2013

SYSTEM DYNAMICS MODELING AS A QUANTITATIVE-QUALITATIVE FRAMEWORK FOR SUSTAINABLE WATER RESOURCES MANAGEMENT: INSIGHTS FOR WATER QUALITY POLICY IN THE GREAT LAKES REGION

Ali Mirchi
Michigan Technological University

Follow this and additional works at: https://digitalcommons.mtu.edu/etds
Part of the Water Resource Management Commons

SYSTEM DYNAMICS MODELING AS A QUANTITATIVE-QUALITATIVE FRAMEWORK FOR SUSTAINABLE WATER RESOURCES MANAGEMENT: INSIGHTS FOR WATER QUALITY POLICY IN THE GREAT LAKES REGION

By

Ali Mirchi

A DISSERTATION

Submitted in partial fulfillment of the requirements for the degree of

DOCTOR OF PHILOSOPHY

In Civil Engineering

MICHIGAN TECHNOLOGICAL UNIVERSITY

2013

This dissertation has been approved in partial fulfillment of the requirements for the Degree of DOCTOR OF PHILOSOPHY in Civil Engineering.

Department of Civil and Environmental Engineering

Dissertation Advisor: *Dr. David W. Watkins*

Committee Member: *Dr. Alex S. Mayer*

Committee Member: *Dr. Daya Muralidharan*

Committee Member: *Dr. Kaveh Madani*

Department Chair: *Dr. David Hand*

Table of Contents

List of Figures

List of Tables

Preface

This dissertation is a compilation of previously published material, including a book chapter and journal articles, in print or in preparation. Chapters 1-5 are co-authored, and the author's contribution to each of these chapters is described below. The order of co-authors reflects their contribution in the form of providing research direction, technical comments, as well as review and editing of the paper.

Chapter 1 is published in Mirchi, A., Watkins, D.W. Jr., Madani, K. (2010). Modeling for watershed planning, Management and decision making. In: Vaughn JC (Ed.), *Watersheds: management, restoration and environmental impact.* Nova Science Publishers, Hauppauge, New York. This chapter was primarily authored by the dissertation author.

Chapter 2 is published in Mirchi, A., Madani, K., Watkins, D.W. Jr., Ahmad, S., (2012). Synthesis of system dynamics tools for holistic conceptualization of water resources problems. *Water Resources Management* 26(9), 2421-2442. The dissertation author conducted the literature review and was the primary author of the article.

Chapter 3 is published in Mirchi, A., Watkins, D.W. Jr., 2012. A systems approach to holistic TMDL policy: Case of Lake Allegan, Michigan. *Journal of Water Resources Planning and Management*, doi: 10.1061/(ASCE)WR.1943-5452.0000292. The dissertation author was the primary author of the article. Coauthor, David W. Watkins provided research direction, reviews and edited the paper.

Chapter 4 is being considered for publication as Mirchi, A., Watkins, D.W. Jr. A high-level simulation-optimization framework for non-point source phosphorus load reduction in the Kalamazoo River watershed. *Science of the Total Environment*. This chapter was primarily authored by the dissertation author, while co-author David W. Watkins provided research direction and assistance with optimization analysis. Furthermore, the co-author reviewed and edited the paper.

Chapter 5 is being considered for publication as Mirchi, A., Watkins, D.W., Meredith Ballard-LaBeau, Daya Muralidharan, and Alex Mayer. Market-based policy instruments for mitigating agricultural phosphorus loads in the Maumee Basin. *Environmental Management*. This chapter was primarily authored by the dissertation author, while co-authors David W. Watkins, Daya Muralidharan, and Alex S. Mayer provided research direction, reviews and edited the paper. Co-author Meredith Ballard-LaBeau provided nutrient load estimates generated by the Upper Midwest SPARROW model.

Acknowledgements

Many wonderful individuals and different funding sources were instrumental in successful completion of my doctoral dissertation. I gratefully acknowledge the support I received from them during the four memorable years of studying at Michigan Tech in the beautiful Upper Peninsula of Michigan. I hope that I will be a good ambassador for the values that Michigan Tech stands for.

My genuine gratitude goes to my PhD advisory committee members who provided me with the encouragement, guidance, and mentorship that it takes to nurture a young researcher. Special thanks to my advisor, Dr. David Watkins, who encouraged me to come to Michigan Tech, and gave me the opportunity to develop my research skills in the field of water resources, as well as in the general area of sustainability. Dave was always very patient with me as I spoke enthusiastically about research ideas that required much time and effort to mature, sending me relevant journal articles, books, and other resources to facilitate my learning. Furthermore, he constantly encouraged me to attend international academic events and introduced me to prominent researchers in the field, which played an important role in my professional development. Dr. Alex Mayer facilitated my education through financial and academic support, allowing me to attend weekly research group meetings that expedited my integration in Michigan Tech academic environment. I learned a lot simply by observing Alex's multi-disciplinary approach to water resources problems, his collegial leadership style when interacting with a diverse group of experts, and his valuable comments. Dr. Daya Muralidharan was very generous with her time, helping me improve my understanding of natural resources economics by allowing me to sit in her classes, and by pointing me to the right direction for finding economic data and literature. Dr. Kaveh Madani was an important figure in shaping my interest in systems thinking and applying the systems approach to water resources problems. Although Kaveh was not on the Michigan Tech campus, he was always an email away with solid comments and discussions streaming in through rapid email communications, as well as online meetings.

