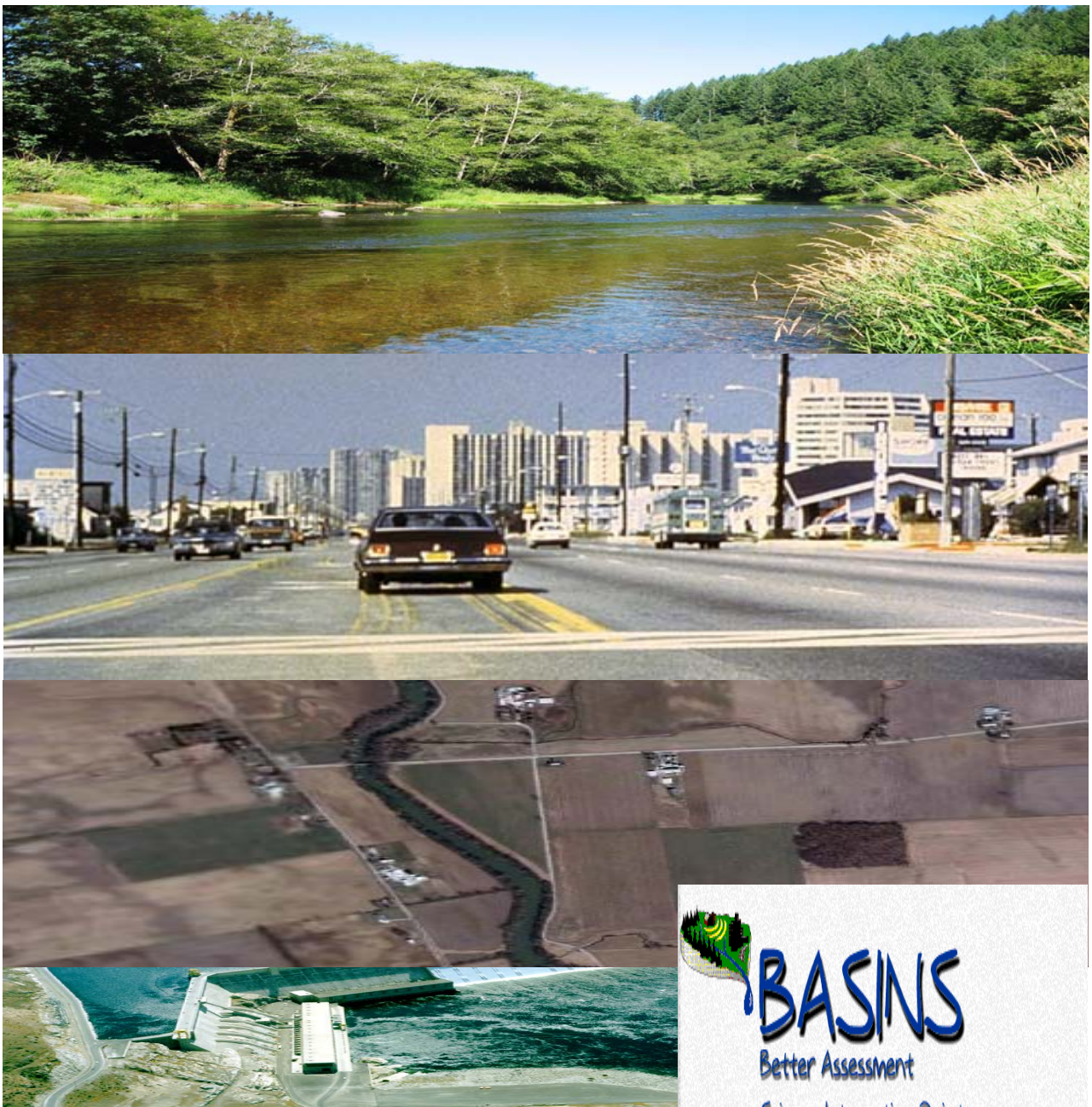**EPA**
United States
Environmental Protection
Agency

Modeling the Impacts of Hydromodification on Water Quantity and Quality

Hydrological Simulation Program - Fortran (HSPF)

BASINS
Better Assessment
Science Integrating Point
and Nonpoint Sources

Modeling the Impacts of Hydromodification on Water Quantity and Quality

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ACKNOWLEDGEMENT

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PART 1. METHOD DEVELOPMENT

MODELING THE IMPACTS OF HYDROMODIFICATION ON WATER QUANTITY AND QUALITY

Abstract: Hydromodification activities are driven by human population growth and resource extraction and consumption including urbanization, agriculture, forestry, mining, water withdrawal, climate change, and flow regulation by dams and impoundments. These anthropogenic activities alter natural flow regimes and lead to reduced downstream water quantity and degraded water quality. Recently, USEPA and states recognized hydromodification as a stressor and a leading source of water quality impairment in streams and rivers. Hydromodification-induced stressors include chemical pollutants, pathogens, nutrients, suspended solids, and flow and habitat alteration. The diverse and interacting nature of hydromodification-induced stressors has made Total Maximum Daily Load (TMDL) development for impaired streams and rivers a major regulatory challenge. Because hydromodification integrates stressors that have combined, cumulative, and synergistic effects on water quantity and quality, TMDL modeling approaches are not well-suited for simulating the impacts of hydromodification. Modeling integrated stressors requires the development and application of predictive models and innovative modeling approaches, such as the Better Assessment Science Integrating Point and Nonpoint Sources (BASINS) modeling framework. Although BASINS has been in use for the past 10 years, there has been limited modeling guidance on its applications for complex environmental problems, such as modeling impacts of hydromodification on water quantity and quality. This report consists of two parts: Part 1 presents the development of a BASINS-based methodology that is applicable to modeling

hydromodification. Part 2 is a case study of how the proposed modeling approach can

forecast the impacts of urbanization on water quantity and quality.

1 INTRODUCTION

Rivers and river-fed lakes are valuable natural resources providing about 61 percent of the nation's drinking water as well as serving riverine habitats to an estimated 40 percent of the fish species and about half of the birds in North America (Whiting, 2002). In addition, rivers store flood waters for groundwater recharge and provide recreational amenities, such as boating, fishing, and swimming (Whiting, 2002) along with navigation, irrigation, power generation, and waste load transport and assimilation (Poff et al., 1997). These valuable ecosystem services are threatened by hydromodification projects such as land resources development for agriculture, energy, mining, forestry, transportation, and residential housing; and water resources development for irrigation, municipal water supply, and flood control. For instance, urban development and efforts to use rivers for transportation, water supply, flood control, irrigation, and power generation often alter flow regimes thus threatening the sustainability of the ecosystem services that rivers and river-fed lakes provide (Poff et al., 1997).

Poff et al. (1997) emphasized the importance of managing the impacts of anthropogenic watershed disturbances and urged scientists to develop management protocols that accommodate economic uses while protecting ecosystem functions. Naiman et al. (2002) stated that forecasting the impacts of changing water regimes is a fundamental challenge for the scientific community. A way to address these challenges and achieve sustainable management of land and water resources at the watershed level is to develop integrative modeling approaches that consider stressors as a system with positive and negative feedback loops, synergies, and interferences (Zimmerman et al. 2009).

A number of factors have delayed the development of integrative modeling approaches and their application to complex hydromodification problems. First, cross-disciplinary, professional boundaries, and different views among hydrologists, engineers, planners, ecologists, and biologists make it difficult to apply a holistic approach to evaluating the impacts of hydromodification. Second, watershed boundaries, which are basic environmental management units, do not usually coincide with local government boundaries where hydromodification decisions are made. Third, water quantity and water quality are mostly regulated separately by various federal and state agencies, even though water quantity strongly influences water quality. Fourth, land use planning, which has strong influence on water quantity and quality, is regulated at the city or county boundary level even though the impacts of land use change on water quantity and quality transcend local boundaries.

Recently, a number of studies have used integrative modeling approaches to assess water allocation options (Letcher et al, 2004), develop hydrologic, agronomic, and economic models for river basin management (Cai et. al. 2003), develop multi-objective evolutionary algorithms for managing ecosystem services (Bekele and Nicklow, 2005), and integrate water allocation and water quality models (Azevedo et al. 2000). Clearly, managing the impacts of hydromodification on water quantity, habitat, and water quality requires modeling approaches that forecast the stressor levels associated with alternative future scenarios (Mohamoud, 2008). Two recent federal government reports on water research and data across federal agencies have emphasized the need for comprehensive watershed management approaches (NRC, 2004; General Accounting Office, 2004). In

this report, integrative modeling approaches are defined as those capable of simulating multiple stressors and their impacts on water quantity and quality.

The dominant hydromodification–induced stressors include flow alteration, water quality degradation, and habitat alteration. At present, EPA's traditional water-quality criteria and standards program do not address the effects of habitat alteration and flow regulation on aquatic life (Jackson and Davis, 1994). Flow alteration due to hydromodification alters water quantity which strongly influences water quality, yet regulatory programs usually do not examine how water quantity changes affect water quality. An area with strong relevance to water quantity and quality is the TMDL program because pollutant load calculations are flow-dependent and are calculated as the product of concentration and streamflow. Flow alterations due to hydromodification may introduce errors in TMDL allocation estimates because relationships of water quantity and quality are not usually examined when developing TMDL plans for impaired water bodies for watersheds impacted by hydromodification.

The proposed modeling approach will allow resource managers to answer some key resource development and management questions that have strong influence on sustainable management of land and water resources. For example, how can managers balance water availability and demand when watershed conditions are continually changing? How can managers jointly forecast flow alteration and water quality degradation due to hydromodification? How can resource managers identify allowable levels of hydromodification using "what-if" scenarios? What ecological risks are associated with different hydromodification categories and levels? What management options are available to resource managers to mitigate the impacts of hydromodification?

Answering these questions would require innovative modeling approaches that can forecast hydromodification-induced stressors, such as flow alteration, water quality degradation, and habitat alteration; and evaluate their impacts on the health of aquatic ecosystems. Given the complex and the interacting nature of hydromodification-induced stressors, a lack of holistic or integrative modeling approaches has been the reason for our inability to manage land use, water quantity and quality, and health of aquatic ecosystems jointly and sustainably. The objective of this study is to show how BASINS can be used to simulate and forecast the impacts of hydromodification on water quantity and quality and discuss ways to assess the cumulative impacts that hydromodification-induced stressors may have on the health of aquatic ecosystems.

2 HYDROMODIFICATION: SOURCE OF INTEGRATED STRESSORS

Hydromodification describes land and water resources development activities that are driven by human population growth and resource consumption. These activities often produce direct or indirect changes to water quantity and quality. USEPA (1993) defines hydromodification as the "alteration of the hydrologic characteristics of coastal and non-coastal waters, which in turn could cause degradation of water resources." According to USEPA (2007), hydromodification consists of channelization and channel modification, construction of dams and impoundments, and streambank and shoreline erosion. In the literature, hydromodification has also been narrowly defined as hydrograph modification.

USEPA (2007) presents hydromodification as a leading source of water quality impairment for streams, lakes, estuaries, aquifers, and other water bodies in the United States. The National Water Quality Inventory Report to Congress (2004) that was released in 2009 identified agricultural nonpoint source (NPS) pollution as the primary (48%) water quality impairment of assessed streams and rivers followed by hydromodification (20%), and habitat alteration (14%) (USEPA, 2009). Figure 1 shows the top ten sources of impairment in assessed rivers and streams in the United States. They are closely linked to human activities that alter the physical structure or the natural function of a water body. Water quality degradations caused by hydromodification include increased sedimentation, higher water temperature, lower dissolved oxygen, degradation of aquatic habitat structure, and loss of fish and other aquatic populations.

	Miles
Agriculture	94,182
Hydromodification	61,748
Unknown	48,957
Habitat Alteration	42,752
Natural/Wildlife	39,120
Municipal Discharges/Sewage	35,301
Unspecified Nonpoint Source	34,556
Atmospheric Deposition	27,522
Resource Extraction	22,691
Urban Runoff/Stormwater	22,559

Percent of Impaired Stream Miles Affected
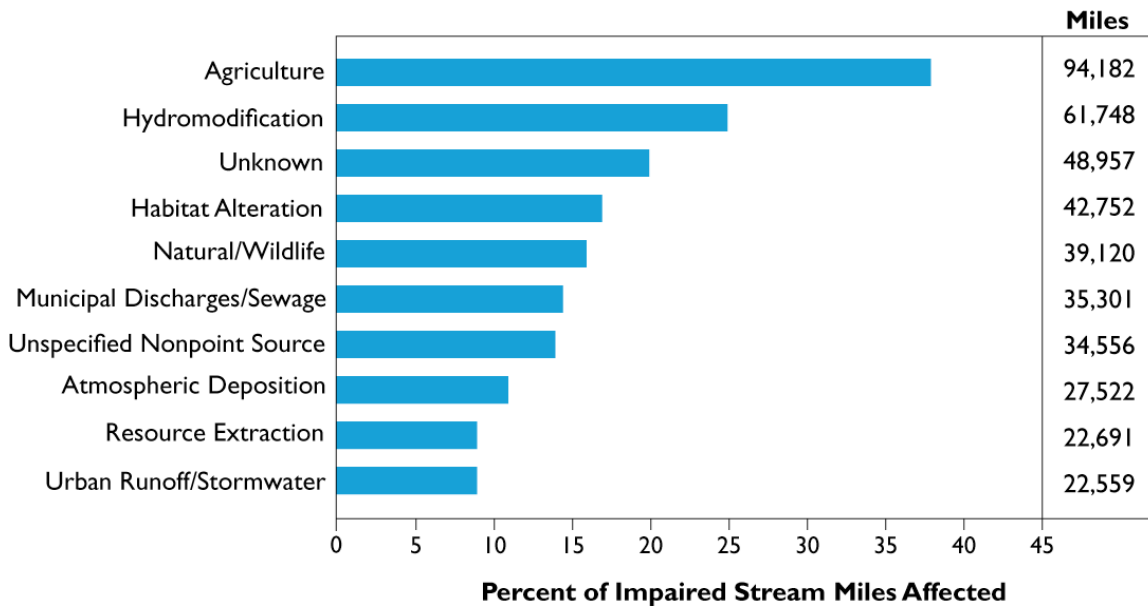
Figure 1. Top 10 sources of impairment in assessed rivers and streams in the United States (Source: USEPA, 2009)

Adding to USEPA's narrow definition of hydromodification, we define hydromodification more broadly to include urbanization, climate change, water withdrawals, and inter-basin transfers. Our intention is to use the term for a wide range of anthropogenic watershed disturbances that alter natural flow regimes and degrade water quality. Addressing the impacts of integrated stressors is more effective than addressing stressors individually one at a time because integrated stressors have integrated effects that are not independent. Furthermore, many watersheds are impaired by integrated stressors and resource managers are unable to identify the cause of impairment. Despite being a major source of impairment in assessed water bodies, modeling approaches that consider the impacts of hydromodification on water quantity and quality are not available. The following sections present selected hydromodification categories and introduce our proposed approach to modeling each category. Note that water quantity, flow, streamflow, and water availability are used interchangeably throughout this report.

