geomorphic context to control the physical and biological response of streams to hydromodification. The basic characteristics of streamflow regimes are typically described in five ways: magnitude, frequency, duration, timing, and rate of change. There is a large body of science linking one or more of these five elements of flow regimes to geomorphic processes, physical habitat, and ecological structure and function. A few examples of linkages with physical habitat are provided in Table 2-1; these linkages describe the mechanisms by which flow changes can impact stream ecology through morphological alterations.

Table 2-1. Examples of Relationships between Flow Regime Attributes and Physical Habitat Characteristics (adapted from Roesner and Bledsoe 2002).

Flow Attribute	Example Relationships with Physical Habitat
Magnitude	• Determines extent to which erosion/removal thresholds for substrate, banks, vegetation, and structural habitat features are exceeded • Determines whether floodplain inundation/exchange occurs • Habitat refugia may become ineffective during extreme events
Frequency	• Flashiness can affect potential for recovery of quasi-equilibrium channel forms between events, bank stability, and streambank/riparian vegetation assemblages • Frequency of substrate disturbance can act as a major determinant of fish reproductive success and benthic macroinvertebrate abundance and composition
Duration	• Determines the impact of a threshold exceeding event, e.g., scour depths • Urbanization frequently increases the duration of geomorphically effective flows which also affect bank vegetation establishment and maintenance • Extended durations of high suspended sediment concentrations can act as chronic and acute stressors on fish communities
Timing	• The temporal sequence of flow events affects channel form and stability as geomorphic systems may be “primed” for abrupt changes. • Stream biota may use flow timing as a life-cycle cue • Predictability of flow can affect utilization of habitat refugia
Rate of Change	• Affects bank drainage regimes (bank stability) and sedimentation processes, e.g., re-suspended fine sediment concentrations during storm hydrographs, embeddedness, armoring • Rapid drawdown can result in stranding of instream biota • Rise and fall rates control riparian water table dynamics and seedling recruitment

The mechanisms of such impacts are also well detailed by Center for Watershed Protection (2003); for example, increased flows are related to a reduction in habitat diversity and simplification of habitat features such as pools; this in turn reduces the availability of deep-water cover and feeding areas.

Many studies support the conclusion that stream biota are also directly impacted by altered flow regimes, independent of channel instability and erosion. Konrad and Booth (2005) identified four hydrologic changes resulting from urban development that are potentially significant to stream ecosystems: increased frequency of high flows, redistribution of water from baseflow to stormflows, increased daily variation in streamflow, and reduction in low flow. They caution that ecological benefits of improving physical habitat and water quality may be tempered by persistent effects of altered streamflow and sediment discharge, and that hydrologic effects of urban development must be addressed for restoration of urban streams. Walsh *et al.* (2007) concluded that low-impact watershed drainage design was more important than riparian revegetation with respect to indicators of macroinvertebrate health. Bioengineered bank stabilization can also have positive effects on habitat and macroinvertebrates, but it cannot completely mitigate impacts of urbanization with respect to stream biotic integrity (*Sudduth and Meyer* 2006). Walters and Post (2011) and Brooks *et al.* (2011) found impacts to benthic macroinvertebrates due to upstream water abstractions, including reductions in total biomass of insects and reductions in abundance respectively.

2.9 Conclusions

Alterations in streamflow and sediment transport as a result of land use change can have severe impacts on streams. Common responses include changes in water balance, surface and near-surface runoff timing and magnitude, groundwater recharge, sediment delivery and transport, channel enlargement, widespread incision, and habitat degradation. The extent and consequences of these impacts depend on stream type, watershed context, and local controls on channel adjustment; as such, stream responses to hydromodification are complex and difficult to predict with any precision. Due to the direct impacts of streamflow modification on vegetation and biota, channel morphology cannot be the sole measure of hydromodification impacts. Thus, mitigation efforts that are narrowly focused on channel stability may be insufficient for sustaining key ecological attributes. Likewise, reach-scale stabilization of streams will not necessarily result in the return of comparable habitat quality and complexity (Henshaw and Booth 2000, Roesner and Bledsoe 2003). Hydromodification management should be considered in the context of an overall watershed-scale strategy that targets maintenance and restoration of critical processes in critical locations in the watershed. Furthermore, it is imperative that monitoring and adaptive management be focused on achieving desired objectives for aquatic life and overall stream “health” in addition to simply measures of geomorphic response.

3. Framework for Hydromodification Management

3.1 Introduction and Overview

The current approach to managing hydromodification impacts on a project-by-project basis is not sufficient to protect beneficial uses of streams. This section outlines a comprehensive, alternative framework that begins with watershed analysis and uses the results to guide the site-based management decisions that are the current focus of most hydromodification management strategies. It also recommends the implementation of a compensatory mitigation program in support of hydromodification management objectives identified in the watershed analysis. Figure 3-1 summarizes this approach and illustrates how current site-based management relates to the larger framework.

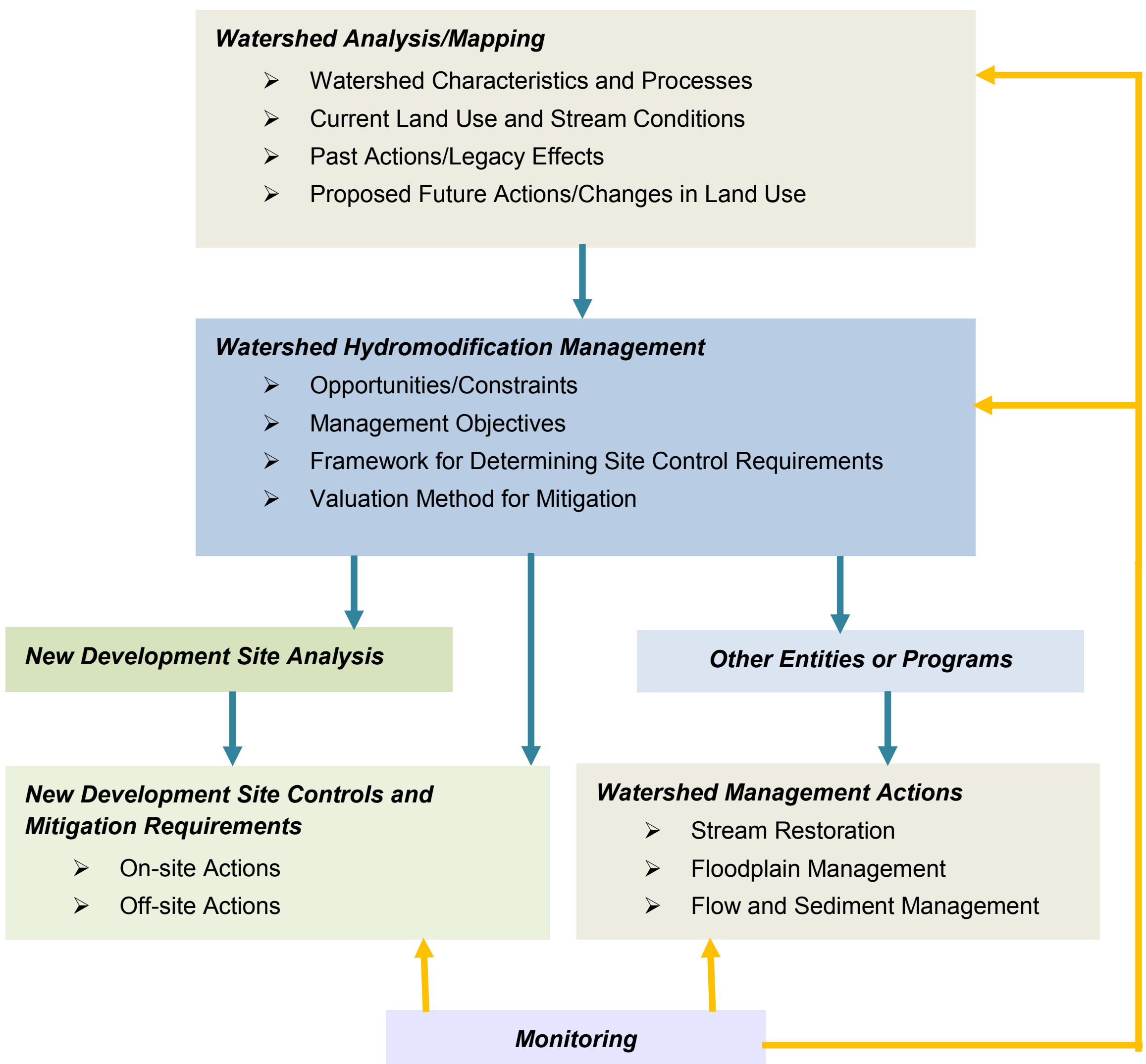


Figure 3-1. Framework for Integrated Hydromodification Management.

This section discusses the details of the integrated framework proposed in Figure 3-1. Key features of this comprehensive approach to hydromodification management are:

- Hydromodification management needs to occur primarily at the watershed scale. The foundation of any hydromodification management approach should be an analysis of existing and proposed future land use and stream conditions that identifies the relative risks, opportunities, and constraints of various portions of the watershed. Site-based control measures should be determined in the context of this analysis.
- Clear objectives should be established to guide management actions. These objectives should articulate desired and reasonable physical and biological conditions for various reaches or portions of the watershed. Management strategies should be customized based on consideration of current and expected future channel and watershed conditions. A one-size-fits-all approach should be avoided.
- An effective management program will likely include combinations of on-site measures (e.g., low-impact development techniques), in-stream measures (e.g., stream habitat restoration), and off-site measures. Off-site measures may include compensatory mitigation measures at upstream locations that are designed to help restore and manage flow and sediment yield in the watershed.
- Management measures should be informed and adapted based on monitoring data. Similarly, monitoring programs should be designed to answer questions and test hypotheses that are implicit in the choice of management measures, such that measures that prove effective can be emphasized in the future (and those that prove ineffective can be abandoned).
- Hydromodification potentially affects all downstream receiving waters; therefore, there generally should be no areas exempted from hydromodification management plans. However, the variety of types and conditions of receiving waters should result in a range of requirements. This also means that objectives, and the management strategies employed to reach them, will need to acknowledge pre-existing impacts associated with historical land uses.

A watershed-based approach to hydromodification management will allow integration of objectives with related programs such as water quality management, groundwater management, and habitat management and restoration through mechanisms such as Integrated Regional Water Resources Management Plans.

Implementation of this approach will likely require changes in the current administration of hydromodification management plans statewide, both in the development and promulgation of regulations by the State and Regional Water Boards and in the administration and execution of those regulations by local jurisdictions (Table 3-1). In the short term, municipalities will need to broaden the approaches to on-site management measures and expand monitoring and adaptive management programs based on the tools described in this document. In the long term, regulatory agencies will need to develop watershed-based programs that allow for implementation of management measures in the locations and manner that will have the greatest impact on controlling hydromodification effects. A watershed-based approach will also allow the integration of hydromodification management objectives with related programs such as water quality management, groundwater management, and habitat management and restoration through mechanisms such as Integrated Regional Water Resources Management Plans.

Table 3-1. Recommendations for implementation of watershed-based hydromodification management, organized by the scale of implementation and the time frame in which useful results should be anticipated.

Time Frame	Programmatic: State and Regional Water Boards	Local: City and County Jurisdictions
Short-term (<10 years)	• Define the watershed context for local monitoring (at coarse scale) • Evaluate whether permit requirements are making positive improvements	• Evaluate whether specific projects/ regulations are meeting objectives • Identify the highest priority action(s) to take
Long-term (1+ decades)	• Define watershed context and setting benchmarks for local-scale monitoring (i.e., greater precision, if/as needed) • Demonstrate how permit requirements can improve receiving-water "health," state-wide (and change those requirements, as needed)	• Evaluate and demonstrate whether actions (on-site, instream, and watershed scale) are improving receiving-water conditions • Assess program cost-effectiveness • Identify any critical areas for resource protection

3.2 Background on Existing Strategies and Why They are Insufficient

Current hydromodification approaches and strategies, such as flow and sediment-control basins, have been long-recognized as insufficient to fully address hydromodification impacts (e.g., Booth and Jackson 1997, Maxted and Horner 1999). Present understanding of the causes and effects of urbanization suggest that such approaches must be expanded to include integrated flow and sediment management at the watershed scale, along with stream corridor/floodplain restoration (NRC 2009).

Flow management has its origins in flood-control basins intended to reduce peak discharge through stormwater detention (Dunne and Leopold 1978). A key shortcoming of these approaches for hydromodification management is that they do not address (and may exacerbate) cumulative erosive forces on the receiving channel because they trap sediment and release sediment-starved water to downstream areas. Simple detention can increase the frequency and duration with which channels are exposed to erosive effects (McCuen and Moglen 1988, Bledsoe *et al.* 2007), resulting in an increase in the downstream impacts of hydromodification.

Since the late 1980's in parts of the US, hydromodification management plans began to explore "flow-duration" control standards as a way to address this shortcoming. These standards require that the post-project discharge rates *and durations* may not deviate above the pre-project discharge rates and durations by more than a specific (and typically quite small) percent, across a broad range of discharges at and above the presumed threshold of instream erosion and sediment transport, as averaged over a multi-year period of measured (or simulated) record. This approach is a dramatic improvement over earlier methods, although it does not adequately address the issues of sediment deficit associated with urbanization (Chin 2006). In addition, current flow-duration standards do not fully account for the effects of flow alteration on in-stream habitat and biological functions (e.g., they do not address the seasonality of peak flows, rates of hydrograph rise and recession, low-flow magnitude and duration) and therefore may not be protective of all beneficial uses of downstream waterbodies.

Current strategies are also insufficient with respect to how municipal stormwater permits apply hydromodification standards. Currently, development triggers are established to determine if a project is subject to the standards. These triggers are generally specified by either project land use type in conjunction with size, or by project size alone (e.g., 20 units or more of single family residential housing, or 10,000 square feet or more of new impervious area). The exemption of many small projects from hydromodification controls can result in cumulative impacts to downstream waterbodies (see Booth and Jackson, 1997, for an example from western Washington of the cumulative effects of a small-project exemption); a move to include LID requirements that apply to all projects, regardless of size, is a positive development to begin to address this issue. There is usually also an exemption for projects discharging to hardened channels or waterbodies; however these exemptions may not be supportive of future stream restoration possibilities, and do not address the impacts of hydromodification on lentic and coastal waterbodies (as yet not fully understood). A further limitation of the current permit structure is that there is no consideration of project characteristics such as position within the watershed, sensitivity of the receiving stream reach, or level of coarse sediment production on the proposed project site. Finally, current programs rely solely on regulating new development and re-development to prevent hydromodification impacts without addressing pre-existing conditions which may limit the effectiveness of future management actions.

Shortcoming of current hydromodification standards that may limit their effectiveness include the exemption of many small projects, which can result in cumulative impacts to downstream waterbodies, and the reliance solely on regulating new development and re-development without addressing pre-existing conditions which may limit the effectiveness of future management actions.

When flow-control measures of whatever regulatory standard have failed to protect streams from erosion, hydromodification “management” typically consists of bank or channel armoring, drop structures, and other hard engineering approaches. Although these methods may reduce local hydromodification impacts, it is typically at the expense of other in-stream or riparian functions or beneficial uses. For example, channel armoring can reduce habitat and water conservation functions and services by direct habitat removal, increased bed scour, and decreased connectivity between the channel and its floodplain. In addition to loss of biological and physical stream function, many armoring solutions degrade or fail over time because they address only the localized channel instability rather than the overarching processes that led to the instability (Kondolf and Piegay 2004). For example, drop structures constructed to stabilize a specific channel reach will tend to shift downstream the consequences of an insufficient sediment load—the reach immediately upstream of the drop structure is "protected," but that immediately downstream is degraded even more severely. In extreme cases, the structure itself can be undermined by downstream erosion and headcutting that is exacerbated by the sudden shift in velocity and associated eddy effects (i.e., hydraulic jump) that often occurs downstream of grade stabilization (Chin 2006). Bank armoring can also fail due to being undermined by erosion at the toe of slope, which can lead to scour (Figure 3-2). In both cases, structural failures often lead to a sequence of incremental increases in the size and extent of the structural solution in an attempt to continually repair increasing channel degradation. In extreme cases, catastrophic failure of bank or grade stabilization can lead to sudden and dramatic changes in channel form, which can be associated with devastating loss of habitat, infrastructure, and property.

Figure 3-2. Undermining of grade control and erosion of banks downstream of structures intended to stabilize a particular stream reach. Left photo is looking upstream at drop structure; right photo is looking downstream from the drop structure.

3.3 Development of Comprehensive Hydromodification Management Approaches

The goal of hydromodification management should be to protect and restore overall receiving water conditions, by maintaining or reestablishing the watershed processes that support those conditions, in the face of urbanization. Achieving these goals will require that hydromodification management strategies operate across programs beyond those typically regulated by NPDES/MS4 requirements. Successful strategies will need to be developed, coordinated, and implemented through land-use planning, non-point source runoff control, and Section 401 Water Quality Certifications and Waste Discharge Requirement programs in addition to traditional stormwater management programs. Thus, all levels of the regulatory framework—federal, state, and local—will need to participate in developing such a program, with program development occurring mainly through regulatory and resource protection agencies and program implementation occurring mainly through local jurisdictions.

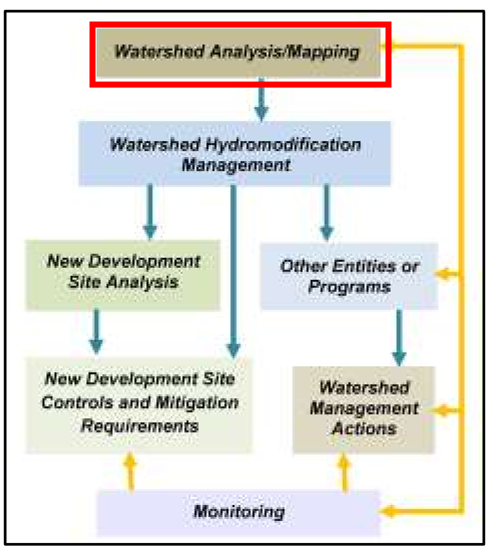
As shown in Figure 3-1, watershed-scale hydromodification management should include all of the following key elements:

- Watershed-wide assessment of the condition of key watershed processes, to understand the natural functioning of the watershed and what has been (or is at risk of being) altered by urbanization.
- Watershed-wide assessment of hydromodification risk, to categorize areas based on the likelihood of hydromodification impacts and to identify opportunities for restoration or protection of key reaches or sub-basins.

The goal of hydromodification management should be to protect and restore overall receiving water conditions, by maintaining or reestablishing the watershed processes that support those conditions, in the face of urbanization.

- Appropriate management objectives for various stream reaches and/or portions of the watershed.
- Process for selecting management actions and mitigation measures for project sites and stream reaches.
- Monitoring program that is consistent with the goals of the HMP so that information generated can be used to improve the HMP over time.

3.4 Watershed Mapping and Analysis – Identification of Opportunities and Constraints



Watershed analysis should be the foundation of all hydromodification management plans. Analysis should identify the nature and distribution of key watershed processes, existing opportunities and constraints in order to help prioritize areas of greater vs. lesser concern, areas.

"Watershed analysis" has several steps, of which the first is mapping. Mapping may occur at the watershed or regional (i.e., multiple watersheds) scale. Mapping should include data layers to facilitate the following analyses. Most of these data layers are freely available as online. Further information on analysis tools is provided in the next section. These maps should be designed for iterative updates over time as new information becomes available:

- Dominant watershed processes – analysis of topography (10-m digital elevation model), hydrology, climate patterns, soil type (NRCS soil classifications) and surficial geology can be used to identify the location and type of dominant watershed processes, such as sediment source areas and areas where infiltration is important or where overland flow likely dominates. This can provide a template for the eventual design of management measures that correspond most closely to the pre-development conditions, which support processes that promote long-term channel health. The Central Coast Hydromodification Control Program (the "Joint Effort"; see Booth *et al.* 2011) provides an example of this type of analysis.

- Existing stream conditions – At a minimum the National Hydrography Database (NHD) can provide maps of streams and lakes in the watershed. Additional information on stream condition should be included to the extent that it is available. This could include major bed material composition, channel planform, grade control locations and condition, and approximate channel evolution stage. These maps can also be used to conduct general stream power evaluations.

- Current (Past) and anticipated future land use - Current land use and land cover plus proposed changes due to general or specific plans. Historical information on past land use practices or stream conditions should be included if it is readily available. Classified land cover (NLCD 2006) is available from the Multi-Resolution Land Characteristics Consortium (MRLC).

- Potential coarse and fine sediment yield areas – methods such as the Geomorphic Land Use (GLU) approach (Booth *et al.* 2010) can be used that to estimate potential sediment yield areas based on geology, slope and land cover.

- Existing flood control infrastructure and channel structures – maps should include major channels, constrictions, grade control, etc. that affect water and sediment movement through the watershed. Any available information on water quality, flood control or hydromodification management basins should also be included.

- Habitat – both upland and in-stream and riparian habitat should be mapped to help determine areas of focus for both resource protection and restoration. This may be based on readily available maps such as the National Wetlands Inventory and National Land Cover Database, aerial photo interpretation, or detailed local mapping.

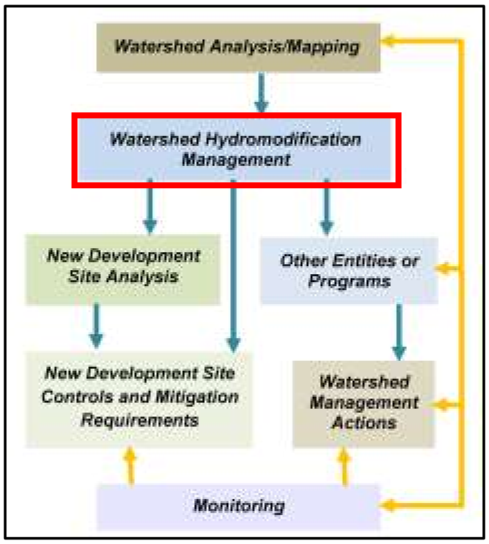
- Areas of Particular Management Concern – these may include sensitive biological resources, critical infrastructure, 303(d) listed waterbodies, priority restoration areas or other locations or portions of the watershed that have particular management needs.

- Economic and social opportunities and constraints – comprehensive watershed management includes consideration of opportunities for improving community amenities associated with streams, economic redevelopment zones, etc. Details on this are beyond the scope of this paper, but emphasize the need to include planning agencies in the development of hydromodification management plans.

Substantial resources will be necessary to implement a watershed analysis approach; therefore, opportunities for joint funding and leveraging of resources should be vigorously pursued.

Watershed analysis will be challenging especially for smaller municipalities with limited resources or where their jurisdiction only encompasses a portion of the watershed. Substantial resources will be necessary to implement this approach; therefore, opportunities for joint funding and leveraging of resource should be vigorously pursued. A cooperative approach should replace the current fragmented efforts among regions and jurisdictions. Furthermore, the State and Regional Water Boards should support completion of these maps and common technical tools as the foundation for future hydromodification management actions.

3.5 Defining Management Objectives



Results of the watershed analysis should be used to determine the most appropriate management actions for specific portions of the watershed. Management strategies should be tailored to meet the objectives, desired future conditions, and constraints of the specific channel reach being addressed.

Decisions should be based on considerations of areas suitable for specific ecosystem services, opportunities, and constraints as described above. Management objectives may be aimed at reducing effects of proposed future land use or mitigating for the effects of past land use, and they may apply to stream reaches or upland areas. Potential management objectives for specific stream reaches may include: protect, restore, or manage as a new channel form.

The specific manifestation of each of these strategies will differ by location, based on constraints of the stream, watershed plan objectives, etc. Decisions about appropriate objectives will need to consider current and future opportunities and constraints in upland, floodplain, and in-stream portions of the watershed. General definitions are provided below as a starting point for case-specific refinement.

Management strategies should be tailored to meet the objectives, desired future conditions, and constraints of the specific channel reach being addressed. Objectives for specific stream reaches may include:

- **Protect**
- **Restore**
- **Manage as a new channel form**

3.5.1 Protect

This approach consists of protecting the functions and services of relatively unimpacted streams in their current form through conservation and anti-degradation programs. This strategy should not be used if streams are degraded, or nearing thresholds of planform adjustment or changes in vegetation community. This strategy may apply following natural disturbances such as floods depending on the condition of the stream reach and the ability for natural rehabilitation to occur (due to how intact watershed processes are). The goal of this strategy is not to create an artificial preserve (such as a created stream running through an urban park) but rather a naturally function river system. Fully channelized systems are not considered in this framework. Examples of specific actions include:

- Preserving intact channel systems through easements, restrictions, covenants, etc. This should be considered in the watershed context to ensure adequate connectivity with upstream and downstream reaches of similar condition, and to ensure that the watershed processes responsible for creating and maintaining instream conditions will persist.
- Providing appropriate space for channel processes to occur (e.g., floodplain connectivity).
- Establishing transitional riparian and upland buffer zones that are protected from encroachment by infrastructure or development.

3.5.2 Restore

There are many definitions of "restoration". For the purposes of this document, restoration is considered re-establishing the natural processes and characteristics of a stream. The process involves converting an unstable, altered, or degraded stream corridor, including adjacent riparian zone (buffers), uplands, and flood-prone areas, to a natural condition. In most cases, restoration plans should be based on a consideration of watershed processes and their ability to support a desired stream type. The watershed analysis discussed above should be used to determine how and where watershed process should be protected or restored in order to best support stream and stream-corridor restoration. This process should be based on a reference condition/reach for the valley type and includes restoring the appropriate geomorphic dimension (cross-section), pattern (sinuosity), and profile (channel slopes), as well as reestablishing the biological and chemical integrity, including physical processes such as transport of the water and sediment produced by the stream's watershed in order to achieve dynamic equilibrium. Design of restoration structural elements must be based on existing and anticipated upstream land uses, and reflect the modified hydrology resulting from these uses. Restoration should apply to streams that are already on a degradation trajectory where there is a reasonable expectation that a more stable equilibrium condition that reflects previously existing conditions can be recreated and maintained via some intervention. Creating a stream system that differs from "natural conditions" is not considered restoration. All elements of the "protection" strategy should also be included once the restoration actions are complete. Examples of specific actions include:

- Floodplain and in-stream measures that restore natural channel form consistent with current and/or anticipated hydrology and sediment yield. Examples include recontouring, biotechnical slope stabilization, soft-grade control features (e.g., woody debris).
- Revegetation of stream banks and beds, including removal of invasive species.
- Preserving intact channel systems through easements, restrictions, covenants, etc. This should be considered in the watershed context to ensure adequate connectivity with upstream and downstream reaches of similar pristine condition.

- Providing appropriate space for channel processes to occur (e.g. channel migration at allowable levels, floodplain connectivity, and development of self-sustaining riparian vegetation).
- Establishing transitional riparian and upland buffer zones that are protected from encroachment by infrastructure or development.

3.5.3 Manage as New Channel Form

Once a stream channel devolves far enough down the channel evolution sequence, it is extremely difficult to recover and restore without substantial investment of resources. If critical thresholds in key structural elements, such as planform or bank height, are surpassed, streams should be allowed to continue progressing toward a new stable equilibrium condition that is consistent with the current setting and watershed forcing functions, if such progress does not pose a danger to property and infrastructure. Substantial alteration of flow or sediment discharge, slope or floodplain width may make it improbable that a stream can be restored to its previous condition. In such circumstances, it may be preferable to determine appropriate channel form given expected future conditions and "recreate" a new channel to match the appropriate equilibrium state under future conditions. For example, a multi-thread braided system may not be the appropriate planform based on new runoff and sediment pattern; instead, a single-thread channel or step-pool structure may be a more appropriate target. Examples of specific actions include:

- In-channel recontouring or reconstruction of channel form.
- Floodplain recontouring or reconstruction that improves connectivity with the channel.
- In extreme circumstances based on channel condition, position in the watershed, etc. this may involve hardening portions of the channel and focusing "mitigation" measures at off-site measures at a different part of the watershed. Off-site mitigation can be informed by "hydromodification risk mapping".
- Re-establishing longitudinal connectivity for sediment transport and ecological linkages.
- Preserving intact channel systems through easements, restrictions, covenants, etc. This should be considered in the watershed context to ensure adequate connectivity with upstream and downstream reaches of similar pristine condition.
- Providing appropriate space for channel processes to occur (e.g. floodplain connectivity).
- Establishing transitional riparian and upland buffer zones that are protected from encroachment by infrastructure or development.

Several authors have previously noted that in urban systems, natural channel state often can no longer be sustained under changed hydrological conditions. Thus, different management goals are probably appropriate for watersheds at varying stages of development (Booth, 2005) and at varying degrees of adjustment (Chin and Gregory 2005). In this context, identifying which channels are suitable for protection, restoration, or alternative channel form can be used to guide restoration and management efforts (Booth *et al.* 2004).