I gratefully acknowledge funding sources that supported my education and research. Department of Civil and Environmental Engineering offered me teaching assistantship positions that helped me improve my teaching and interaction with students. A significant proportion of my research was done while I was a research assistant supported through the US National Science Foundation (NSF) under Grant No. 0725636. I appreciate the Graduate School's Finishing Fellowship, offered to me in the summer 2013. I acknowledge Michigan Tech's Center for Water and Society (CWS), and Graduate Student Government (GSG) for providing travel grants.

I would also like to thank many people who provided comments and suggestions on my research: Drs. Marty Auer, James Mihelcic, Qiong Zhang, Julie Zimmerman, Fuzhan Nasiri, and Valerie Fuchs. I acknowledge the help I received at different stages of the work from Drs. Meredith Labeau, Rabi Gyawali, Sinan Abood, Azad Henareh, Agustin Robles-Morua, and Foad Yousef, as well as Katelyn Watson, Ehsan Taheri, and Meysam Razmara. Furthermore, I appreciate the people whose support and friendship was an asset, facilitating acclimation and integration of a young Iranian couple in the Keweenaw: Nancy and Dianne Sprague, Megan and Thomas Werner, and my dear friends at NOSOTROS, GSG, Canterbury House, Global City, and Iranian Community at Michigan

Tech. I will not be able to name everyone in this short acknowledgement section. However, I want to emphasize that I am indebted to all my professors, colleagues, and friends at Michigan Tech, who positively affected my life one way or another.

I owe deep gratitude to my sacrificing parents, Mina Nikfarjam and Nasrollah Mirchi, to my supportive brothers Mohammad and Saeed Mirchi, and to my dear friends back home and around the globe whose love, support, and encouragement never fell short. I am honored to have such great parents, brothers, and friends whom I have missed a lot during the years of staying in the US.

Last, but in no way least, I would like to thank my lovely wife, Sara Alian, who was an interminable source of emotional and moral support during ups and downs of my journey as a graduate student. I can never thank you enough for accompanying me in this journey, and creating the most dependable of support systems that I could ever have asked for. I admire your unconditional love, unyielding devotion, and quiet patience while I was consumed in work. I am thrilled to see how our love has grown and solidified over these years. I love you, and wholeheartedly thank you for standing by me.

Abstract

Early water resources modeling efforts were aimed mostly at representing hydrologic processes, but the need for interdisciplinary studies has led to increasing complexity and integration of environmental, social, and economic functions. The gradual shift from merely employing engineering-based simulation models to applying more holistic frameworks is an indicator of promising changes in the traditional paradigm for the application of water resources models, supporting more sustainable management decisions. This dissertation contributes to application of a quantitative-qualitative framework for sustainable water resources management using system dynamics simulation, as well as environmental systems analysis techniques to provide insights for water quality management in the Great Lakes basin.

The traditional linear thinking paradigm lacks the mental and organizational framework for sustainable development trajectories, and may lead to quick-fix solutions that fail to address key drivers of water resources problems. To facilitate holistic analysis of water resources systems, systems thinking seeks to understand interactions among the subsystems. System dynamics provides a suitable framework for operationalizing systems thinking and its application to water resources problems by offering useful qualitative tools such as causal loop diagrams (CLD), stock-and-flow diagrams (SFD), and system archetypes. The approach provides a high-level quantitative-qualitative modeling framework for "big-picture" understanding of water resources systems, stakeholder participation, policy analysis, and strategic decision making. While quantitative modeling using extensive computer simulations and optimization is still very

important and needed for policy screening, qualitative system dynamics models can improve understanding of general trends and the root causes of problems, and thus promote sustainable water resources decision making.

Within the system dynamics framework, a growth and underinvestment (G&U) system archetype governing Lake Allegan's eutrophication problem was hypothesized to explain the system's problematic behavior and identify policy leverage points for mitigation. A system dynamics simulation model was developed to characterize the lake's recovery from its hypereutrophic state and assess a number of proposed total maximum daily load (TMDL) reduction policies, including phosphorus load reductions from point sources (PS) and non-point sources (NPS). It was shown that, for a TMDL plan to be effective, it should be considered a component of a continuous sustainability process, which considers the functionality of dynamic feedback relationships between socio-economic growth, land use change, and environmental conditions.

Furthermore, a high-level simulation-optimization framework was developed to guide watershed scale BMP implementation in the Kalamazoo watershed. Agricultural BMPs should be given priority in the watershed in order to facilitate cost-efficient attainment of the Lake Allegan's TP concentration target. However, without adequate support policies, agricultural BMP implementation may adversely affect the agricultural producers. Results from a case study of the Maumee River basin show that coordinated BMP implementation across upstream and downstream watersheds can significantly improve cost efficiency of TP load abatement.

Chapter 1- Background and objectives[1]

1.1. Introduction

Water resource systems are modeled to facilitate well-studied designs and informed management decisions. In engineering and management practices, it is important to understand complex interactions occurring today as well as predict impacts years, perhaps even decades, into the future. In recent years, watershed management practices that were once praised for their broad benefits to society have become the focus of harsh criticisms for their adverse and unexpected environmental or socioeconomic impacts. River channelization (Shen et. al, 1994; Langler and Smith, 2001), dam construction (Tullos, 2009), irrigation development (Dokhuvny and Stulina, 2001; Cai et. al., 2003; Schlüter et. al., 2006; Yoshinobu et. al., 2006), inter-basin water transfer (Madani and Marino, 2009), and hydraulic mining of rivers (Wright and Schoellhamer, 2004) are some examples of numerous cases of deteriorating environmental conditions caused by lack of understanding of dynamic interactions of various watershed subsystems.