8

3 HYDROMODIFICATION CATEGORIES AND MODELING CHALLENGES

3.1 Urbanization

As the total impervious area in a watershed increases, peak flow rates and flow volumes increase (Arnold and Gibbons, 1996; Tang et al., 2005) and baseflow and groundwater recharge decrease (Rose and Peters, 2001). Such alterations of natural flow regimes affect the distribution of surface water and baseflow components of streamflow. Hydrologic imbalances caused by urbanization have serious consequences for water availability. In many parts of the world, incidence of water supply shortages due to land use change have been reported for communities in water-rich areas (Okun, 2002). In the United States, frequent, severe droughts and urbanization-induced water shortages have been observed in some parts of New England, in the southeast (Atlanta), and in areas in the west coast (Seattle and Portland) (Sehlke, 2004).

Urbanization not only alters natural flow regimes, it also degrades water quality. For sub-basins in east of Melbourne, Australia, Hatt et al. (2004) found that water quality loads were correlated with imperviousness and drainage connections. Other studies have linked urbanization and associated imperviousness to increased sediment, bacteria, and nutrient loads (Schueler, 1995; Gove et al., 2001; Mallin et al. 2001

Our approach to forecasting the impacts of urbanization on water quantity and quality is presented in Part 2 of this report.

3.2 Water Withdrawals and Interbasin Transfers

Water withdrawn directly from rivers and streams alters the natural flow regime. Withdrawals and interbasin transfers are water management options that allow managers to balance water demand and water availability by issuing water withdrawal permits. However, without knowing the available water levels under future development scenarios, issuing water withdrawal permits would not balance water demand and water availability. It may, however, lead to violation of downstream environmental flow targets and minimum flows required by the Endangered Species Act (ESA). Managing the effects of water withdrawals and interbasin transfers not only requires the use of hydrologic and water quality models to forecast scenario-specific water quantity and quality changes, but also water allocation models to ensure that sufficient quantity and quality of water is available to all water users. In general, water allocations are simulated with state-specified models that have no hydrologic and water quality simulation capabilities. These models include the Water Rights Analysis Package (WARP) used in Texas (Wurbs, 2005) and MODSIM: River Basins Management Decision Support System used in Colorado (Labadie, 2005).

Our approach to modeling the impacts of water withdrawals on streamflow uses HSPF to simulate water quantity and quality projections for alternative future scenarios. Under each scenario, the effects of water withdrawal levels on downstream flow targets and water quality impaired streams are evaluated, and the scenario that most closely matches water availability with water demand and minimizes the overall impacts of hydromodification is selected.

3.3 Channel Modification and Streambank Erosion

Streams and rivers, which are important habitats to many aquatic organisms, are affected by channel modification. Channel modifications include direct channel operations such as dredging, widening, and straightening, or indirect modifications caused by flow alteration. Today, many streams and rivers are degraded by hydromodification-induced stressors such as flow alteration, unsanitary discharge, and channelization projects (Leblanc et al.,1997). A number of investigators have examined the ecological impacts of hydromodification by assessing the biotic integrity of streams using multi-metric indices, such as the indices of biotic integrity (IBI) (Karr 1991; Fitzpatrick et al. (2004) or bioindicator approaches (Adams, 2005). To link urbanization-induced stressors to stream habitat degradation, a number of investigators have examined relationships between total impervious area (TIA) and biological integrity (Morse et al., 2003; Booth et al. 2004). Other investigators have related hydrologic metrics (estimated from simulated or observed streamflow data) to biological integrity (Claussen and Biggs, 1997; Booth et al., 2004; Konrad et al., 2005). For example, Richter et al. (1996) identified flow magnitude, frequency, duration, timing, and rate of change as ecologically relevant hydrological indicators.

Our approach to modeling channel modification and streambank erosion consists of three steps. First, modelers use HSPF to simulate changes in water quantity and quality due to hydromodification. Second, modelers develop hydrologic indicators (e.g., Q_2) from simulated long-term streamflow time series. In general, Q_2 is defined as the flow that corresponds with the two-year recurrence interval. Third, modelers evaluate the impacts of simulated stressors such as increased flow velocity, shear stress, and

streambed and streambank erosion on stream habitat quality. Specifically, our approach to modeling the impacts of channel modification on stream habitat is to use simulated hydrologic, hydraulic, and water quality metrics that link hydromodification-induced stressors to stream habitat quality. In addition, hydromodification impacts on stream habitat can be evaluated by linking HSPF to a habitat suitability model such as PHABSIM (Milhous et al. 1984).

3.4 Flow Regulation by Dams and Impoundments

Trends in urbanization, population growth, and increased water demand and usage have led to extensive damming of rivers and streams. In the United States, more than 85% of the inland waterways are now artificially controlled (NRC, 1992), including nearly 1 million km of rivers that are affected by impoundments (Echeverria et al. 1989). Dams and impoundments control flooding, generate electric power, and provide irrigation, navigation, recreation, and municipal water needs, but in some cases, their benefits to society are outweighed by their adverse environmental impacts. Dams and impoundments cause flow alteration, and inundate wetlands and riparian areas. They also tend to reduce or eliminate downstream flooding, block fish migration routes, increase turbidity and sedimentation during construction, and retain sediment after construction.

Flow regulation by dams and impoundments profoundly alters natural flow regimes, which results in degraded river ecosystems (Ward and Stanford, 1995; Ligon et al. 1995; Power et al., 1996). Fishery managers have long argued that maintaining the pre-development natural flow regime is essential to the composition and structure of native riverine ecosystems and associated biodiversity (Richter et al. 2000). To mitigate the ecological impacts of flow regulation, several mitigation measures have been proposed.

These measures include: establishing minimum flow releases (Colby, 1990), offering controlled flushing flows (Collier et al 1997), or maintaining natural flow regimes to flow levels observed before the regulation project (Stanford et al., 1996; Poff et al. 1997).

Our approach to modeling the impacts of flow regulation on water quantity and quality is to set up a flow regulation scenario by placing a hypothetical dam and impoundment at different locations in the watershed. The first step is to build an HSPF hydraulic function table known as the FTable with the desired elevation-area-storage-discharge relationships to represent the dam and the impoundment. To evaluate the effect of flow regulation on natural flow regimes, modelers can compare pre-regulation and post-regulation observed and simulated streamflows for each future scenario. Based on these comparisons, they select scenarios that maintain the pre-development hydrologic condition of the watershed. Resource managers can also examine levels of stressors associated with different flow regulation scenarios to select a scenario that minimizes the integrated effects of flow regulation on water quantity, quality, and the health of aquatic ecosystems.

3.5 Climate Change

General circulation models (GCMs) are used to make future climate change projections that account for increasing levels of CO_2 in the atmosphere. Mean global surface temperature is expected to increase in the range of 1.5 to 5.8 $^{\circ}$C by 2100 (Houghton et al., 2001). For the United States, mean temperature and precipitation projections are about 4.5 $^{\circ}$c increase in temperature and 7.5 % increase in precipitation. In general, GCM model projections are used to estimate changes in precipitation and temperature. Four commonly used GCM models are: The Goddard Institute of Space

Studies (GISS), Geophysical Fluid Dynamics Laboratory (GFDL), the United Kingdom Meteorological Office (UKMO), and the Oregon State University (OSU) models. Climate change projections vary across regions, but most regions are projected to have increased frequency of intense storms, increased soil erosion and sedimentation, and increased sea-level. Consequently, climate change can be expected to have serious effects on water quantity, water quality, and the health of aquatic ecosystems.

Our approach to modeling climate change impacts on water quantity and quality is to use GCM temperature and precipitation projections. Model users can select scenarios that are similar to mean GCM projections for the United States or scenarios that differ from mean GCM projections. For example, a modeler could select a scenario that has 10% increase in annual precipitation, with a 10% increase in the frequency of high precipitation events, and increased return frequency of storms of particular magnitudes. For temperature increases, the modeler could select a corresponding scenario with two degree increases during the cool season months, and four degree increases during the warm months.

Note that climate change is only one driver of hydromodification, and it can be addressed separately or concurrently with urbanization, flow regulation, and channel modification. For more discussion on generating climate change scenarios for the HSPF model in BASINS, interested readers should refer to the Climate Assessment Tool (CAT) manual, which can be accessed at http://cfpub.epa.gov/ncea/cfm/recordisplay.cfm?deid=203460.

4 THE NEED FOR HYDROMODIFICATION MODELING FRAMEWORK

The proposed hydromodification modeling approach makes the watershed the unit of management and regulatory focus. This is based on the concept that every land area belongs to a watershed and that the integrated effects of all hydromodification activities that occur in a watershed have a measurable impact at the watershed outlet or at some downstream point of interest. Watershed-based approaches also address the complexities in modeling hydromodification impacts and allow resource managers to forecast scenario-specific hydromodification stressor levels, evaluate their impacts on water quantity and quality, and develop management plans that mitigate the impacts of hydromodification-induced stressors. The proposed approach is applicable to restoration efforts, but is more suitable for protecting least developed watersheds or pristine watersheds that require judicious management and development protocols. To guide management decisions that protect or restore watersheds, we present some key hydromodification related questions that resource managers must answer before hydromodification project plans are implemented in a watershed (Figure 2).

The watershed-based concept presented in Figure 2 resembles the TMDL process in the sense that TMDLs are intended to reduce pollutant loads to levels that can be assimilated by a water body in order to meet EPA's water quality standards. In general, the TMDL process has a very limited scope because it addresses water quality only, and not water quantity. In addition, TMDLs often target a single stressor and a single water body. Unlike TMDL approaches, the approach described in this report assesses a watershed's capacity to assimilate not only pollutant loads, but also all

hydromodification-related stresses. Managing hydromodification impacts on water quantity and quality at the watershed level has many benefits but it requires cooperation among stakeholders. Cooperation among stakeholders is essential because watershed boundaries often cross multiple jurisdictions. The New Jersey Department of Environmental Protection (NJDEP) recognized that pollutant loads, water withdrawals, and land use required new approaches that could not be addressed by regulatory programs alone (EPA, 2004). Regulatory programs such as TMDLs have limited scope and cannot be used to address integrated stressors such as flow alteration, habitat alteration, and water quality degradation.

The proposed approach was developed in recognition of the need for comprehensive watershed management approaches that link economic development to ecosystem sustainability. For instance, using the proposed modeling approach, resource managers can identify where to institute land preservation, and where to place hydromodification projects in a watershed (Figure 2).

Can land use planners, resource managers, and regulatory staff agree on allowable stressor levels before implementing hydromodification project plans in a watershed?

What management options are available to mitigate the impacts of hydromodification projects at the watershed scale?

Water Withdrawal

Dam and Impoundment

Urbanization

Channel Modification

What levels of flow alteration and water quality degradation or loss of ecosystem services due to hydromodification can be assimilated by a watershed?

Figure 2. Hydromodification activities and watershed management questions that resource managers must answer before hydromodification projects are implemented.

4.1 Essential Components of a Hydromodification Modeling Framework

Figure 3 illustrates a conceptual hydromodification modeling framework that consists of four components. The first component characterizes present and future hydromodification levels using "what-if" scenarios that have different percentages of urban land and percent impervious cover, different distributions of land use across a watershed, number and location of dams and water impoundments, and water withdrawal and inter-basin transfer amounts.

The second component identifies and quantifies scenario-specific levels of hydromodification-induced stressors, i.e. flow and habitat alterations and water quality degradation, and assesses how integrated stressors may affect the health of aquatic ecosystems.

The third component identifies and selects best management practices (BMPs) that mitigate the impacts of hydromodification on water quantity and quality. Mitigating the impacts of hydromodification requires a watershed-scale hydromodification management plan that may include of BMPs for controlling runoff at the source (i.e. those that enhance infiltration). Local governments in coastal California and other western states developed hydromodification management plans to control nonpoint source pollution from urban watersheds.