Upland objectives should be established to support management objectives for stream reaches. These objectives will have direct implications and will influence site-specific control requirements (discussed below). Potential management objectives for upland areas may include:

- *Conserve open space for infiltration*: Infiltration reduces the magnitude and duration of runoff to the stream channel and allows flow to re-enter the stream through diffuse overland flow, shallow subsurface flow, or groundwater recharge. This in turn reduces the work (energy) on the channel bed and banks and helps promote stability.
- *Conserve open space for stream buffers*: Buffers allow many of the same infiltration processes discussed above to occur. In addition, they provide space for channel migration and overbank flow, both of which function to reduce energy and allow the channel to better withstand potentially erosive forces associated with high flow events.
- *Conserve open space for coarse sediment production*: Course sediment functions to naturally armor the stream bed and reduce the erosive forces associated with high flows. Absence of coarse sediment often results in erosion of in-channel substrate during high flows. In addition, coarse sediment contributes to formation of in-channel habitats necessary to support native flora and fauna.
- *Encourage development on poorly-infiltrating soils*: The difference between pre and post development runoff patterns is less when development occurs on soils that have low infiltration rates and functioned somewhat like paved surfaces. Focusing development on these areas reduces changes in hydrology associated with transition to developed land uses.
- *Encourage urban infill*: Urban infill reduces the effect on watershed processes by concentrating development on previously impacted areas. This reduces disruption of hydrology and sediment process compared to developing on open space or other natural areas.

3.6 Selecting Appropriate Management Objectives

The combination of expected force acting on the stream channel (in terms of higher flow and less sediment) and estimated resistance (in the form of channel and floodplain condition) can be used to inform selection of an appropriate management objective for a specific stream reach, as shown in Figure 3-3. This figure represents a conceptual approach to selecting appropriate management objectives, in which modifications to runoff and sediment are compared against stream reach conditions. By weighing these factors within the context of watershed opportunities, constraints and resources, management objectives and specific actions can be determined. More complete decision support systems or guidance will need to be developed for individual

Selection of appropriate management objectives should consider changes to runoff and sediment, and existing stream reach conditions, within the context of watershed opportunities, constraints and resources.

hydromodification management plans that account for other considerations such as upstream and downstream conditions, cost, infrastructure constraints, availability of floodplain area for restoration, presence of downstream sensitive resources, etc. All decisions should be made in the context of the watershed position of a project site relative to existing opportunities and constraints as discussed above.

A number of tools are available to be used in conjunction with watershed mapping to inform this prioritization process. For example, GLU mapping (Booth *et al.* 2010) and hydromodification risk mapping can be used to assign high, medium or low ratings to watershed resistance (i.e., susceptibility to change). Similarly, field based tools such as the hydromodification screening tool (Bledsoe *et al.* 2010) or European tools such as Fluvial Audit or River Habitat Survey can be used to assign a rating of high, medium or low at the reach scale. In addition to geomorphic assessments, habitat assessments such as the California Rapid Assessment Method (CRAM; Collins *et al.* 2008) or biological evaluations via an index of biotic integrity (IBI; e.g., Ode *et al.* 2005) should be used as measures of biological condition to provide a more complete stream assessment. The next section provides an overview of hydromodification assessment and prediction tools, as well as further details on specific tools to support the selection of management objectives.

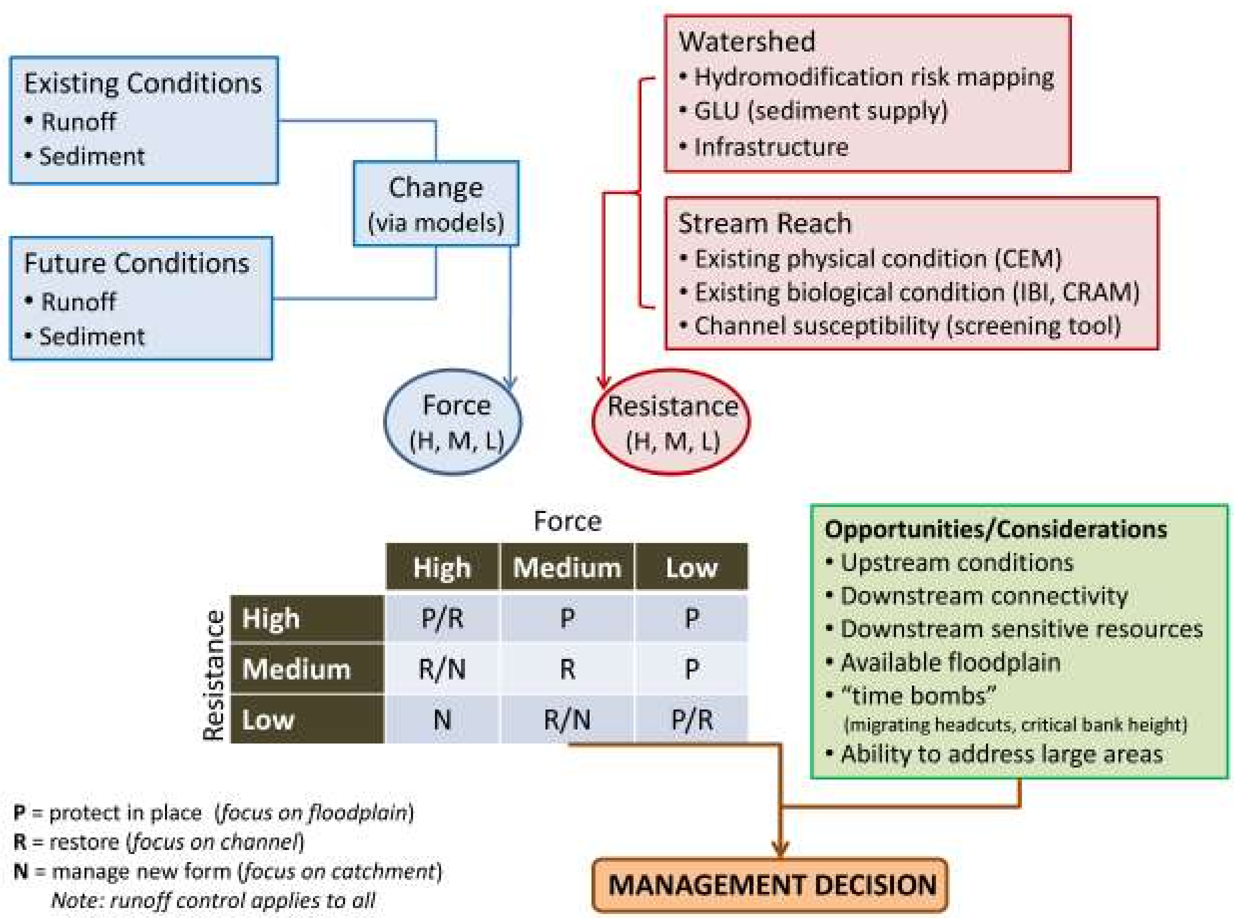
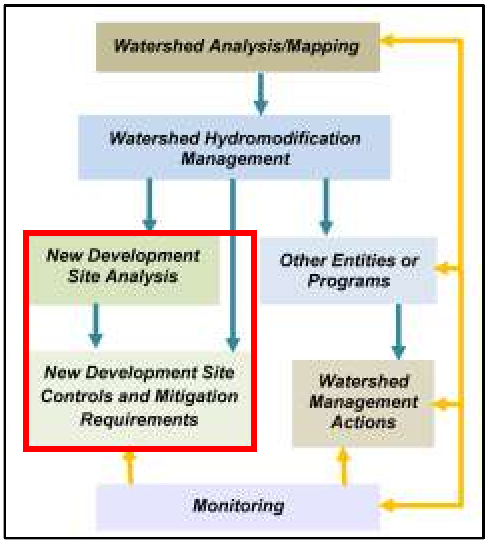


Figure 3-3: Example of a hydromodification management decision-making process.

3.7 Framework for Determining Site-Specific Control Requirements



Once the watershed analysis is complete and opportunities, constraints and management objectives have been identified for both upland areas and stream reaches, a framework should be developed for site-specific project analyses and control requirements. The level of detail required for the analysis of proposed projects should be based on a combination of factors including project size, location within the watershed, and point of discharge to receiving waterbody.

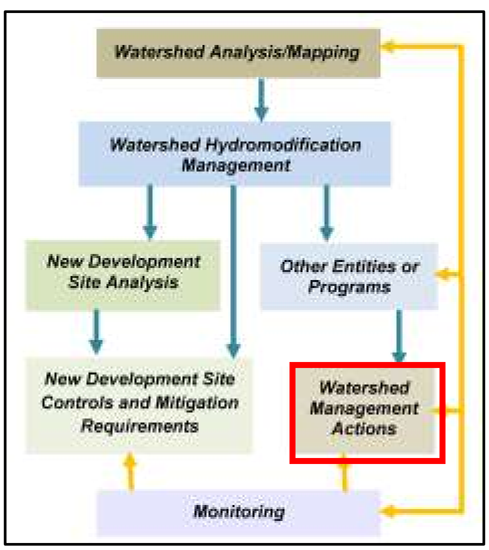
The HMP should specify how these factors will be evaluated within the context of the identified management objectives to determine analysis requirements. The HMP should also ideally contain scalable BMP designs (based on conservative assumptions and consistent with prevailing watershed conditions) that can be applied by small projects where appropriate to avoid overly burdensome requirements for site-specific analysis. The framework should include the following components:

- A set of standard on-site management measures/BMPs that should apply to all projects; no projects should be exempted from these measures as they will have broader water quality benefits beyond helping to control the effects of hydromodification. These management actions consist of reducing the effects of urbanization on catchment runoff and sediment yield. On-site management measures should attempt to reduce excess runoff, maintain coarse sediment yield (if possible) and provide for appropriate discharge to receiving streams to support in-stream biological resources. In some cases, common features or facilities may be able to accommodate these objectives. In other cases, separate features or facilities will be necessary to deal with distinct objectives. On-site measures should ***generally be applied in all cases*** as allowed by site-specific geotechnical constraints, with specific management practices informed by the watershed processes most important at particular locations in the watershed, as well as by the nature of downstream receiving waters:
    - Low impact development (LID) practices.
    - Disconnecting impervious cover through infiltration, interception, and diversion.
    - Coarse sediment bypass through avoidance of sediment yield areas or measures that allow coarse sediment to be discharged to the receiving stream.
    - Flow-duration control basins to reduce runoff below a threshold value.

- Specification of the level of analysis detail and design requirements for the project, depending on project location, discharge point, and project size. Levels of analysis and design requirements may include:
    - Application of scalable, standardized designs for flow control based on site-specific soil type and drainage design. The assumptions used to develop these scalable designs should be conservative, to account for loss of sediment and uncertainties in the analysis and our understanding of stream impacts.
    - Use of an erosion potential metric, based on long-term flow duration analysis and in-stream hydraulic calculations. Guidelines should specify stream reaches where in-stream controls would and would not be allowed to augment on-site flow control.
    - Implementation of more detailed hydraulic modeling for projects of significant size or that discharge to reaches of special concern to understand the interaction of sediment supply and flow changes.
    - Analysis of the water-balance for projects discharging into streams with sensitive habitat. This may include establishment of requirements for matching metrics such as number of days with flow based on the needs of species present.
- Guidelines for prioritization of on-site or regional flow and sediment control facilities. Watershed analysis will help identify opportunities for regional flow or sediment control facilities, which may help to mitigate for existing hydromodification impacts.

Appendix A provides detailed guidance on the appropriate application of tools to meet site control requirements.

3.8 Off-site Compensatory Mitigation Measures



In some cases, on-site control of water and sediment will not be sufficient to offset the effects of hydromodification on receiving waters. In these cases, off-site compensatory mitigation measures will be necessary (similar to the concepts used in the Section 401/404 permitting programs). Off-site measures could be implemented by project proponents or through the use of regional mitigation banks or in-lieu fee programs.

Off-site mitigation may be necessary for several reasons:

- Off-site measures may be more effective at addressing effects or at achieving desired management goals. This may be particularly true for sites near the bottom of a watershed where upstream measures may be preferred

- Off-site measures may be necessary to supply compensation for residual project impacts where on-site measures are limited by site constraints or solutions are beyond the scope of what can be accomplished on an individual site.
- Off-site measures may be necessary where accomplishing specified management objectives is not practical using on-site measures alone. Off-site measures may be desired to remedy legacy effects of prior land use or to achieve desired beneficial uses.

Performance monitoring and adaptive management must be a part of compensatory mitigation given its inherent uncertainty.

In cases where on-site control of water and sediment will not be sufficient to offset the effects of hydromodification on receiving waters, off-site compensatory mitigation measures will be necessary. Implementation of this approach will require that the State Water Board develop a valuation method to help determine appropriate off-site mitigation requirements in light of the anticipated impacts of hydromodification on receiving streams.

The location and type of mitigation should be determined in the context of the watershed analysis and should account for the size and nature of the impact, location in the watershed, pre-existing conditions in the watershed, and uncertainty associated with the success of the proposed mitigation actions. In some cases these measures may be near the project site (e.g., restoring a stream reach downstream of the project site), but in other cases the off-site mitigation may be in the form of in-lieu fee or "mitigation bank" type contributions to a project located in a different portion of the watershed (e.g. upstream grade control, protection of sediment source areas). Such off-site mitigation relatively far from the site will only be possible if conducted in the context of an overall watershed plan, as discussed above. Off-site measures may include:

- Stream corridor restoration
- Purchase, restoration and protection of floodplain/floodway habitat
- Purchase and/or protection of critical sediment source or transport areas
- Regional basins or other retention facilities
- Upstream or downstream natural/bio-engineered grade control
- Retrofit or repair of currently undersized structures (e.g. culverts, bridge crossings)
- Removal or hydrologically disconnecting impervious surfaces

A valuation method will be necessary for assigning appropriate mitigation requirements in light of the anticipated impacts of hydromodification on receiving streams. The valuation method should be developed by the State Water Board.

To support the management approaches discussed above, HMPs should provide general guidance for application of models and other tools based on the questions being asked and the desired outcomes of the HMP. Models can also be used to help communicate levels of uncertainty in particular management actions and to guide restoration / in-channel management actions. Modeling and other tools are discussed in detail in Section 4 and Appendices A and B.

Management endpoints should articulate the desired physical and biological conditions for various reaches or portions of the watershed. To the extent possible, these desired conditions should be expressed in numeric, quantifiable terms to avoid ambiguity.

Finally, management endpoints should articulate the desired physical and biological conditions for various reaches or portions of the watershed. To the extent possible, these desired conditions should be expressed in numeric, quantifiable terms to avoid ambiguity. Additionally, since regulatory strategies will invariably rely on quantifiable measures to determine whether stormwater management actions achieve these desired conditions, identifying appropriate numeric objectives will support determinations of regulatory compliance. As desired physical and biological watershed conditions are expressed in quantifiable terms to the extent possible, a similar need would apply to site control requirements. Control measures should be linked to, a) a desired condition (or goal), b) the parameter(s) that best define that condition, and c) quantifiable measures that serve to evaluate performance of the control measure. Direct measures (e.g., volume of runoff to be retained) as well as indirect or surrogate measures (IBI scores) are appropriate if they are quantifiable.

4. Overview of Assessment and Prediction Tools

4.1 Introduction

The previous section discussed a number of potential actions for managing hydromodification impacts. These ranged from high-level watershed-scale characterization to the site-specific design of a proposed development. This section provides an overview of the current and emerging assessment and prediction tools available to inform these management actions. An organizing framework helps explain the appropriate application of these tools, as well as their strengths and weaknesses. Specific tools that support the selection of management objectives are also discussed. Examples of “suites” of tools that are commonly used together to predict stream responses and formulate management prescriptions for channels of varying susceptibility are presented in Appendix B. Appendix A provides detailed guidance on the appropriate application of tools to meet site control requirements.

Municipalities are the primary audience for this section, as they select and incorporate these tools into their HMPs. However, the State and Regional Water Boards should be aware of the overall capabilities, appropriate uses, and gaps in our current toolbox. The development of new and improved tools should ideally be coordinated at the State level for optimum cost effectiveness and widest applicability. The table below identifies the key actions necessary at both the programmatic and local level to address the considerations discussed above, within the context of the goals of the framework described in Section 3.

Table 4-1. Recommendations for the application and improvement of tools in support of the proposed management framework.

Time Frame	Programmatic: State and Regional Water Boards	Local: City and County Jurisdictions
Short-term (<10 years)	• Develop quality control and standardization for continuous simulation modeling • Perform additional testing and demonstration of probabilistic modeling for geomorphic response • Pursue development of biologically- and physically-based compliance endpoints	• Work cooperatively with adjacent jurisdictions to implement hydromodification risk mapping at the watershed scale • Implement continuous simulation modeling for project impact analysis
Long-term (1+ decades)	• Improve tools for sediment analysis and develop tools for sediment mitigation design • Develop tools for biological response prediction • Improve tools for geomorphic response prediction	• Expand use of probabilistic and statistical modeling for geomorphic response • Apply biological tools for predicting and evaluating waterbody condition

4.2 Background

In the context of hydromodification, tools and models are typically used to help answer one or more of the following questions involving an assessment of natural and human influences at various spatial and temporal scales:

- How does the stream work in its watershed context?
- Where is the stream going? For example, have past human actions induced channel changes? What are the effects on sediment transport and channel form? What is the magnitude of current and potential channel incision following land use conversion?
- How will the stream likely respond to alterations in runoff and sediment supply?
- How can we manage hydromodification and simultaneously improve the state of the stream?

Previous sections have underscored the variability and complexity of relationships among land use, the hydrologic cycle, and the physical and ecological conditions of stream systems. It follows that the process of assessing stream condition and predicting future conditions is highly challenging and subject to uncertainty. Therefore it is important to understand the inherent strengths and limitations of the available tools, especially with respect to prediction uncertainty and how it is expressed for various tools. Considerable judgment is needed to choose the appropriate model for the question at hand. In addition to prediction uncertainty, considerations in choosing the right model for a particular application include appropriate spatial and temporal detail, cost of calibration and testing, meaningful outputs, and simplicity in application and understanding (NRC 2001; Reckhow 1999a,b).

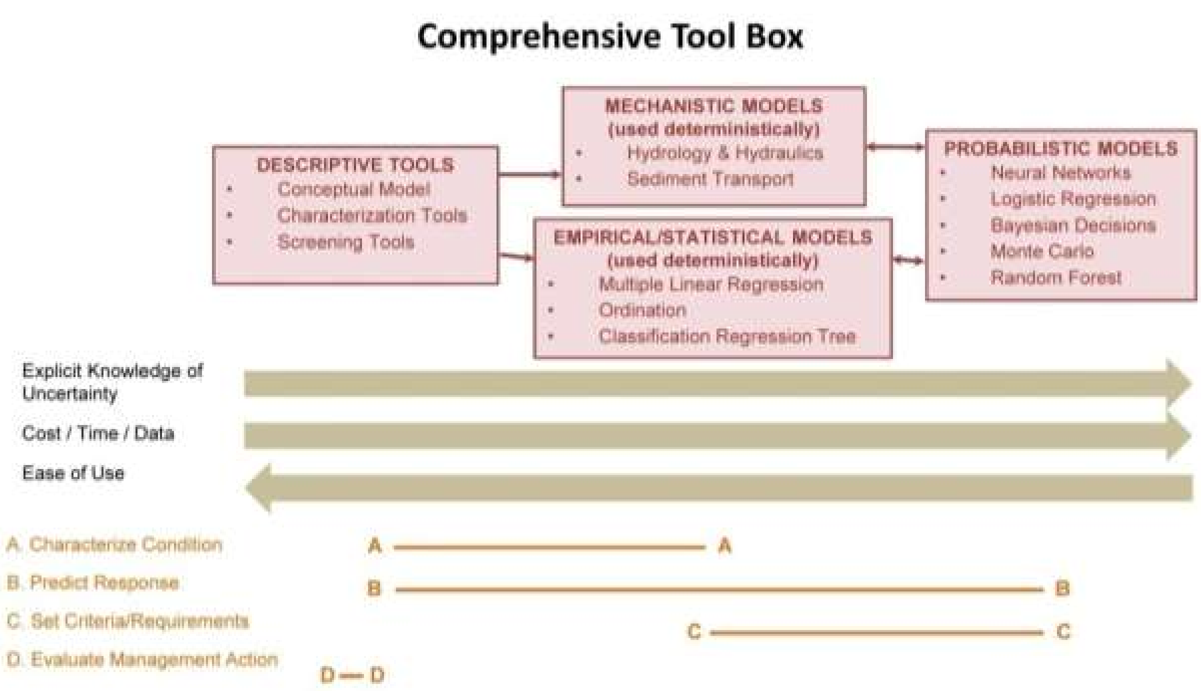


Figure 4-1. Organizing Framework for understanding hydromodification assessment and management tools.

4.3 Organizing Framework

Figure 4-1 presents an organizing framework by which to understand the available tools that may be applied in support of hydromodification management and policy development. Tools fall into three major categories: descriptive tools, mechanistic and empirical/statistical models that are used deterministically, and probabilistic models/predictive assessments with explicitly quantified uncertainty. The organizing framework relates these categories to the types of question the tools are designed to answer, specifically: characterization of stream condition, prediction of response, establishment of criteria/requirements, or evaluation of management actions. The framework also characterizes the tools according to the following features: intensity of resource requirements (i.e., data, time, cost), and the extent to which uncertainty is explicitly addressed. Subsequent sections of this section discuss each of the three major categories in turn, highlighting examples of specific tools within each category.

Given the uncertainty associated with predicting hydromodification impacts, probabilistic models should be incorporated into analysis and design, particularly where resource values or potential consequences of impacts are high.

Tools required to support the management framework presented in Section 3 include watershed characterization and analysis tools and project analysis and design tools. The level of resolution that is required will depend on the point in the planning process. At early stages, descriptive tools will be sufficient, but more precise tools will be required toward the design phase. Currently, most projects rely solely on deterministic models. However, given the uncertainty associated with predicting hydromodification impacts, probabilistic models should be incorporated into analysis and design, particularly where resource values or potential consequences of impacts are high.

4.3.1 Descriptive Tools

Descriptive tools include conceptual models, screening tools, and characterization tools. These tools are used to answer the question: *What is the existing condition of a stream or watershed?* Although descriptive tools are not explicitly predictive, they can be used to assess levels of susceptibility to future stressors by correlation with relationships seen elsewhere. The application of some type of descriptive tool, such as a characterization tool, is almost always necessary before applying a deterministic model. In particular, descriptive tools can aid in understanding the key processes and boundary conditions that may need to be represented in more detailed models.

Conceptual Models. A conceptual model, in the context of river systems, is a written description or a simplified visual representation of the system being examined, such as the relationship between physical or ecological entities, or processes, and the stressors to which they may be exposed. Conceptual models have been used to describe processes in a wide range of physical and ecological fields of study, including stream-channel geomorphology (Bledsoe *et al.* 2008). For example, Channel Evolution Models (CEMs) are conceptual models which describe a series of morphological configurations of a channel, either as a longitudinal progression from the upper to the lower watershed, or as a series at a fixed location over time subsequent to a disturbance. The incised channel CEM developed by

Schumm *et al.* (1984) is one of the most widely known conceptual models within fluvial geomorphology. This CEM documents a sequence of five stages of adjustment and ultimate return to quasi-equilibrium that has been observed and validated in many regions and stream types (ASCE 1998, Simon and Rinaldi 2000). The Schumm *et al.* (1984) CEM has been modified for streams characteristic of southern California, including transitions from single-thread to multi-thread and braided evolutionary endpoints (Hawley et al., in press).

Conceptual models also include planform classifications of braided, meandering and straight, and other general geomorphic classifications, which categorize streams by metrics such as slope, sinuosity, width-to-depth ratio, and bed material size. The qualitative response model described by Lane's diagram (1955), and discussed earlier in this report, is also a conceptual model.

***Characterization Tools*.** Examples of characterization tools include baseline geomorphic assessments, river habitat surveys, and fluvial audits. A fluvial audit uses contemporary field survey, historical map and documentary information and scientific literature resources to gain a comprehensive understanding of the river system and its watershed. Fluvial audits, along with watershed baseline surveys are a standardized basis for monitoring change in fluvial systems. These types of comprehensive assessments are comprised of numerous, more detailed field methodologies, such as morphologic surveys, discharge measurements, and estimates of boundary material critical shear strength through measurements of resistance (for cohesive sediments) or size. Baseline assessments may also draw on empirical relationships such as sediment supply estimation models.

***Screening Tools*.** Screening tools can be used to predict the relative severity of morphologic and physical-habitat changes that may occur due to hydromodification, as a critical first step toward tailoring appropriate management strategies and mitigation measures to different geomorphic settings. However, assessing site-specific stream susceptibility to hydromodification is challenging for several reasons, including the existence of geomorphic thresholds and non-linear responses, spatial and temporal variability in channel boundary materials, time lags, historical legacies, and the large number of interrelated variables that can simultaneously respond to hydromodification (Schumm 1991, Trimble 1995, Richards and Lane 1997).

Screening tools can be used to predict the relative severity of morphologic and physical-habitat changes due to hydromodification, as a critical first step toward tailoring appropriate management strategies and mitigation measures to different geomorphic settings.

Despite the foregoing difficulties, the need for practical tools in stream management have prompted many efforts to develop qualitative or semi-quantitative methods for understanding the potential response trajectories of channels based on their current state. For example, predictors of channel planform can be used to identify pattern thresholds and the potential for planform shifts (e.g., van den Berg 1995, Bledsoe and Watson 2001, Kleinhans and van den Berg 2010).

In addition, regional CEMs (discussed above) can partially address the needs of the hydromodification management community by providing a valuable framework for interpreting past and present response trajectories, identifying the relative severity of potential response sequences, applying appropriate models in estimating future channel changes, and developing strategies for mitigating the impacts of processes likely to dominate channel response in the future (Simon 1995).

More recent screening-level tools for assessing channel instability and response potential, especially in the context of managing bridge crossings and other infrastructure, have borrowed elements of the CEM approach and combined various descriptors of channel boundary conditions and resisting vs. erosive forces. For example, Simon and Downs (1995) and Johnson *et al.* (1999) developed rapid assessment techniques for alluvial channels based on diverse combinations of metrics describing bed material, CEM stage, existing bank erosion, vegetative resistance, and other controls on channel response. Although based on a strong conceptual foundation of the underlying mechanisms controlling channel form, these specific examples are either overly qualitative with respect to the key processes, or developed with goals and intended applications (e.g., evaluating potential impacts to existing infrastructure such as bridges or culverts) that differ from what is needed by current hydromodification management programs.

SCCWRP has recently proposed a general framework for developing screening-level tools that help assess channel susceptibility to hydromodification, and a new region-specific tool for rapid, field-based assessments in urbanizing watersheds of southern California (Booth *et al.* 2010, Bledsoe *et al.* 2010). The criteria used to assign susceptibility ratings are designed to be repeatable, transparent, and transferable to a wide variety of geomorphic contexts and stream types. The assessment tool is structured as a decision tree with a transparent, process-based flow of logic that yields four categorical susceptibility ratings through a combination of relatively simple but quantitative input parameters derived from both field and GIS data. The screening rating informs the level of data collection, modeling, and ultimate mitigation efforts that can be expected for a particular stream-segment type and geomorphic setting. The screening tool incorporates various measures of stream bed and bank erodibility, probabilistic thresholds of channel instability and bank failure based on regional field data, integration of rapid field assessments with desktop analyses, and separate ratings for channel susceptibility in vertical and lateral dimensions.

An example of a specific analysis component that predicts changes in post-development sediment delivery, and that can be applied within this screening tool framework, is a GIS-based catchment analyses of “Geomorphic Landscape Units” (GLUs). A GLU analysis integrates readily available data on geology, hillslope, and land cover to generate categories of relative sediment production under a watershed’s current configuration of land use. Those areas subject to future development are identified, and corresponding sediment-production levels are determined by substituting developed land cover for the original categories and reassessing the relative sediment production. The resultant maps can be used to aid in planning decisions by indicating areas where changes in land use will likely have the largest (or smallest) effect on sediment yield to receiving channels.

Effective screening tools for assessing the susceptibility of streams to hydromodification necessarily rely on both field and office-based elements to examine local characteristics within their broader watershed context. Proactive mapping of flow energy measures (e.g., specific stream power) throughout drainage networks has the potential to complement field-based assessments in identifying hotspots for channel instability and sediment discontinuities as streamflows change with land use. Such analyses may partially guide subsequent field reconnaissance; however, this approach also has limitations in that some geomorphic settings are inherently difficult to map using widely available digital elevation data. In particular, maps of stream power in narrow entrenched valleys and low gradient valleys (ca. <1%) with sinuous channels should be carefully field-truthed and used with a level of caution commensurate with the accuracy of the input data.

Moreover, spatial variability in channel boundary materials and form cannot be accurately mapped at present using remotely sensed data. Thus, boundary materials and channel width are typically assumed in watershed-scale mapping efforts, thereby introducing potential inaccuracies. Coupling desktop analysis with a field-based assessment when using such an approach can help resolve variation in site-specific features such as the erodibility of bed and bank materials, channel width, entrenchment, grade control features, and proximity to geomorphic thresholds.

4.3.2 Mechanistic and Empirical/Statistical Models with Deterministic Outputs

Mechanistic/deterministic models are simplified mathematical representations of a system based on physical laws and relationships (*link to next*). Empirical/statistical models use observed input and output data to develop relationships among independent and dependent variables. Statistical analyses determine the extent to which variation in output can be explained by input variables. Both types of models are typically used to generate a single output or answer for a given set of inputs. These tools can be used to help answer such questions as: *What are the expected responses in the stream and watershed given some future conditions? What criteria should be set to prevent future hydromodification impacts?* However, hydromodification modeling embodies substantial uncertainties in terms of both the forcing processes and the stream response. Deterministic representations of processes and responses can therefore mask uncertainties and be misleadingly precise, unless prediction uncertainty is explicitly characterized as described later in this section.