The watershed has been widely acknowledged to be the appropriate unit of analysis for many water resources planning and management problems (e.g., McKinney et. al., 1999). However, many of the environmental processes and socioeconomic activities

[1] The content of this chapter is based on the book chapter: Mirchi, A., Watkins, D.W. Jr., Madani, K., (2010). Modeling for watershed planning, management and decision making. In: Vaughn, J.C. (Ed.) Watersheds: Management, restoration and environmental impact. Nova Science Publishers, Hauppauge, New York. Reprinted with permission from Nova Science Publishers, Inc.

occurring within a watershed system are simply too complex, dynamic, and spatially variable to be precisely monitored and thoroughly understood. As population grows, continued human encroachment into natural systems seems inevitable, with expanding communities needing increased water supplies to carry on various development activities in the watershed. Paradoxically, both water shortage (drought) and overabundance (flooding) will become even more problematic for many communities, yet expectations will remain high for using water as a means of socioeconomic development and ecosystem conservation and enhancement. It is unlikely that these expectations can be met without the aid of analytical tools such as computer watershed models.

Models help us predict future impacts of projects and management policies, which in turn contributes to improved water resources system design, planning, and operation, and thus more sustainable water resources management. They provide mathematical representations of watershed processes and affected socioeconomic and environmental systems. Models have become a fundamental and integrated element of any engineering project or management practice that is deemed to alter diverse natural processes. Models help us gain insights into hydrological, ecological, biological, environmental, hydrogeochemical, and socioeconomic aspects of watersheds (Singh and Woolhiser, 2002), and thus contribute to systematized understanding of how watershed subsystems function (Lund and Palmer, 1997), which is essential to integrated water resources management and decision making (Madani and Marino, 2009).

Water resources modeling for planning, management and decision making requires a holistic approach. Development and management of water resources systems almost always involves a host of different objectives advocated by a multitude of stakeholder groups, which often have conflicting interests. Failing to recognize the need for holistic planning and management of water resources may lead to unsustainability in the socioeconomic or environmental systems. A chronological synthesis of watershed modeling provides an overview of how modeling goals have evolved from describing only physical processes to the integration of social, economic, and environmental objectives in support of decision making. Identifying appropriate frameworks, which can facilitate the transition of water resources management towards holism, remains an area of research among water resources scholars.

1.2. Chronological synthesis of watershed modeling

For decades, water resources professionals have been developing and applying models to address watershed problems, yet watershed models are still evolving in terms of approach, application, and ability to provide users with a comprehensive and reliable understanding of problems. Watershed modeling efforts before 1960 were aimed mostly at quantitative representation of individual hydrologic processes (see reviews by Singh and Woolhiser, 2002; Chen, 2004; Crawford and Burges, 2004). Various components of the hydrologic cycle, such as surface runoff, infiltration, groundwater flow, and evapotranspiration, were modeled separately (Singh and Woolhiser, 2002), but a lack of data and computing capability hindered more integrated analysis (Freeze and Harlan, 1969; Chen, 2004).

Watershed modeling was revolutionized after the advent of computers in the 1960s. Development of the Stanford Watershed Model in 1966 (Crawford and Linsley, 1966) initiated a prolific era of modeling efforts that incorporated snowmelt runoff, streamaquifer interaction, reservoir and channel flow routing, and water quality into watershed models such as Hydrologic Simulation Program FORTRAN (Johanson, et al., 1984; Singh and Woolhiser, 2002) and HEC rainfall runoff and river hydraulics models (USACE, 1989).

Early attempts to develop an integrated approach to planning and design of water resources systems can be traced back to 1955 when the Harvard Water Program brought together a group of professors with engineering, economics, and political science backgrounds to integrate economic theory and engineering practice through a multidisciplinary environment (Maass, et al., 1962; Reuss, 2003). In the late 1960s and early 1970s, economic water demand curves were used to establish a conceptual framework for regional scale integrated water management models that maximize the net benefits of water allocation (Harou et al., 2009). Following these early economic modeling efforts, many researchers have contributed to build hydroeconomic models of watershed systems by linking hydrological, hydrogeological, hydraulic, and biogeochemical processes to economic principles to facilitate integrated planning and management of watersheds (Brouwer and Hofkes, 2008). However, watershed planning and management decisions may not only rely on economic and hydrologic aspects of the system. In 1990s and 2000s, a plethora of research has been carried out on hydroeconomic models (Heinz et al., 2007; Brouwer and Hofkes, 2008; Harou et al., 2009), along with consideration of social and political aspects of watershed systems (Griffin, 1999; Korfmacher, 2001; Beck et al., 2002; Bagheri, 2006; Madani and Marino, 2009), which demonstrates a trend towards more holistic modeling approaches.

Since the time of development of Stanford Watershed model, the computational capacity to run sophisticated models has continuously increased at an overwhelming rate (Singh and Frevert, 2006). Over the same period, watershed models have evolved from purely engineering/economic models to more integrated tools that are capable of addressing various planning, design, and management problems with a desired level of detail. Growing computational capabilities, together with integration of data processing and management tools such as Geographic Information Systems (GIS) and data-base management systems with the watershed models (Singh and Woolhiser, 2002), has allowed for detailed spatial and temporal analyses of watershed systems. Likewise, great technological advances in remote sensing, satellites, and radar applications, combined with GIS techniques and an enhanced ability to perform field measurements, has allowed for more spatially distributed modeling of watersheds (Kite and Pietroniro, 1996; Fortin et al., 2001; Chen, 2004). Figure 1.1 schematically illustrates how watershed models are becoming more comprehensive and sophisticated thanks to increasing data processing capabilities and adoption of an interdisciplinary approach to address a wide spectrum of problems ranging from strategic level decisions to development of design alternatives.