The fourth component uses adaptive management to iteratively evaluate how model simulation results agree with data obtained after future scenarios are implemented in the watershed. By continuously updating the model and comparing scenario-specific model forecasts with post-implementation observed data, resource managers can determine

when stressor levels reach unsustainable levels and modelers can validate model

simulations and minimize predictive uncertainties..
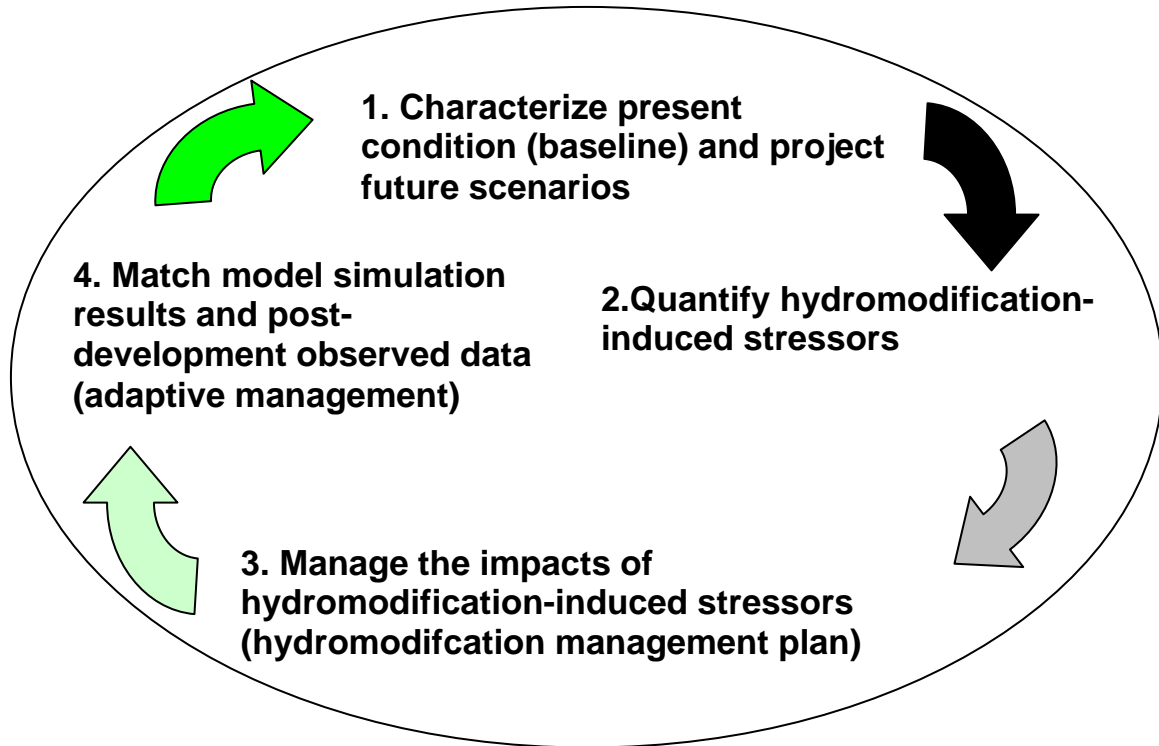
Figure 3. Components of a hydromodification modeling framework

4.2 BASINS Modeling Framework: A Background

The Clean Water Act was established to restore and maintain the chemical, physical, and biological integrity of the nation's waters. In the 1970s and 1980s, EPA successfully regulated point source pollution through the National Pollutant Discharge Elimination System (NPDES) permit program. However, managing nonpoint source pollution from terrestrial ecosystems has proven to be more difficult. To manage this problem efficiently, EPA adopted an approach that makes the watershed the unit of regulatory focus (USEPA, 1998; Whittemore and Beebe, 2000). Adoption of the watershed approach as the management and regulatory unit has created a need for watershed models and modeling approaches. Today, watershed models are widely used to manage nonpoint source pollution, particularly through the development of TMDL plans for water quality impaired water bodies. To manage land and water resources in a sustainable manner, resource managers need modeling approaches that also forecast the impact of hydromodification on water quantity and quality. In 2007, EPA released a guidance document on managing nonpoint source pollution caused by hydromodification (USEPA, 2007). This document listed a number of models applicable to hydromodification modeling, but did not present specific hydromodification modeling guidance. HSPF and AQUATOX, which are part of the BASINS modeling framework, were among the models listed.

In 1996, USEPA released BASINS ver. 1.0 modeling and decision support system (EPA, 1996a). BASINS integrates Geographic Information System (GIS) tools, national databases (elevation, hydrography, meteorological, land use, and soil), assessment tools (target, assess, and data mining), data management and graphing programs (WDMUtil

and GenScen), models (HSPF, SWAT, PLOAD, and AQUATOX), and analysis tools including the Climate Analysis Tool (CAT).

Figure 4 matches BASINS' modeling capabilities with hydromodification drivers, stressors, and impacts. As shown in Figure 4, using GIS tools and databases, BASINS provides access to information about soils, topography, and land use and land cover of a watershed. In addition, BASINS provides information on hydromodification projects already present in the watershed and their distribution in the landscape. Resource managers can use BASINS to identify priority areas for preservation and development. As stated earlier, hydromodification activities are driven by population growth and the accompanying need for resource extraction and consumption. The watershed management goal is to minimize the impacts of economic development projects on water quantity and quality by reducing hydromodification-induced stressors to levels that can be assimilated by the watershed or mitigated through BMPs. Here, we briefly discuss some of the models available in BASINS that apply to modeling hydromodification.
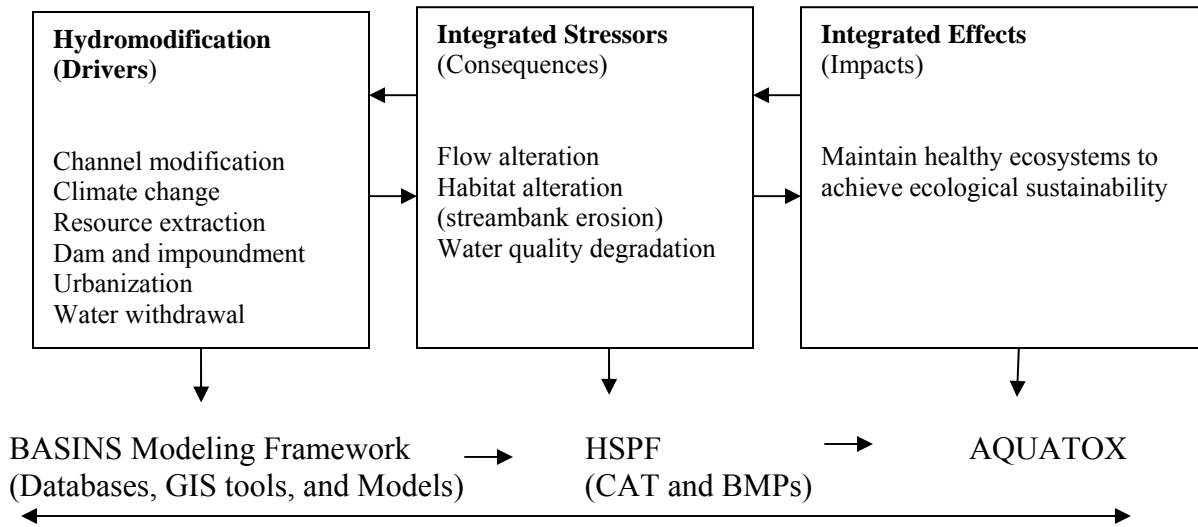
Figure 4. **Matching framework modeling capabilities with hydromodification drivers, integrated stressors, and integrated effects.**

HSPF is EPA's premier watershed hydrology and pollutant transport model (Whittemore and Beebe, 2000) and is the core watershed model in BASINS. HSPF simulates hydrologic processes and water quality for a range of types of user-defined scenarios. Because of its extensive water quality simulation capabilities, HSPF is frequently used for TMDL plan development for impaired water bodies. In addition to water quality and hydrologic process simulations, HSPF also serves as a water allocation model. It has been used to assess the effects of land use change on streamflow (Brun and Band, 2000; McColl and Agett, 2007), effects of water withdrawal on streamflow (Zariello and Reis, 2000), and effects of climate change on water quantity (Middlekoop et al. 2000; Goncu and Albek, 2009). Furthermore, HSPF has been used for developing hydrological and biological indicators of flow alteration in the Puget Sound low land streams (Cassin et al. 2005). Although HSPF is the core model, BASINS has other models and tools that are applicable to modeling hydromodification. Models and tools that are relevant to the objectives of this study are AQUATOX, climate analysis tool (CAT), and HSPF's BMP Toolkit. Whittemore and Beebe (2000) reviewed BASINS and

noted that its success would depend on efforts to share technical experiences and solutions to problems.

The BASINS Climate Analysis Tool generates climate change scenarios for HSPF. CAT users need information on climate change projections for the region of interest within the United States. Based on Global Circulation Models (GCMs) projections of the region of interest, CAT users can develop climate change scenarios by changing temperature, precipitation, and evapotarnspiration data. Note that CAT is not available in early versions of BASINS, but is available in BASINS 4.

Representing best management practices and simulating their effectiveness in controlling nonpoint source pollution is a desirable feature in regulatory models. In general, models employed for TMDL plan development are not well-suited for plan implementation because many models lack BMP representation and simulation capabilities; Whittemore and Beebe (2002) emphasized these aspects of the HSPF model. According to Endreny (2002), HSPF has no explicit simulation of storm sewer networks and lacks the capability to simulate common water quality BMPs. A recently developed HSPF BMP Toolkit represents some types of BMPs, and it can be used to evaluate their effectiveness in controlling runoff and pollutant loads. HSPF BMP related limitations have been addressed and HSPF is now one of the few publicly available watershed models with capabilities to represent and simulate vegetative, storage, and infiltration BMPs. Currently, the BMP Toolkit is a web-based tool and is not part of BASINS, but modelers can easily work with the BMP toolkit to modify their model input data. .

AQUATOX is an ecological effects model that can be used to evaluate past, present, and future direct and indirect effects from various stressors, including nutrients, organic

wastes, sediments, toxic organic chemicals, flow and water temperature in aquatic ecosystems (Park et al. 2008). AQUATOX uses HSPF output as input, including results of simulations that include hydromodification-influenced stressors. Based on the magnitudes of stressors generated by HSPF in different scenarios, AQUATOX can simulate how hydromodification-induced stressors may affect biota in aquatic ecosystems.

4.3 Hydromodification Decision-making Example

Figure 5 presents a flow chart describing how to apply the BASINS modeling framework to watersheds that are likely to experience hydromodification. As an example, we selected two hydromodification categories: urbanization and flow regulation by dams and impoundments. For each, we present a list of hydromodification-induced stressors and scenarios. As shown in Figure 5, the first step is to characterize the current hydromodification levels, then determine if additional hydromodification projects are planned for the watershed. First, modelers use HSPF to simulate streamflow and water quality; modelers then calibrate and validate the model under present conditions. After successful calibration and validation, model users then simulate flow and water quality for the future scenarios. Comparisons of flow and water quality for pre-hydromodification (current watershed condition) and post-hydromodification determines flow alteration and water quality degradation levels associated with each scenario.

In the second step, after simulating flow and water quality for baseline and future scenarios, model users set up HSPF as a water allocation model to allocate and track how different hydromodification categories and scenarios affect water use and availability. HSPF uses a schematic network of nodes and links to track changes in inflows, storages,

and outflows over time. To minimize flow alteration impacts, resource managers may seek to identify scenarios that closely match pre-hydromodification water availability levels.

In the third step, model users simulate how alternative future may alter stream habitats and water quality. Models in this case are used to evaluate how simulated changes due to hydromodification are likely to affect existing TMDL allocations and/or the health of aquatic ecosystems. As an example, modelers can employ the AQUATOX model to explore whole-system stressor-response relationships. Modelers can also extract hydrologic and water quality indicators or metrics from simulated streamflow and water quality data, then use these metrics to explore system stressor-response relationships. Based on simulated scenario comparisons, resource managers can select hydromodification management plans that minimize undesirable impacts on aquatic resources. Finally, to address model predictive uncertainties, resource managers may employ an adaptive management approach to compare and update model simulations based upon data collected after initiation of hydromodification projects.

Characterize the current hydromodification level of the watershed

Are there any planned hydromodification projects?

Yes

No

What type of hydromodification projects are planned for the watershed?

Develop a baseline model for the watershed

Maintain current water quantity an quality condition of the watershed

Urbanization

Construction of Dams and Impoundments

Future build-out scenarios
- Set hypothetical "what-if" build-out scenarios with different total impervious area (TIA) levels

Future scenarios
- ✓ Build a new dam/reservoir
- ✓ Test different water withdrawal levels
- ✓ Test different reservoir operations and water allocation options

Management Plan Evaluation
Are flow alterations and water quality degradation controlled? Are water availability and demand balanced? Are ecosystem services maintained at prepoject levels?

Yes

Hydromodifi-cation impacts are minimized

Stressor forecasting (HSPF Model)
- Forecast flow alteration
- Forecast water quality degradation
- Forecast habitat alteration
- ✓ Forecast water availability
- ✓ Forecast water demand

No

Link HSPF to AQUATOX Model
Assess the ecological risks associated with alternative future scenarios

Hydromodification Management Plan
Simulate HSPF with/without BMPs, select suitable best management practices (BMPs) using HSPF BMP Toolkit