Although valuable, deterministic representations (such as those derived from continuous simulation modeling) of processes and responses can mask uncertainties and be misleadingly precise unless prediction uncertainty is explicitly characterized.

Hydrologic Models are used to simulate watershed hydrologic processes, including runoff and infiltration, using precipitation and other climate variables as inputs. Some models, such as the commonly-used HEC-HMS, can be run for either single-event simulations or in a continuous-simulation mode which tracks soil moisture over months or years. Other hydrologic models that are commonly used for event-based and continuous simulation modeling include HSPF and SWMM. It is widely accepted that continuous simulation modeling, rather than event-based modeling, is required to assess long term changes in geomorphically-significant flow events (Booth and Jackson 1997; Roesner *et al.* 2001).

Several HSPF-based continuous simulation models have been developed specifically for use in hydromodification planning. These include the Western Washington Hydrology Model (WWHM) and the Bay Area Hydrology Model (BAHM). Hydromodification Management Plans (HMPs) in Contra Costa County, San Diego County and Sacramento County have developed sizing calculators for BMPs based on modeling done using HSPF models. To illustrate the point about uncertainly in mechanistic models, HSPF contains approximately 80 parameters, only about 8 of which are commonly adjusted as part of the calibration process.

Hydraulic Models are used to simulate water-surface profiles, shear stresses, stream power values and other hydraulic characteristics generated by stream flow, using a geometric representation of channel segments. The industry standard hydraulic model is the HEC River Analysis System (HEC-RAS).

Coupled Hydrologic and Hydraulic Models represent a valuable tool in hydromodification management. Because the streamflow regime interacts with its geomorphic context to control physical habitat dynamics and biotic organization, it is often necessary to translate discharge characteristics into hydraulic variables that provide a more accurate physical description of the controls on channel erosion potential, habitat disturbance, and biological response. For example, a sustained discharge of 100 cfs could potentially result in significant incision in a small sand bed channel but have no appreciable effect on the form of a larger channel with a cobble bed. By converting a discharge value into a hydraulic variable (common choices are shear stress, or stream power per unit area of channel relative to bed sediment size), a "common currency" for managing erosion and associated effects can be established and applied across many streams in a region. Such a common currency can improve predictive accuracy across a range of stream types. As opposed to focusing on the shear stress or stream power characteristics of a single discharge, it is usually necessary to integrate the effects of hydromodification on such hydraulic variables over long simulated periods of time (on the order of decades) to fully assess the potential for stream channel changes. By using channel morphology to estimate hydraulic variables across a range of discharges, models like HEC-RAS provides a means of translating hydrologic outputs from continuous simulations in HEC-HMS, SWMM, or HSPF into distributions of shear stress and stream power across the full spectrum of flows.

Sediment Transport Models such as HEC-6T, the sediment transport module in HEC-RAS, CONCEPTS, MIKE 11 and FLUVIAL12, use sediment transport and supply relationships to simulate potential changes in channel morphology (mobile boundary) resulting from imbalances in sediment continuity. This means that hydraulic characteristics are calculated as channel form and cross-section evolve through erosion and deposition over time. Such models have high mechanistic detail but are often difficult to apply effectively. Although it is not a mobile boundary model, the SIAM (Sediment Impact Analysis Method) module in HEC-RAS represents an intermediate complexity model designed to predict sediment imbalances at the stream network scale and to describe likely zones of aggradation and degradation.

Statistical Models use descriptive tools and empirical data to develop relationships that quantify the risk of specific stream behaviors. For example, Hawley (2009) developed a statistical model to explain variance in channel enlargement based on measures of erosive energy and channel features such grade control and median bed sediment size. Such models often include independent variables based on input from the mechanistic models described above; however, a key difference is that statistical models do not explicitly represent actual physical processes in their mathematical structure. Instead, these models simply express the observed correlations between dependent and independent variables. Like mechanistic models, the output from these models is commonly treated as precise results in management decisions, despite the fact that predictions from most statistical models could be readily (and more accurately) expressed in terms of confidence intervals with a range of uncertainty.

Probabilistic/Risk-based Models integrate many of the tools discussed above, using modeled changes in hydrology as input to hydraulic models, which in turn provide input to various types of statistical models to predict response. However, the predictions are not represented as deterministic outputs, instead, the range of (un)certainty in the likelihood of the predicted response is explicitly quantified. Although not commonly used for hydromodification management at this time, there are well established models based on these principals currently in use in other scientific disciplines. An example of a probabilistic approach that has been used for hydromodification management is a logistic regression analysis that was used to produce a threshold "erosion potential metric" that can be used to quantify the risk of a degraded channel state. More details on this approach are provided in Appendix B.

Risk-based modeling in urbanizing streams provides a more scientifically defensible alternative to standardization of stormwater controls across stream types, and can inform management decisions about acceptable levels of risk.

Risk-based modeling in urbanizing streams provides a more scientifically defensible alternative to standardization of stormwater controls across stream types. A probabilistic representation of possible outcomes also improves understanding of the uncertainty that is inherent in model predictions, and can inform management decisions about acceptable levels of risk.

Predictive Tools for Habitat Quality and Stream Biota. The tools discussed above focus on physical stream impacts; however, as discussed in the preceding chapter, it is recognized that maintenance of stream "stability" does not necessarily conserve habitat quality and biological potential. In general, the knowledge base for biota/habitat associations is not generally adequate to allow for prediction of how whole communities will change in response to environmental alterations associated with urbanization. Making such predictions deterministically requires a thorough knowledge of species-specific environmental responses, as well as an adequate (accurate) characterization of habitat structure and habitat dynamics (both of which are modified by urbanization). However, recent studies have demonstrated that the effects of hydrologic alterations induced by urbanization on selected stream biota can be quantitatively described without a full mechanistic understanding, using stressor-response type relationships and empirical correlations from field-measured conditions (Konrad and Booth 2005, Konrad *et al.* 2008, DeGasperi *et al.* 2009).

In moving beyond a narrow focus on linkages between flow alteration and channel instability, scientific understanding of hydrologic controls on stream ecosystems has recently led to new approaches for assessing the ecological implications of hydromodification. The essential steps in developing quantitative "flow-ecology relationships" have been recently described in the Ecological Limits of Hydrologic Alteration (ELOHA) process (Poff *et al.* 2010), a synthesis of a number of existing hydrologic techniques and environmental flow methods. ELOHA provides a regional framework for elucidating the key hydrologic influences on biota of interest, and translating that understanding into relationships between hydromodification and biological endpoints that can be used in management decision making. This requires a foundation of hydrologic data provided by modeling and/or monitoring, and sufficient biological data across regional gradients of hydromodification. Although hydrologic–ecological response relationships may be confounded to some extent by factors such as chemical and thermal stressors, there are numerous case studies from the US and abroad in which stakeholders and decision-makers reached consensus in defining regional flow standards for conservation of stream biota and ecological restoration (Poff *et al.* 2010; http://conserveonline.org/workspaces/eloha).

4.3.3 Strengths, Limitations and Uncertainties

The Organizing Framework shown in Figure 4-1 shows the applicability of the three major categories of tools in support of various management actions. This section addresses a range of issues relating to strengths, limitations and uncertainty of the tools discussed above. Detailed analysis of individual models is beyond the scope of this document, but EPA/600/R-05/149 (2005) contains an extensive comparison of functions and features across a wide range of hydrologic and hydraulic models.

> **Explicit consideration, quantification, and gradual reduction of model uncertainty will be necessary to advance hydromodification management.**
>
> **The uncertainty inherent to hydromodification modeling underscores the need for carefully designed monitoring and adaptive management programs.**

General Considerations. The well-known statistician George Box famously said that "all models are wrong, some are useful." The usefulness of a model for a particular application depends on many factors including prediction accuracy, spatial and temporal detail, cost of calibration and testing, meaningful outputs, and simplicity in application and understanding. There is no cookbook for selecting models with an optimal balance of these characteristics. Models of stream response to land-use change will always be imperfect representations of reality with associated uncertainty in their predictions. In addition to the prediction errors of standard hydrologic models, common limitations and sources of uncertainties include insufficient spatial and/or temporal resolution, and poorly known parameters and boundary conditions. Ultimately, the focus of scientific study in support of decision making should be on the decisions (or objectives) associated with the resource and not on the model or basic science. Each model has limitations in terms of its utility in addressing decisions and objectives of primary concern to stakeholders. Prediction error, not perception of mechanistic correctness, should be the most important criterion reflecting the usefulness of a model (NRC 2001; Reckhow 1999a,b). The predictive models discussed above may be thought of as predictive scientific assessments; that is, a flexible, changeable mix of small mechanistic models, statistical analyses, and expert scientific judgment.

Region-Specific Considerations. Because all models are vulnerable to improper specification and omission of significant processes, caution must be exercised in transferring existing models to new regional conditions. For example, mobile boundary hydraulic models are mechanistically detailed but not generally well-suited to many southern California streams given the prevalence of near-supercritical flow, braiding and split flow (Dust 2009). In addition, bed armoring and channel widening resulting from both fluvial erosion and mass wasting processes are key influences on channel response in semi-arid environments. These processes are not well-represented and constrained in current mobile boundary models. Accordingly, the appropriateness of existing models for addressing a particular hydromodification management question should be empirically tested and supported with regionally appropriate data from diverse stream settings.

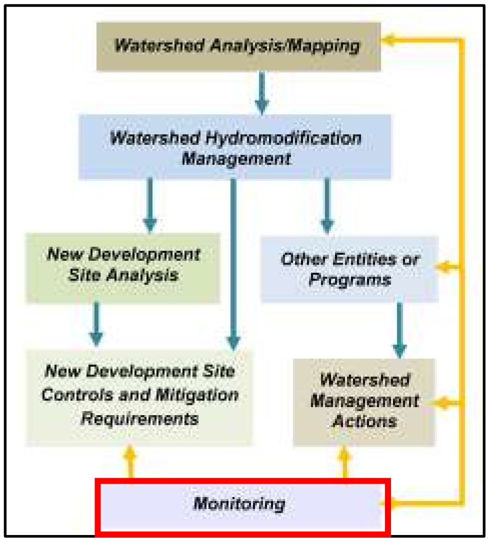
Managing Uncertainty. To date, hydromodification management has generally relied on oversimplified models or deterministic outputs from numerical models that consume considerable resources but yield highly uncertain predictions that can be difficult to apply in management decisions. Numerical models are nevertheless an important part of the hydromodification toolbox, especially in characterizing rainfall-response over decades of land-use change. It is challenging to rigorously quantify the prediction accuracy of these mechanistic numerical models; however, their utility of can be enhanced by addressing prediction uncertainties in number of ways (Cui *et al.* 2011). Candidate models can be subjected to sensitivity analysis to understand their relative efficacy for assessment and prediction of hydromodification effects. Moreover, it should also be demonstrated that selected models can reasonably reproduce background conditions before they are applied in predicting the future. Modeling results that are used in relative comparisons of outcomes are generally much more reliable than predictions of absolute magnitudes of response.

Hydromodification modeling embodies substantial uncertainties in terms of both the forcing processes and stream response. Deterministic representations of processes and responses can mask uncertainties and can be misleading unless prediction uncertainty is explicitly quantified. Errors may be transferred and compounded through coupled hydrologic, geomorphic, and biologic models. Accordingly, explicit consideration, quantification, and gradual reduction of model uncertainty will be necessary to advance hydromodification management. This points to two basic needs. First, there is a need to develop more robust probabilistic modeling approaches that can be systematically updated and refined as knowledge increases over time. Such approaches must be amenable to categorical inputs and outputs, as well as combining data from a mix of sources including mechanistic hydrology models, statistical models based on field surveys of stream characteristics, and expert judgment. Second, the uncertainty inherent to hydromodification modeling underscores the need for carefully designed monitoring and adaptive management programs, as discussed in Section 5.

A risked-based framework can provide a more rational and transparent basis for prediction and decision-making by explicitly recognizing uncertainty in both the reasoning about stream response and the quality of information used to drive the models. Prediction uncertainty can be quantified for any of the types of models described above; however, some types are more amenable to uncertainty analysis than others. For example, performing a Monte Carlo analysis of a coupled hydrologic-hydraulic model is a very demanding task. A simple sensitivity analysis of high, medium, and low values of plausible model parameters is much more tractable and still provides an improved understanding of the potential range of system responses. Such information can be subsequently integrated with other model outputs and expert judgment into a probabilistic framework. For example, Bayesian probability network approaches can accommodate a mix of inputs from mechanistic and statistical models, and expert judgment to quantify the probability of categorical states of stream response. Such networks also provide an explicit quantification of uncertainty, and lend themselves to continual updating and refinement as information and knowledge increase over time. As such, they have many attractive features for hydromodification management, and are increasingly used in environmental modeling in support of water quality (Reckhow 1999a,b) and stream restoration decision-making (Stewart-Koster *et al.* 2010).

Sediment Supply. As described above, a reduction in sediment supply to a stream may result in instability and impacts, even if pre- and post-land use change flows are perfectly matched. Thus, there is a need to develop management approaches to protect stream channels when sediment supply is reduced, and to refine and simplify tools to support these approaches. This continues to prove challenging because, the effects of urban development on sediment supply in different geologic settings are not well understood and poorly represented in current models. As a starting point, models used to analyze development proposals that reduce sediment supply could be applied with more protective assumptions with respect to parameters and boundary conditions (inflowing sediment loads). Effects of altered sediment supply on stream response could be addressed in a probabilistic framework by adjusting conditional probabilities of stream states to reflect the influence of reductions in important sediment sources due to land use change.

5. MONITORING



"Monitoring" can cover a tremendous range of activities in the context of stormwater management in general, and of hydromodification in particular. For example, the NPDES Phase 2 general permit for California (SWRCB, 2003 (*www.swrcb.ca.gov/**water**_issues/.../storm**water**/.../final_ms4_**permit**.p...)*, National Pollutant Discharge Elimination System (NPDES) General Permit No. CAS000004, p. 11) notes that the objectives of a monitoring program may include:

- Assessing compliance with the General Permit.
- Measuring and improving the effectiveness of stormwater management plans.
- Assessing the chemical, physical, and biological impacts on receiving waters resulting from urban runoff.
- Characterizing storm water discharges.
- Identifying sources of pollutants.
- Assessing the overall health and evaluating long-term trends in receiving water quality.

These objectives span multiple goals, ranging from verifying of compliance, evaluating effectiveness, characterizing existing conditions, and tracking changes over time. Each would likely require different monitoring methods, duration of measurement, and uses of the resulting data (Table 5-1). This variability emphasizes what we consider the key starting point of any monitoring program: to answer the questions, "What is the purpose of monitoring? How will the data be used?" Even secondary considerations can exert great influence over every aspect of the design of a monitoring program: "How quickly do you need to have an answer?" And, perhaps most influential of all, "What are the resources available to provide that answer?"

Table 5-1. The recommended purpose(s) of monitoring associated with hydromodification control plans, organized by the scale of implementation and the time frame in which useful results should be anticipated.

Time Frame	Programmatic: State and Regional Water Boards	Local: City and County Jurisdictions
Short-term (<10 years)	• Define the watershed context for local monitoring (at coarse scale) • Evaluate whether permit requirements are making positive improvements	• Evaluate whether specific projects/ regulations are meeting objectives • Identify the highest priority action(s) to take
Long-term (1+ decades)	• Define watershed context and setting benchmarks for local-scale monitoring (i.e., greater precision, if/as needed) • Demonstrate how permit requirements can improve receiving-water "health," state-wide (and change those requirements, as needed)	• Evaluate and demonstrate whether actions (on-site, instream, and watershed scale) are improving receiving-water conditions • Assess program cost-effectiveness • Identify any critical areas for resource protection

5.1 The Purpose of Monitoring

In the context of hydromodification assessment and management, we propose three interrelated purposes for monitoring that will guide the discussion and recommendations in this section:

- **Characterizing the conditions** of receiving waters downstream of urban development (including any trends in those conditions over time).
- **Evaluating the effectiveness** of hydromodification controls at protecting or improving the conditions of downstream receiving waters (and modify them, as needed).
- **Setting priorities** on the wide variety of hydromodification control practices, as promulgated by the State and Regional Boards and as implemented by local jurisdictions.

These needs give rise to several interrelated types of monitoring, all common to many watershed and stormwater monitoring programs. They are typically executed at different spatial and temporal scales, and if well-designed and executed they can collectively help guide management actions. We define them here, using terms and definitions that are common to the monitoring literature:

- **Performance monitoring,** by which is normally meant the evaluation of a particular stormwater facility relative to its intended (or designed) performance, but independent of whether that intended design is actually beneficial for downstream receiving waters.
- **Effectiveness monitoring,** by which we mean the assessment of how well specific management actions or suites of actions reduce or eliminate the direct impacts of stormwater on receiving waters. This type of monitoring can answer a question common to stormwater management: does a particular facility actually achieve its intended goal (e.g., flow releases from a stormwater facility protect the stream channel downstream from erosion)? More broadly, monitoring can evaluate the "effectiveness" of a suite of measures or an overall program designed to produce

beneficial outcomes (or avoid negative ones) in downstream receiving waters. In this context, the precise boundaries division between effectiveness monitoring and other types are blurry and unnecessarily artificial.

- **Trends monitoring,** by which we mean an integrative assessment of whether our "endpoint" indicators (physical, chemical, or biological) are showing any consistent, statistically significant change over time. Such monitoring rarely "proves" the direct impacts of a specific stressor on a receiving water, but it is critical to setting and evaluating progress towards integrative assessment endpoints at a regional scale. If well-designed, trend monitoring commonly provides useful information at smaller spatial scales as well, particularly in evaluating response to recent management actions or recovery from a prior disturbance.
- **Characterization monitoring**, by which is commonly meant the identification and (or) the quantification of various parameters in stormwater or a receiving-water body. Characterizing the condition of an outflow discharge or a water body at a particular time and place is always an outcome of the other kinds of monitoring; when it is called out as a goal in-and-of itself, however, it is can be useful to prioritize actions—but only if there is a preexisting standard for what constitutes a "good" or "acceptable" condition (also termed "status monitoring"), and a program to implement (or at least to set the priority for implementing) actions to improve the condition of waterbodies found to be "not good" or "unacceptable."

Without a context for evaluation, characterization monitoring is prone to generate large quantities of rarely used data. We strongly encourage that the purpose of any "characterization" monitoring be clearly articulated in hypothesis testing, priority setting, or systematic trend evaluation. As noted by NRC (2009, p. 508) with respect to this type of monitoring, "…monitoring under all three (NPDES municipal, industrial, and construction) stormwater permits is according to minimum requirements not founded in any particular objective or question. It therefore produces data that cannot be applied to any question that may be of importance to guide management programs, and it is entirely unrelated to the effects being produced in the receiving waters." We seek to proactively avoid this problem.

Monitoring should occur at two scales:

- **Regional or state-wide scale- this will require a time frame of one to several decades**
- **Local scale – this is required to evaluate the performance and effectiveness of specific management measures.**

In this sub-section, we focus our discussion on two interrelated scales at which these various types of monitoring should be applied as outlined in Table 5-1 at the beginning of this section. The first, which here and elsewhere in this document is termed "programmatic," has a regional or state-wide spatial scale; many of its key actions will require a time frame of one to several decades. Monitoring data from this scale should inform the broadly construed "health" of receiving waters to assess whether the range of hydromodification strategies being implemented is maintaining desired conditions across the (state-wide) range of physiography, climate, land-use change, and regulatory approaches of the regional boards. They should be used to identify particularly promising (or particularly ineffective) combinations of control strategies and landscape conditions. Finally, they should provide regionally tailored benchmarks for what constitutes "healthy watersheds” and “healthy receiving waters” so regulators and permittees alike know what still needs to be done, where it should be done, and how urgently it needs to happen.

The second scale of monitoring data we term “local.” It comprises the generation of monitoring data to evaluate the performance and effectiveness of specific management measures (be they structural or nonstructural) at reducing the negative consequences of hydromodification on downstream receiving waters. Useful information at this scale will normally be generated in the time frame of an NPDES permit cycle (i.e., ~5 years) and should provide direct guidance on whether the evaluated management strategies are working, need refinement, or should be abandoned altogether. They should also provide guidance on the degree to which management efforts should be prioritized where regulatory flexibility exists, given the conditions (and, perhaps, the potential responsiveness) of downstream receiving waters. Over longer time frames, monitoring at this scale can also provide public demonstration of the value of regulatory and programmatic efforts, and it can also help identify the most cost-effective mix of publically funded projects and regulatory protection to achieve (or maintain) receiving-water health.

5.2 Programmatic Monitoring at the Regional Scale

5.2.1 Defining Watershed Context

Although not “monitoring” in the strictest sense of this word, establishing a watershed context for the measurement and evaluation of receiving waters is a hallmark of virtually all recommended monitoring strategies (e.g., Beechie *et al.* 2010, Brierley *et al*. 2010). Monitoring programs should be consistent with the watershed perspective that forms the basis for the management framework discussed in Section 3. In California (as in most other states), this can only be executed at a supra-jurisdictional scale, because most watersheds cross one or more city and/or county boundaries. This presents the long-term challenge that many jurisdictions do not have authority over parts of the landscape that can affect the quality of rivers and streams that pass through their boundaries; more immediately, however, it makes an inclusive watershed assessment almost impossible to execute at a local level.

5.2.2 Determining the Effectiveness of Permit Requirements

A second, more challenging contextual need at the regional scale is the definition of thresholds or endpoints against which to compare the results of monitoring or modeling. Both of these “assessment tools” can guide the application of hydromodification control strategies, evaluate their real or likely success, and predict the consequences of hydromodification on downstream receiving waters. However, they provide little insight into the question, “how good is good enough?” Answering this question requires a definition of “assessment endpoints” (borrowing the term from NRC 1994), which in turn requires objective, quantifiable criteria for evaluating progress or outright success.

Most existing HMPs require the permitted municipalities to develop programs and policies to assess the potential effects of hydromodification associated with new development and redevelopment, to include management measures to control the effects of hydromodification, and to implement a monitoring program that assesses the effectiveness of HMP implementation at controlling and/or mitigating the effects of hydromodification. Yet the appropriate objectives of such management measures, or a basis to evaluate success or failure of the HMP through monitoring data, are rarely provided in consort. Setting these endpoints is beyond the capacity of any but the largest municipalities—and even for those, neither the field of watershed science nor the arena of public policy is so clear that an unequivocally "correct" answer is likely to emerge without much additional work. Any such finding would also lack state-wide applicability; California is far too physically and ecologically diverse for an assessment endpoint developed in one part of the state to transfer everywhere without careful consideration.

For these reasons, we consider this aspect of monitoring at the regional scale to be a long-term, state-wide effort. This reflects the challenge of conducting meaningful characterization (or "status") monitoring: it requires a benchmark against which the measured condition can be compared, and to which an absolute rating ("good," bad," etc.) can be assigned.

In contrast, "trends" monitoring requires no such benchmark, only equivalent measurements undertaken at multiple times coupled with an understanding of what direction of change is desirable. For this reason, evaluating whether permit requirements are making positive improvements is a reasonable (and probably critical) short-term effort, one that can be conducted locally (see below). It should also be integrated and compiled at a regional level, however, the better to inform the continued development of hydromodification requirements.

5.3 Monitoring at the Local Scale

The needs of a monitoring program for local jurisdictions should complement those being satisfied at a regional scale. Showing net improvement is critical to maintaining support for regulatory actions and capital expenditures, but any monitoring program must reflect the typical constraints of showing rapid results while acknowledging constraints on staff resources and expertise (Scholz and Booth 2001). No less urgent is the need to identify what to do "next"—not necessarily establishing a multi-year capital improvement plan, but at least identifying key problems with one or two associated actions that would likely result in significant improvements in receiving-water conditions. Watershed characterization, as discussed above and applied to a specific jurisdiction, can provide useful guidance for such identification; even without it, local knowledge is commonly sufficient in-and-of itself. Targeted monitoring can normally confirm (or refute) such inferences in short order, which is why we place this monitoring application in the "short-term" category.

However, a monitoring program can also provide longer term guidance to local jurisdictions. When supported by the regional context of receiving-water conditions, local monitoring data can demonstrate trends over time that can lend support to (or indicate necessary changes to) hydromodification control plans. In combination with economic data, they can show long-term cost-effectiveness. Finally, site-specific monitoring data, when analyzed in the context of an appropriate scale of watershed characterization, can guide the stratification of less developed and undeveloped watershed areas into those where more assertive protection (or restoration) will be most worthwhile. None of these outcomes depend solely on collecting monitoring data, which is why none of them are presumed to be credible "short-term" applications of monitoring data. However, they have found expression in other parts of the country having long-term monitoring efforts, and they should provide similar benefits to California as well.

5.4 Developing a Monitoring Plan

"Monitoring" the effects of a management action, whether it is a new regulation, a change in operational procedures, or a constructed project, is commonly included by design or required by regulation. The collection of monitoring data may be seen as a worthwhile activity in its own right, but this discussion uses a more restrictive, implementation-based definition: any "monitoring" needs to demonstrate a direct connection to management actions, such that the results of monitoring are translated into on-the-ground management actions (or changes in management actions). This focus on the *use* of monitoring data requires clear linkages between a management action, the uncertainties associated with that action, the ways in which the effects of that action are expressed (and can be measured) in the world, and the management changes that should be implemented if monitoring results provide unanticipated (or equivocal) resolution to those uncertainties. This is the basis for establishing an "adaptive management" approach to hydromodification monitoring, discussed in more detail in Appendix C. Here, we discuss the design of a monitoring program and outline the variety of measurements that can be made, under the assumption that the intended use(s) of the monitoring data have already been established.

"Stormwater management would benefit most substantially from a well-balanced monitoring program that encompasses chemical, biological, and physical parameters from outfalls to receiving waters" (NRC 2009, p. 257). In pursuit of a comprehensive monitoring program we might also add regular documentation of weather and climate conditions and land-cover changes. As a practical matter, however, monitoring at a site scale is almost never coordinated with other equivalent efforts at other locations, nor placed in a broader spatial context being developed as part of a regional effort. For monitoring data to have greatest value, however, such coordination and context-setting is needed.

Stormwater management would benefit most substantially from a well-balanced monitoring program that encompasses chemical, biological, and physical parameters… (NRC, 2009)

5.4.1 Design of a Monitoring Plan

As noted at the beginning of this section, the overarching question that must be asked and answered at the beginning of any monitoring design effort is "What is its purpose?" The considerations enumerated below cannot be addressed without an explicit answer to this question, because the outcome of those considerations will depend on how the data are to be used. For certain common application of monitoring data we suggest guidance that will be widely appropriate, but there are no recommendations in this section (or any other monitoring guidance document) that apply universally.

Multiple authors have condensed their guidance for designing a monitoring plan into a short list of steps that should precede the first instance of field data collection (e.g., Shaver *et al.* 2007). Although all differ in details and intended audience, they share significant commonalities that can be distilled as follows:

- Articulate the purpose of the monitoring (the "management question").
- Identify key constraints, in particular the geographic range and scale over which the monitoring can occur, financial/staff resources available, and the time frame in which results must be generated.
- Evaluate existing information, model outputs, and/or regulatory requirements to identify promising metrics and specific sites appropriate to the management question.
- Identify the specifics of the monitoring plan: what parameter(s), where, for how often and for how long. This may include multiple iterations, wherein the guidance of Step 3 must align with the constraints of Step 2.

Most such guidance is written with site-specific, "local" monitoring in mind—the existing literature provides less direction for monitoring that is herein recommended to occur at a regional scale over the next one or more decades. However, the basic principles are the same at all scales: a coherent, explicit purpose needs to be articulated, resource constraints need to be acknowledged, and a credible strategy needs to be developed with its specifics fleshed out. Below we discuss some of the primary considerations in this last step, because they are common across a wide range of monitoring purposes, programmatic constraints, and indicator types.

5.4.2 Constraints (Step 2 of the Monitoring Plan)

Ideally, a monitoring program should be designed to detect trends, assess effectiveness and allow for source identification.

Scale. Ideally, a monitoring program should encompass multiple, nested scales of monitoring that are determined by the question(s) being addressed. For hydromodification applications, the broadest scale of monitoring is that of the integrated effect of stormwater impacts and stormwater management on receiving waters. *Trends monitoring* (and characterization monitoring, if regionally appropriate ranges of quality have been determined) addresses these questions, and it also allows stormwater and resource managers to measure the broad benefits obtained from management investments. Site-specific conditions normally cannot be traced back to specific generators of pollution (NRC 2009), and so monitoring at the broadest scales (i.e., many tens of square miles and larger) should not attempt to do so. Instead, identifying overall conditions and trends requires a broad spatial scale over long time frames (i.e., multiple years), the essence of trends monitoring. This level of effort is recommended as a regional responsibility, because the area(s) of interest will normally far exceed the geographic limits of any single jurisdiction.

If trends monitoring (or long-standing prior knowledge) indicates that there are impacts on beneficial uses, a second (and more site-specific) scale is invoked, that of *effectiveness monitoring:* which of our many stormwater-management actions are achieving the greatest reduction in downstream impacts (and which are not)? On the whole, such stormwater control measures, both structural and nonstructural, vary by land use—the measures suitable for a residential neighborhood will likely be impractical or ineffective (or both) in an industrial setting. We therefore anticipate that most effectiveness monitoring will be stratified by land use and conducted by individual jurisdictions (see, for example, such an approach in the Nationwide Stormwater Quality Database, which contains water-quality data from more than 8600 events and 100 municipalities throughout the country).