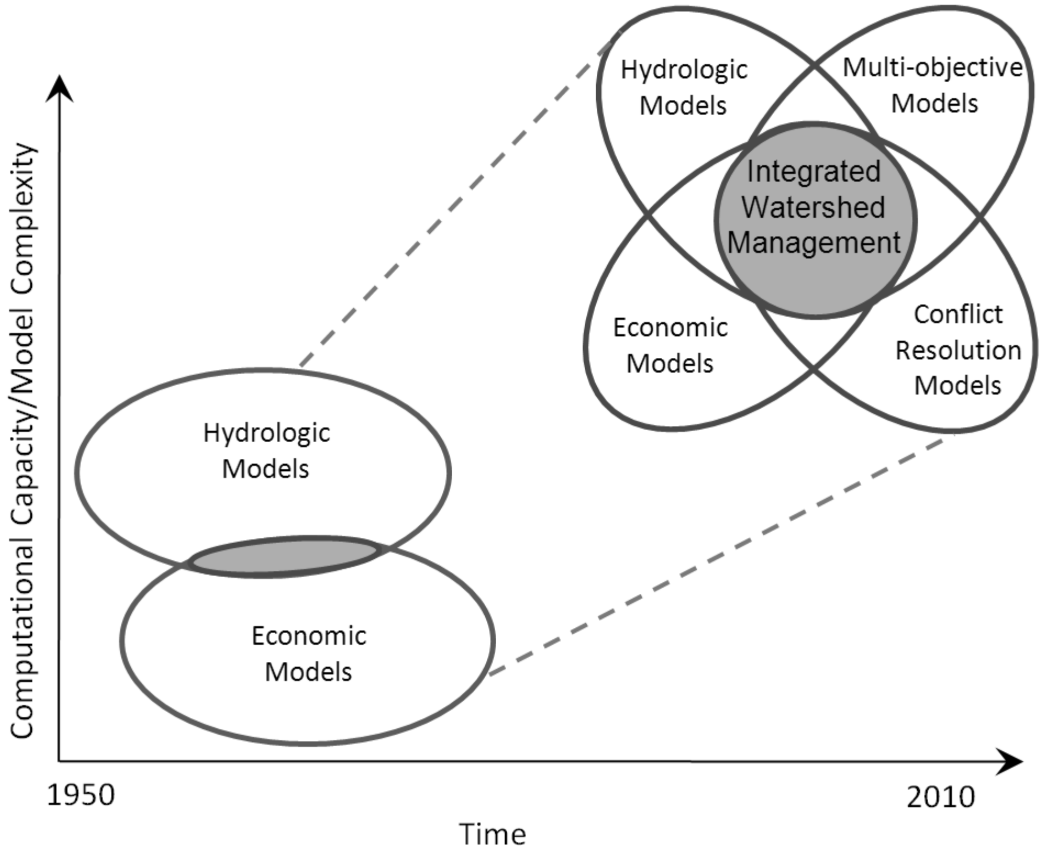


Figure 1.1.Integrated watershed modeling evolution over time.

Although the inherent complexity of water resources systems, coupled with lack of data and insufficient computational capacities, has often led to artificial compartmentalization of natural processes and human behavior for ease of modeling, the last few decades have seen a marked shift towards multi-disciplinary and integrated systems modeling (Estes, 1993; MacKenzie, 1996; Schultz, 2001; Madani and Mariño, 2009; Simonovic, 2009). The gradual shift from merely employing engineering-based simulation models to applying integrated hydroeconomic models, and more recently multi-criteria/multi-objective decision making and conflict resolution models, is an indicator of promising changes in the traditional paradigm for the application of water resources models. More holistic understanding of watershed systems, consideration of multiple stakeholder values, objectives and behavior, and improved abilities to predict and plan for future impacts are likely to lead to more sustainable water resources planning and management decisions. Figure 1.2 depicts the chronological evolution of water resources planning and management approaches.