Adaptive Management
Select different scenarios and re-evaluate the decision-making process
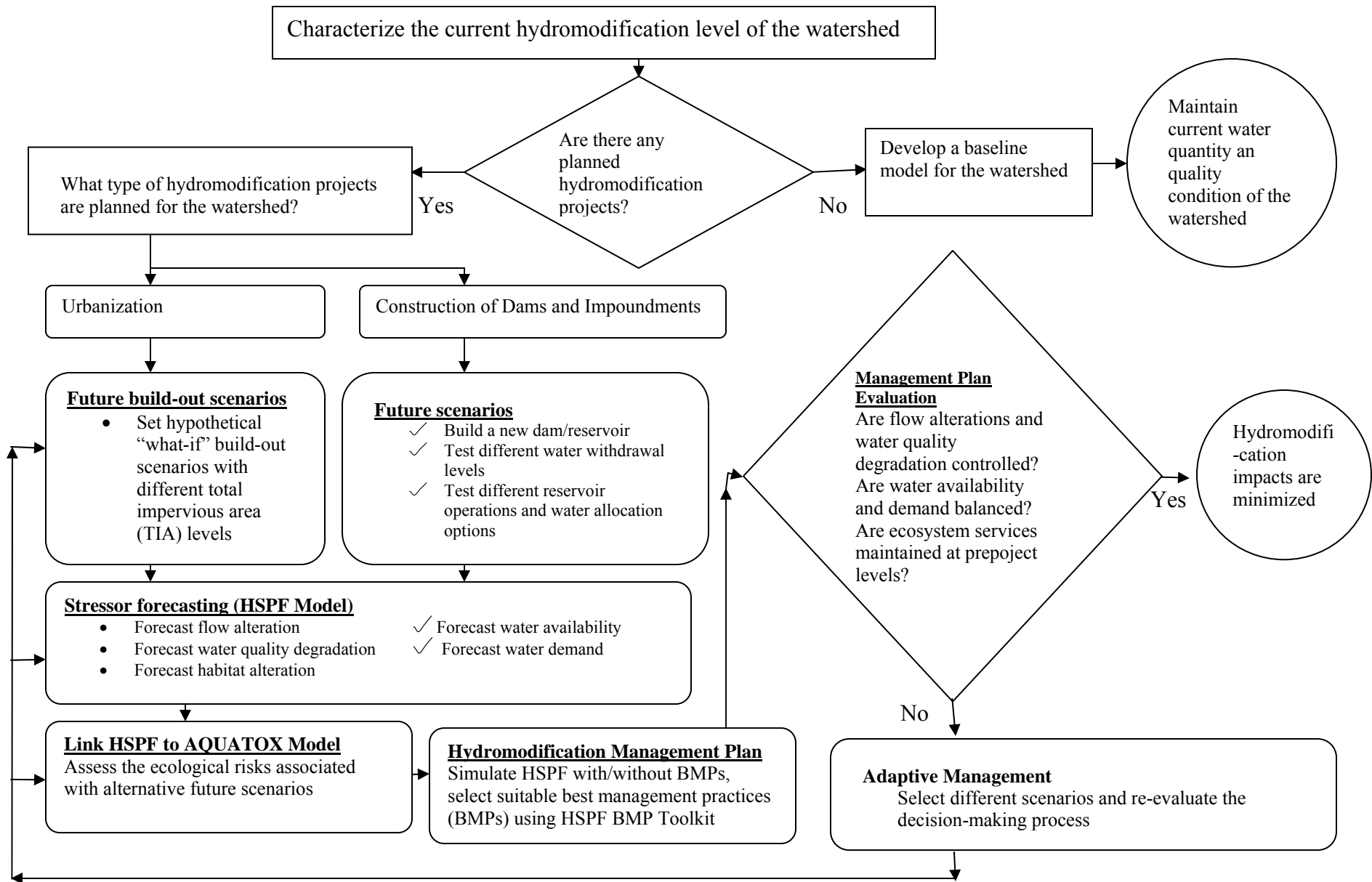
Figure 5. Decision-making framework for watersheds receiving hydromodification projects

4.4 Linking Integrated Stressors to Aquatic Ecosystem Impacts

As stated earlier, AQUATOX is an ecological risk assessment model, which is part of the BASINS modeling framework. AQUATOX simulates the effects of multiple simultaneous stressors that may include nutrients, organic toxicants, temperature, suspended sediment, and flow, on the health of aquatic ecosystems. Potential applications of the AQUATOX model include estimation of fish recovery after pollutant loads are reduced, ecosystem responses to invasive species, effects of mitigation measures, and changes in ecosystem services. By forecasting hydromodification-induced stressors using HSPF as the load-generating model, and assessing the resulting ecological impacts of the simulated stressors with AQUATOX, resource managers can evaluate the biological ramifications of hydromodification projects before the projects are implemented. Although linking integrated stressors to aquatic ecosystem impacts is beyond the scope of the current study, our suggested approach to establishing stressor-response relationships is to link HSPF to the AQUATOX model.

4.5 Model Application Example

In Part 2 of this report, we present a case study demonstrating how BASINS can be used to forecast urbanization-induced stressors, namely flow alteration and water quality degradation under various urban development scenarios. We include in this example a brief discussion of how urbanization-induced stressors affect stream channel erosion. We also attempt to link urbanization-induced stressors to impacts on the health of aquatic ecosystems using hydrological indicators or metrics. As a test watershed, we selected a

medium-sized headwater watershed, which is a tributary of the Yaquina Watershed in

Oregon, USA.

PART 2: APPLICATION EXAMPLE

5. FORECASTING URBANIZATION IMPACTS ON WATER QUANTITY AND QUALITY: CASE STUDY OF THE YAQUINA WATERSHED, OREGON, USA

Abstract: Protecting ecosystem services provided by headwater watersheds increasingly is becoming an important land and water management objective. To allow resource managers to minimize future watershed degradation and eliminate the need for restoration, we present a method to forecast potential impacts of urbanization before urban development plans are implemented in a watershed. The method establishes both a baseline that represents the pre-development condition, and build-out scenarios that represent future development. In this report, we arbitrarily selected three future build-out scenarios that represent 15, 45, and 85 percent total impervious area (TIA), where impervious cover was used as a landscape indicator to represent the effects of development. We employed the Hydrological Simulation Program – FORTRAN (HSPF) watershed model to simulate streamflow, total suspended solids (TSS), and nitrate in the baseline scenario and for the three build-out scenarios. Comparisons of simulated results in the baseline scenario to those forecasted for the three build-out scenarios show increased peak flows and decreased baseflows for the build-out scenarios. Nitrate simulation results show increased nitrate concentrations for the three build-out scenarios. Suspended sediment concentrations were forecasted to increase with increasing TIA from 15 to 45 percent, but the 85 percent TIA scenario resulted in lower TSS concentrations. Our forecasts also indicate that mean channel wetted width, flow depth, and flow velocity decrease with increasing percent TIA. The proposed method serves as an exploratory

approach in which resource managers and land use planners forecast urbanization-

induced stressors before urban development plans are implemented in the watershed. The

impacts of these stressors on aquatic ecosystems and services can then be simulated.

5.1 Introduction

For nearby receiving water bodies, urbanization has been found to lead to altered hydrologic regimes, deteriorated water quality, losses of habitat and biodiversity, beach closings, fishery declines, and fish consumption advisories (Nixon, 1995; Richardson, 1997; Rabalais et al., 1996; Boesch at al., 2001; Elofson et al., 2003; Niemi et al., 2004). Many of the adverse ecological impacts of urbanization are closely linked to increases in impervious area (Paul and Meyer, 2001; Allan, 2004). The percent of total impervious area (TIA) is that portion of a watershed area covered by built surfaces such as paved roads, parking lots, sidewalks, driveways, and rooftops. Percent TIA in a watershed has been used as a gross indicator of urbanization and associated impacts on streamflow (Ng and Marsalek, 1989), water quality (Schueler, 1994; Arnold and Gibbons, 1996; Conway, 2007), and the health of aquatic ecosystems (Klein, 1979; Morse et al., 2003).

In general, as the percent TIA increases, the fraction of precipitation that is able to infiltrate the soil and recharge groundwater decreases, and the fraction that becomes overland runoff increases (Schueler, 1994). Without major mitigation measures, urbanization causes reduced baseflow and declining water tables, and increased flood flow magnitude and frequency (Arnold and Gibbons, 1996; Paul and Meyer, 2001), with resulting changes in stream channel morphology and in-stream suspended sediment from the increased scouring, and consequent stream habitat degradation (Booth, 1990; Paul and Meyer, 2001).

Urbanization also causes increased nutrient (Creed and Band, 1998; Wernick et al., 1998; Donner et al., 2004), suspended sediment (Nelson and Booth, 2002; Wotling and Bouvier, 2002) and other pollutant inputs (e.g. metals, pesticides, road salts) from

terrestrial areas to receiving water bodies, along with increased water temperature (Niemi et al., 2004). Such urbanization-induced changes cause stresses to aquatic organisms that impair the overall health of aquatic ecosystems. Increased nitrate export to coastal waters leads to estuarine eutrophication, with changes in biotic community structure and diversity (Turner and Rabalais, 1991; Vitousek et al., 1997; Boesch et al., 2001). Compton et al. (2003) found positive correlation between nitrate concentrations and broadleaf cover dominated by red alder (alnus rubra) in forested watersheds in the Oregon Coast Range. Increased sediment inputs cause reduced light penetration and may directly impede the reproductive processes of aquatic organisms (Wotling and Bouvier, 2002; Nelson and Booth, 2002).

Land use planners and decision-makers need guidance on how to forecast the impact of urban development on hydrologic processes (e.g., increased flood frequency and loss of groundwater recharge), on nutrient and sediment concentrations, and on ecological integrity (e.g., loss of biodiversity and wildlife habitat). Much of what is known about the effects of urbanization-induced stressors on aquatic ecosystems was obtained through field observations (Morse et al., 2003; Konrad and Booth, 2005). Monitoring urban streams alone cannot, however capture the full impact of urbanization because impacts observed in urban streams are often moderated by existing best management practices (BMPs). In addition, monitoring urban streams usually does not account for impacts that have occurred during construction but before mitigation measures were fully implemented. Stream monitoring is a reactive approach that offers limited insights to land use planners and decision-makers. The ability to forecast future conditions reliably in

urban streams before implementation of development plans obviously would be more helpful.

Many investigators acknowledge the interdependence of structure and function of stream ecosystems and natural flow variability (Poff and Allan, 1995, Power et al., 1996, Richards et al., 1997). For example, flow alterations often result in changes to the ecological organization of aquatic and riparian systems that lead to changes in physiology and behavior of individuals, populations, community composition, and food web structure (Poff et al., 1997; Bunn and Arthington, 2002; Poff et al., 2006). Building on such established relationships, hydrologic metrics with strong ecological relevance have been developed for natural flows (Poff and Ward, 1989; Poff et al., 1997) and for altered flows (Richter et al., 1996; Olden and Poff, 2003). However, approaches that link urbanization-induced stressors explicitly to specific ecological effects, such as loss of aquatic life or physical habitat, are lacking.

A proactive approach is needed for forecasting urbanization-induced stressor levels and establishing links between them and the resulting condition of aquatic ecosystems. An alternative to monitoring streams to assess the ecological impacts of urbanization is to forecast urbanization impacts with hypothetical build-out scenarios or actual development plans. Using hypothetical build-out scenarios, resource managers employ comprehensive watershed models to forecast urbanization-induced stressor levels before implementation of urban development plans and any associated mitigation measures in a watershed. We propose herein using the Hydrological Simulation Program - FORTRAN (HSPF) (Bicknell et al., 2001) to simulate streamflow, nitrate, TSS, and channel morphology changes.

The objective of this study is to present a modeling approach to forecast potential impacts of alternative urban development scenarios by using watershed total percent impervious area as an indicator of urban development impacts on hydrology, flow width, flow depth, flow velocity, nitrate concentration, and TSS concentration.

5.1.2 Watershed Description

The study watershed is a headwater tributary located upstream of the United States Geological Survey's gaging station near Chitwood, Oregon (44° 39' 29" N, 123° 50' 15" W) (Figure 6). The study watershed is characterized by wet winters, relatively dry summers, and mild temperatures that are typical of the Coastal Range areas of Oregon. The long-term average annual precipitation for Newport is 1767 mm. Precipitation rainfall comes from moist air masses from the Pacific Ocean. A large portion of the precipitation occurs in November, December, and January. Conversely, the warmest and driest months are July, August, and September (Figure 7).
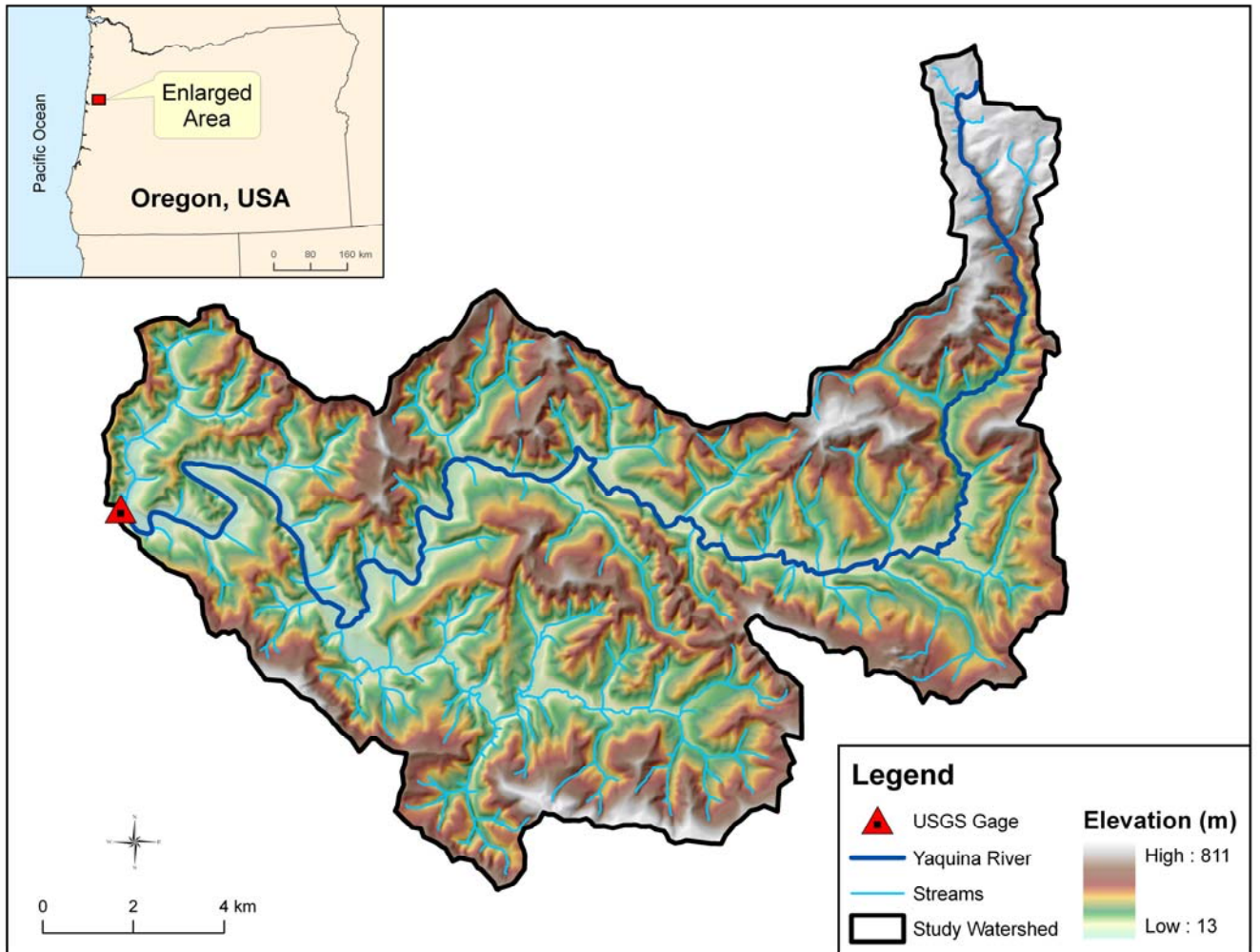
Figure 6. Location of the study watershed.
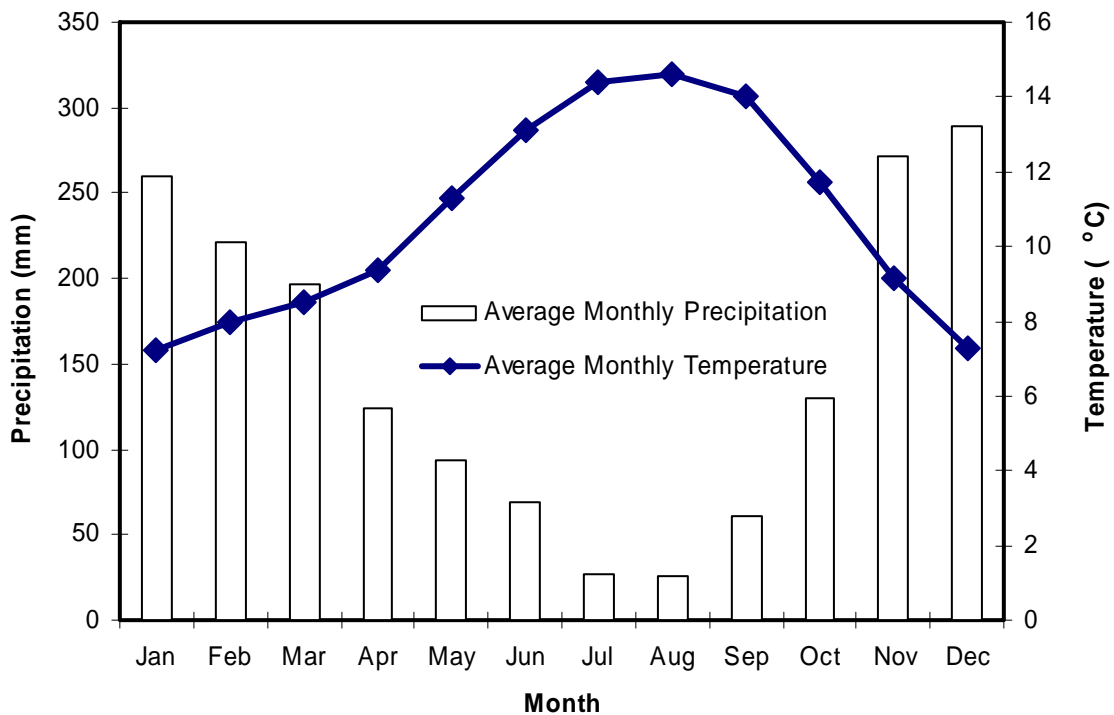
Figure 7. Distribution of long-term monthly precipitation and temperature for Yaquina Watershed