The finest scale of monitoring is that of *source identification*, a form of characterization monitoring: what specific locations and which parts of the landscape generate stormwater of sufficiently deleterious quantity and (or) quality to cause impacts to beneficial uses, be they direct or indirect effects? This question is widely posed in stormwater management programs, and a number of existing monitoring programs seek to provide answers. The science of stormwater already suggests where the greatest attention is probably warranted (NRC 2009), namely a particular focus on areas of well-connected (or "effective") impervious area, high vehicular traffic, and exposure to toxic chemicals. We therefore suggest these categories should define areas of highest priority for this type of targeted investigation, allowing even a resource-constrained jurisdiction to conduct a useful, well-focused monitoring effort with good efficiencies.

Siting. Site selection is most commonly guided by the location of the management action being evaluated while dictated by more mundane considerations of property ownership and access logistics. In general, sites need to meet a few following basic criteria.

- Appropriate scale: the upstream area should be dominated by, or at least significantly affected by, the management action of interest.
- Responsiveness: at the chosen location, the parameters being measured should be amenable to change in response to the management action (e.g., monitoring for geomorphic change in a concrete channel is ill-advised).
- Representativeness: the results at the chosen location should be credibly extrapolated to "similar" sites, and those sites in aggregate should constitute a widespread (or otherwise important) subset of the landscape as a whole.
- Access: the site should be easily reached by the appropriate personnel and equipment, and with a cost of doing so consistent with the frequency of measurements being made. Any equipment left unattended needs to be secure (or well-hidden).

There are institutional considerations in site selection as well. Multiple programs implement monitoring or impose monitoring requirements, and coordination can provide mutual benefits and efficiencies to all. In particular, monitoring driven by management actions at a particular location (i.e., a local scale) will always benefit from information from one or more regional-scale reference sites that can characterize natural or background variability. Local studies will rarely have resources to execute such an effort themselves, again emphasizing the importance of a nested (and coordinated) hierarchy of monitoring programs.

Time and Variability. Evaluating the effectiveness of management actions requires a preliminary judgment of the time frame over which effects can be recognized. For water-quality parameters, storm-specific grab samples or continuous flow-weighted sampling has been most common; for changes in geomorphic form or in the population attributes of benthic macroinvertebrates, one-time annual sampling that presumes to integrate the effects of the past year are typical. Flow metrics are normally extracted from “continuous” (i.e., 5-, 15-, or 60-minute) measurements of discharge. However, every measurement has some degree of variability, a consequence of “natural” variability, measurement errors, and induced change (i.e., the effects of the management action we are trying to perceive). Separating these components is a matter of statistical analysis (see next section) based on repeated measurements, either in time or in space (or both).

We note that many practices common to past monitoring efforts, particularly the use of individual grab samples to characterize stormwater quality, have yielded results with little to no subsequent value: “…to use stormwater data for decision making in a scientifically defensible fashion, grab sampling should be abandoned as a credible stormwater sampling approach for virtually all applications” (NRC 2009, p. 330).

The duration of a monitoring program is commonly determined by the desire for “timely” answers, although normally the ability to generate statistically significant results is a function of the system being evaluated and the indicators being measured. This often creates a conflict between the intended “mission” of the monitoring program and its ability to produce defensible results, a conflict that can only be avoided by a design that identifies meaningful variables to measure, conducts sufficiently frequent measurements to dampen random variability, and must persist~~s~~ for long enough to allow a management “signal” to emerge from the data. This is the essence of the iteration noted above in Step 4 of monitoring-plan design above.

The monitoring program design must persist long enough to a allow management "signal" to emerge from the data. Consequently, long-term records (i.e., one to several decades) will be needed to detect all but the most dramatic of trends in biological indicators.

In one of relatively few quantitative studies of variability in biological indicators, Mazor *et al.* (2009) found that year-to-year variability for the same site sampled in the same season showed a variability (i.e., $\pm 1\sigma$) was typically about 10 points for a benthic IBI. With average scores for their 5 sites ranging from 28–51 (on a 100-point scale), this reflects a coefficient of variation of about 25%. Individual metrics were even more variable. This emphasizes that long-term records (i.e., one to several decades) will be needed to detect all but the most dramatic of trends in biological indicators.

The duration of monitoring also needs to capture the events that are most important to the anticipated responses of the measured system. For evaluating the effects of hydromodification, frequent storms (i.e., those that are normally expected to occur one to several times per year) are commonly judged important and their effects would normally be captured by a monitoring effort of even just one or a few years’ duration. Particularly in more semi-arid regions of the state, however, significant channel-altering events may occur only after many decades of relative quiescence and stability, and noticeable (or documentable) response of streams to hydromodification may only occur under certain circumstances or following specific combination of events. Therefore, the lack of channel response on an annual basis may not necessarily indicate that management actions are effective. Thus a long-term, ongoing monitoring effort is necessary to capture the responses to infrequent, stochastic events, but determining the likely duration of such a program requires some knowledge (or assumptions) of the critical drivers of those responses. It therefore requires a well-posed set of management questions underlying the monitoring effort as well.

For management questions concerning the effectiveness of hydromodification controls, monitoring will almost always benefit from long-term flow monitoring at multiple sites, especially those in the mid to upper watershed (and key tributaries, depending on the scale of the effort). Local rainfall measurements are nearly as essential, since flow data without rainfall data resolved at a similar spatial and temporal scale are useless at best, misleading at worst. Baseline (pre-project) monitoring normally is also invaluable. However, each of these elements will normally require some combination of a multi-scale, long-term, coordinated monitoring program with an investment of at least several years' duration in anticipation of (and follow-up after) a specific management action at a specific location. Despite the value for evaluating the effects of hydromodification (and hydromodification control efforts), such monitoring almost never occurs to this degree. To the extent this remains a practical constraint on implementation, the range of management questions needs to be commensurately narrowed as well.

Statistical Considerations. The statistical design of a monitoring program is beyond the scope of this section, because the range of possible requirements and approaches is tremendously broad. Several general principals are worth articulating, however, because they apply almost universally (and are commonly ignored):

- Although trends can be "suggested" by monitoring data, only statistically rigorous results can be offered as "proof." Thus, ignoring this dimension of monitoring program design severely limits future applicability of the results.
- Most natural parameters display high variability when measured outside a laboratory, and thus the magnitude of change caused by a management action also needs to be great before it can be recognized. There is a trade-off between the relative magnitude of change and the number of samples required to recognize it (i.e., large relative changes require fewer samples), but many monitoring efforts pay little attention to this basic fact. Where sampling can only occur during specified storm conditions or once during the same season each year, the duration of a monitoring campaign sufficient to detect even large changes in naturally variable parameters is likely to be a decade or longer. For many management applications, this is tantamount to generating no useful information at all (but is significantly more costly).
- The level of effort needed can be estimated *a priori* to help guide final monitoring design, but only if the degree of variability and the magnitude of change to be perceived are known or estimated ahead of time. One such example is given below, where the diagonal lines are labeled with the number of independent samples needed to achieve a typical level of statistical power for various combinations of permissible error from the "true" value (x axis), and the intrinsic variability in values across the population being measured (y axis) in Figure 5-1 below.

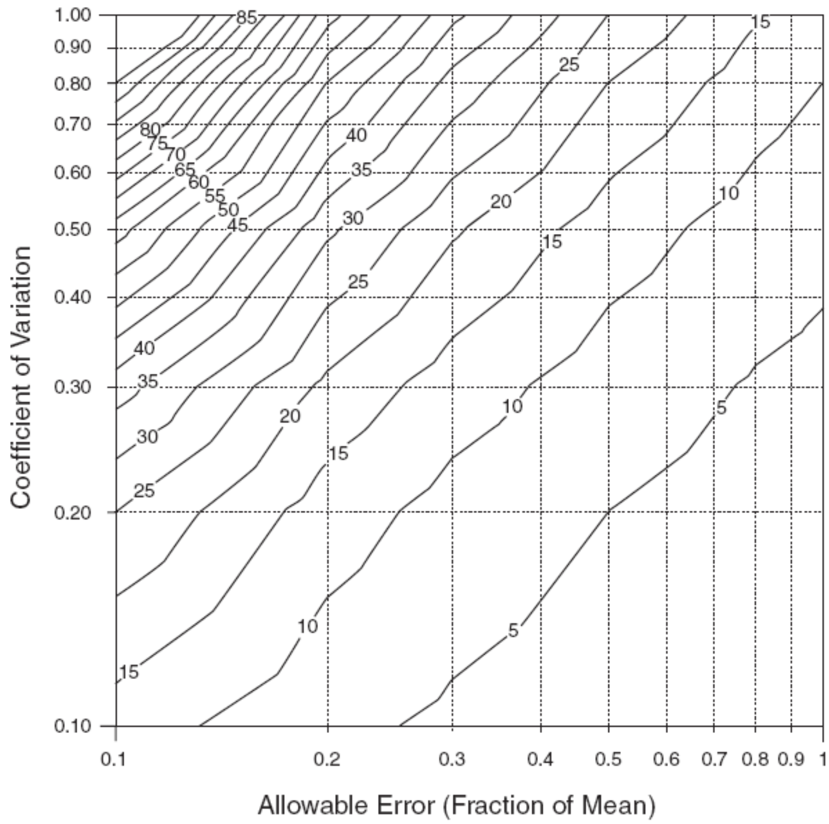


Figure 5-1. Sample requirements for confidence of 95% (α = 0.05) and power of 80% (β = 0.20). Figure from Pitt and Parmer 1995.

5.4.3 What to Monitor (Step 3 of the Monitoring Plan)

The choice of "what to monitor" follows from the choice of assessment endpoints, which in turn depends on the choice of management goals: for example, if "stable stream channels" is the intended outcome of an HMP, then measurement of the physical form of a channel over time would be appropriate. If diagnosing the cause of observed changes is also desired, then some evaluation of potential causal agents (e.g., hydrology, sediment input, or direct disturbance) would also be needed. Because management goals are now commonly (and appropriately) cast more broadly, however, they can embrace less clearly defined endpoints such as "watershed health" or "biological integrity." Many such endpoints fail the test of quantifiable objectivity.

However, these goals invoke a broad scope of concern, embracing not only physical stream conditions but also a range of chemical, hydrologic, and biological attributes. They encompass a broader catalog of receiving waters that may need to be evaluated. Finally, they emphasize the importance of looking more broadly to identify the cause of observed changes—both spatially, to conditions throughout a watershed that may have influence downstream; and temporally, to recognize ongoing adjustments to past disturbance (i.e., legacy effects) and to future environmental changes (e.g., climate change) that commonly lie well beyond the ability of local watershed managers to address. The imprecision of these goals should not obscure the importance of broadening the scope of stormwater and hydromodification assessments to include not only the traditionally emphasized characterization of selected water-quality constituents and channel stability, but also more integrative measures.

These considerations suggest two broad categories of assessments, which largely but not entirely align with the two scales of implementation (i.e., "programmatic" and "local") defined in Table 4–1:

- Integrative: defining an overall level of "health" of the watershed, as expressed in the condition(s) of its receiving waters. Current scientific consensus suggests that biological indicators are best suited to this scale of evaluation (Karr and Chu 1999), insofar as they integrate the consequences of multiple stressors on aquatic systems and because many management goals (and regulatory requirements) are cast in biological terms. To be meaningful, however, any such indicators need to be suitably chosen and stratified for their particular geo-hydro-climatological region (e.g., "ecoregions"; Omernick and Bailey 1997).
- Targeted: demonstrating the achievement of an established regulatory standard or a designated threshold (typically, a measured or modeled pre-development condition) by a particular parameter, commonly one or more chemical constituents or a specific hydrologic metric of flow. This can be evaluated at the outfall of a single stormwater facility, at the discharge point for a site, or in the receiving water itself. Many of these thresholds are important in their own right—to protect human health, to preserve riparian property from erosion, to avoid flooding of previously non-inundated lands. However, they should be recognized as providing only one-dimensional views of a much broader system. Thus, targeted monitoring can supplement but should not replace more integrative measures.

Integrative assessment endpoints require multiple lines of evidence to characterize receiving-water conditions. At their most comprehensive, they should include measures of flow, geomorphic condition, chemistry, and biotic integrity (Griffith *et al.* 2005, Johnson and Hering 2009). However, biological criteria are generally key to integrative assessment: "In general, biological criteria are more closely related to the designated uses of waterbodies than are physical or chemical measurements" (NRC 2001, p. 8). In most applications, such assessments are compared to one or more reference sites where conditions have been independently judged as "excellent," or where human disturbance is minimal and so best-quality conditions are assumed.

Integrative assessment endpoints require multiple lines of evidence to characterize receiving-water conditions. At their most comprehensive, they should include measures of flow, geomorphic condition, chemistry, and biotic integrity.

The task of identifying and quantifying reference conditions in California streams is presently being carried out by the Reference Condition Management Program (RCMP) of the State Water Board's Surface Water Ambient Monitoring Program (SWAMP; see 2009 Recommendations). About 600 sites have been recognized by this program as "reference" based on having minimal human disturbance, and they have been geographically stratified into the 12 Level III ecoregions mapped for the state of California (by USEPA 2000). The metrics chosen to characterize their biologic conditions should provide an appropriate list for the evaluation of impaired (or potentially impaired) streams.

An equivalent set of reference sites and conditions for other receiving-water types does not presently exist. California also presently lacks a systematic basis for defining relative categories of "poor," "fair," "good," or "excellent" based on numeric values of biological indicators, such as exists in parts of the Pacific Northwest. Several regions, however, now have multimetric biological indicators with defined reference conditions (see below).

Elsewhere, however, there is as yet no context for setting assessment endpoints for biological indicators in California receiving waters. Such an effort is in progress, at least for streams, and its eventual completion to support the management application of more local monitoring results is a key recommendation of this report. Biological assessment endpoints will need to be established region by region on an as-needed basis; in the interim, locally collected data can be very useful for trend monitoring of receiving water but not for defining existing levels of "health."

5.4.3.1 An Example from Washington State

The Puget Sound region of western Washington State provides an instructive example for identifying indicators and establishing desired assessment endpoints. Multiple agencies over the last two decades have sought to measure the overall ecological health of the region and to define targets for recovery. Following the most recent three-year process, the lead agency for the current effort released its set of 20 "dashboard indicators" designed both to express scientific understanding of conditions needed for ecological health and to communicate that understanding in a public-accessible manner (http://www.psp.wa.gov/pm_dashboard.php; accessed September 5, 2011). They cover physical, chemical, and biological indicators: all expressed in terms of relative improvement or quantified conditions to be reached by the year 2020.

This level of target-setting is possible only after extensive study and public discussion; it falls far beyond the scope of the present document. It is instructive for the state of California, however, in several regards as it looks to the future:

- The physiographic scope of the indicators and their target values is well-constrained to a particular geographic region with broadly similar geologic, hydrologic, and climatological attributes. Multiple parallel efforts would almost certainly be needed for a more diverse region (such as the entire state).
- Each indicator has a strong scientific basis for inclusion and at least some scientific basis for specific targets. Their communication value with the public was also an explicit criterion for inclusion.
- The most numerous indicators are biological, and they address multiple levels of the trophic chain from top predators to plants (a planktonic metric, however, was rejected as requiring too much additional scientific study and offering little communication value to the general public).
- Although emphasizing biology, the indicators are broadly distributed amongst biological, chemical, and physical metrics; most are broadly integrative in nature (e.g., reference to "bug populations" (the Puget Sound B-IBI) and a "freshwater quality index").

- The set of physical indicators is most parsimonious for instream conditions, and excluding marine nearshore and estuary conditions is restricted to a single hydrologic metric (chosen for its presumed influence on fish). This stands in stark contrast to most existing hydromodification monitoring plans, which emphasize measures of channel geomorphology and a wider range of hydrologic metrics. Such indicators may provide useful performance measures, but they should not be mistaken for more integrative measures of ecosystem or watershed "health."
- Although each indicator has a specified, numeric goal to be reached by 2020, there are no articulated changes to the current management plan if any of those goals are not reached (or if interim measures suggest that they will not be reached). This is a recognized shortcoming of the present plan but there is no mechanism yet in place to address it. As such, it does not currently meet the test for "adaptive management" (see Appendix C).

In California, such a list of integrative assessment indicators (let alone quantified endpoints for those indicators) cannot presently be defined, except in a few specific localities where data collection and analysis have been ongoing for many years. Thus, we recognize the value of such targets but must guide the present development of monitoring in recognition of their near-complete absence. Rectifying this shortcoming is the central recommendation for long-term program development; in the interim, short-term monitoring at both the regional and local levels need to acknowledge the absence of an integrative context in which to interpret their results.

In California, a list of integrative assessment indicators (let alone quantified endpoints for those indicators) cannot presently be defined, except in a few specific localities. Rectifying this shortcoming is the central recommendation for long-term program development.

Regulatory standards are established on the assumption that "clean water" will result in "healthy streams," but the elements of a watershed are far too complexly interrelated to permit such a simplistic perspective. Although the inverse ("polluted water results in unhealthy streams") is almost always true, the challenge for inferring causality from typical monitoring data is that *many* such stressors can all yield the same, degraded outcome. For this reason, targeted monitoring can provide useful diagnostic information and demonstrate regulatory compliance, but it cannot provide sufficient information to address integrative assessment endpoints.

5.4.3.2 Indicators from Existing Programs

We now turn to some of the most common indicators used in monitoring programs today, recognizing that their suitability in any given application depends on the questions being asked, the characteristics of the natural system being measured, and the practical constraints imposed on the monitoring program.

Hydrologic Indicators. Historically, the effects of urbanization on flow were characterized exclusively in terms of peak flow increases (e.g., Leopold 1968, Hollis 1975). Study since those early works has emphasized the degree to which other attributes of a stream hydrograph are changed by watershed imperviousness, and the importance of assessing the *duration* of moderate flows that are capable of transporting channel sediments and the frequency with which those geomorphically active flows occur

(Section 2). Thus, monitoring relevant to a particular hydromodification management application will likely include a variety of flow metrics (e.g., Konrad and Booth 2005, Degasperi *et al*. 2009).

In moving beyond a narrow focus on linkages between watershed urbanization, flow alteration, and in-stream effects, scientific understanding of hydrologic controls on stream ecosystems has recently led to new approaches for assessing the ecological implications of hydromodification. For example, the ecological limits of hydrologic alteration (ELOHA) framework is a synthesis of a number of existing hydrologic techniques and environmental flow methods that allows water-resource managers and stakeholders to develop socially acceptable goals and standards for streamflow management (Poff *et al.* 2010). The central focus of the ELOHA framework is the development empirically testable relationships between hydrologic alteration and ecological responses for different types of streams. This requires a foundation of hydrologic data provided by gaging and/or monitoring, and sufficient biological data across regional gradients of hydromodification. Although hydrologic–ecological response relationships may be confounded to some extent by factors such as chemical and thermal stressors, there are numerous case studies from the US and abroad in which stakeholders and decision-makers have reached consensus in defining regional flow standards for conservation and ecological restoration of streams and rivers (Poff *et al.* 2010).

Hydrologic monitoring provides essential information needed for establishing flow–geomorphology–ecology relationships, validating conceptual models, and assessing effectiveness of management actions in developing watersheds. Implementing regional flow standards should proceed in an adaptive management context, where collection of monitoring data or targeted field sampling data allows for testing of flow alteration–geomorphic–ecological response relationships. This allows for a fine-tuning of flow management targets based on improved understanding of the actual mechanisms; however, such monitoring can be expensive and it may take many years to adequately characterize the full spectrum of streamflows. Thus, hydrologic monitoring programs should be carefully planned and executed so that they are cost-effective and address the key uncertainties In this paper we primarily focus on indicators that do not require additional, extensive data collection.

Hydrologic indicators provide essential information needed for establishing flow–geomorphology–ecology relationships, validating conceptual models, and assessing effectiveness of management actions in developing watersheds.

Geomorphic indicators have been long-recognized as simple, easy-to-measure, and relatively responsive indicators of changes to the flow regime or sediment supply of a river or stream.

Biological indicators provide an integrative view of river condition, or river health.

Hydrologic monitoring is feasible in the context of a short-term program only if the purpose is to evaluate the engineering performance of a particular facility. For most applications, however, at least two (and commonly many more) years are necessary to measure a range of variable conditions sufficient to capture significant geomorphic and/or biological effects. Measurement of precipitation, generally a less cost-intensive effort than flow monitoring, must occur in consort for the data to be useful. In an effort to minimize the cost of continuous long-term flow modeling, a hydrologic model may be calibrated on one or two years of actual data and then used *in lieu* of further data to predict flow conditions. Whether the level of imprecision so introduced is appropriate will depend on the management questions being asked, but in general such an approach is normally judged more appropriate for comparative results (e.g., did a specified flow magnitude increase in frequency or duration?) than for absolute results (what is the magnitude of the 2-year discharge?).

Geomorphic Indicators. Geomorphic indicators have been long-recognized as simple, easy-to-measure, and relatively responsive indicators of changes to the flow regime or sediment supply of a river or stream (e.g., Leopold 1968). They require little specialized equipment, many commonly can be measured "in the dry" (or close to it), they typically change little from week-to-week (and so are often measured only once per year), and the morphologic features of interest provide the physical template on which a wide range of biological conditions are expressed.

Scholtz and Booth (2000) recognized five geomorphological "channel features" commonly measured as part of monitoring programs:

- Channel geometry (cross sections, longitudinal profile).
- Channel erosion and bank stability.
- Large woody debris.
- Channel-bed sediment.
- In-stream physical habitat (pools, riffles, etc.).

To this list, others have also added:

- Floodplain connectivity.
- Channel planform (meandering, braiding, rates of channel shifting).

Each metric has well-defined methods for field (or, in some cases, airphoto) measurements that need not be repeated here. However, despite broad agreement on *how* to measure each parameter, there is substantially less agreement on the meaning of particular measurements, or indeed under what circumstances (if any) such measurements should be made at all. Most contentious are the various protocols for assessing instream physical habitat (#5 above)—seemingly the most "relevant" for a host of biological applications and for evaluating restoration success. However, a variety of studies have documented a high level of uncertainty imposed by observer bias:

> "Habitat-unit classification was not designed to quantify or monitor aquatic habitat. At the level necessary for use as a stream habitat monitoring tool, the method is not precise, suffers from poor repeatability, cannot be precisely described or accurately transferred among investigators, can be insensitive to important human land-use activities, is affected by stream characteristics that vary naturally and frequently, and is not based on direct, quantitative measurements of the physical characteristics of interest. Relying on habitat-unit classification as a basis for time-trend monitoring is time-consuming, expensive, and ill-advised." (Poole *et al*. 1997, p. 894)

Other geomorphic metrics, in contrast, can provide a robust, albeit coarse, characterization of the channel boundaries. Some changes, particularly if consistently expressed by multiple adjacent cross-sections, can provide clear documentation of systematic channel changes over time that can be credibly associated with upstream changes (e.g., increased discharge from urbanization leading to channel enlargement). Other changes, however, may have a more indirect or uncertain association with upstream conditions (e.g., grain-size changes) because of the potential for rapid, ill-described changes over time without a corresponding human "cause." This emphasizes the importance of having a well-crafted purpose for the monitoring program into which the utility of any chosen parameter can be clearly described.

Biological Indicators. Biological indicators have been long-applied in society's evaluation of stream conditions, but historically that application has been rather informal. Observation of major fish kills, for example, is the application of a "biological indicator," but it provides little diagnostic or discriminatory information except in those streams where conditions are so poor that even casual awareness is inescapable. As a more refined assessment tool, however, their application to freshwater streams is only a few decades old. As such, the science is still under construction and some basic principles are still debated.

The rationale behind using biological indicators, however, is relatively undisputed. Karr (1999) has provided a useful summary of that rationale, of which the key elements are:

- Biological monitoring and biological endpoints provide the most integrative view of river condition, or river health.
- Biological monitoring is essential to identify *biological* responses (emphasis added) to human actions.
- Communicating results of biological monitoring to citizens and political leaders is critical if biological monitoring is to influence environmental policies.

Some of the earliest references to biological monitoring are associated with the development of RIVPACS, the River Invertebrate Prediction and Classification System, developed by the Centre for Ecology and Hydrology in the United Kingdom and now applied in a number of countries worldwide to predict instream biological conditions from a suite of watershed and channel variables. Since that beginning, other approaches have been advanced and practiced (e.g., the US Environmental Protection Agency's *Rapid Bioassessment Protocols*) that provide alternative, but likely near-equivalent results (e.g., Herbst and Silldorf 2004).

In this section we compare several biological indicators recently applied in various regions of California. This not intended as a comprehensive comparison of all available approaches potentially applicable to California; rather, it simply provides a few examples that illustrate the differences, and the similarities, of the various approaches. As the comparisons demonstrate, there is no "right" approach—but all share commonalities that are likely to be valuable elements of any biological monitoring program. We focus exclusively on benthic macroinvertebrates (BMI), because these have seen the longest and most widespread application (both in California and worldwide) given their species diversity and their relative geographic immobility. However, a variety of other biological metrics (particularly fish and periphyton) have relevance to biological monitoring and strong advocates in the scientific community. Their omission here is not a judgment on their value, merely a reflection of the broader applicability and richer scientific development of BMI-based indicators.

Multimetric indices are presently completed for four areas of the state (Eastern Sierra, North Coast, Central Valley, and Southern Coast). They are not standardized or calibrated state-wide (nor should they necessarily be), and they do not provide statewide coverage. In addition, the City of Santa Barbara (Ecology Consultants 2010) has sponsored development of its own BMI index (geographically embedded within the Southern Coast region), with both commonalities and differences between it and the others.

Eastern Sierra Nevada. Herbst and Silldorf (2009) developed an IBI based on streams from the upper Owens River north to the Truckee River. Their purpose was both to provide a region-specific IBI for future use and to evaluate the results of such an approach with others that also make use of BMIs to assess stream conditions. They evaluated the performance of 12-, 10-, and 8-metric indices, recommending the 10-metric index as providing the best overall performance included in the 12-metric index were these 10 and also predator richness and EPT% abundance:

- % tolerant percent richness (% of taxa with TV= 7,8,9,10).
- Richness (total number of taxa).
- Chironomidae Percent Richness (% of taxa that are midges).
- Ephemeroptera (E) Richness (number of mayfly taxa).
- Plecoptera (P) Richness (number of stonefly taxa).
- Trichoptera (T) Richness (number of caddisfly taxa).
- Dominance 3 (proportion of 3 most common taxa)
- Biotic Index (modified Hilsenhoff, composite tolerance).
- Acari richness (number of water mite taxa).
- Percent shredders (% of total number that are shredders).

A statistical analysis suggests that as many as 10 distinct classes can be discriminated using this IBI, although their recommended application uses only five categories of quality.

North Coast. Rehn *et al.* (2005) developed an IBI based on coastal-draining streams from Marin County north to the Oregon border. They evaluated 77 individual metrics, testing them for responsiveness to human disturbance and redundancy, and ultimately settled on eight:

- EPT richness.
- Coleoptera richness.
- Diptera Richness.
- Percent intolerant individuals.

- Percent non-gastropod scraper individuals.
- Percent predator individuals.
- Percent shredder taxa.
- Percent non-insect taxa.

Their statistical analysis indicated that five categories of quality could be discriminated; response was driven most strongly by watershed land cover (natural vs. unnatural) and percent of substrate that was sand-sized or finer. They also suggested a set of thresholds for rejecting potential "reference" sites (Rehn *et al.* 2005; Table 5-2), which was also used in the Southern Coast study (Ode *et al.* 2005; see below):

Table 5-2. Thresholds for rejecting potential "reference" sites.

Stressor	Threshold
Percentage of unnatural land use at the local scale	> 5%
Percentage of urban land use at the local scale	> 3%
Percentage of total agriculture at the local scale	> 5%
Road density at the local scale	> 1.5 km/km^2
Population density (2000 census) at the local scale	> 25 ind./ km^2
Percentage of unnatural land use at the watershed scale	> 5%
Percentage of urban land use at the watershed scale	> 3%
Percentage of total agriculture at the watershed scale	> 5 %
Road density at the watershed scale	> 2.0 km/km^2
Population density (2000 census) at the watershed scale	> 50 ind./ km^2

Central Valley. Rehn *et al.* (2008) also developed an IBI for Central Valley streams, evaluating 80 candidate metrics to yield a final list of five:

- Collector richness.
- Predator richness.
- Percent EPT taxa.
- Percent clinger taxa.
- Shannon diversity (a composite measure of taxonomic richness and evenness of abundance).

They found that reach-scale physical habitat variables were more critical in their data set than water chemistry or land use. They also presented their findings with greater caution than with other regions of the state, noting the difficulty of identifying truly "unimpaired" reference conditions and the geographic concentration of much of their source data.

Southern Coast. Ode *et al.* (2005) developed a BMI index of biological integrity based on 61 potential metrics from reference sites drawn from relatively undisturbed coastal-draining watersheds from Monterey Bay south to the Mexican border. They included seven final metrics:

- Percent tolerant taxa.
- Percent collector-gatherer + collector-filterer individuals.
- Predator richness.
- Percent intolerant individuals.
- EPT richness.
- Percent noninsect taxa.
- Coleoptera richness.