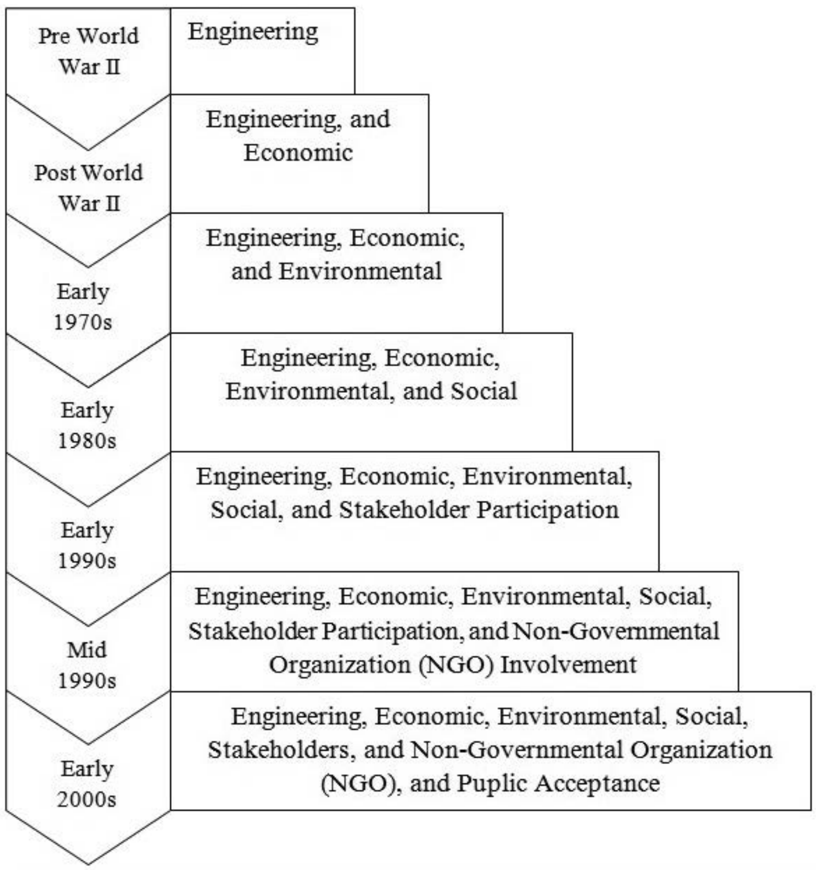


Figure 1.2.Chronological evolution of water resources planning and management approaches (Adapted from Arshady, 2010).

1.3. Water resources modeling methods and approaches

Water resources modeling methods and approaches have been categorized in different ways according to the types of problems they address and their method of finding a preferred solution. When categorized according to their solution method, models are classified either as simulation or optimization models. While there is a clear distinction between these two, as will be described below, many water resources studies involve a combination of simulation and optimization to analyze watershed systems and develop effective management policies. Alternatively, water resources models may be classified according to their scope and purpose into the following categories: engineering-based watershed process models, hydroeconomic models, multi-criteria (multi-objective) decision making models, and conflict resolution models. Each of these categories of models is briefly described below.

1.3.1. Modeling methods: Simulation and optimization

There are some key differences in the philosophy of these two modeling methods, and proper understanding of these differences is crucial to selection and application of the appropriate model. Depending upon the type and nature of the water resources planning and management problem being addressed, modelers have used either simulation or optimization models as the primary methods to study and analyze watersheds. However, optimization and simulation modeling are not mutually exclusive. In many studies, they are used in complementary fashion to support decision making. For example, following the preliminary screening of alternatives, feasible alternatives generated by optimization can be simulated for detailed analysis and impact prediction (Loucks and van Beek, 2005).

Simulation models take physical parameters and engineered designs, or management plans, as inputs and generate detailed predictions of outcomes. Simulation is widely applied in the detailed design phase of projects for quantitative performance and impact analysis of a limited number of alternative designs. The method is suitable for sensitivity (or "what if") analysis under a number of scenarios of interest. For example, a modeler may wish to use a simulation model to evaluate the performance of alternative designs under drought, normal, and flood scenarios. If performance of each alternative is unacceptable, new alternatives must be developed and evaluated. Engineering-based simulation is thus considered as an alternative-focused method in which the modeler intends to reach the best possible alternative design or quantitative representation of natural systems through a trial and error process (Makowski et. al., 1996; Garbrecht, 2006).

Optimization methods are geared towards creating alternatives based on selecting values for decision variables that provide the best value of an objective function, subject to a set of mathematical constraints (equations or limits that need to be satisfied in order for a particular alternative to be feasible). Understandably, expressing operational objectives and constraints in a mathematical form that can be solved by a computer often requires simplification of physical and socioeconomic relationships. Some advantages of optimization models are that they can help to screen a large number of potential alternatives, generate new alternatives that otherwise may have been overlooked, and provide an intuitive means of trade-off analysis. Also, optimization results need to be interpreted carefully, as the "optimal" outcomes may be overly optimistic and not achievable in practice. Table 1.1 compares some main aspects of simulation and optimization models.

Table 1.1.Simulation versus optimization.

Modeling method	Simulation	Optimization
Key question addressed	What if?	What's best?
Development effort	Low	High
Computational efficiency	High	Low
Transparency/ acceptability to the stakeholders	High	Low

1.3.2. Modeling approaches: Scope and problems addressed

1.3.2.1. Watershed process models

Watershed process simulation models are used for quantitative analysis, or prediction, of natural processes occurring at the watershed scale, to understand watersheds' natural behavior or their response to human- engineered alterations (Singh and Woolhiser, 2002). The structure of watershed process models varies depending upon modeling objectives, but in general they are built using a series of mathematical equations that describe the components of hydrologic or biogeochemical cycles, such as surface water hydrology, hydrogeology, soil chemistry, and limnological processes, to name a few. Presently, there exists a large number of generalized watershed simulation models that include, among others, rainfall-runoff processes, river hydraulics, groundwater hydraulics, and water quality processes (Wurbs, 1998). By focusing on natural processes, these models are often able to provide a detailed representation of one or more watershed subsystems. Engineering-based watershed process models are frequently applied in watershed planning and management to help raise the decision makers' awareness of technical nuances of proposed design alternatives, and predict the potential impacts of projects prior to their implementation. Watershed process models have been used in a wide range of studies, including rainfall-runoff prediction, flood mitigation design, water supply development, safety assessment of water infrastructure, land use planning, irrigation planning, hydropower operations, and surface and groundwater quality protection.