The watershed has a drainage area of 184 km^2. Presently, the study watershed is 97 percent covered by evergreen forest, about two percent covered by agricultural land, and about one percent covered by medium density urban residential land (Table 1). The dominant land cover is coniferous forest because the original complex forest has been replaced by single species silviculture and opportunistic pioneer species. In riparian areas along streams, disturbed sites are frequently occupied by pioneer broad-leaf trees (e.g., red alder) (Ohmann and Gregory, 2002).

The dominant soil of the study watershed is Bohannon soil series found on the western slopes of the Oregon Coast Range areas. The soils of the watershed are generally

well drained with poorly developed horizons (Ohmann and Gregory, 2002). Geological

formations of the watershed contain massively to thinly-bedded tuffaceous siltstones,

sandstones, basalt breccias, and augite-rich tuff that provide a constant source of silica

and suspended sediment (Snavely and Wagner, 1963).

Table 1. Baseline and hypothetical future land use scenarios

Watershed Area (Hectares) by Land use Category			Impervious Area at Build-out (%)
Scenarios	Forest (ha)	Agriculture (ha)	Urban (ha)	
Baseline	18057	261	25	---
Scenario15	12946	261	5136	15
Scenario45	7443	261	10639	45
Scenario85	106	261	17976	85

5.2 Methods

The method proposed to forecast urbanization-induced stressors consists of the following steps. More detailed discussion of each step is given in subsequent parts of the methods section.

Step 1: Characterize the current land use, hydrology, and water quality of the watershed (baseline analysis and characterization).

Step 2: Simulate flow, TSS, and nitrate under baseline conditions.

Step 3: Calibrate and validate flow, TSS concentration, and nitrate concentration simulated by the model for the baseline condition using observed flow, TSS, and nitrate data.

Step 4: Project future development scenarios using actual land use plans if available or hypothetical build-out scenarios if plans are not available for the watershed.

Step 5: Simulate flow, TSS, and nitrate for selected future build-out scenarios using the model parameters calibrated under the baseline condition.

Step 6: Develop urbanization-induced stressor indicators from simulated flow, TSS, and nitrate data for each scenario. To establish cause and effect relationships between forecasted urbanization-induced stressors and their ecological impacts, we used hydrological alteration indicators. For each scenario, we estimated 7Q10 (7-day average low-flow with a recurrence interval of 10 years), bankfull flow, baseflow index (total volume of base flow divided by the total volume of runoff for a period) (Wahl and Wahl, 1995), and indicators of hydrological alteration (IHA) from scenario-specific simulated streamflow. Indicators of hydrologic alteration (IHA) captures five flow characteristics

that are related to biological integrity: magnitude, duration, frequency, timing, and rate of change (Richter et al., 1996). Other flow-related forecasted physical habitat indicators include shear stress, depth, and flow velocity. In addition, we estimated nitrate and TSS concentrations for each scenario. Calculating indicators from forecasted streamflow, TSS, and nitrate offers to land use planners an exploratory approach where potential ecological impacts of urbanization can be predicted before urban development plans are implemented in a watershed.

Step 7: Establish linkages between urbanization-induced stressors and ecological impacts (hydrologic, hydrologic, and water quality indicators)

8. Link HSPF to AQUATOX model and assess the ecological impacts of build-out scenarios

5.2.1 Model Setup and Input Data

HSPF is a calibrated-parameter model which uses observed data for its calibration and validation. For this study, observed streamflow, TSS, and nitrate data were obtained at the Chitwood gaging station (USGS 14306030). Daily streamflow data from 1972 to 1991 were obtained from a USGS website (USGS, 2007), supplemented by data from the Oregon Water Resources Department that is available at http://www.wrd.state.or.us/OWRD/SW/streamflow_midco.shtml. Daily observed streamflow averages 7.1 m^3/sec, although it can vary from 0.06 m^3/sec during late summer low flow conditions, to 186 m^3/sec at peak flow conditions in late fall and winter (October to March).

Nitrate data were collected weekly from December 1999 through December 2001, and monthly from January 1999 to December 2002, at the USGS stream gaging station near

Chitwood. Water samples collected for dissolved nutrient measurements were filtered

(Cole Parmer 142 mm nylon filter 0.45 μm membranes) and frozen for later analysis by

the Marine Science Institute (MSI), University of California, Santa Barbara, CA using a

Lachat QwickChem 8000 Autoanalyzer for simultaneous determination of nitrite and

nitrate + nitrite. Analytical information, including blank procedures, sample replicate

results, and other quality assurance details are available at www.msi.ucsb.edu/Analab.

Because nitrite generally comprised less than 2% of the nitrogen species at these sites, we

treated nitrate + nitrite as nitrate. Daily TSS was obtained from a USGS website (USGS,

2006); it was available for the period from October 1, 1972 through September 30, 1974.

The Hydrologic Simulation Program – FORTRAN (HSPF), part of U.S.EPA's Better

Assessment Science Integrating Point and Nonpoint Sources (BASINS) (USEPA, 2001),

was selected to simulate streamflow, TSS, and nitrate for a baseline or reference scenario

and build-out scenarios. HSPF is a comprehensive, conceptual, continuous simulation

model that simulates flow and water quality constituents originating from pervious and

impervious land surfaces, in streams and well-mixed impoundments (Bicknell, et al.,

2001). HSPF uses the PERLND and the IMPLND application modules to represent

processes that occur on pervious and impervious land surfaces, respectively, and it uses

the RCHRES module to represent processes that occur in water bodies. HSPF uses

BASINS to extract soil, land cover, and geomorphological data and parameter values

from geographic information databases, such as the Digital Elevation Model (DEM),

State Soil Geographic (STATSGO) database, and National Land Cover Data (NLCD)

databases, using the BASINS geographic information systems (GIS) analysis tools

(USEPA, 2001). The study watershed was delineated into eight subwatersheds; each

subwatershed has its own land use, soil, and topography and is considered a homogeneous hydrological response unit.

The model uses meteorological data such as precipitation, air temperature, potential evapotranspiration, solar radiation, wind speed, dew point temperature, and cloud cover. Our climate data were obtained from the Hatfield Marine Science Center in Newport, Oregon (www.weather.hmsc.orst.edu), a weather station in Nashville, Oregon, and BASINS databases. Potential evapotranspiration was estimated from meteorological data using the Hamon (1961) method.

5.2.2 Model Calibration and Validation

After necessary input data were prepared, the model was calibrated using historical streamflow, TSS, and nitrate data corresponding to the baseline. The parameter values that gave the best calibration results for streamflow, TSS, and nitrate were retained. We used those parameter values to forecast streamflow, nitrate, and TSS concentrations for the three build-out scenarios. The hydrological calibration periods were 1973 through 1977 and 2000 through 2001; the nitrate calibration period was from October 2001 through September 2002; and the TSS calibration period was from October 1973 through September 1974. We used one year for TSS calibration and the other for TSS validation. Because both TSS and nitrate are affected by the hydrological calibration performance, we performed two hydrologic calibrations. One coincided with the period when observed TSS data were available and the other coincided with the period when observed nitrate data were available. After successful hydrological, TSS, and nitrate calibrations, the model was validated using observed streamflow data from periods that were not used for

model calibration. We employed HSPF calibrations using observed and simulated daily data, but we also did some monthly and annual calibration comparisons for the baseline and the build-out scenarios.

5.2.3 Model Performance Evaluation Criteria

There are no widely accepted quantitative HSPF model calibration criteria to determine if the model predictions are acceptable; there are, however, some generally accepted HSPF calibration and validation guidelines. For most HSPF streamflow calibrations, acceptable model calibration performance is achieved when the correlation coefficient between monthly simulated and observed streamflow is greater than 0.85 (Donigian, 2002). Because sediment and nitrate calibrations are affected by the hydrological calibration performance, the TSS and nitrate calibration criteria are far less stringent than the hydrologic calibration criteria. Donigian (2002) states that HSPF annual and monthly hydrology, sediment, and nutrient simulations are considered "good" when the percentage differences between simulated and observed data are between 10 and 15 for flow, between 20 and 30 for sediments, and between 15 and 25 for nutrients.

HSPF calibration and validation procedures consist of matching simulated streamflow, nitrate, and TSS concentrations with observed data. We used goodness-of-fit measures such as coefficient of determination (R^2), root mean square error (RMSE), and coefficient of model-fit efficiency (E) (Nash and Sutcliffe 1970) to evaluate the model's performance in predicting observed streamflow, TSS, and nitrate. The coefficient of determination is written as:

$$R^2 = \left\{ \frac{\sum_{i=1}^{N}(Q_{obs} - \overline{Q}_{obs})(Q_{sim} - \overline{Q}_{sim})}{\sum_{i=1}^{N}[(Q_{obs} - \overline{Q}_{obs})^2]^{0.5}[\sum_{i=1}^{N}(Q_{sim} - \overline{Q}_{sim})^2]^{0.5}} \right\}^2 \qquad (1)$$

where Q_{obs} is the observed streamflow, Q_{sim} is the corresponding simulated streamflow,

and $\overline{Q}_{obs}$ and $\overline{Q}_{sim}$ are average observed and average simulated streamflow, respectively,

and N is the number of data points used in the average calculation. The coefficient of

model fit efficiency can be written as:

$$E = 1.0 - \frac{\sum_{i=1}^{N}(Q_{obs} - Q_{sim})^2}{\sum_{i=1}^{N}(Q_{sim} - \overline{Q}_{obs})^2} \qquad (2)$$

where E is the Nash-Sutcliffe coefficient and other terms are as defined in (1). The root

mean square error (RMSE) is written as:

$$RMSE = [\sum_{i=1}^{N}(Q_{obs} - Q_{sim})^2]^{0.5} \qquad (3)$$

where Q_{obs} and Q_{sim} are defined as in (1) and N is the number of data points used for the

comparison period.

5.2.4 Future Build-out Scenario Development

The sequential nature of urban development projects introduces uncertainties in land

use projections, making it difficult to project changing land use across an entire

watershed. Land use change models that use economic and social drivers can project land

use change. These include the California Urban Futures Model (Landis, 1995) and the

Land Use Evolution and Impact Assessment Model (LEAM) (Deal, 2001). The LEAM

model forecasts specific land use change projections using watershed-specific drivers, such as economic, population, social, geography, transport, and open space.

Our approach can use information from a land use change model, but since land use change information is not usually available at the watershed scale, we present a simple, generic method that uses hypothetical build-out scenarios. In areas where information on future land use change is available from existing zoning and master plans, watershed managers can use actual land use plans instead of hypothetical scenarios. This allows resource managers to set an allowable TIA level that corresponds to a watershed's unknown future build-out. It assumes that a watershed will be developed eventually and will have a threshold TIA at the end (build-out condition). Build-out is defined as an estimate of the amount and location of potential development for an area. It is used by land use planners who evaluate the potential impacts of urban development using build-out analysis. Generally, build-out analysis, as currently used by land use planners, does not employ watershed-scale models. The proposed method allows land use planners to set different impervious area levels for build-out scenarios, and simulate how different scenarios alter hydrology and water quality. Although there is no limit on the number of build-out scenarios selected and impacts simulated, in this study we limited our analysis to three build-out scenarios that correspond with 15, 45, and 85% total impervious levels.