They note that the last two on the list are not common in other multimetric B-IBIs but were statistically appropriate for their data set. They judge that this "SoCal B-IBI" can discriminate 5 categories of condition, using 5 categories evenly divided along a 100-point scale. Particularly strong correlations amongst all seven metrics were displayed in comparison to road density and percent "watershed unnatural."

A portion of the Southern Coast region has also been the subject of independent IBI development over the past decade (Ecology Consultants 2010, 2011). The region of study spans the Santa Barbara coastal streams from the Ventura County line west about 45 miles to Gaviota Creek. Their work led to the development of an IBI using the following 7 metrics:

- # of insect families
- # of EPT families
- % EPT minus Baetidae
- % PT
- Tolerance value average
- % sensitive BMIs
- % predators + shredders

In the course of this work, tolerance values were adjusted for certain taxa based on local observations of presence/absence relative to the level of watershed disturbance. With these changes, they found strong statistical basis for discriminating five categories of biological quality. They also found that considering both watershed-level land use patterns and localized physical habitat conditions were necessary to achieve the best prediction of biological integrity.

Summary. A compilation of the various metrics (Table 5-3) demonstrates only broad commonalities between the various regional IBI's presently available for specific parts of California, suggesting that additional work needs to be done before comprehensive recommendations for biological monitoring can be made. At present, perhaps half(?) of the state's area is covered by existing multimetric indices as noted above, and for these areas they provide the best (indeed, the only) guidance for meaningful collection and interpretation of biological data. Elsewhere, however, only a few general points can be made:

- Biological monitoring in un-assessed regions of the state cannot be used to identify absolute conditions of biological health (i.e., "status" monitoring). However, they will likely be useful for "trends" monitoring, where only the change relative to a prior state is being sought.
- Despite the variability in metric choices amongst the various regions (Table 5-2), some broad commonalities are apparent. In particular, several types of metrics are likely to provide useful indicators of change in a known direction (i.e., an increase or decrease in the metric can be confidently assigned to a change in quality in a known direction):
    - One or more measures of tolerance or intolerance
    - One or more measures of predator prevalence
    - One or more measures of EPT taxa or taxa richness

This list does not purport to describe a true multimetric B-IBI, nor to provide a basis to evaluate instream biological health on an absolute scale (i.e., from "poor" to "excellent"). In the absence of any region-specific guidance, however, changes in one or more of these metrics are each likely to provide some initial, useful indication of temporal trends in biological health until such time as the types of studies referenced above can be conducted.

Table 5-3. Compilation of metrics used in the five regional B-IBI's described in the text.

METRIC	Eastern Sierra	North coast	Central Valley	Southern coast	Santa Barbara
Percent intolerant individuals		X		X	X
% tolerant (% of taxa with TV= 7,8,9,10)	X			X	
Tolerance value average					X
# of insect families					X
Percent non-insect taxa		X		X	
Percent shredders (% of total number that are shredders)	X	X			
Percent predator individuals		X			
% predators + shredders					X
Predator richness			X	X	
Collector richness			X		
Percent non-gastropod scraper individuals		X			
Percent clinger taxa			X		
Percent collector-gatherer + collector-filterer individuals				X	
EPT richness		X		X	X
Percent EPT taxa			X		
% EPT minus Baetidae					X
% PT					X
Ephemeroptera (E) Richness (number of mayfly taxa)	X				
Plecoptera (P) Richness (number of stonefly taxa)	X				
Trichoptera (T) Richness (number of caddisfly taxa)	X				
Coleoptera richness		X		X	
Diptera Richness		X			
Chironomidae Percent Richness (% of taxa that are midges)	X				
Richness (total number of taxa)	X				
Dominance 3 (proportion of 3 most common taxa)	X				
Biotic Index (modified Hilsenhoff, composite tolerance)	X				
Acari richness (number of water mite taxa)	X				
Shannon diversity index			X		

5.5 Recommendations

Based on this review of monitoring theory, current applications, and current needs, the following steps are recommended to advance a state-wide program of monitoring to support the management of hydromodification control plans.

5.5.1 Programmatic Monitoring

Over the next several years, the following actions should be implemented at the state and/or regional level:

- Executing broad-scale, GIS-based watershed characterization;
- Identifying a set of representative indicator watersheds, and a basic suite of regular measurements that are suitable for establishing trends in physical, chemical, and biological indicators;
- Identifying (and multi-metric monitoring within) a relatively small set of watersheds that have implemented recent hydromodification control plans to initiate the long-term evaluation of downstream trends.

Over the course of the next several NPDES permit cycles (i.e., one or more decades), the following actions should also be undertaken as a regional responsibility:

- Setting regionally appropriate endpoints for biological health of receiving waters;
- Identifying particularly promising (or particularly ineffective) combinations of control strategies across a range of different landscape conditions;
- Providing supplemental data collection at reference sites to support trends monitoring by local jurisdictions;
- Compiling local results to guide development and refinement of regionally appropriate hydromodification control strategies.

5.5.2 Local Monitoring

Over the next several years, the following actions should be implemented by local jurisdictions at a local scale:

- Implementing a program of source identification at one or more high-risk locations (e.g., high vehicular traffic, high imperviousness, toxic chemical storage/transport);
- Demonstrating the hydrologic performance of one or more representative hydromodification control facilities;
- Monitoring trends at one or more representative receiving waters, ideally at a regionally identified site (see the second bullet under “Programmatic monitoring,” above);

- Conducting a synoptic evaluation of waterbodies, stratified by watershed type (see the first bullet under “Programmatic monitoring,” above), to identify highest priority systems for protection or rehabilitation, if not already known.

Over the course of the next several NPDES permit cycles, the following long-term actions should also be undertaken as a local responsibility:

- Monitoring representative conditions to evaluate whether management actions are improving overall receiving-water health;
- Evaluating cost-effectiveness of implemented hydromodification control measures;
- Identifying critical areas for resource protection by virtue of existing high-quality conditions.

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APPENDIX A: GUIDANCE FOR APPROPRIATE APPLICATION OF HYDROLOGIC AND HYDRAULIC ANALYSES

Information contained in this document is intended solely for guidance purposes only. It is not intended to be an instruction manual and use of any of the guidance provided herein is at the risk of the user. No other person or entity shall be entitled to rely on the services, opinions, recommendations, plans or specifications provided in the document.

1. INTRODUCTION AND PURPOSE

The purpose of this Appendix is to provide technical guidance on hydrologic and hydraulic analyses, including the use of Continuous Simulation (Hydrologic) Modeling (CSM), in support of hydromodification assessment and mitigation. CSM is the industry standard developed since the early 2000s for use in the assessment and mitigation of hydromodification. The fundamental difference between CSM and peak flow hydrologic modeling, is that CSM considers the full range of flow events over a long period of record, typically 30 years or more, to develop flow duration curves, whereas peak flow hydrologic modeling generally considers synthetically (usually calibrated to measured data) produced event-based hydrographs (2-, 10-, 50-, 100- and 200-year return frequency events). CSM allows flow duration curves and other derived hydraulic metrics to be compared between existing and proposed conditions in order to assess hydromodification impact potential and to develop mitigation strategies. The guidance provided in this appendix is the product of the experience gained in the application of hydromodification management strategies to multiple urban development projects. This appendix is not intended to be an instruction manual but to provide guidance to engineers, planners and regulatory staff on specific modeling elements involved with HMPs.

MODELING METOHDOLOGY REVIEW

Modeling Approaches

A common approach to mitigating hydromodification impacts from development projects is to construct best management practices (BMPs) which capture, infiltrate and retain runoff, where possible. In such cases, the water is detained and released over a period of time at rates which more closely mimic pre-project hydrology. Methods commonly used to size hydromodification BMPs include hydrograph matching (matching pre and post-project flow regimes), volume control and flow duration control. Hydrograph matching is most traditionally used to design flood detention facilities for a specific storm recurrence interval, such as the 100-year storm, whereby the outflow hydrograph for a project area matches the pre-project hydrograph for a design storm. Volume control matches pre- and post-project runoff volume for a project site; however, the frequency and duration of the flows are not controlled. This can result in higher erosive forces during storms. Flow duration control matches both the duration and magnitude of a range of storm events for pre- and post-project runoff. The complete hydrologic record is taken into account, and runoff magnitudes and volumes are matched as closely as possible.

It is generally accepted that flow duration control matching is the most appropriate method to be used in the design of hydromodification BMPs. The flow duration control approach has been used in at least half a dozen HMPs in California, all of which used a CSM to match flow durations. However, differences exist in how the continuous simulation modeling is used between programs.

OVERVIEW OF APPENDIX

This appendix covers the following specific topics, addressed in the order in which they would arise as part of a hydromodification analysis for a major development project:

Section 2 addresses calculation of a flow control range, including identification of an acceptable low flow value, based upon critical flow for incipient motion of the channel material. .

Section 3 addresses the development of evaluation criteria to assess the effectiveness of a proposed mitigation design, including a discussion of flow duration matching and the erosion potential metric.

Section 4 addresses CSM, including precipitation data requirements, hydrologic time steps, model calibration and validation, and other modeling considerations and tips.

2. METHOD FOR SELECTION OF A FLOW CONTROL RANGE

INTRODUCTION TO FLOW CONTROL

Most hydromodification plans (HMPs) in California have adopted a flow control approach, which establishes a range of flow magnitudes discharging from the proposed site that must be controlled. The magnitude of the flow range is commonly expressed in terms of a percentage of the return period flow to which it is equivalent; for example: from 10% of the Q2 to 100% of the Q10. Flow magnitudes within the prescribed range must not occur more frequently under the proposed condition than they do in the existing (or pre-project) condition. Another way of expressing this is that the long term (decadal) cumulative duration of these flows must not be longer in the post-project condition compared to the pre-project condition. Generally, a small exceedance tolerance is allowed. For example, the following is a typical criterion that has been used in HMPs:

> For flow rates ranging from 10% of the pre-project 2-year recurrence interval event (XQ2) to the pre-project 10-year runoff event (Q10), the post-project discharge rates and durations shall not deviate above the pre-project rates and durations by more than 10% over and more than 10% of the length of the flow duration curve. The specific lower flow threshold should be influenced by results from the channel susceptibility assessment.

The rationale behind setting an upper limit is the understanding that when less frequent, high intensity/volume precipitation events occur, the watershed reaches a saturation level and responds in a similar manner for undeveloped and developed conditions. Furthermore, while these less frequent, high magnitude events do induce significant geomorphic change, they occur so infrequently that over a long time period, they comprise only a small portion of the work done on a channel. For example GeoSyntec (2007) used a hydro-geomorphic model to assess cumulative sediment transport on Laguna Creek (near Sacramento) and determined that 95% of the total erosion and sediment transport in the creek is accomplished by flow rates less than Q_{10}.

The purpose of determining a low flow range is one of practical design consideration when meeting a requirement for flow duration matching. The requirement to match flow durations between a pre- and post-project condition requires that runoff be detained and infiltrated within a BMP (e.g. open basin or underground vault). If flow matching is required to be achieved for all flows down to zero, the BMP volume will be significantly larger (and therefore more costly) than if there were some low flow below which runoff could be discharged at durations longer than in the pre-project condition. A key assumption underlying the concept of a low-flow discharge is that the increase in discharge durations below this rate will not increase channel erosion because the flows are too small to initiate movement of channel materials to any significant extent. Another critical assumption in the flow duration matching approach is that a single discharge value is valid across the range of grain sizes and geometries in the streams to which that low flow value applies.

For a specific set of hydraulic conditions (e.g., cross sectional shape, channel slope, bed and bank roughness), the flow rate can be calculated where the critical shear strength value is reached. Thus with an estimate of the critical shear strength of the materials composing a channel's bed or banks, and the hydraulic conditions occurring at the same location, the critical flow rate can be determined at which transport (or erosion) begins. This critical flow rate (Q_c) can then be compared to the magnitude of a flood peak which occurs every two years (Q_2) to establish the estimate of percent Q_2 to be used as the lower flow threshold.

Thus in order to calculate the lower flow threshold as expressed by a percentage of Q_2, three values must be determined for each analysis location (described in further detail below):

- The critical shear strength (τ_c) of bed and bank materials;
- The critical flow rate (Q_c) at which this critical shear strength is reached and exceeded;
- The magnitude of a flood peak which occurs every two years (Q_2).

In contrast, when using an erosion potential (Ep) metric (rather than flow duration matching) for BMP sizing, the Ep analysis incorporates channel geometry to estimate shear stresses generated at various flow rates, and then compares these to estimated critical shear stresses (i.e., shear stress required to initiate transport) for the grain size distribution within the stream. However, for either flow duration matching or for erosion potential analysis, the first step is to determine the critical shear stress for incipient motion of channel materials.

DETERMINATION OF CRITICAL SHEAR STRESS

The composition and condition of the bed and banks of a stream channel are the best indicators of how a channel will react (i.e., its susceptibility) to hydrologic changes resulting from development projects (i.e., hydromodification). Channels composed of materials more resistant to erosion are less susceptible to excessive erosion due to hydromodification than channels composed of less resistant materials. Channel material type can vary widely between, as well as within, watersheds. Figure 2-1 **Error! Reference source not found.**a. and b. illustrate stream incision through (a) relatively loosely consolidated, non-cohesive sand and gravels, and (b) relatively cohesive silty-clays. The resistance of bed and bank materials is quantified by their critical shear strengths, (τ_c) that is, the value where entrainment or transport begins.

Typical stream erosion in a southern California stream (granular, non-cohesive
(Photo courtesy of Eric Stein, SCCWRP)

Typical stream erosion in a northern California stream (generally cohesive silty clays)
(Photo cbec, inc.)

Notes:		*Guidance on the Use of Continuous Simulation Hydrologic Modeling* **Examples of Stream Erosion in Southern and Northern California**		
		Project: 11-1001	Created By: CBB	**Figure 2-1**

Figure 2-1. a. Example of a loosely consolidated, non-cohesive sand and gravel stream bed. b. Example of a relatively cohesive silty-clay stream bed.

Several methods are available for the estimation of critical shear stress, including laboratory studies (e.g., flume studies) and field measurements, with different methods utilized for cohesive materials and non-cohesive materials.

Estimating Critical Shear Stress for Non-Cohesive Materials

The most common method for determining the critical shear stress of a non-cohesive material is through the application of the Shields relationship. This relationship is applicable to the calculation of critical shear stress for a uniform size mixture of sediment with a known particle size and specific gravity. Since it was originally proposed by Shields in 1936, the relationship has been tested and further investigated by several other researchers, resulting in a variety of modifications, primarily through variation of the Shields parameter. The original value of the Shields parameter proposed by Shields was 0.06, however, values from 0.03-0.06 have been suggested, with 0.045 acknowledged as a good approximation. Recent research has demonstrated that a value of 0.03 may be more appropriate for estimating incipient motion in streams with gravel beds (Neill 1968, Parker et al. 2008, Wilcock et al. 2009), where D50 estimates are based upon data collected via pebble count. The decision of what value of Shields parameter is used can have a large influence on the resulting τ_c estimate. For example, if a value of 0.06 is used, it results in twice as large of an estimate of τ_c than if a value of 0.03 is used.

While the Shields relationship was developed for a mixture of uniform sized sediment, it can be applied to a mixture of sediment with varying sizes as long as the distribution is uni-modal and does not have a high standard deviation of grain sizes (Wilcock 1993). In contrast, for sediment mixtures which are bimodal (e.g., if there is a large amount of sand in addition to gravel), a different approach (e.g., Wilcock and Crowe 2003) is recommended. For a more in depth discussion of sediment transport and incipient motion, the reader is referred to Wilcock et al. (2009).

In order to apply the Shields relationship to determine τ_c, the median grain size (d50) present on the channel surface must be determined. River channels are often armored; meaning that coarser material is present on the surface than is present underneath the armor layer. However to access and transport the finer material beneath, the surface layer must first be mobilized. The median grain size is determined by analysis of a particle size distribution.

A particle size distribution can take the form of: 1) a cumulative *frequency* distribution which is determined by way of a pebble count or photographic analysis, or 2) a cumulative *weight* distribution. For a cumulative frequency distribution a subset of particles present on the surface are measured, and the frequency of particles within different size class bins is used. **Error! Reference source not found.** shows a sample particle size distribution graph developed from a pebble count. For a cumulative weight distribution, a bulk sample of the surface material is collected, and then sorted using a set of sieves with different screen sizes. The amount of material retained by each sieve is weighed and then used to plot the cumulative weight distribution. Both approaches have advantages and disadvantages.

A pebble count is a relatively straightforward field technique that is easily applied in streams which are wadable. **Error! Reference source not found.** shows photographs of pebble counts being conducted in the field. They can be performed relatively quickly, which means more samples can be collected to better characterize the conditions present in a reach. However, there are a variety of ways a pebble count can be conducted, and there is tremendous opportunity to introduce bias to the measurement. Furthermore, while studies often cite Wolman (1954) as the method employed in data collection, strict adherence to this protocol is not always achieved. Rather than the method suggested by Wolman (1954), a refined, more regimented approach has been suggested by Bundte and Abt (2001a), and is recommended. In addition, it should be noted that pebble counts generally do a poor job of characterizing sand and smaller sized material. In addition to pebble counts, software can be used to process a digital image of an area of the bed. The software samples a subset of particles present in the image, and using assumptions regarding the amount of given particle that is visible, is able to provide a cumulative frequency distribution.

Collecting a bulk sample for sieve analysis is another method frequently employed to determine values for typical characteristic indices of a particle size distribution. In this method a sample is collected from the channel surface, and then the sample is segregated into various size classes with sieves. One advantage of this approach is that it utilizes all the data available from the sampled area (as opposed to a pebble count which uses a subset of the entire population, e.g., ~100 particles as opposed to thousands), however the sampled area is typically smaller than the area sampled within one pebble count. One disadvantage is the size of sample that is necessary. Because the resulting particle size distribution is based upon weight, the largest particles present can have a very large influence on the resulting particle size distribution. Research has suggested that the weight of the entire sample must exceed 100x the weight of the largest particle present to escape this possible bias. This means large (volume and weight) samples are often required. Some sieving can occur on site through the use of shaker sieves, but typically some portion of the sample is also taken back to the lab for further analysis. Thus, bulk samples typically require more effort and equipment to establish a particle size distribution, however they provide a much more accurate estimate, especially when a large fraction of the sample is sand sized (2mm) and smaller.

For a more in depth discussion of sampling methods to determine particle size distributions in wadable streams, the reader is referred to Bunte and Abt (2001).

Estimating Critical Shear Stress for Cohesive Materials

The methods described above are not appropriate for cohesive materials, which due to chemical cohesion between particles exhibit larger τ_c values than would be estimated by consideration of particle size/weight in isolation (i.e., cohesive properties not considered). One method that allows for the determination of τ_c *in situ* is the application of a jet test (ASTM 2007). The jet-testing apparatus and analytical methods were developed by researchers at the USDA Agricultural Research Station (Hanson and Cook 1999; Hanson et al. 2002; Hanson and Cook 2004; ASTM 2007). The method uses a submerged impinging jet of water directed perpendicularly at the material surface, in order to erode the material. As erosion occurs, a scour hole is created. The depth of this hole is measured periodically as time progresses through the test. As the scour hole increases in depth, the strength of the jet is reduced because it is travelling longer distance through water from the jet orifice to the soil surface. Eventually, the energy of the jet is dissipated enough that it no longer has energy in excess of the material's shear strength and erosion stops. **Error! Reference source not found.** shows a photograph of a jet testing rig deployed in a stream bank.

In addition to jet testing, *in situ* testing of shear strength can be obtained through the application of a field vane shear test (ASTM 2008). This method provides τ_c values based upon the assumption that the bed or bank will fail via large blocks (composed of thousands of particles), as opposed to erosion occurring particle by particle. As such, the values measured by a shear vane are often several orders of magnitude larger than those obtained via testing with the jet-device.

Estimating Critical Shear Stress Through the Use of Literature Values

An alternative to the measurement/calculation of τ_c, is the use of values found in the literature. Indeed, several HMPs have several HMPs have been developed through assumption of material resistance properties found in the literature based upon literature based upon a textural description of the material. An often-cited reference is Fischenich (2001), which provides a (2001), which provides a summary (compiled from the relevant literature) for critical shear strength values for various values for various materials. An extract from this reference is provided in

Figure 2-5.

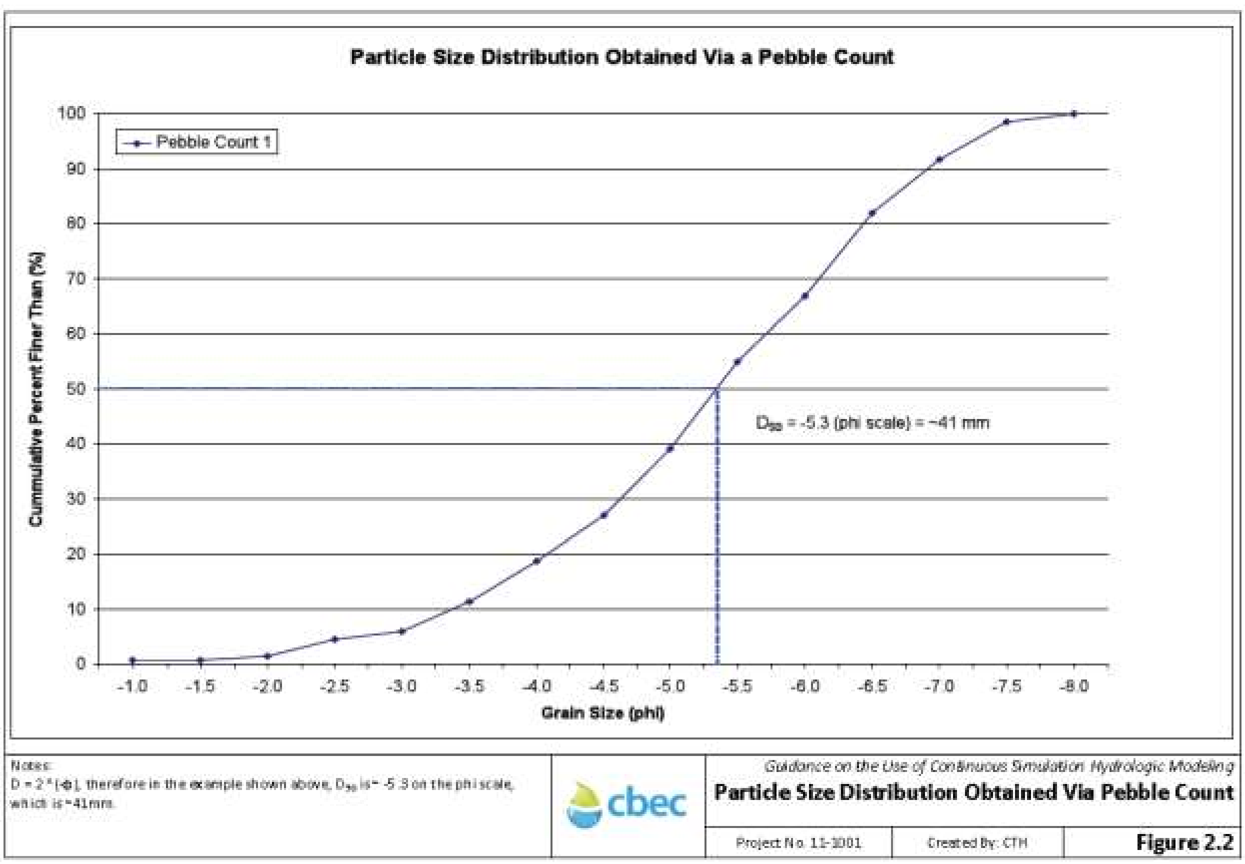

Figure 2-2. Particle Size Distribution Graph Developed from a Pebble Count

Notes:	cbec	*Guidance on the Use of Continuous Simulation Hydrologic Modeling* **Pebble Counts Being Conducted in the Field**		
		Project: 11-1001	Created By: CBB	**Figure 2-3**

Figure 2-3. Pebble Counts Being Conducted in the Field

Typical installation of jet testing equipment in stream bank

Hole created in cohesive bank material by jet impinging on surface

Notes:		*Guidance on the Use of Continuous Simulation Hydrologic Modeling* **Jet Testing Equipment Deployed in a Stream Bank**		
		Project: 11-1001	Created By: CBB	**Figure 2-4**

Figure 2-4. Jest Testing Equipment Deployed in a Stream

Table 2. Permissible Shear and Velocity for Selected Lining Materials[1]

Boundary Category	Boundary Type	Permissible Shear Stress (lb/sq ft)	Permissible Velocity (ft/sec)	Citation(s)
Soils	Fine colloidal sand	0.02 - 0.03	1.5	A
	Sandy loam (noncolloidal)	0.03 - 0.04	1.75	A
	Alluvial silt (noncolloidal)	0.045 - 0.05	2	A
	Silty loam (noncolloidal)	0.045 - 0.05	1.75 – 2.25	A
	Firm loam	0.075	2.5	A
	Fine gravels	0.075	2.5	A
	Stiff clay	0.26	3 – 4.5	A, F
	Alluvial silt (colloidal)	0.26	3.75	A
	Graded loam to cobbles	0.38	3.75	A
	Graded silts to cobbles	0.43	4	A
	Shales and hardpan	0.67	6	A
Gravel/Cobble	1-in.	0.33	2.5 – 5	A
	2-in.	0.67	3 – 6	A
	6-in.	2.0	4 – 7.5	A
	12-in.	4.0	5.5 – 12	A
Vegetation	Class A turf	3.7	6 – 8	E, N
	Class B turf	2.1	4 – 7	E, N
	Class C turf	1.0	3.5	E, N
	Long native grasses	1.2 – 1.7	4 – 6	G, H, L, N
	Short native and bunch grass	0.7 - 0.95	3 – 4	G, H, L, N
	Reed plantings	0.1-0.6	N/A	E, N
	Hardwood tree plantings	0.41-2.5	N/A	E, N
Temporary Degradable RECPs	Jute net	0.45	1 – 2.5	E, H, M
	Straw with net	1.5 – 1.65	1 – 3	E, H, M
	Coconut fiber with net	2.25	3 – 4	E, M
	Fiberglass roving	2.00	2.5 – 7	E, H, M
Non-Degradable RECPs	Unvegetated	3.00	5 – 7	E, G, M
	Partially established	4.0-6.0	7.5 – 15	E, G, M
	Fully vegetated	8.00	8 – 21	F, L, M
Riprap	6 – in. d_{50}	2.5	5 – 10	H
	9 – in. d_{50}	3.8	7 – 11	H
	12 – in. d_{50}	5.1	10 – 13	H
	18 – in. d_{50}	7.6	12 – 16	H
	24 – in. d_{50}	10.1	14 – 18	E
Soil Bioengineering	Wattles	0.2 – 1.0	3	C, I, J, N
	Reed fascine	0.6-1.25	5	E
	Coir roll	3 - 5	8	E, M, N
	Vegetated coir mat	4 - 8	9.5	E, M, N
	Live brush mattress (initial)	0.4 – 4.1	4	B, E, I
	Live brush mattress (grown)	3.90-8.2	12	B, C, E, I, N
	Brush layering (initial/grown)	0.4 – 6.25	12	E, I, N
	Live fascine	1.25-3.10	6 – 8	C, E, I, J
	Live willow stakes	2.10-3.10	3 – 10	E, N, O
Hard Surfacing	Gabions	10	14 – 19	D
	Concrete	12.5	>18	H

[1] Ranges of values generally reflect multiple sources of data or different testing conditions.

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Notes:	cbec	*Guidance on the Use of Continuous Simulation Hydrologic Modeling* **Permissible Shear Strength and Velocity for Selected Lining Materials**		
		Project No: 11-1001	Created By: CBB	**Figure 2-5**

Figure 2-5. Permissible Shear and Velocity for Selected Lining Materials

DETERMINATION OF CRITICAL FLOW (Q_c)

For a specific set of hydraulic conditions at a location (i.e., cross sectional shape, channel slope, bed and bank roughness), the flow rate at which critical shear values are reached can be calculated. These calculations can be made with a programed spreadsheet analysis, or with a hydraulic model (e.g., HEC-RAS, Brunner 2010). Because of their ease of use and the ease at which multiple flow rates can be assessed (in order to determine when τ_c is reached), hydraulic models are typically employed for this part of the analysis. Average boundary shear stress is calculated with the following equation:

$$\tau = \rho g R s$$

where *p* represents the density of water, *g* represents the gravitational constant, R represents the hydraulic radius (defined as the wetted area dived by the wetted perimeter), and *s* represents the slope. For wide channels the value of the hydraulic radius is approximately equal to the average depth of the cross section. The hydraulic model calculates the value for R for a given discharge based on the channel dimensions.

Typically one-dimensional approximations are used for this analysis, which means that the value of Q_c determined is that where the cross sectional average of τ_c is reached, not the highest value which is occurring at the deepest point of the cross section. This is typically considered reasonable because the grain size is determined for the bed of the cross section, not just the shallow or deep area.

Analyses can be conducted at a station, or in other words just looking at one cross section in isolation using normal depth calculations, or within a larger hydraulic model constructed for the entire reach (i.e., multiple distributed cross sections upstream and downstream of the location of interest). The advantage of looking at the cross section of interest within the context of the entire reach is that conditions downstream (e.g. a constriction which causes a backwater condition) may affect the flow depth (or hydraulic radius), yielding different results than would be obtained if the cross-section was analyzed in isolation.