1.3.2.2. Hydroeconomic models

Apart from its life-sustaining role, water has economic value for various in-stream and off-stream uses such as domestic use, agriculture, industry, transportation, recreation, waste assimilation, and ecosystem maintenance (Gibbons, 1986). While physically-based watershed process models can capture the natural hydrologic behavior of watersheds, they have traditionally neglected the economic aspect of watershed modeling. However, water scarcity manifested by drought-induced economic downturn and intensified by growing demands for water necessitates consideration of appropriate economic factors in a robust watershed modeling framework to devise economically justifiable watershed management plans. Hydroeconomic models, often based on optimization methods, possess the advantage of facilitating economic studies by maximizing or minimizing some specified economic objective function subject to a series of constraints.

Harou et al. (2009) describe hydroeconomic models as solution-oriented tools that foster formulation of new strategies to promote water-use efficiency and transparency of decision making, thus contributing to integrated water resources management. However, maximizing the economic value of water use serves as the only driver of decisions in hydroeconomic models as economic valuation of many social, political and environmental objectives remains difficult. Integrated modeling of watershed-scale hydrological, environmental, and economic aspects of water use often requires simplified representation of natural processes (Heinz et al., 2007). Thus, water resources management decisions which are solely based on hydroeconomic models may not be comprehensive and a holistic model and approach is required for integrated water resources management. Hydroeconomic models have been applied to analyze water resources management practices and potential economic and environmental impacts, to address trade-offs and interactions among various stakeholder groups, to evaluate long term drought management and flood mitigation plans, to improve water resources operation policies and strategies, to suggest climate change adaptation strategies, and to identify economically promising resources for environmental restoration (i.e., to improve water quality and quantity for ecosystems).

1.3.2.3. Multi-objective decision making models

Water resources planning and management decisions must almost always consider multiple goals, many of which are conflicting. Often it is impossible to aggregate the goals into a single criterion or performance measure in the alternative ranking and selection process (Makowski et. al., 1996). Thus, multi-criteria (or multi-objective) decision support methods are widely applied for water policy planning and evaluation, as well as infrastructure development (Hajkowicz and Collins, 2007). In the context of optimization modeling, these methods seek to generate solutions that are "non-dominated," meaning that performance with respect to one objective cannot be improved without decreasing performance with respect to another objective. For example, reservoir operators need to consider the trade-off between water supply and flood mitigation benefits, as increasing the reliability of meeting a target supply (i.e., storing more water in a reservoir) would impose additional flood risk. By using optimization, all dominated solutions may be screened out, and the non-dominated solutions evaluated for trade-offs, allowing the decision maker to focus on a smaller set of potentially preferred alternatives (Hajkowicz and Collins, 2007). For water resources systems, MCDM methods may consider quantitative and qualitative criteria such as engineering standards and expected performance, environmental integrity, investment and operating costs, equity, and aesthetics (Hipel, 1992).

1.3.2.4. Conflict resolution models

The multitude of watershed planning and management objectives inevitably leads to conflicts among watershed stakeholders, or interest groups. In many cases, however, different stakeholder groups share common interests (e.g., a homeowner along a river may be primarily concerned about flood risk reduction but may also value the riverine ecosystem), or they may be able to reach compromise agreements (e.g., development of one portion of the floodplain may be offset by enhancing wetlands in another portion). Conflict resolution models essentially seek to promote compromise through holistic understanding of technical, socioeconomic, political, and environmental aspects of the problem (Lund and Palmer, 1997). Conflict resolution models have served as flexible tools for quantitative and qualitative analysis of watershed systems to suggest, given the circumstances, what would happen to the system based on detectable trends, stakeholders' interests, concerns, and behavior. Unlike the traditional "win-lose" or "zero-sum" conflict resolution approach, water resources conflict resolution models seek to lead the parties involved in the conflict towards a "win-win" situation or a " positive-sum", socially feasible solution (Nandalal and Simonovic, 2003).

Conventionally, most multi-criteria decision making models tend to transform multi-objective problems to a single composite objective (e.g. economic benefit, environmental integrity, social welfare), assuming that stakeholders will perfectly cooperate to reach the system's optimal solution (Madani, 2009). However, such an assumption may result in unrealistic results. Therefore, other conflict resolution models such as game theory models have been used in water resources management, which are capable of generating a more realistic simulation of stakeholders' and decision makers' behaviors by accounting for their concern to maximize their own benefit (Madani, 2010). By creating a platform for collaborative modeling and constructive negotiation, conflict resolution models can enhance stakeholders' and decision makers' understanding of the problem and aid in the definition of solution objectives and constraints. Collaborative modeling can facilitate the development of feasible alternatives, as well as the evaluation of alternatives' performance and impacts. Proper use of conflict resolution models has been found to increase technical confidence in the solution agreed upon (Lund and Palmer, 1997).

1.4. Objectives and organization

Water resources systems may be considered as hotspots for the sustainability process as they lie at the intersection of socioeconomic and environmental subsystems. As the need for comprehensive and reliable understanding of the consequences of natural and anthropogenic alteration of watersheds has grown, so has interest in water resources systems modeling to facilitate well-informed planning, and provide insights for decision making. This dissertation will contribute to application of a quant-qualitative framework for sustainable water resources management. It will focus on fundamentals of the systems approach to holistic water resources management with application to water quality management planning. Systems thinking and system dynamics simulation, as well as environmental systems analysis techniques are applied to provide insights for water quality management of example cases in the Great Lakes basin. The objectives of the dissertation are as follows:

- Illustrate the role of systems thinking paradigm in water resources planning and decision making;
- Demonstrate qualitative, as well as quantitative capabilities of system dynamics modeling in facilitating holistic water resources modeling and policy making;
- Identify and simulate the system structure driving the long-term eutrophication-recovery trend of Lake Allegan, Michigan to provide insights into policy leverages for mitigating impairment;

- Develop a framework for applying the systems approach for implementation of the concept of total maximum daily load (TMDL) for reducing non-point source (NPS) total phosphorus (TP) emission in the Kalamazoo River watershed, Michigan;
- Investigate market-based policy options for mitigating total phosphorus loads in the Maumee Basin.