Table 1 lists the area covered by each land use category and the percentages of TIA used for the baseline (i.e., pre-development) and hypothetical future build-out scenarios. The three build-out scenarios correspond to low density residential (scenario15), high density residential (scenario45), and commercial development (scenario85). Selection of 15, 45, and 85 percent TIA levels is arbitrary and these levels primarily demonstrate the

model's sensitivity to different levels of imperviousness. Scenario85 (in particular) was derived only for comparative purposes since only a few urban watersheds ever reach such high TIA levels. Impervious area may not reach 85% because some areas of the watershed are unbuildable due to soil and slope characteristics or are zoned as committed open spaces. The Ballona Watershed in Los Angles, CA is one of the few watersheds with impervious cover that reaches 85%. Land use planners should select percent TIA levels that realistically reflect their future land use plans so they can set threshold TIA limits for a watershed.

For each scenario, we kept a minimum of13 percent of the urban land area as committed open spaces to include land occupied by detention basins, parks and other recreational areas, wetlands, and riparian buffer zones. When developing scenarios with this approach, watersheds can be subdivided, and different TIA levels can be assigned to different subwatersheds. Questions to answer are what will be the total allowable impervious levels at build-out for a watershed and what stressor levels are associated with different build-out scenarios.

5.2.5 Forecasting Flow, Nitrate, and TSS Alterations

Modeling streamflow, nitrate, and TSS alterations under the three build-out scenarios makes certain assumptions. For example, calibrated model parameter values obtained from the baseline condition for each land use category were unchanged when modeling the altered scenarios. In addition, to forecast streamflow, habitat, and water quality conditions for the future scenarios, we converted existing pervious forest land to impervious urban land -- but retained the historical precipitation inputs for all simulations since it is not possible to forecast future precipitation. The baseline and build-out scenario

streamflow, TSS, and nitrate were all simulated for a 30-year (1973 to 2002) period. The 30-year streamflow, TSS, and nitrate projections for the three build-out scenarios reflect the hydrologic and water quality degradations that would have been observed in the watershed through 2002 if the watershed had reached 15, 45, or 85 percent urban TIA at build-out in 1973.

As already noted, the 30-year simulations assumed that future precipitation and climate would be similar to those observed over the previous 30 years. In addition, the HSPF model treated all impervious areas as connected even though, in reality, not all of them are. This latter assumption can lead to runoff-water and contaminant input over-prediction because it ignores infiltration that occurs when runoff from an impervious area passes over an adjacent pervious area.

5.2.6. Linking Stressors and to Impacts

To project the ecological impacts of urbanization, indicators of urbanization-induced stressors were derived from forecasted flow data. For each scenario, we derived indicators from long-term simulated flow, TSS, and nutrient data. For example, indicators of flow alterations include the indicators of hydrological alterations (IHA) (Richter et al., 1996), baseflow index, bankfull flow (Q_2), and 7Q10. We estimated the baseflow index using a recursive digital filter (Eckhardt, 2005); specifically, we examined indicators that are closely related to peak flow and baseflow changes under the three build-out scenarios. These indicators can be used to establish cause and effect relationships between urbanization-induced stressors and changes in abundance, diversity, and fitness of aquatic communities. To assess the ecological impacts of simulated urbanization-induced

stressors, one can link HSPF to AQUATOX or to the Physical Habitat Simulation model (PHABSIM), a major component of the Instream Flow Incremental Methodology (IFIM) (Milhous et. al., 1984). Linking HSPF to aquatic ecosystem models and simulating ecological impacts of urbanization-induced stressors with AQUATOX AND PHABSIM is beyond the scope of this study.

5.3 Results and Discussion

5.3.1 Model Calibration and Validation Results

Hydrologic Calibration and Validation. Statistical and graphical visualization

calibration tools available in BASINS were used to calibrate and validate HSPF for the

baseline scenario. Table 2 shows goodness-of-fit measures for the calibrations. For the

two hydrologic calibration periods, the results were within acceptable calibration range:

daily calibrations had coefficients of determination (R^2) of 0.8 and 0.87, and monthly

calibrations had R^2 values of 0.91 and 0.95. Validation performance was slightly lower

than that for the calibration. For the two hydrologic validation periods, the R^2 values were

0.72 and 0.82 for the daily validations, and 0.83 and 0.89 for the monthly calibrations.

The RMSE and Nash-Sutcliffe Efficiencies (E) values are also provided in Table 2. As

shown in Figures 3a and 3b, simulated streamflow compared well to observed daily

streamflow for both hydrologic calibration periods. All the parameters adjusted as part of

the hydrologic, TSS, and nitrogen calibrations are listed in Table 3.

Table 2. Model calibration and validation results

	R^2	RMSE	E	R^2	RMSE	E
Hydrologic Calibration and Validation						
	Calibration Period(1973 to 1977)			Validation Period (1977 to 1979)		
Daily	0.80	5.8	0.77	0.72	6.4	0.71
Monthly	0.95	2.2	0.94	0.89	3.0	0.87
	Calibration Period (2000 to 2001)			Validation Period (1999 to 2000)		
Daily	0.87	3.6	0.82	0.69	7.8	0.62
Monthly	0.91	4.1	0.81	0.83	3.5	0.79
TSS Calibration and Validation						
	Calibration Period (1973 to 1974)			Validation Period (1972 to 1973)		
Daily	0.52	39.36	0.46	0.56	19.1	0.44
Monthly	0.89	11.78	0.82	0.78	7.7	0.59
Nitrate Calibration and Validation						
	Calibration Period (2000 to 2001)			Validation Period (1999 to 2000)		
Daily	0.75	----	----	0.39	----	---

Table 3. List of parameter values adjusted during calibration

Parameter	Description	Calibrated value
Hydrology		
LZSN	Lower Zone Nominal Storage (cm)	20.32
UZSN	Upper Zone nominal Storage (cm)	2.36
INFILT	Infiltration parameter (cm/hr)	0.17
INTFW	Interflow inflow parameter	6.50
IRC	Interflow Recession Parameter (per day)	0.70
AGWRC	Daily recession constant of groundwater flow (per day)	0.98
DEEPFR	Fraction of groundwater inflow (inactive)	0.00
BASETP	Fraction of E-T from baseflow	0.00
AGWETP	Fraction of E-T from active groundwater	0.00
LZETP	Lower zone E-T parameter	0.45
CEPSC	Interception storage capacity (cm)	0.51
Sediment Simulation		
KRER	Coefficient-detachment equation	1.14
JRER	Exponent-detachment equation	1.00
KSER	Coefficient-detached washoff equation	3.00
JSER	Exponent-detached washoff equation	5.80
KGER	Coefficient-matrix scour equation	5.80
JGER	Exponent-matrix scour equation	5.20
KSAND	Coefficient-sandload power function	0.01
EXPSND	Exponent-sandload power function	3.90
W (cm∕s)	Fall velocity in still water	0.14
M (kg∕m2 day)	Erodibility coefficient	1.05
TAUCD (kg∕m2)	Critical bed shear for deposition	2.44
TAUCS (kg∕m2)	Critical bed shear for scour	6.35
Nitrogen Simulation		
KTAM20 (per hour)	Nitrification rates of ammonia	0.015
KTO220 (per hour)	Nitrification rates of ammonia and nitrite	0.002
TCNIT (per hour)	Temperature correction coefficient for nitrification	1.50
KNO320 (per hour)	Nitrate denitrification rate at 20 degrees C	0.02
TCDEN (none)	Temperature correction coefficients for denitrification	1.00
DENOXT (mg/l)	Dissolved oxygen concentration threshold for denitrification	0.50

Sediment Calibration and Validation. Suspended sediment concentration calibration and validation performances generally are impacted negatively by the lack of available long-term historical data with appropriate temporal resolution. In this study, only two years of daily TSS concentration data were available for model calibration and validation. As shown in Table 2, the coefficients of determination (R^2) for the sediment calibration were 0.52 for the daily and 0.89 for the monthly simulations. Sediment validations had coefficients of determination (R^2) of 0.56 for the daily and 0.78 for the monthly simulations. Graphical comparisons of the observed versus simulated TSS concentrations indicated that the model slightly under-predicted the higher TSS values, but overall sediment simulations generally followed the runoff hydrograph trend (Figure 8c). Despite under-predicting the higher TSS values, overall model calibration and validation performances for TSS concentrations were acceptable.

Nitrate Calibration and Validation. Nitrate concentration simulation performance depends on the quality and the quantity of observed data. Observed concentration data had many missing data points, which created model calibration problems. To address this problem, we compared observed and simulated nitrate only on days when observed nitrate concentrations data were available. Despite using only limited observed versus simulated comparisons, our nitrate concentration calibrations had coefficients of determination (R^2) of 0.75 for the daily simulations (Table 2). The nitrate validations had lower coefficients of determination of 0.39 for daily simulations (Table 2). Lack of continuous observed daily nitrate data at daily (or shorter) time intervals may have contributed to the lower model validation performance. Both observed and simulated

nitrate concentrations closely followed the seasonal hydrologic trend, except during the

first winter storm, suggesting that nitrate export from terrestrial areas is strongly

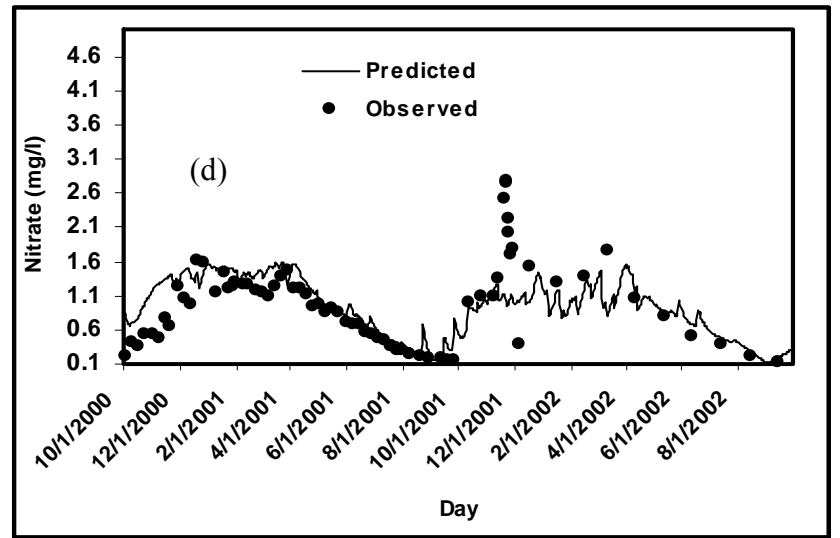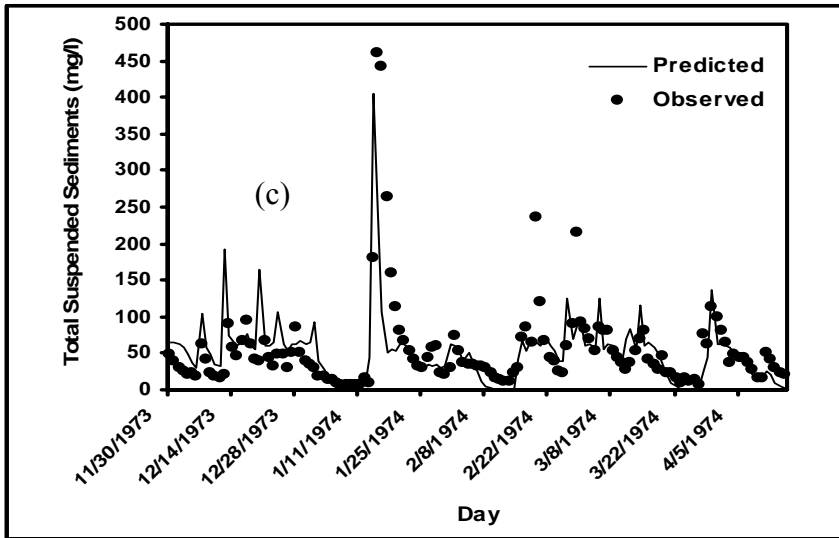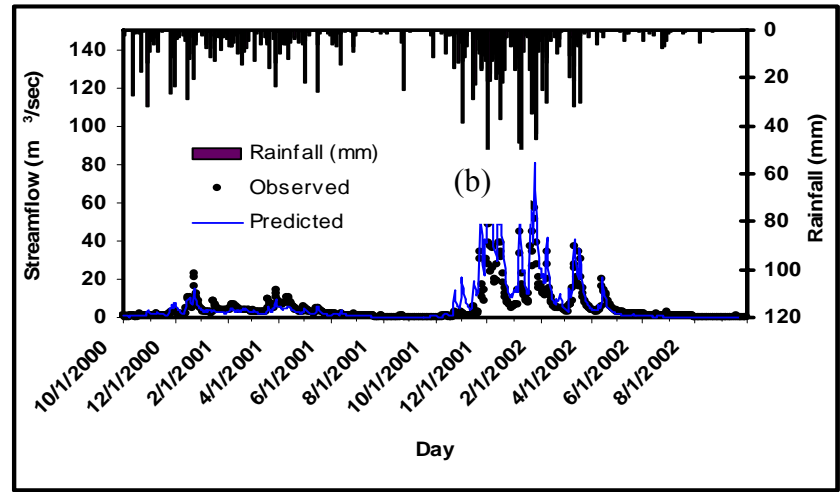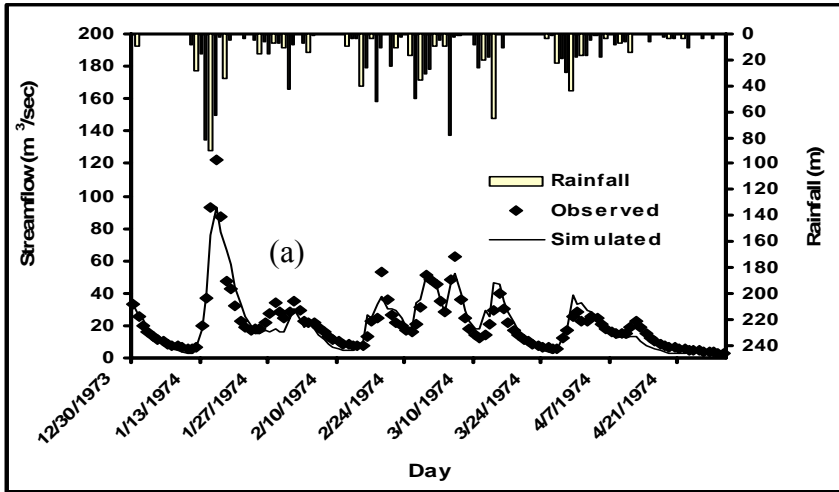influenced by nitrate build-up and storm wash-out mechanisms (Figure 8d).