It is important that the determination of τ_c (via pebble count or other means) and the hydraulic calculations to determine Q_c, occur at the same location. Typically the analysis is undertaken at a riffle because these are the high points of a long profile and are what are controlling incision in the system. Bed material characterization in a pool is much more difficult (because of the depth of water), in addition the resulting calculated shear values are typically much higher, because of the added depth.

If HEC-RAS is used (which is typical), the way the bank markers are set can have a dramatic influence on the calculated shear results. The bank markers are used to delineate differences in roughness across the channel and flood plain (typically higher values are used on the lateral margins to include the influence of vegetation roughness in the resulting depth calculations). The shear values calculated by HEC-RAS are segregated by these bank markers, and thus may include values for each of the floodplains as well as the channel. If bank markers are set too wide, and the shear stress calculation may include a portion of the floodplain too, and subsequently the conditions in the actual channel will be greatly underestimated. Remember that the model is essentially using the average depth for the entire cross section (as limited by the bank markers), so including floodplain with shallow depths greatly influences the average depth and thus the resulting calculated shear value.

DETERMINATION OF Q_2

The determination of a value of Q_2 is the third and final piece of the equation used to determine what percent of Q_2 the lower threshold should be. As with the other two pieces, several options are available, and again the decision on what method is used can have a profound influence upon the final results. Q_2 can be determined through the results of a calibrated and validated hydrologic model (e.g., HEC-HMS, HSPF, SWMM, etc.) which uses precipitation, sub basin area, soil conditions, etc. to calculate a runoff hydrograh. This type of model can be used in one of two ways, to simulate a single precipitation event or to simulate a long term (e.g., 50 year) precipitation record. The first approach produces a single runoff hydrograph resulting from a "design" storm, from which the peak magnitude can be determined. As such the results are largely controlled by the precipitation hyetograph, so a good understanding of how that was developed is important. This method has been used considerably less than the approach detailed below. The advantage of this method is that, if any existing model has already been developed (e.g., SacCalc; DFCE 2001), it will be cheaper and easier for an agency to review. However, it can yield different values for Q_2, due to differing assumptions employed in the modeling.

The second method uses a long-term precipitation record for simulation which results in a flow record containing a large number of runoff events of varying magnitudes (i.e., which are subsequently analyzed to determine the magnitude of the 2 year recurrence interval event). This method is more typical for HMP assessments, but again methodical decisions can have a large influence on the results. The rigor of the model calibration and validation has a strong influence. If the model is not representing through simulation what is actually occurring, then the simulation results are questionable.

Assuming the model has been calibrated and satisfactorily validated or verified, the manner in which the simulated runoff record is analyzed is important. The first basic distinction is whether an annual maximum series (AMS) or a partial duration series (PDS) is used. In an AMS analysis, just the single largest flood peak of any given year is used in the analysis, and the second and third largest events of the year are ignored. This is the method typically utilized when analyzing the flood frequency of large, less frequently occurring flood events. In the second approach, PDS, multiple flood events are considered in any given year. This is important when the second or third largest flood events in one year are greater than the annual maximum of another year. Because more large events are included, the resulting estimate of the given return period event (e.g., Q_2) is larger. For example, Langbein (1960) showed that a 1.45 year event determined with PDS is the same magnitude as a 2 year event with an AMS, and a 2 year event determined with PDS is a 2.54 year event with an AMS. Thus the value of Q_2 determined by PDS is larger than the value of Q_2 determined by AMS. While significant differences are apparent for smaller magnitude, more frequently occurring events (e.g., Q_2), for return periods greater than 10 years, there is almost no difference between the results obtained from the AMS and PDS.

When compiling a PDS for a recurrence interval analysis, the manner in which events are identified as independent can also have an effect upon the results. One typical method is to include all flood peaks above a certain base magnitude. This base value is often selected as equal to the lowest annual maximum flood of record, however can also be chosen such that the PDS only contains as many peaks as there are years of record. Some analysts have established a base value (e.g., 0.002 cfs/acre), and then added a duration below this base value as well (i.e., flow must be below 0.002 cfs/acre for at least 24 hours for events to be considered independent). One additional method is to identify individual events by extracting the highest peak (not just the maximum value) within a moving time window (e.g., 3 days), and therefore determine independence through time, rather than the discharge rate receding to a non-storm condition. With all of these options available, and no prescribed standard, the use of a PDS can have different Q_2 results even if an identical flow time series is used.

SUMMARY

The determination of the lower flow threshold, defined as a percentage of Q_2, is heavily influenced by three primary inputs: τ_c, Q_c, and Q_2. The determination of each of these values is sensitive to a variety of factors determined by the particular methodology. To demonstrate the sensitivity of the lower flow threshold to methodological decisions, a few examples are provided below.

- If 0.06 is used rather than 0.03 for Shields parameter in Shields relationship, τ_c increases, subsequently Q_c increases and ultimately the lower limit increases
- If bank markers are set too wide (including the floodplain and not just the channel) in the hydraulic analysis, a larger value for Q_c is calculated (because of a reduction of the hydraulic radius due to the inclusion of extensive shallow floodplain areas), resulting in an increase of the lower limit.
- If an annual maximum series is used in place of a partial duration series, the calculated Q_2 will be less than that obtained by a PDS analysis, and the ratio of Q_c to Q_2 will be higher if the AMS is used.

3. DEVELOPMENT OF EVALUATION CRITERIA

FLOW DURATION CONTROL AND PEAK FLOW CURVE MATCHING

Flow Duration Control (FDC) and Peak Flow Curve (PFC) matching criteria in their current form for many counties in CA are similar in form to the curve matching criteria from WA (WADOE, 2001). The curve matching criteria typically include a goodness of fit or variance due to the difficulty in achieving a precise match across the range of flows. The criteria are typically applied at the subwatershed scale based on continuous simulation flow results for pre- and post-project conditions to size individual BMP or LID features. In this instance, flow matching at the subwatershed scale assumes that there are no routing or timing effects in the treated runoff when it rejoins the receiving waterbody; however, this may not be true in all cases. For example, if treated runoff is delayed and rejoins the upstream runoff such that there is an increase in flow rates and durations or an increase in the peak flows in the receiving waterbody, then there is the potential to impair the receiving waterbody. To address this potential concern, the FDC and PFC criteria could be applied to the routed flows in the receiving waterbody as a check.

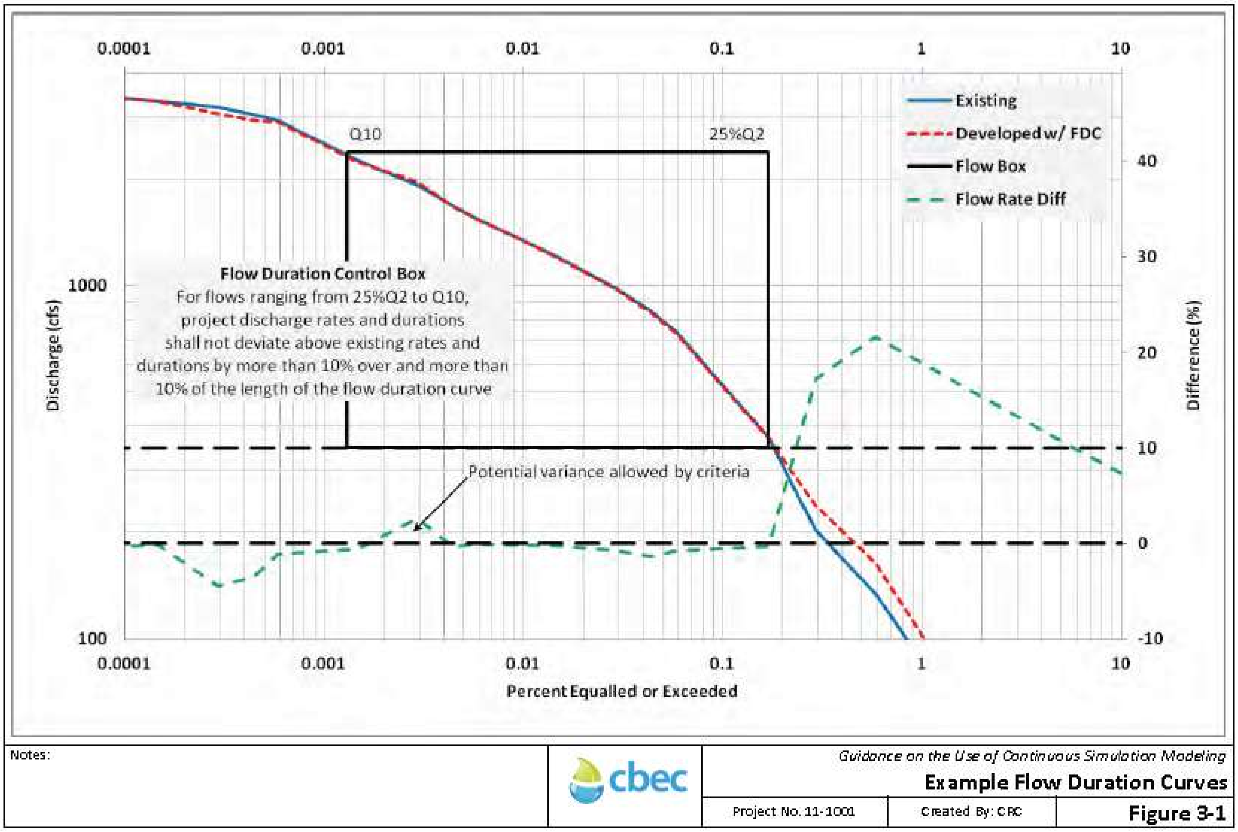

Figure 3-1 shows an example of FDC matching on the routed flows within a receiving waterbody with an example of the variance allowed by the criteria. However, it is cautioned that the FDC variance (e.g., "…by more than 10 percent over and more than 10 percent of the length…") may need to be reduced to something less than 10 percent (perhaps based on a ratio of watershed areas) to account for cumulative effects if there remain the potential for continued development in the watershed.

EROSION POTENTIAL

Erosion Potential (EP) is an index to indicate the impact of increased flows on stream stability and is based on bed mobility and an integration of work (as a function of velocity and excess shear stress in the channel only) over time, expressed as a ratio of post-project work divided by pre-project work in the receiving waterbody. Total work is based on integrating effective stream power as:

$$W = \sum_{i=1}^{n} (\tau_i - \tau_c)^e \cdot V_i \cdot \Delta t_i$$

where *W* is the total work done (ft-lbf/ft^2), τ is the average channel shear stress, τ_c is the critical shear stress to initiate erosion, *e* is an exponent varying from 1 to 2.5 to account for the exponential rise in stream power with flow, *V* is the velocity (ft/sec), and Δt is the numerical time step (sec). The EP index is then calculated as the ratio of W_{dev} / W_{ex} where W_{ex} and W_{dev} is the total work for existing and developed conditions, respectively. EP can be calculated at any location in the waterbody based on continuous simulation time series of flow, velocity, and excess shear stress in the channel as derived from hydraulic model outputs.

EP criteria are not widely integrated into HMPs. Notably Santa Clara Valley Urban Runoff Pollution Prevention Program (SCVURPPP) included EP criteria in their HMP, but in so much as it was used to inform their overall management objective (i.e., post-project runoff shall not exceed estimated pre-project rates and/or durations) and the development of their FDC / PFC criteria. In the SCVURPPP (2005) final HMP, an EP ratio <= 1.0 was recommended as the instream target value to be maintained for stream segments downstream of the point of discharge for HMP management. From a risk management perspective, the chance of a stream becoming unstable at an EP of 1.0 is 9%, meaning that 1 in 11 streams could become unstable even with controls (SCVURPPP, 2005). As such, instream EP must be evaluated considering the effects of the cumulative changes that have or may take place in the watershed.

Even though EP criteria are not widely promoted in county HMPs, that does not preclude analyses based on EP from being used, especially when instream measures permit more robust geomorphic analyses (e.g., SCVURPPP final HMP; SSQP draft HMP). While EP analyses are more time and data intensive, there is the potential outcome to discharge runoff at higher rates and durations than FDC / PFC criteria would allow, thus resulting in possibly smaller onsite measures. The time and data intensiveness of EP analyses stem from the need to evaluate the hydraulic and geomorphic conditions of the receiving waterbody to be protected at multiple locations based on continuous simulation hydraulic model outputs and geomorphic data. Potential hydraulic model considerations when performing EP calculations are addressed below.

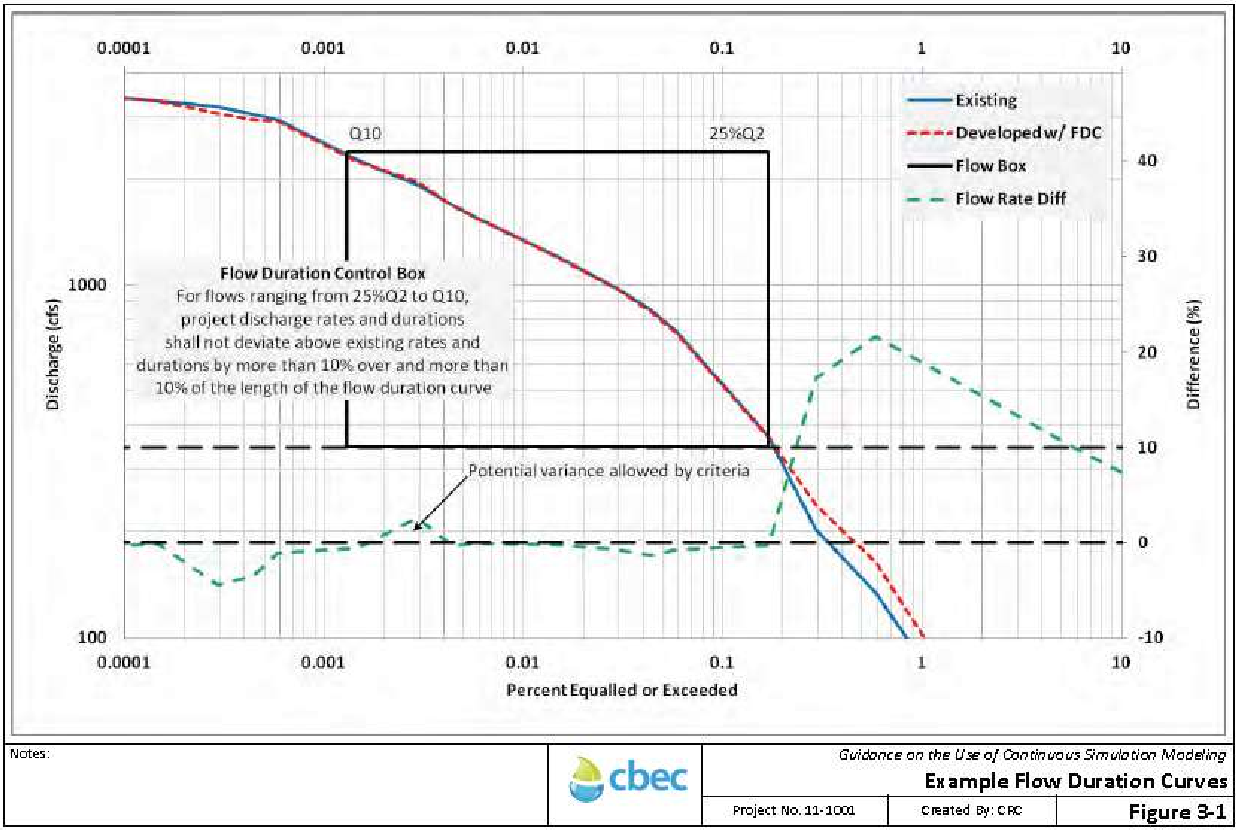

Figure 3-1. Example Flow Duration Curves

4. DATA REQUIREMENTS FOR CSM AND HYDRAULIC ANALYSIS

Hydrologic models capable of performing long-term continuous simulation to support HMPs include, but are not limited to, HSPF, HEC-HMS soil moisture accounting (SMA) method, and other hydrology models, such as the Bay Area Hydrology Model (BAHM). The first two are public domain software models and the third is a proprietary software model customized for specific counties that uses HSPF as its computational engine. A fourth modeling tool based on continuous simulation results, and also using HSPF as its computational engine, are the suite of BMP sizing calculators specifically designed for HMP management for select counties. These have been developed for Contra Costa and San Diego County and Sacramento County (in draft form). All four suites of models use site conditions (i.e., topography, soils, vegetation, and land use) and long-term precipitation data to calculate the various components of the hydrologic cycle (i.e., infiltration, surface runoff, soil moisture, evapotranspiration, percolation, interflow, and groundwater). Specific details about each model and model comparisons (e.g., TetraTech, 2011) are not discussed here, but can be reviewed in available literature.

Following model selection, hydrologic models are created for existing and project conditions based on various considerations, some of which are discussed in subsequent sections. For project conditions, county specific HMP measures need to be specified to manage project runoff to meet the evaluation criteria identified above. The BMP sizing calculators and BAHM-type hydrology models do have optimization routines to size BMP and LID measures. Automatic sizing allows for efficient and quick sizing of such features based on county specific, model specific (e.g., the sizing calculator for San Diego and Contra Costa County is based on pre-defined sizing factors such that site specific continuous simulations do not need to be performed, and is limited to drainage management units of less than 100 acres), and user-defined (e.g., the BAHM-type hydrology models require site specific continuous simulation with a wide selection of measure configurations) assumptions and limitations. As standalone models, HSPF and HEC-HMS offer flexibility as it relates to model configuration, model inputs, and user-defined parameters. However, these models do not have optimization routines to size various BMP and LID measures, thus requiring manual iteration to achieve a satisfactory solution.

PRECIPITATION DATA

Long-term precipitation data in the range of 30 to 50 years is typically needed to generate a sufficiently long flow record from which FDC and PFC analyses and/or subsequent hydraulic analyses can be performed. The precipitation data observation interval should ideally be no coarser than hourly, and if available, can be sub-hourly (e.g., 15 minutes) to coincide with a finer continuous simulation time step.

The precipitation data should ideally be located near the project site, and if needed, scaled to the project site based on a ratio of mean annual precipitation as derived from county specific mapping or regional sources (e.g., PRISM [http://www.prism.oregonstate.edu/]) and reviewed to ensure that it captures key IDF characteristics from county specific mapping or regional sources (e.g., NOAA Atlas 14 [http://www.nws.noaa.gov/oh/hdsc/index.html]). A variety of precipitation data sources exist, and include, but are not limited to:

- ALERT system for individual counties (e.g., Sacramento [http://www.sacflood.org/])
- Western Region Climate Center (WRCC [http://www.wrcc.dri.edu/])
- NOAA National Climatic Data Center (NCDC [http://www.ncdc.noaa.gov/])
- California Irrigation Management Information System (CIMIS [http://wwwcimis.water.ca.gov/])

HYDROLOGIC SIMULATION TIME STEP

The continuous simulation time step and output reporting interval for the four models identified above has traditionally has traditionally been hourly. However, an hourly time step is often significantly larger than the time of concentration for concentration for developed subwatersheds relative to existing subwatersheds, especially those commonly configured commonly configured developed subwatersheds that are limited to less than 100 acres. The sizing calculator and BAHM-type calculator and BAHM-type models are hardwired at hourly, but the public domain software still affords the user to go to a the user to go to a finer time step. As such, a sub-hourly time step and output reporting interval is preferred in order to preferred in order to adequately resolve and sample flow from developed subwatershed elements where time of where time of concentrations are typically less than one hour. As shown by

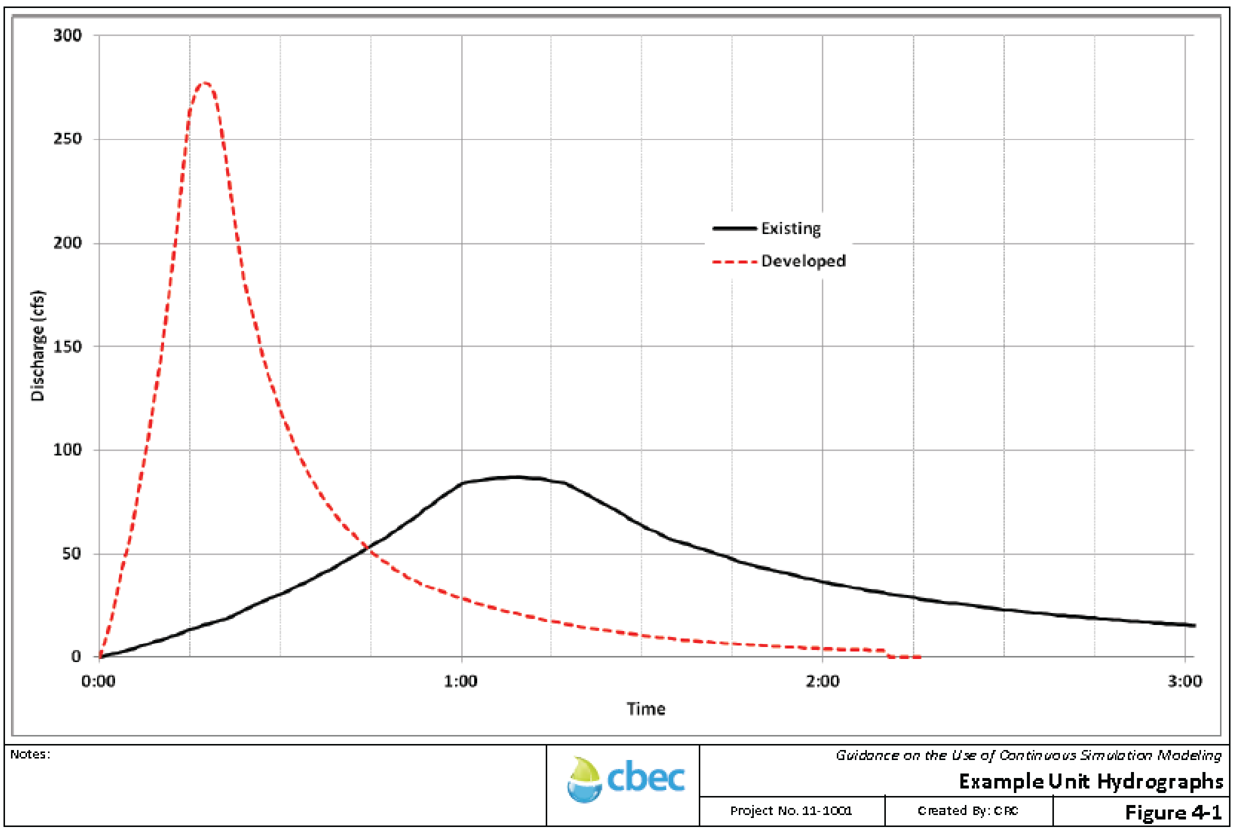

Figure 4-1 for a typical developed subwatershed, the unit hydrograph for developed conditions is flashier, peaks quicker (well within one hour), and the recession limb becomes small quickly. While a sub-hourly time step and output reporting interval may not be desirable due to the volume of model output that will be generated, it is possible to bias the results in favor of the developed condition due to under sampling of the flashier and larger developed flows under an hourly time step.

HYDROLOGIC MODEL CALIBRATION AND VALIDATION

In developing continuous simulation models, the model parameters describing soil characteristics, land use descriptions, and evapotranspiration should be derived from published data (e.g., soil survey, local studies, county standards, etc.). These parameters should be calibrated and validated, where applicable, by comparing modeled flows to measured or observed flows with the receiving waterbody for specific overlapping periods when there is adequate precipitation, evapotranspiration, and flow data. In the absence of site-specific data for calibration and validation, calibrated model parameters from neighboring watersheds within the region could be used so long as proper justification is provided that said parameters are appropriate. However, it is not recommended that local studies rely upon calibrated parameters from other regions where soil characteristics and land use descriptions are markedly different. Rather, when calibration cannot be performed, general review and comparison of continuous simulation model outputs (e.g., hydrograph shape, AMS, etc.) to standardized event-based approaches could be performed to demonstrate that continuous simulation results are generally consistent with local standards and methodologies.

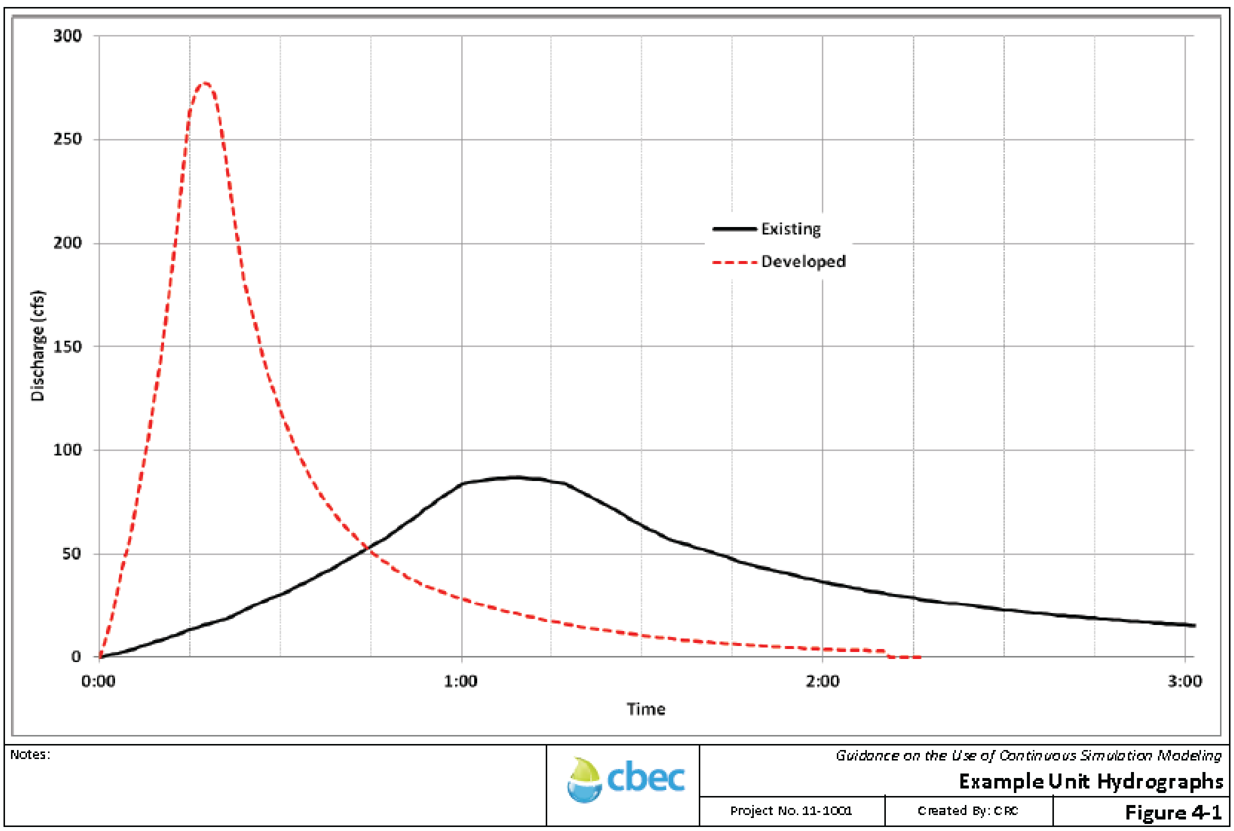

Figure 4-1. Unit Hydrograph Method

For example, continuous simulation modeling in Sacramento County for some developments has relied up conversion of SacCalc (HEC-1 pre- and post-processor) event-based models to the SMA method within HEC-HMS. This conversion often involves retaining the surface infiltration rate determined by SacCalc based on accepted land use descriptions, but parameterizing the subsurface based on soil survey information and local studies, using local potential evapotranspiration data, and reviewing model hydrographs for reasonableness.

HYDRAULIC MODEL CONSIDERATIONS

Sometimes hydraulic models are needed since the basic flow routing within the hydrologic models is not adequate to characterize the potential changes to the hydraulic and geomorphic character of the receiving waterbody, especially when instream measures are suggested or EP is used as the evaluation criteria. Potential considerations and issues encountered when developing and using hydraulic models for continuous simulation include:

1. Low flow instabilities can introduce anomalies into model output (which is commonly encountered in HEC-RAS), so careful hydraulic model selection is important for accuracy and efficiency
2. The sensitivity of the hydraulic model outputs (i.e., velocity and shear stress) to accurate hydraulic description of the receiving waterbody (i.e., cross section geometry (i.e., is it based on LiDAR influenced by vegetation or ground survey), proper definition of channel transitions, proper definition of channel bank markers, appropriate Manning's n-values, etc.)
3. Selection of appropriate compliance points that are representative of the reach and capture flow changes (e.g., downstream of points of discharge and not in backwater areas).

All of these issues have the potential to introduce error and subjectivity into long-term hydraulic analyses and care should be taken to systematically address each source of error.

GENERAL TIPS

A series of general tips are provided as follows. These can be used to increase efficiency and accuracy when performing CSM.

- To shorten the simulation time, the precipitation record can be truncated to only the rainy season (e.g., October through May) by removing the dry summer months from the simulation, especially in ephemeral systems where applicable.
- Hourly precipitation data does not prohibit the continuous simulation model from being run at a sub-hourly time step.
- Subwatershed delineation between existing conditions and developed conditions can often result in relatively large existing subwatersheds compared to relatively small developed subwatersheds. It is commonly known that smaller subwatersheds have flashier flows, so making existing and developed conditions subwatershed sizing consistent is recommended to provide a more meaningful comparison.