This dissertation is organized in six chapters. The first chapter, as was presented in the preceding sections, provides an introduction, giving background information about how water resources models have become more holistic over the last decades. The fundamentals of systems thinking and system dynamics as a suitable framework for integrated analysis of water resources problems are discussed in Chapter 2. Furthermore, an application of the systems approach to a water quality management problem is presented in Chapter 3. The fourth and fifth chapters of the dissertation are devoted to insights from application of the systems approach to water quality policy in the Great Lakes Region. A simulation-optimization framework for guiding total maximum daily load (TMDL) implementation in the Kalamazoo River watershed is presented in Chapter 4. Chapter 5 investigates a number of policy instruments for reducing TP loads in the Maumee Basin, which covers parts of the three states of Indiana, Michigan, and Ohio. The conclusions and potential areas of future research are given in Chapter 6.

1.5. References

Arshady, M. (2010). A systemic analysis of hydro energy and agriculture sectors' vulnerability to water scarcity in the Great Karoon Basin. M.Sc. Thesis, Department of Water Resources Engineering, Faculty of Agriculture, Tarbiat Modares University, Tehran, Iran. In Persian.

Bagheri, A. (2006). Sustainable development: implementation in urban water systems. Dissertation, Lund University, Lund, Sweden <http://luur.lub.lu.se/luur?func=downloadFile&fileOId=546537> (accessed Oct. 22, 2009).

Beck, M.B., Fath, B.D., Parker, A.K., Osidele, O.O., Cowie, G.M., Rasmussen, T.C., Patten, .C., Norton, B.G., Steinmann, A., Borrett, S.R., Cox, D., Mayhew, M.C., Zeng, X.-Q., Zeng, W. (2002). Developing a concept of adaptive community learning: case study of a rapidly urbanizing watershed. *Integrated Assessment* 3 (4), 299-307.

Brouwer, R., Hofkes M. (2008). Integrated hydro-economic modelling: approaches, key issues and future research directions. *Ecological Economics* 66, 16-22.

Cai, X., McKinney, D. C., Lasdon, L. S. (2003). Integrated hydrologic-agronomic-economic model for river basin management. *Journal of Water Resource Planning and Management* 129, 4-17.

Chen, Y.D. (2004). Watershed modeling: where are we heading? *Environmental Informatics Archives* 2, 132-139.

Crawford, N. H., Burges, S. J. (2004). History of the Stanford watershed model. *Water Resources Impact* 6, 3–5.

Crawford, N.H., Linsley, R.K. (1966). Digital simulation in hydrology: Stanford Watershed Model IV. Technical report No. 39, Department of Civil Engineering, Stanford University, p. 210.

Fortin, J. P., Turcotte, R., Massicotte, S., Moussa, R., Fitzback, J., and Villeneuve, J. P. (2001). A distributed watershed model compatible with remote sensing and GIS data. I: Description of model. *Journal of Hydrologic Engineering* 6 (2), 91–99.

Freeze, R. A., Harlan, R. L. (1969). Blueprint for a physically-based, digitally-simulated hydrologic response model. *Journal of Hydrology* 9, 237-258.

Garbrecht, J. D. (2006). Comparison of three alternative ANN designs for monthly rainfall–runoff simulation. *Journal of Hydrological Engineering* 11, 502–505.

Gibbons, D.C. (1986). The Economic Value of Water. Resources for the Future, Washington, D.C.

Hajkowicz S, Collins, K. (2007). A review of multiple criteria analysis for water resources planning and management. *Water Resources Management* 21, 1553–1566.

Harou, J.J., Pulido-Velazquez, M.A., Rosenberg, D.E., Medellin-Azuara, J., Lund, J.R., Howitt, R. (2009). Hydro-economic models: concepts, design, applications and future prospects. *Journal of Hydrology* 375, 334-350.

Heinz, I., Pulido-Velasquez, M., Lund, J., Andreu, J. (2007). Hydro-economic modeling in river basin management: implications and applications for the European Water Framework Directive. *Water Resources Management* 21, 1103–1125.

Hipel, K. W. (1992). Multiple objective decision making in water resources. *Water Resources Bulletin* 28, 3-12.

Johanson, R. C., Imhoff, J. C., Kittle, J. L., Donigian, A. S. (1984). Hydrologic Simulation Program - FORTRAN (HSPF). User's manual for release 8. Report No. EPA-600/3- 84-066, U.S. EPA Environmental Research Lab, Athens, GA.

Kite, G. W., Pietrorino, A. (1996). Remote sensing applications in hydrological modelling. Hydrological Sciences Journal 41(4), 563– 587.

Korfmacher, K.S. (2001). The politics of participation in watershed modeling. *Environmental Management* 27(2), 161–176.

Langler, G.J., Smith, C. (2001). Effects of habitat enhancement on 0-group fishes in a lowland river. *Regulated Rivers: Research and Management* 17, 677–686.

Loucks, D.P., van Beek, E. (2005). Water resources systems planning and management: an introduction to methods, models and applications. In: Studies and reports in hydrology. UNESCO Publishing, Paris.