Figure 8. Model calibration results: streamflow (a and b), TSS (c), and nitrate (d) calibrations.

5.3.2 Forecasting Stressor Levels for the Build-out Scenarios

Forecasting Flow Alterations. Flow alterations for build-out future scenarios show significant changes as a function of rising percent of TIA levels (Figure 9). To document these scenario-specific changes, we compared: daily streamflows simulated for the baseline scenario to streamflows forecasted for scenario15, scenario45, and scenario85 (Figure 9a); simulated annual water balances for all scenarios (Figure 9b); simulated flow duration curves at low flow conditions, i.e., for flows with greater than 95% exceedances (Figure 9c); and simulated flow duration curves at high flow conditions, i.e., for flows with less than 5% exceedances (Figure 9d). Comparisons of the flow duration curves show that low flows get smaller with increased percent TIA (Figure 9c) and high flows get larger (Figure 9d). Comparisons of the annual average water balance components under the baseline versus the build-out future scenarios for a 30-year simulation period, using the same historical precipitation data, showed increased surface runoff and decreased interflow, baseflow, and evapotranspiration for scenario15, scenario45, and scenario85 (Figure 9b).

Peak Flow. The simulation results clearly show increased flow flashiness associated with increases in percent TIA levels (Figures 4a and 4d); specifically, low intensity storm events usually did not generate runoff for the baseline and scenario15, but did so for scenario45 and scenario85. Thus, the baseline and scenario15 were characterized by smooth runoff hydrographs, whereas scenario45 and 85 exhibited high peak flow rates (Figure 9a). The data suggest the existence of a threshold percent TIA above which peak flows greatly increase with increased percent TIA level. In addition, comparisons of flow

duration curves with exceedances less than 5% show small peak flow differences between the baseline scenario and scenario15, but relatively large peak flow differences among the three build-out future scenarios (Figure 9d).

Baseflow and Groundwater Recharge. To examine the effect of increased imperviousness on baseflow and groundwater recharge, we calculated baseflow index (BFI) from observed streamflow (historical condition), streamflow simulated for the baseline condition, and streamflow forecasted for the three future build-out scenarios. Baseflow indices from the observed and simulated data for the baseline scenario were 0.62 and 0.64, respectively, whereas indices forecasted for scenario15, scenario45, scenario85 were 0.59, 0.46, and 0.28, respectively. This decreasing BFI trend indicates how baseflow and groundwater recharge decrease when TIA levels increase.

To further examine the relationship between increased imperviousness, baseflow and groundwater recharge further, we compared flow duration curves for flows with exceedance probability of greater than 95% for the baseline scenario and for the three build-out scenarios over 30-years (Figure 9c). These comparisons reveal little difference between the baseline condition and scenario15 simulations, but show drastic baseflow reductions for the scenario45 and scenario85 simulations (Figure 9c). In this report, we also calculated 7Q10 flow values from streamflow simulated for the baseline, scenario15, scenario45, and scenario85 as 0.16, 0.15, 0.10, and 0.03 m^3/sec, respectively.
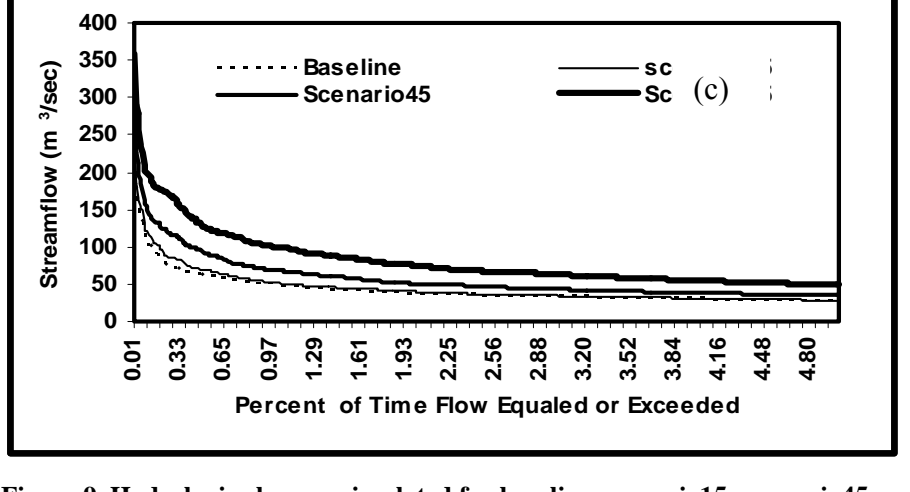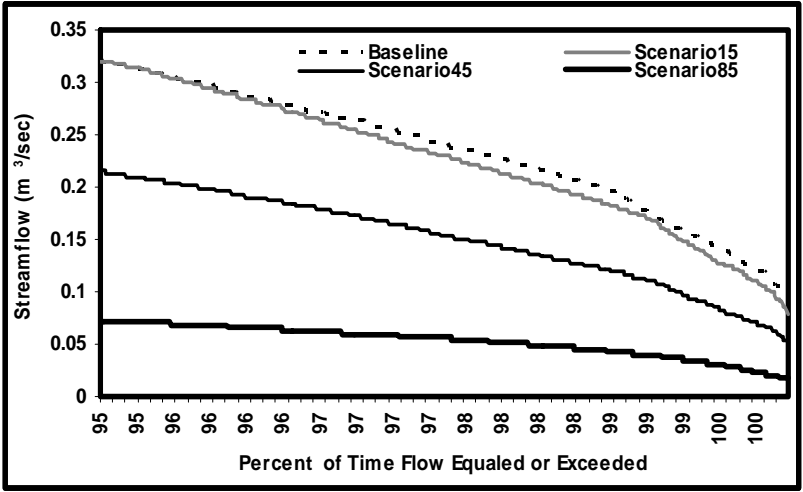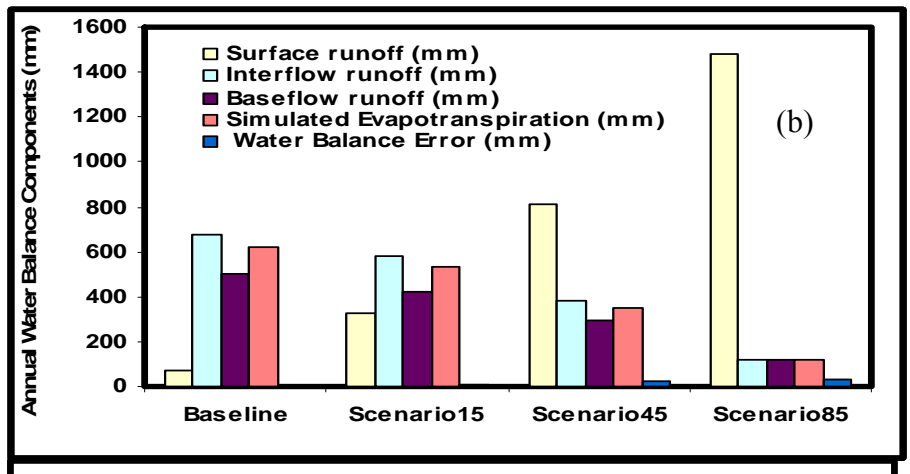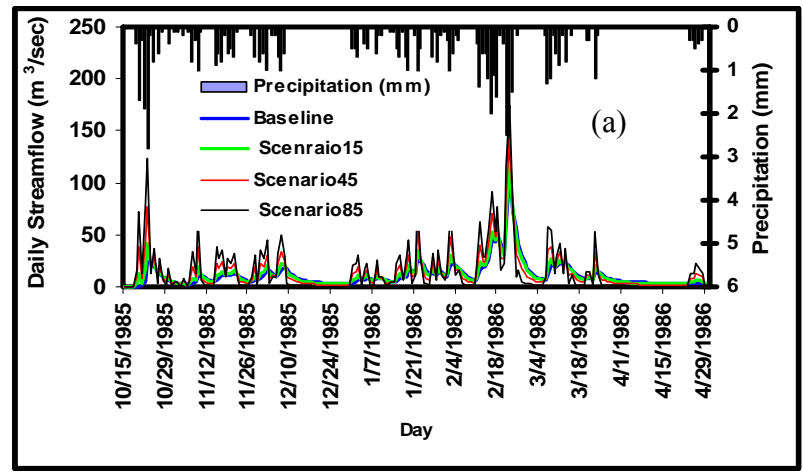
Figure 9. Hydrologic changes simulated for baseline, scenario15, scenario45, and scenario85: (a) daily streamflows, (b) annual water balances (c) flow duration curves at low flow conditions, and (d) flow duration curves at high flow conditions

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Forecasting TSS Concentrations. Suspended sediment concentrations forecasted for scenario15, scenario45, and scenario85 over a 30-year simulation period did not yield a continuously increasing function with percent TIA level. Indeed, our scenario projections suggest that sediment concentrations are controlled by available sediment supply and by transport capacity. Specifically, during some elevated flow periods, scenario45 projections had higher TSS concentrations than scenario85, even though scenario85 had higher simulated flows, thus higher sediment transport capacity (Figure 9b). One explanation is that there is a TIA threshold value beyond which sediment loads transported from land surfaces may not increase with increased impervious area. This implies that when sediment supply is available, scenario85 would have produced the highest simulated TSS concentrations because it has the greatest sediment transport capacity. However, as the landscape becomes increasingly impervious, sediment supply becomes limited and increases in sediment transport capacity do not translate directly into increased sediment export.

An analysis of forecasted daily TSS concentrations for the build-out scenarios indicates a threshold percent TIA value somewhere between 45 and 85 TIA levels where increased percent TIA does not translate to increased TSS concentrations. In this report, we forecasted 30-year average annual sediment loads of 146, 211, 452, and 241 (kg/ha-yr) for the baseline, scenario15, scenario45, and scenario85, respectively.

Forecasting Nitrate Concentrations. For the build-out scenarios, the model forecasted higher nitrate concentrations at low flows and lower concentrations at elevated flows (Figure 9a). This is because nitrate simulations are strongly influenced by flow

alterations, particularly baseflow reduction and increase in surface runoff. Conversely, at baseline condition, higher nitrate concentrations were observed during elevated flow conditions and lower concentrations at low flow conditions, suggesting that, at baseline, nitrate accumulates in the upper soil layers during dry periods and is washed out during wet periods (Figure 9). This observed nitrate build-up - wash-out cycle under the baseline scenario was severely disrupted by urbanization, which alters the dominant runoff generation mechanisms and flow pathways.

Model results simulated for the build-out scenarios show increased nitrate concentrations during dry periods and lower concentrations at elevated flow periods. Such interdependence between flow alteration and nitrate concentrations might also impact soil denitrification processes. As stated previously, our nitrate projections for the three build-out scenarios produced nitrate concentrations in the baseflow that increased with increased percent TIA level. Even ignoring other nitrogen sources, such as lawn fertilizers and atmospheric deposition, our results indicate that flow alteration alone can increase nitrate concentrations that may cause eutrophication threats to coastal waters. In summary, our forecasted 30-year average annual nitrate loads were 13, 14, 16, and 23 kg-NO3-N/ha-yr for the baseline, scenario15, scenario45, and scenario85, respectively.
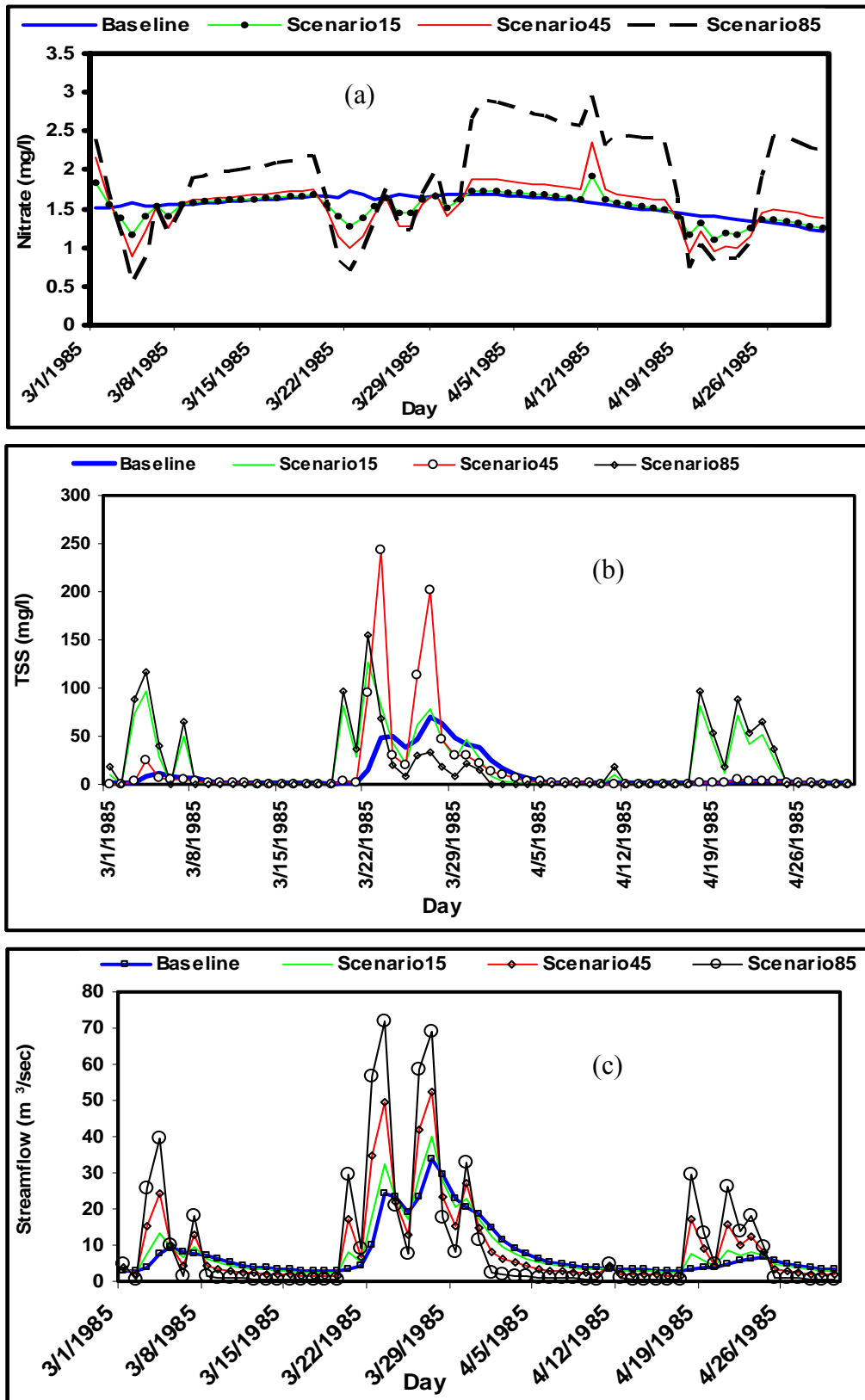
Figure 10. HSPF simulated streamflow, TSS, and nitrate concentrations for baseline, scenario15, scenario45, and scenario85