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APPENDIX B: APPLICATION OF SUITES OF MODELING AND ASSESSMENT TOOLS

Introduction. This appendix provides a discussion of four example "suites of tools" that can be used to perform predictive scientific assessments and address specific questions related to hydromodification assessment and management. The suites are changeable mixes of mechanistic models, statistical analyses, and expert scientific judgment that incorporate a number of the tools discussed in Chapter 4, combined in various ways. For example, some suites apply a series of cascading models, in which the output from one is used as input to the next; other suites apply a number of models in parallel to develop an assessment based on the weight of evidence. The suites of tools discussed below are used to perform a baseline stability assessment, a channel forming discharge analysis, an erosion potential analysis, and a sediment transport analysis. Most of these standard tools (with the exception of the erosion potential suite) have been widely employed in a variety of stream management activities for decades, and are considered essential components of the broader fluvial geomorphology toolbox. This is far from a comprehensive list of tools, as there are many other important tools (focused on both geomorphic and biologic endpoints) relevant to hydromodification management (Kondolf et al. 2003; Poff et al. 2010); however, the purpose of this appendix is to briefly illustrate how several standard tools can be integrated to answer key questions about stream responses and to provide a stronger technical basis for hydromodification management.

Application of these tools provides basic geomorphic data and knowledge that are typically needed to manage a stream for some desired future state in a watershed with changing land uses. This critical information comes at a cost—the tools require substantially more time and effort to apply than has been the norm in hydromodification management because they involve examining streams within their watershed context with a deeper level of geomorphic analysis. Stormwater management programs typically have made the "practical" assumptions that stream reaches can be managed in isolation from the larger systems of which they are a part, and that effective management prescriptions can be formulated with little or no substantive geomorphic analysis. ***These assumptions are in direct conflict with current understanding in fluvial geomorphology and stream ecology, which indicates that protection of stream integrity is often predicated upon careful assessments of geologic and historical context, performing detailed hydraulic and sedimentation analyses where appropriate, and developing basic understanding of streamflow-ecology linkages***. If hydromodification management policies are to have a reasonable chance of actually achieving their aims, then it will most likely be necessary to reject these simplifying assumptions and instead rely on approaches rooted in current scientific understanding of stream systems.

The suites of tools described below go beyond screening level assessments that are designed, in part, to identify which streams lend themselves to relatively straightforward management prescriptions versus which streams do not. For streams that do not lend themselves to generic management prescriptions, the level of analysis performed with these tools should increase with the level of risk and geomorphic / biologic susceptibility of the streams. This does not mean that every stream will require in-depth analysis by local permitting agencies. It is not possible to carry out sufficient geomorphic analyses with the tools illustrated below on a permit-by-permit basis, and local governments may lack the resources and/or technical capacity to effectively apply these tools. Instead, ***the vital information provided by these tools***

will need to be obtained through proactive regional studies that involve baseline assessments followed by progressively more in-depth analyses as necessary to provide local governments with a sound basis for effective project-by-project decision-making within a broader watershed management framework.

1. **Baseline Stability Assessment**. This suite of tools is designed to answer the following key questions:
   - What is the trajectory of the stream's form over time?
   - How has the channel form responded to changes in water and sediment supply over the years?
   - Is the channel close to a geomorphic threshold that could result in rapid, significant change in response to only minor flow alteration?
   - How can past channel responses provide insight into potential responses to future watershed change, and so aid in prediction of future hydromodification-induced changes?
   - What level of subsequent geomorphic analysis is appropriate given the complexity of the situation and the susceptibility of the streams of interest?

   The goals of a baseline stability assessment are to:
   - Document the historical trends of the system;
   - Establish the present stability status of the system and identify the dominant processes and features within the system;
   - Provide the foundation for projecting future trends with and without proposed project features;
   - Provide critical data for calibration and proper interpretation of models; and
   - Provide a rational basis for identification and design of effective alternatives to meet project goals.

   The key tools that comprise this suite include:
   - GIS mapping of topography, soils, geology, land use/land cover across the contributing watershed (e.g., Thorne 2002)
   - Analysis of hydro-climatic data, e.g. streamflow gauge records, changes in stage-discharge relationships over time (e.g., Thorne 2002)
   - Analysis of aerial photos and historical data (e.g., Thorne 2002)
   - Field reconnaissance (e.g., Thorne 1998)
   - Qualitative response (e.g., Lane 1955b, Schumm 1969, Henderson 1966 relations)
   - Classification systems - (e.g., Thorne 1997; Schumm et al. 1982; and channel evolution model developed for S CA by Hawley et al. in press)
   - Relationships between sediment transport and hydraulic variables
   - Regional hydraulic geometry (e.g., Hawley 2008; Haines in prep)
   - Regional planform and stability predictors (e.g., Hawley et al. in press, Bledsoe et al. in press, Dust and Wohl 2010)

- Bank stability analysis (e.g., BSTEM http://www.ars.usda.gov/Research/docs.htm?docid=5044, Hawley (2009), Bledsoe et al. in press, Osman and Thorne 1988; Thorne et al. 1998)
- Sediment budgets (Booth et al. 2010; Reid and Dunne 1996)
- Fluvial audit (Thorne 2002 – a comprehensive framework for performing baseline assessments)

A baseline assessment is completed by integrating information from all the available data sources and analytical tools. Analysis with each of the individual tools may yield a verdict of aggradation, degradation, or dynamic equilibrium with respect to the channel bed, and stable or unstable with respect to the banks. The individual assessments can produce contradictory results. In this case, one should assign a level of confidence to the various components based on the reliability and availability of the data, and the analyst's own experience level. As is often the case in the management of fluvial systems, there is no "cookbook" answer, and we must always incorporate sound judgment.

A process-based channel evolution model (CEM) is a particularly useful element of the baseline assessment process. A CEM aids in identifying the dominant processes and trends of channel change and provides a framework for subsequent, more detailed modeling (ASCE 2008). In some locations, CEMs have already been developed and calibrated with regional data. For example, the CSU / SCCWRP Screening Tool (Bledsoe et al. 2010) grew out of a regional CEM (Hawley et al. in press) and integrates several baseline assessment tools including regionally-calibrated braiding, incision, and bank stability thresholds, and sediment supply analysis with "Geomorphic Landscape Units" (Booth et al. 2010). In locations where a CEM has not been sufficiently defined, the baseline assessment suite of tools can provide the data and understanding needed to develop a regionally calibrated CEM.

The following are example outputs from a baseline stability assessment, including channel stability and bank stability diagrams associated with key geomorphic thresholds of management concern in the channel evolution sequence (i.e. braiding, incision, and bank failure):

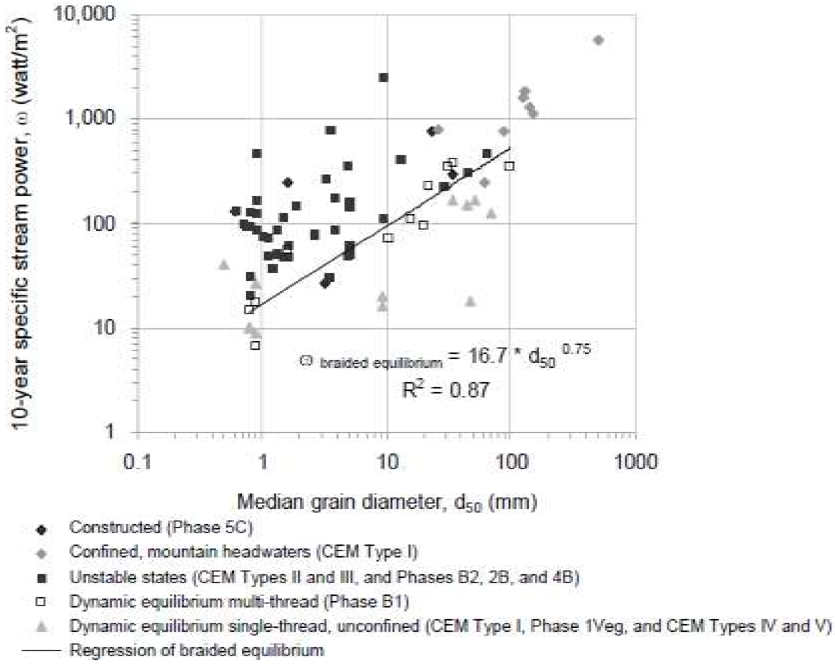

Figure B-1. Stability thresholds for channel types of southern CA, as identified through the development of a regional CEM (Hawley et al., in press).

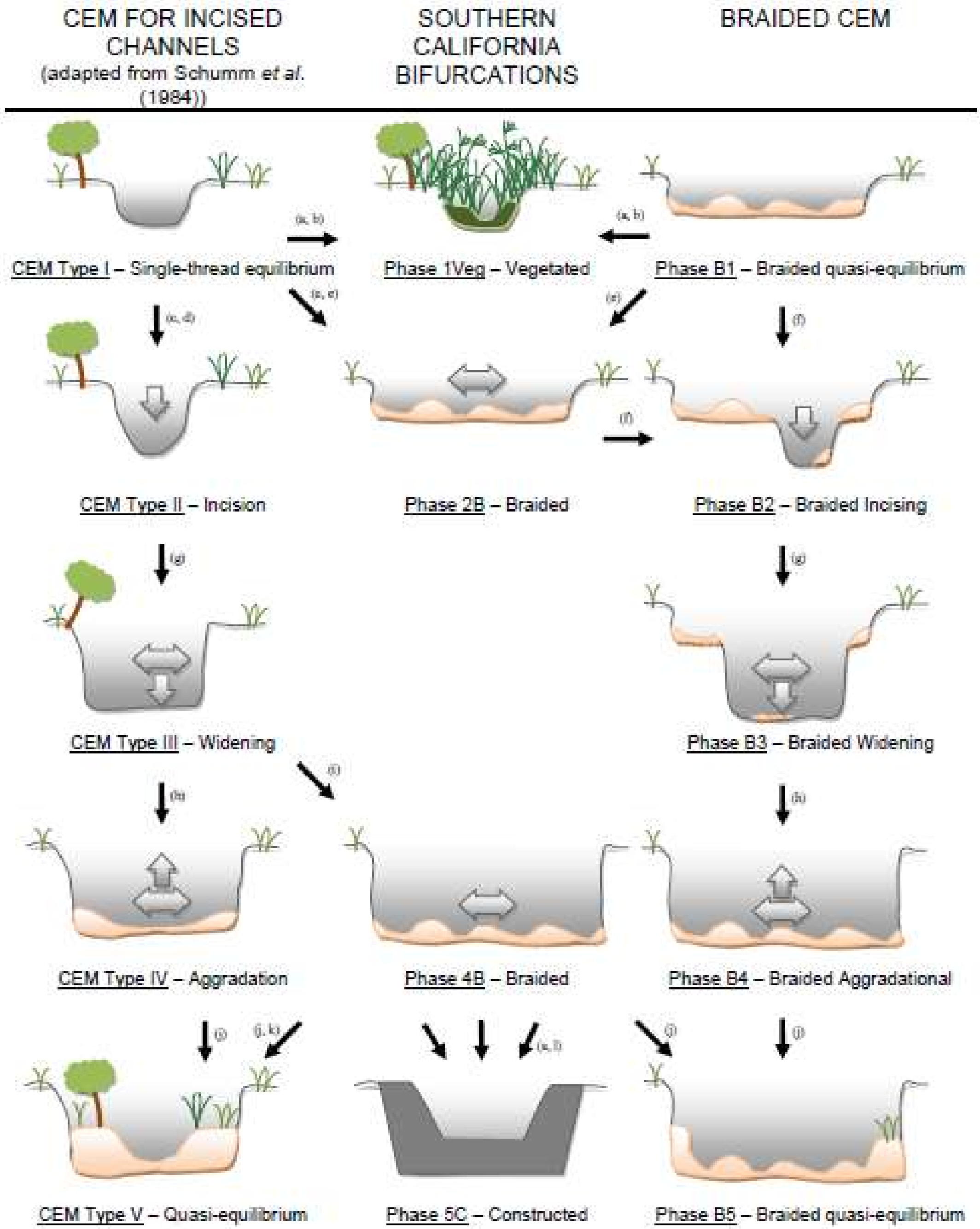

Figure B-2. Channel evolution model of response to hydromodification in southern California (Hawley et al. in press). Red and blue ovals highlight geomorphic thresholds that may be quantified using the baseline assessment suite of tools. By developing a general physical understanding of channel evolution sequences commonly observed in urbanizing watersheds of southern CA, two braiding thresholds and a bank stability threshold of management concern were identified. Channels may shift from single thread to braided planforms if widening is the dominant mode of initial adjustment. Alternatively, single thread channels may become braided after an initial period of incision that triggers geotechnical instability and failure of the banks. Quantitative predictors of these thresholds of braiding, incision, and bank failure can be developed in the baseline assessment process to evaluate the proximity of streams to these critical stages of channel evolution and instability.

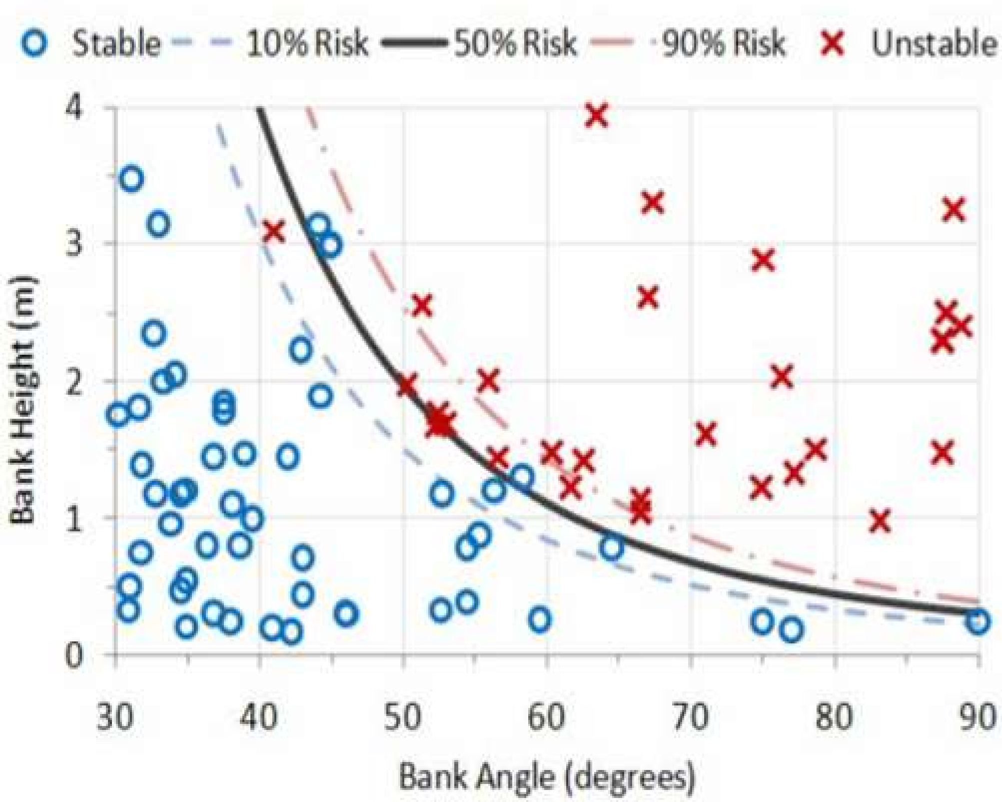

Figure B-3. Bank stability threshold for mass wasting identified through analysis of field data from southern California streams with stable and unstable banks (Bledsoe et al., in press).

2. **Channel-forming discharge suite of tools.** This suite of tools is designed to answer the following key questions:
   - What ranges of discharges are most influential in controlling channel form and processes over decadal time scales?
   - What channel-forming discharge should be used in sediment transport analyses to identify sediment transport capacity, equilibrium slope and geometry, etc.?

   The tools that comprise this suite include the following:
   - Effective discharge computations (e.g., Soar and Thorne 2001; Biedenharn et al. 2000; GeoTools – Bledsoe et al. 2007). An effective discharge analysis directly quantifies the range of discharges that transport the largest portion of the annual sediment yield over a period of many years.
   - Field identification of high water elevations, depositional surfaces, and “bankfull” features
   - Flood frequency analysis
   - Un-gauged site analysis (e.g. USGS StreamStats) http://water.usgs.gov/osw/streamstats/california.html; Hawley and Bledsoe (2011), regional flow duration curve extrapolation – Biedenharn et al. 2000)

This suite incorporates a number of parallel analyses that can be used to establish likely upper and lower bounds to the range of influential discharges, and that can be assessed through a weight-of-evidence evaluation. The following is an example output from the channel forming discharge suite of tools:

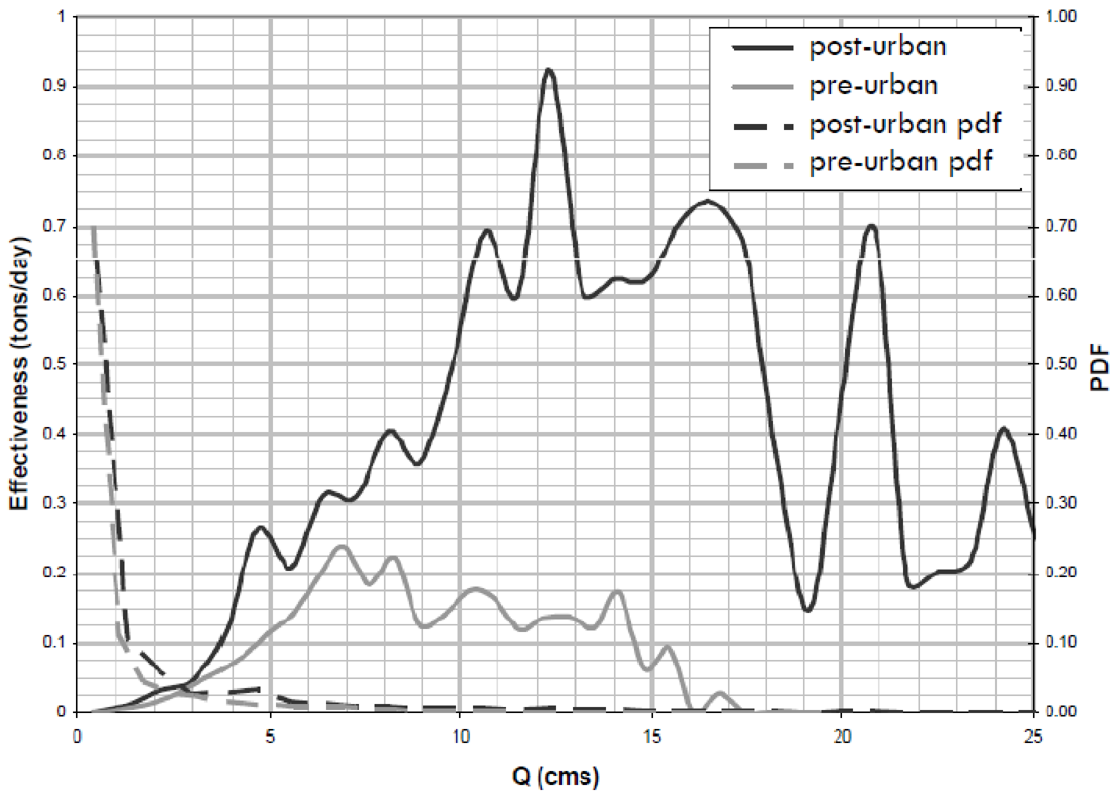

Figure B-4. Flow effectiveness curves for continuous series of pre-urban and post-urban discharges (Biedenharn et al. 2000; Bledsoe et al. 2007). Cumulative sediment yield is approximated by the area under the respective curves. If the stream bed is the most erodible channel boundary, the ratio of areas under these curves would be the erosion potential metric described below in the next suite of tools.

3. **Erosion potential suite of tools.** This suite of tools is designed to answer the following key questions:
   - How do proposed land-use changes or channel alteration affect the capacity of a channel to transport the *most erodible material in its boundary* over a period of many years (erosion potential – Ep)?
   - Do proposed mitigation approaches match the pre- vs. post- development erosion potential over the full spectrum of erosive flows?
   - Do past changes in erosion potential correspond to different states of channel stability and degradation in this region?
   - Does a proposed change in streamflow make it more likely that a channel will enter an alternative / degraded state?

The underlying premise of the erosion potential approach advances the concept of flow duration control (discussed in Chapters 2 and 3) by addressing in-stream processes related to sediment transport. An erosion potential calculation combines flow parameters with stream geometry to assess long term (decadal) changes in the sediment transport capacity. The cumulative distribution of shear stress, specific stream power and sediment transport capacity across the entire range of relevant flows can be calculated and expressed using an erosion potential metric, Ep (e.g., Bledsoe, 2002). This erosion potential metric is a simple ratio of post- vs. pre-development sediment transport capacity over a period of many years. The calculated capacity to transport sediment can be based on the channel bed material or the bank material, depending on which one is more erodible.

This Ep suite of tools has been applied in two primary ways:

a) At a project-level analysis, it has been applied to answer the first two questions above. A municipal stormwater permit may require a project design to achieve an erosion potential (Ep) value of 1.0. This means that a project must be designed so that the long-term erosion potential of the site's stormwater discharge is equal to the erosion potential of the pre-development condition. Section 3.1 below explains the process by which this analysis is conducted.
b) At a regional level, this suite of tools can be applied to answer the third and fourth questions above and to provide further guidance to project-level assessments. For example, practical engineering considerations generally require that a tolerance be permitted around a target design value. It is unlikely that a project design can match an Ep target of 1.0 across all conditions and through all stream reaches, due to variations in a multitude of contributing factors. The selection of an acceptable tolerance or variance from 1.0 is a management decision that should be informed by regional data presented in a risk-based format. Section 3.2 below explains how such a study has been conducted, using the Santa Clara Valley example from northern California.

3.1. *Project-Level Analysis.* As applied to the analysis of project impacts and mitigation design, the steps and associated tools that comprise this suite include the following (Figure B-5):

- Perform continuous simulation of hydrology (e.g. SWMM, HEC-HMS, HSPF) for the project site, for both pre-project condition and post-project condition with the proposed mitigation design.
- Convert discharges and field surveys to hydraulic parameters (shear stress and specific stream power) – e.g., for uniform flow analysis use Manning's equation, GeoTools; for varied flow analysis use HEC-RAS
- Convert hydraulic parameters into sediment transport capacity – e.g., at-a-station hydraulic geometry, HEC-RAS, GeoTools, sediment transport relationships (bedload and total load)
- Integrate Ep over time – e.g., GeoTools

- Compare Ep values for pre-development and post development to determine if the proposed mitigation design is adequate. Adjust stormwater controls as necessary to meet target Ep.

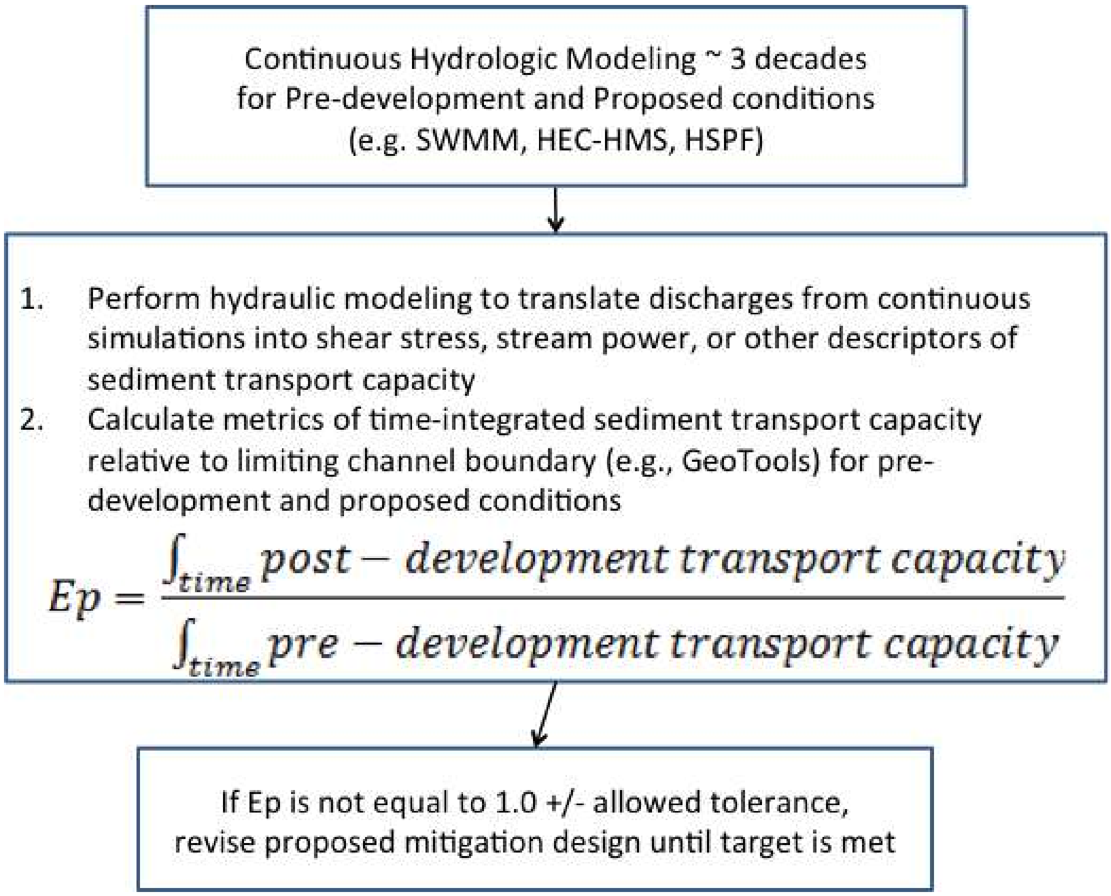

Figure B-5: Steps involved in a project-level Erosion Potential analysis

3.2. *Risk-Based Regional Analysis*. Risk-based modeling estimates the probability of stream geomorphic states. Decision-makers can then choose acceptable risk levels based on an explicit estimate of prediction error. The foundation of risk-based modeling in the context of hydromodification management is the integration of hydrologic and geomorphic data derived from the output of continuous hydrologic simulation models to generate metrics describing expected departures in the most important stream processes. These physical metrics are provided as inputs to probabilistic models that estimate the risk of streams shifting to some undesirable state. Because the decision endpoint is often categorical (e.g., stable, good habitat) the statistical tools of choice are often logistic regression, classification and regression trees (CART), and/or Bayesian probability networks.

The steps below are used to develop a risk-based framework (Fig. B-6) for assessing how hydromodification may impact streams within a region, and for understanding the relationships between deviation from an Ep of 1.0 and the likelihood of channel instability. Illustrating figures are taken from a risk-based approach was used in the development of the Santa Clara Valley Urban Runoff Program Hydromodification Management Plan (www.SCVURPPP.org). This study demonstrated that a time-integrated index of erosion potential based on continuous hydrologic simulation and an assessment of stream power relative to the erodibility of channel boundary materials could be used to distinguish between channels of a particular regional type that are stable vs. degraded by hydromodification in urban watersheds.

- Perform project-level analysis as described in section 3.1 above for existing developments throughout the study watersheds.
- Perform stream surveys throughout the study watersheds to characterize condition (i.e., stable, unstable)
- Create statistical relationships between Ep and different channel states – e.g., logistic regression in R, SAS, Statistica, Minitab, etc. Note that standard regression techniques are applied when the dependent variable and the explanatory variables are quantitative and continuous. To analyze a binary qualitative variable (e.g., 0 or 1, stable or unstable, healthy or degraded) as a function of a number of explanatory variables, alternative techniques must be used. The regression problem may be revised so that, rather than predicting a binary variable, the regression model predicts a continuous probability of the binary variable that stays within 0–1 bounds. One of the most common regression models that accomplishes this is the logit or logistic regression model (Menard, 1995; Christensen, 1997).

Continuous Hydrologic Modeling ~ 3 decades
Pre-development and Current Conditions
(e.g. SWMM, HEC-HMS, HSPF)

1. Survey stable / unstable channels across a gradient of hydromodification
2. Perform hydraulic modeling to translate discharges from continuous simulations into shear stress, stream power, or other descriptors of sediment transport capacity
3. Calculate metrics of time-integrated sediment transport capacity relative to limiting channel boundary (e.g., GeoTools) at each site

$$Ep = \frac{\int_{time} post-development\ transport\ capacity}{\int_{time} pre-development\ transport\ capacity}$$

Perform risk-based / logistic regression analysis linking Ep with probability of acceptable vs. unacceptable channel states

Figure B-6: Steps involved in a Risk-Based Erosion Potential analysis

The variables included in risk-based models of stream response are not limited to erosion potential. Additional multi-scale controls could be included. For example, simple categories of physical habitat condition and ecological integrity could be predicted by augmenting erosion potential metrics with descriptors of the condition of channel banks and riparian zones, geologic influences, floodplain connectedness, hydrologic metrics describing flashiness, proximity to known thresholds of planform change, and BMP types. Furthermore, although most of the emphasis to date has been on predicting geomorphic endpoints, the risk-based approach can be extended to the prediction of biological states in urban streams if the necessary data are available.

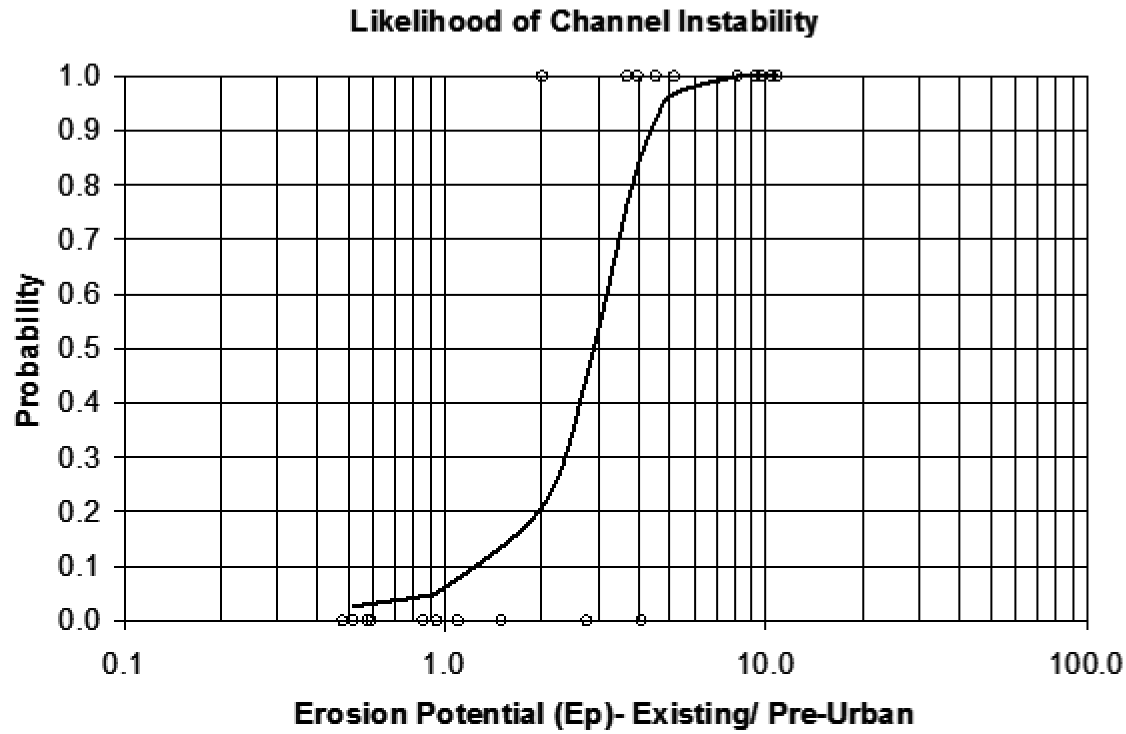

Figure B-7: Example of a logistic regression analysis of stable vs. unstable channels (Bledsoe and Watson, 2001; Bledsoe et al., 2007). The vertical axis represents the probability of stream instability which increases rapidly for channels with sediment transport capacity increased by urban hydromodification (Ep > 1).

3.3. *Strengths and Limitations.* The Erosion Potential approach combines a sound physical basis with probabilistic outputs and requires a substantial modeling effort. Such an effort is necessary to adequately characterize the effects of hydromodification on the stability of streams that are not armored with very coarse material such as large cobbles and boulders. Although policies based on this approach should reduce impacts to channel morphology, they may still fail to protect stream functions and biota. Key simplifying assumptions and prediction uncertainty in the inputs (hydrologic modeling, assumptions of static channel geometry in developing long term series of shear stresses or stream powers, assumptions of stationarity in sediment supply, etc.) have not been rigorously addressed. Its effectiveness also depends on careful stratification of streams in a region such that fundamentally different stream types are not lumped together (e.g. labile sand channels vs. armored threshold channels with grade control) in developing general relationships for instability risk. Endpoints to date have been rather coarse, e.g. stable vs. unstable; as such, they do not provide sufficient resolution for envisioning future stream states. However, the Erosion Potential approach provides promise as an important tool for hydromodification management; it is recommended that it be refined to address sediment supply changes and to provide more finely resolved endpoints for improved predictive capabilities.

4. **Sediment transport analysis suite of tools.** This suite of tools is designed to answer the following questions:
   - Do I need to incorporate sediment transport analysis in predicting channel response to hydromodification, i.e. what is the sensitivity of channel slope and geometry to inflowing sediment load?
   - At what discharges are different fractions of bed material mobilized in a particular stream segment?
   - What is inflowing sediment load to a stream segment, i.e. what is the water discharge Q(t) and sediment supply rate Qs(t) and grain size D(t) delivered to the upstream end of the channel segment of interest?
   - How will the available flow move the supplied sediment through the segment of interest?
   - What is the new equilibrium slope given some change in streamflow, and how much incision would be necessary to achieve this new slope?
   - What is the sediment transport capacity of the segment of interest *relative to* the inflowing sediment load from *upstream* supply reaches?
   - What is the sediment transport capacity of the segment of interest *relative to* the capacity of *downstream* reaches?
   - At the network scale, where are zones of low vs. high energy, aggradation vs. degradation potential, and coarse sediment constriction located?

   The primary tools that comprise this suite include the following:
   - Tools for estimating watershed sediment supply (Reid and Dunne 1996), including the RUSLE (Renard et al. 1997; http://www.ars.usda.gov/Research/docs.htm?docid=5971) and WEPP (Laflin et al. 1991; http://www.ars.usda.gov/Research/docs.htm?docid=10621) models.
   - Effective discharge analysis (see above)
   - Incipient motion analysis (tractive force, e.g. ASCE 2008; Brown and Caldwell 2011; Buffington and Montgomery 1998; Lane 1955a)
   - Sediment continuity analysis at single dominant discharge with an appropriate sediment transport relation – e.g., HEC-RAS, Bedload Assessment for Gravel-bed Streams (BAGS -Pitlick et al. 2009; GeoTools)
   - Equilibrium slope / geometry analysis e.g., HEC-RAS – Copeland et al. 2001, iSURF-NCED 2011)
   - Sensitivity to inflowing sediment load analysis e.g., Copeland's method in HEC-RAS, iSURF-NCED 2011)
   - Sediment continuity analysis over the entire flow frequency distribution e.g., Capacity-Supply Ratio of Soar and Thorne (2001), BAGS, GeoTools

- Network scale sediment balance – Sediment Impact Analysis Methods (SIAM) module in HEC-RAS

Movable bed / mobile boundary models also provide a mechanistic tool for estimating the trend and magnitude of changes in channel geometry due to hydromodification. However, a recent study evaluated the potential applicability of various movable bed and/or boundary models to streams in southern CA (Dust 2009), including HEC-RAS, CONCEPTS (Langendoen, 2000), and FLUVIAL 12 (Chang, 2006). The results of tests performed on urban streams in southern CA indicate that these models are difficult to apply and have high prediction uncertainty due to flows near critical, split flow conditions, and lack of fidelity to complex widening, bank failure, and armoring processes.

The following figures depict example outputs from an application of the sediment-transport suite of tools:

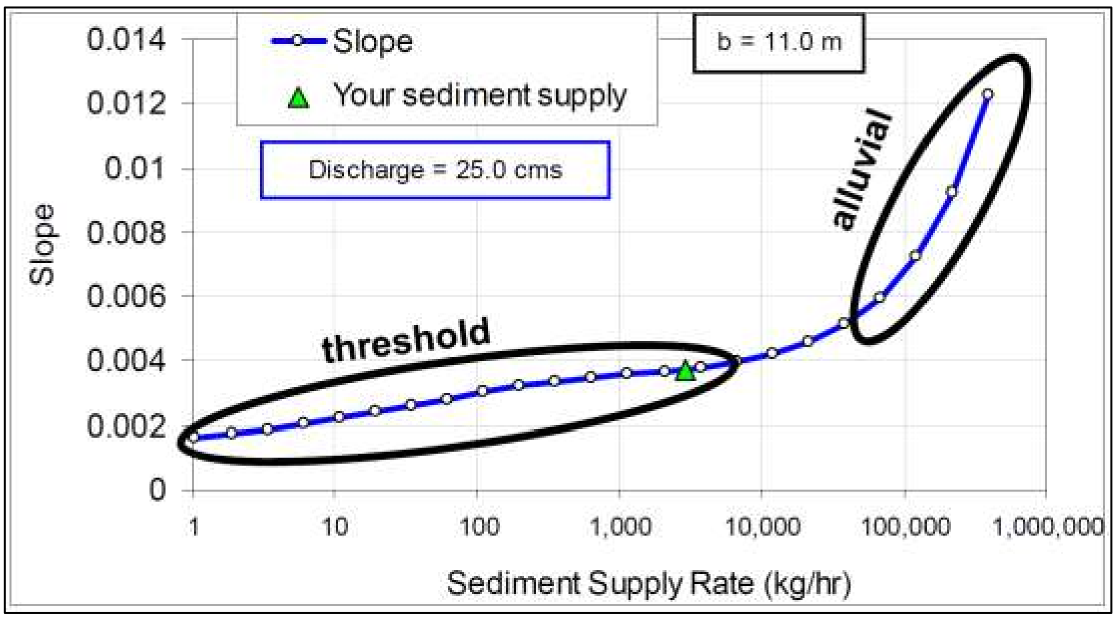

Figure B-8. Sensitivity analysis of equilibrium channel slope to inflowing sediment load (from iSURF, NCED 2011). Slopes of alluvial channels with high sediment supply are much more sensitive than threshold channels with relatively low sediment supply. Channels with beds composed of sand and fine gravels are generally much more geomorphically sensitive to hydromodification than threshold channels in which coarse bed sediments are primarily transported at relatively high flows.

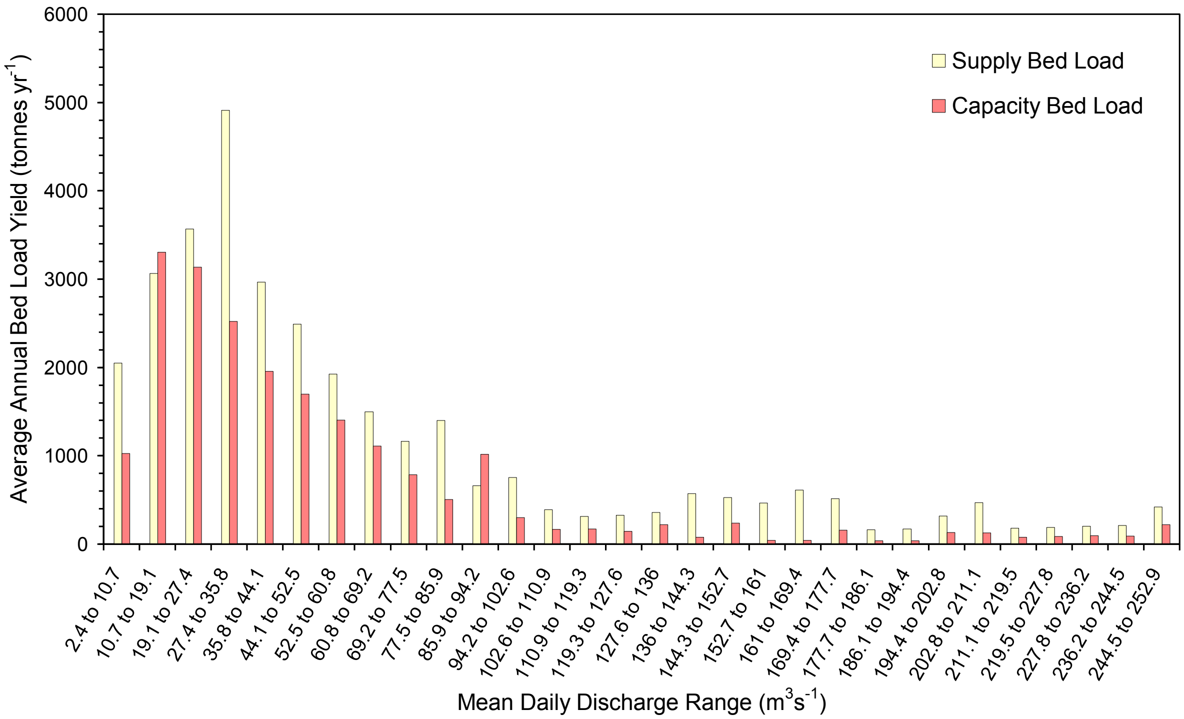

Figure B-9. Analysis of sediment transport capacity vs. inflowing sediment load over the full spectrum of stream discharges (capacity-supply ratio; Soar and Thorne 2001). In this case, the time-integrated capacity to transport bedload is 64% of the supplied bedload and significant aggradation is expected.

5. **Relationship to Management Framework**. These suites of tools could be applied to establish project-specific requirements for hydromodification assessment and mitigation, as recommended in the Management Framework presented in Chapter 3. In the example shown in the diagram below, results of the Baseline Assessment are used as a screening tool to assign high, moderate or low risk levels for stream reaches, in conjunction with the proposed land-use changes. Thus, the Baseline Assessment suite of tools is crucial in determining whether a detailed survey-level assessment and additional suites of tools are necessary for an adequate analysis. The need to apply additional suites of tools in formulating a management approach is commensurate with the level of risk and susceptibility of the stream. More complex and rigorous analysis with multiple suites of tools is necessary in predictive assessments for relatively susceptible stream types such as alluvial channels with sand beds.

   Although a stream may have relatively low susceptibility for overall geomorphic change, it may nevertheless have ecological attributes that are highly susceptible to hydromodification. Thus, suites of tools focused on both geomorphic and biological endpoints must be used to fully assess stream susceptibility to hydromodification. More work will be required to develop tools for prediction of biological response to flow alterations throughout California, as noted in Chapter 3 (see Poff et al., 2010 and http://conserveonline.org/workspaces/eloha).

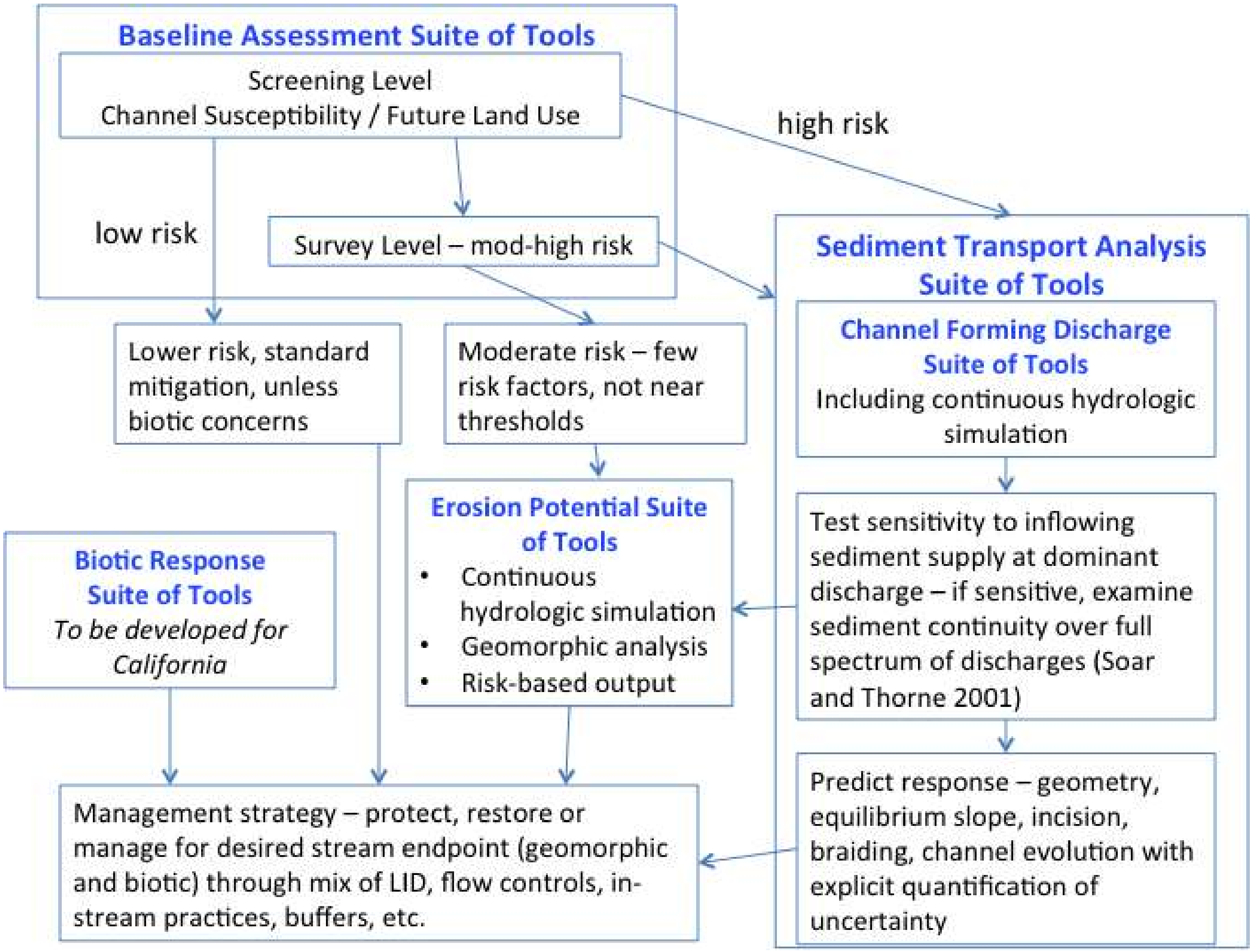

Figure B-10. Conceptual diagram showing relationships among the four suites of existing tools and biotic response tools to be developed in the future. Additional analyses will be required for engineering design.

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APPENDIX C: ADAPTIVE MANAGEMENT

WHAT IS ADAPTIVE MANAGEMENT

Adaptive management is a formalized approach for overcoming the inescapable difficulty in predicting ecological outcomes resulting from natural-resource management actions. It accomplishes this by treating all "management actions" (whether intentional or not) as experimental components within the larger structure of a monitoring program (Holling 1978, Walters 1986, Lee 1999, Ralph and Poole 2003). In other words, specific management actions that may affect ecological processes and functions are systematically evaluated, via "monitoring," to provide the data to affirm or refute the expected outcomes. To the extent that the monitoring results indicate a need to revise the scientific understanding or the management actions built on that understanding, establishing the mechanism to change management actions is a precursor, not an afterthought, of the monitoring program.

Adaptive Management was first articulated over 30 years ago (Holling, 1978) and more recently embraced through various conservation efforts worldwide. Fundamental to this approach is the integration of management and monitoring, recognizing that any management action in the context of a complex ecological system is ultimately experimental, requiring feedback to make progress.

The process of adaptive implementation is iterative and continuous; new knowledge is actively incorporated into revised experiments, a practice best described as "learning while doing" (Lee 1999). The key difference between this approach and other commonly implemented environmental management strategies is the application of scientific principles, such as hypotheses-testing,[is used] to explicitly define the relationships between policy decisions, management actions, and their measured ecological outcomes. Furthermore, this approach provides a means to understand and document these cause-and-effect relationships; it can also point to alternative actions that may produce more desirable outcomes. Uncertainty is embraced and serves as a focal point for defining ever-more specific evaluations.

Scientifically credible and relevant information can only be generated when the management "experiments" are designed with clear hypotheses about the effects of proposed actions or prescriptions. These hypotheses must be testable at multiple scales using available technology and methods (Conquest and Ralph 1998; Currens *et al*. 2000). Hypotheses that cannot be tested, are trivial (e.g., "water flows downhill"), are not credible ("water flows uphill"), or only account for site-specific conditions are not useful in considerations of the singular or cumulative effects of management actions.

In order to retain clear linkages between key questions, hypotheses, and monitoring protocols, the experimental approach must be designed before determining which goals and endpoints are appropriate (Ralph and Poole 2003) since appropriate goals should be *outcomes* of the effort, not a precondition; and the approach must explicitly tie stated hypotheses to the key ecological questions.

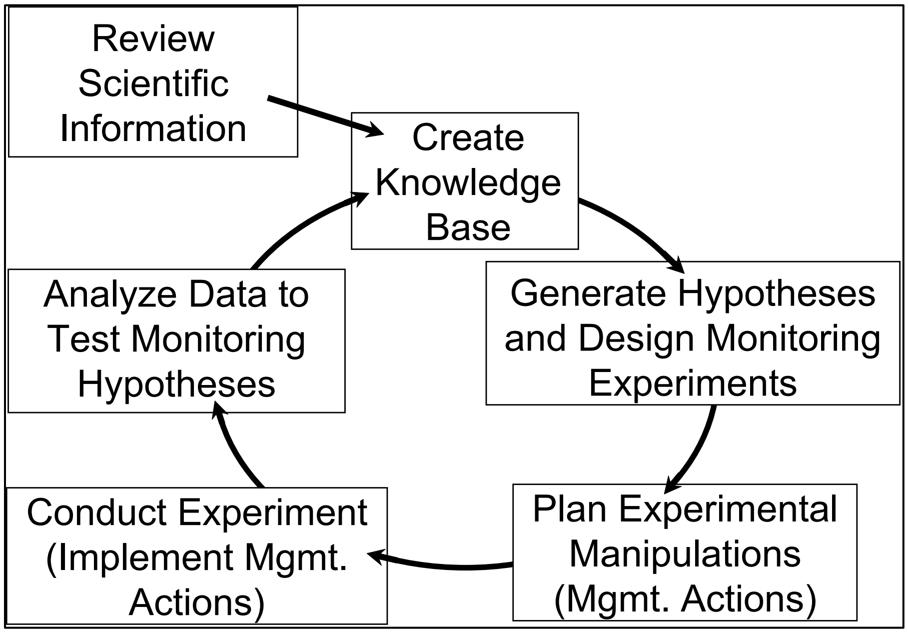

Figure A-1. Framework for an adaptive management program. The key feature of this cycle is the foundation of scientific principles and hypothesis generation; design of the management actions and the monitoring to evaluate their effects are integrated and designed to test assumptions, improve understanding, and reduce uncertainty (modified from Ralph and Poole 2003, Figure 3).

Wagner (2006) asserts that [stormwater] regulatory programs in the past often failed because they were designed in ways that ignored technological and scientific limitations. "Science-based" does not simply mean the monitoring of status and trends followed by responding to imposed benchmarks and goals, but rather that scientific principles must be the foundation of regulatory program design, and that these programs must rely on scientific methods to demonstrate results. Wagner suggests that regulations can still be designed despite incomplete or developing knowledge, but that gaps and limitations must be acknowledged and used to inform ongoing investigations. His argument clearly echoes those of scientists who insist that monitoring experiments and testable hypotheses must frame management decisions and land-use objectives.

WHAT IS *NOT* ADAPTIVE MANAGEMENT, AND WHY IS IT SO PROBLEMATIC?

Unlike the experimental approach embodied by adaptive management, an alternative process traditionally dominates in natural resource management: (1) a problem is identified, but a cause is simultaneously presumed (e.g., "increased sediment inputs into a stream are negatively impacting salmonid survival"); (2) a solution or set of solutions is proposed (e.g., timber harvest is restricted and riparian buffer width is increased), but the prescription is not translated into a testable hypothesis associated with the problem or question; and (3) if the problem is not solved within an arbitrarily reasonable period of time (e.g., a few years) then a different solution is proposed (e.g., "augmented upland and riparian restoration must be implemented"). Although simplified, this outline displays its divergence from adaptive management and from the basic principles of the scientific process—the resulting process is perpetually reactive.

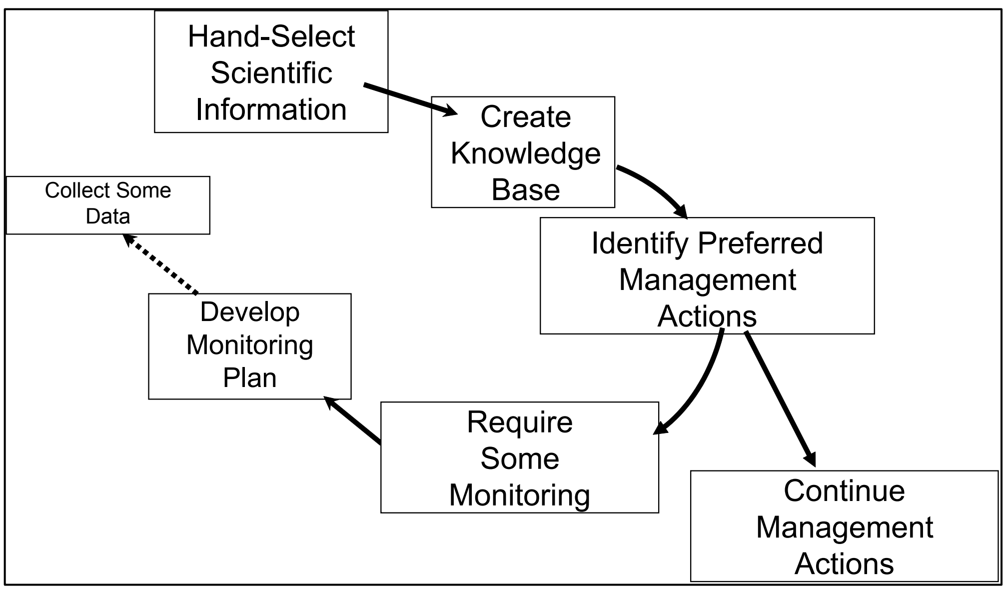

Figure A-2. Common framework for monitoring outside of an adaptive management structure. Management actions are chosen with a presumptive effect on ecological systems, and monitoring is conducted without any feedback to future actions. Even where monitoring is intended to "inform" future management actions, the absence of an explicit experimental design normally limits the utility of any monitoring data to provide meaningful insights.

In its best form, this paradigm has been termed passive adaptive management:

> Restoration planners' current management approach has been described as a "passive" adaptive management approach: science is used to

> develop best-guess predictive models, make policies according to these models, and revise them as data become available. The National Academies advise that every effort be made to take a more "active" adaptive management approach by developing alternative hypotheses for the expected consequences of a particular project and then design the project so the hypotheses can be experimentally tested" (from the summary to *Adaptive Monitoring and Assessment for the Comprehensive Everglades Restoration Plan*, 2003, National Academies Press, 122 pp.).

Ralph and Poole (2003) have aptly named this approach "socio-political adaptive management" (i.e., SPAM).

BARRIERS TO IMPLEMENTING "ACTIVE" ADAPTIVE MANAGEMENT

Although the virtues of active adaptive management are readily articulated, the framework is surprisingly rare in practice. Some of these barriers are practical or logistical, and they include such issues as:

- Longevity and long-term institutionalization of monitoring;
- Effective data management systems that allow managers to readily access data;
- Ability to differentiate effects from natural variability and events, such as flood and fire;
- Cost and technical limitations of necessary data collection.

The most severe impediments, however, are not scientific but social: "We suggest that watershed-scale adaptive management must be recognized as a radical departure from established ways of managing natural resources if it is to achieve its promise... Adaptive management encourages scrutiny of prevailing social and organizational norms and this is unlikely to occur without a change in the culture of natural resource management and research" (Allan et al. 2008).

While science can provide defensible and replicable insights regarding the ecological outcomes of management prescriptions, it cannot offer absolute certainty. Policy can be and should be informed by science but is ultimately based on a variety of considerations that are not always amenable to the spatial, temporal, and technological limitations of the scientific process (Van Cleave et al. 2004). This is an uncomfortable truth for agency managers and elected officials to acknowledge, and it commonly results in funding decisions and public pronouncements using the "language" of science but not its substance.

Although efforts to build large, collaborative programs are commonly characterized by increasing stakeholder involvement and outreach, greater participation does not necessarily mean that true adaptive management is occurring, or that scientific principals are being applied to either the choice of management actions or their evaluation. These efforts, however, do reflect a movement to extend natural resource management decision-making processes beyond just technical experts in order to reflect evolving social values (Pahl-Wostl *et al*. 2007). If they are successful, this approach can open a path to achieving the best of both realms, namely scientific rigor with a broad base of community support.

ATTRIBUTES OF USEFUL HYPOTHESES FOR AN ADAPTIVE MANAGEMENT PROGRAM

A key element of any adaptive management approach is the set of hypotheses that guide both the management actions and their associated monitoring. Because these management actions are recognized as "experimental" (because in a complex system most outcome(s) cannot be predicted with absolute certainty), their selection must be guided by assumptions about what *might happen,* or what is *expected* to happen. This defines the first attribute of a useful hypothesis: it is **credible**, typically because it is based on prior knowledge or scientific understanding of the system. Indeed, some hypotheses may already be so well evaluated and understood (e.g., "Stormwater runoff from freeways carries measurably elevated concentrations of toxic pollutants") that there is little point in framing them in this structure at all—as new monitoring programs to address such hypotheses are highly unlikely to result in new information or knowledge and might be perceived as an unwise expenditure of scarce monitoring resources.

The second attribute of a useful hypothesis stems from the scientific reality that any experiment, whether conducted in the laboratory or across the landscape, provides value only insofar as its outcomes are measured and the effects are distinguishable from the influence of other, unrelated factors. Thus, the hypothesis that guides the experiment should not only be credible but also **testable**. Otherwise, why bother making measurements at all?

Lastly, these actions and measurements and analyses do not occur in a vacuum. Thus, the final guiding principle for any hypothesis in an adaptive management approach is that it be **actionable**, or that different outcomes, as revealed by monitoring, can (and will) result in different management responses. If no difference occurs, then clearly there is no reason to have made the effort in the first place.

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