5.4 Determining Ecologically Relevant Urbanization-Induced Stressor Indicators

Hydrologic Indicators. Flow alterations have an adverse impact on biological integrity, but quantifying that impact has been a challenge. Richter et al. (1996) present indicators of hydrologic alteration (IHA) calculated from observed historical streamflow data. In this study, we estimated IHAs from simulated streamflow for the baseline and for the build-out scenarios (Table 4). Simulated magnitude of average monthly flows vary with the month of year. For some months, the monthly flows decrease with increase in TIA levels whereas for other months monthly flows decrease with increase in TIA levels. As the TIA levels increased, model results show a decrease in magnitude and duration of annual extreme flows for build-out scenarios. Conversely, model results show increased frequency and duration of average high and low flow pulses as well as increased average rate and frequency of flow changes (Table 4). Simulated IHAs show that urbanization does not alter the timing of average annual extreme flows.

Determining IHA values for build-out scenarios with different percent TIA levels, and establishing quantitative links between urbanization-induced stressors and their ecological impacts, are essential to urban ecosystem protection and restoration planning. Because the amount of water available in a stream defines the suitability of a habitat to aquatic organisms, flow alteration, especially low flows, creates unfavorable conditions for native species (Poff et al., 2006). To ensure that sufficient water is available for aquatic organisms, some states set 7Q10 flow values as an in-stream flow requirement that must not be violated when issuing water withdrawal permits for irrigation and municipal water supply.

Table 4. Indicators of hydrologic alteration (IHA) calculated from streamflow simulated for the baseline and the three build-out scenarios for a 30-year period.

IHA Parameter Group	Baseline	Scenario15	Scenario45	Scenario85
Group #1: Magnitude of average monthly flows (m³/sec)				
October	0.47	0.51	0.41	0.19
November	4.29	6.16	6.93	5.49
December	11.35	11.04	11.20	6.12
January	9.91	10.54	10.16	5.40
February	9.12	9.88	8.00	3.70
March	8.45	8.52	8.48	5.75
April	6.20	6.13	4.95	2.35
May	3.36	3.27	2.14	0.76
June	1.98	1.85	1.27	0.40
July	1.14	0.98	0.64	0.19
August	0.67	0.58	0.38	0.11
September	0.47	0.46	0.29	0.09
Group #2: Magnitude and duration of annual extreme flows (m³/sec)				
1-day minimum	0.32	0.27	0.18	0.05
3-day minimum	0.32	0.28	0.18	0.05
7-day minimum	0.33	0.31	0.20	0.06
30-day minimum	0.42	0.47	0.37	0.13
90-day minimum	0.76	0.90	0.98	0.87
1-day maximum	67.60	84.61	117.06	172.90
3-day maximum	50.60	57.54	73.07	94.16
7-day maximum	39.00	41.70	47.59	60.54
30-day maximum	21.65	22.59	24.86	29.65
90-day maximum	15.11	5.81	17.90	21.03
Days with zero flow	0.00	0.00	0.00	0.00
Base flow index	0.05	0.043	0.02	0.01
Group #3:Timing of annual extreme flows (Julian Day)				
Date of minimum	277.00	277.00	275.00	275.00
Date of maximum	347.00	338.00	338.00	338.00
Group #4: Frequency and duration of average high and low pulses				
Low pulse count	1.00	8.00	9.00	8.00
Low pulse duration	62.25	5.75	5.00	5.50
High pulse count	8.50	14.00	27.50	35.00
High pulse duration	8.50	3.00	2.00	2.00
Low Pulse Threshold	30.65	32.30	22.20	7.50
High Pulse Threshold	288.00	307.00	327.00	310.00
Group #5: Rate and frequency of flow changes				
Rise rate	30.75	98.75	239.10	445.00
Fall rate	-2.53	-6.45	-6.42	-5.35
Number of reversals	64.00	111.50	119.00	121.00

Hydraulic and water quality indicators. High peak flow rates associated with high percent TIA levels cause high flow velocities and associated shear stresses that move sediments (Poff et al., 2006), and displace benthic invertebrates (Poff and Ward, 1991), and small fish (Harvey, 1987). High shear stresses also increase the depth of scour of bed sediments to induce higher mortality of benthic invertebrates (Palmer et al., 1992; Townsend et al., 1997). To forecast the risks of increased stream channel erosion, we used Q_2, often referred to as the bankfull flow or channel forming flow, which corresponds to a 2-year return period. Wolman and Miller (1960) reported that flows with 1 to 5 year return periods are important to stream geomorphology because these flows move sufficient amounts of sediment. To forecast the impacts of urbanization on streambank erosion, we examine how different build-out scenarios alter the frequency of bankfull or channel forming flows (Q_2). Using Q_2 as an indicator of increased channel erosion, we forecasted the number of times the baseline scenario Q_2 flow value was exceeded by Q_2 flows forecasted for the three build-out scenarios. We found 22, 91, and 274 days with flows greater than the baseline scenario Q_2 for the 30-year projections for scenario15, scenario45, and scenario85, respectively.

Establishing stressor-response relationships between changes in shear stress and Q_2 values under future build-out scenarios and their impacts on aquatic organisms is key to developing biologically relevant indicators. Although some stressor indicators may have biological relevance, additional field research is needed to establish cause-and-effect relationships between stressor indicators and ecological responses.

The HSPF model simulates many physical habitat variables closely related to channel hydraulic geometry. To establish links between forecasted habitat variables and the health

of aquatic ecosystems, we propose the use of habitat models such as the Physical Habitat Simulation model (PHABSIM), a major component of the Instream Flow Incremental Methodology (IFIM) (Milhous et. al., 1984), and ecological models such as AQUATOX. We suggest importing HSPF forecasted habitat variables and stressors, such as scenario-specific simulated hydraulic, temperature, nutrient, and contaminant data, as inputs into the PHABSIM and AQUATOX models to compute the impact these stressors have on the health of aquatic ecosystems.

Comparisons of simulated flow velocity for the baseline scenario to velocities forecasted for the three build-out scenarios indicate significant baseflow velocity reductions with increasing percent TIA level (Figure 10). For instance, scenario85, which has the highest percent TIA, had the lowest baseflow velocity, followed by scenario45 and scenario15. The baseline scenario, with the lowest percent TIA, had the highest baseflow velocity. As presented earlier, the flow velocities fluctuated between baseflow and peak flows within each scenario. However, these fluctuations increased with increase in percent TIA; such that peak flows and baseflows had an inverse order relationship to percent TIA; that is, scenario85 had the highest peak flows followed in order by scenario45, scenario15, and baseline.
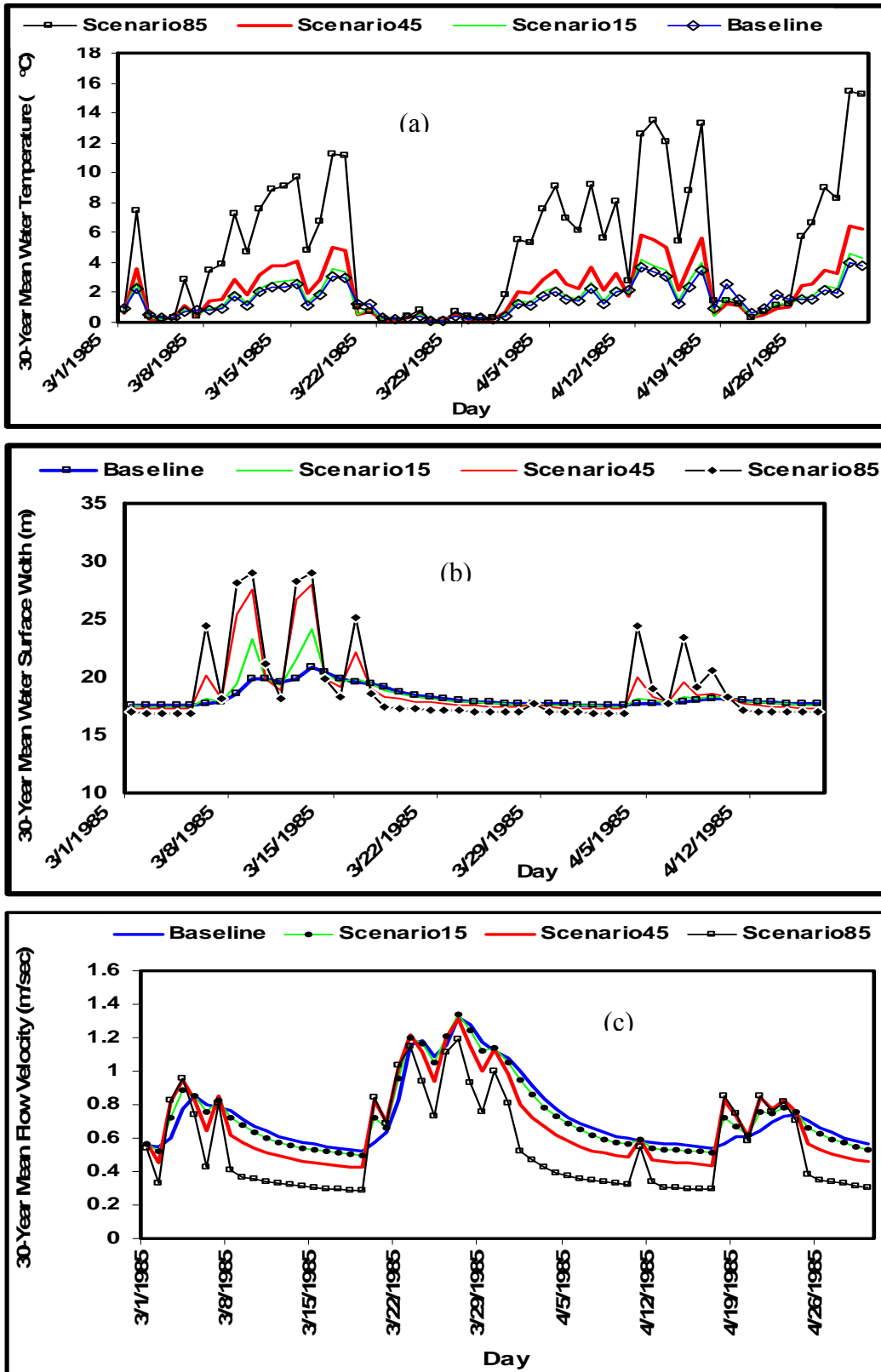
Figure 11. HSPF simulated 30-year mean water temperature (a), water surface width (b), and flow velocity (c) for baseline, scenario15, scenario45, and scenario85.

6 CONCLUSION

Controlling nonpoint source pollution requires the use of integrative modeling approaches that simulate the interactions among integrated stressors. However, current models and modeling approaches cannot represent integrated stressors and many available models do not match the complexities of the system that is modeled. Only a very few watershed models can represent the various hydromodification types and scenarios and can simulate scenario-specific flow alterations and water quality degradation. In this report, we present a modeling approach that is suitable for assessing how hydromodification projects alter natural flow regimes and degrade water quality. The proposed BASINS-based modeling framework enables resource managers to select alternative future scenarios, simulate scenario-specific stressors, and select scenarios that minimize the impacts of hydromodification on water quantity and quality.

Urbanization has been closely linked to the degradation of aquatic ecosystems. Methods to quantify urbanization-induced stressors from future development scenarios have not been generally available. In this study, we present an application of a method to forecast urbanization-induced stressors and discuss ways to develop indicators that link stressors to their ecological impacts. The proposed approach will enable land use planners and resource managers to forecast urbanization-induced stressors from forecasted streamflow and water quality data. Forecasted urbanization-induced stressors include flow alteration (e.g., altered peak flows and baseflows), water quality degradation (e.g., increased nitrate and TSS concentrations), and geomorphic or habitat alterations (e.g., changes in flow depth, wetted surface width, and flow velocity).

The ultimate utility of the proposed forecasting approach depends on successfully developing ecological response indicators that capture the responses of aquatic ecosystems to the stressors of concern. Ecological response indicators can be derived possibly from data on changes in native species richness or on the composition and functional organization of fish and invertebrates.

Future research should focus on developing ecological response indicators that quantitatively reflect aquatic biota responses to forecasted stressors. Specifically, such research should assess the effects of single or integrated stressors on the abundance, diversity, and fitness of aquatic communities. Future modeling in this area ideally will also simulate how different best management practices moderate the ecological effects of urbanization to restore important ecological services and functions.

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