Lund, J.R., Palmer, R.N. (1997). Water resource system modeling for conflict resolution. *Water Resources Update* 108, 70-82.

Maass, A., Hufschmidt, M. A., Dorfman, R., Thomas, Jr., H. A., Marglin, S. A., Fair, G. M. (1962). Design of water-resource systems: New techniques for relating economic objectives, engineering analysis, and governmental planning. Harvard University Press, Cambridge, Massachusetts.

Madani, K. (2009). Climate change effects on high-elevation hydropower system in California. Ph.D. Dissertation, Department of Civil and Environmental Engineering, University of California, Davis. <http://cee.engr.ucdavis.edu/faculty/lund/students/ MadaniDissertation.pdf> (accessed Nov.15, 2009).

Madani, K. (2010). Game theory and water resources. *Journal of Hydrology* 381(3-4), 225- 238.

Madani, K., Hipel, K. W. (2007). Strategic Insights into the Jordan River Conflict. Proceeding of the 2007 World Environmental and Water Resources Congress, Tampa, Florida, Edited by Kabbes K. C., pp. 1-10, American Society of Civil Engineers, doi: 10.1061/40927(243)213.

Madani, K., Mariño, M.A. (2009). System dynamics analysis for managing Iran's Zayandeh-Rud River basin. *Water Resources Management* 23(11), 2163-2187.

Makowski, M., Somlyody, L., Watkins, D. (1996). Multiple criteria analysis for water quality management in the Nitra Basin. *Water Resources Bulletin* 32, 937–947.

Nandalal , K.D.W. , Simonovic, S. P. (2003). State-of-the-art report on systems analysis methods for resolution of conflicts in water resources management. UNESCO-IHP Publication, PCCP Series, p. 20.

Reuss, M. (2003). Is it time to resurrect the Harvard Water Program? *Journal of Water Resources Planning and Management* 129, 357-360.

Singh, V.P., Frevert, D.K., eds. (2006). Watershed Models. CRC Press, Boca Raton, Florida.

Singh, V. P., Woolhiser, D. A. (2002). Mathematical modeling of watershed hydrology. *Journal of Hydrological Engineering* 7(4), 270-292.

Tullos, D. (2009). Assessing the influence of environmental impact assessments on science and policy: An analysis of the Three Gorges Project. *Journal of Environmental Management* 90(Sup. 3), 208–223.

U.S. Army Corps of Engineers (1989). Water Control Software: Forecast and Operations. Hydrologic Engineering Center, Davis, CA.

Wright, S.A., Schoellhamer, D.H. (2004). Trends in the sediment yield of the Sacramento River, California, 1957–2001. San Francisco Estuary and Watershed Science (on line serial), 2(2). <http://repositories.cdlib.org/jmie/sfews/vol2/iss2/art2/> (accessed Oct.15, 2009).

Wurbs, R. A. (1998). Dissemination of generalized water resources models in the United States. *Water International* 23, 190–198.

Wurbs, R.A., James, W.P. (2002). Water resources engineering. Prentice-Hall.NJ.

Yoshinobu, K., Tomohisa, Y., Toshimasa, H., Sadahiro, Y., Koji, I. (2006). Causes of farmland salinization and remedial measures in the Aral Sea basin-Research on water management to prevent secondary salinization in rice-based cropping system in arid land. Agricultural Water Management 85, 1–14.

Chapter 2- Synthesis of system dynamics tools for holistic conceptualization of water resources problems[2]

2.1. Abstract

Out-of-context analysis of water resources systems can result in unsustainable management strategies. To address this problem, systems thinking seeks to understand interactions among the subsystems driving a system's overall behavior. System dynamics, a method for operationalizing systems thinking, facilitates holistic understanding of water resources systems, and strategic decision making. The approach also facilitates participatory modeling, and analysis of the system's behavioral trends, essential to sustainable management. The field of water resources has not utilized the full capacity of system dynamics in the thinking phase of integrated water resources studies. This chapter advocates that the thinking phase of modeling applications is critically important, and that system dynamics offers unique qualitative tools that improve understanding of complex problems. Thus, this chapter describes the utility of system dynamics for holistic water resources planning and management by illustrating the fundamentals of the approach. Using tangible examples, the chapter provides an overview of Causal Loop and Stock and Flow Diagrams, reference modes of dynamic

---

[2] This chapter is a reprint of the article: Mirchi, A., Madani, K., Watkins, D.W. Jr., Ahmad, S., (2012). Synthesis of system dynamics tools for holistic conceptualization of water resources problems. Water Resources Management 26(9), 2421-2442. Reprinted with permission from Springer.

behavior, and system archetypes to demonstrate the use of these qualitative tools for holistic conceptualization of water resources problems. Finally, the chapter presents a summary of the potential benefits as well as *caveats* of qualitative system dynamics for water resources decision making.

2.2. Introduction

An event-oriented view of the world or linear causal thinking cannot address complex problems adequately (Forrester, 1961 and 1969; Richmond, 1993; Sterman, 2000). Figure 2.1 illustrates this unidirectional thinking paradigm, which is grounded on the intuitive assumption that outputs or events are shaped by the collective effect of a series of inputs or causes acting sequentially (Sterman, 2000). One artifact of this type of thinking is that many problems, manifested by discrepancies between the present state and an expected or desired state, are singled out and treated in isolation from the surrounding environment. Consequently, no in-depth understanding of root causes of problems is obtained. Thus, managing complex water resources systems using uni-directional, mechanistic models may be doomed to provide unrealistic, or at least, questionable results (Hjorth and Bagheri, 2006).

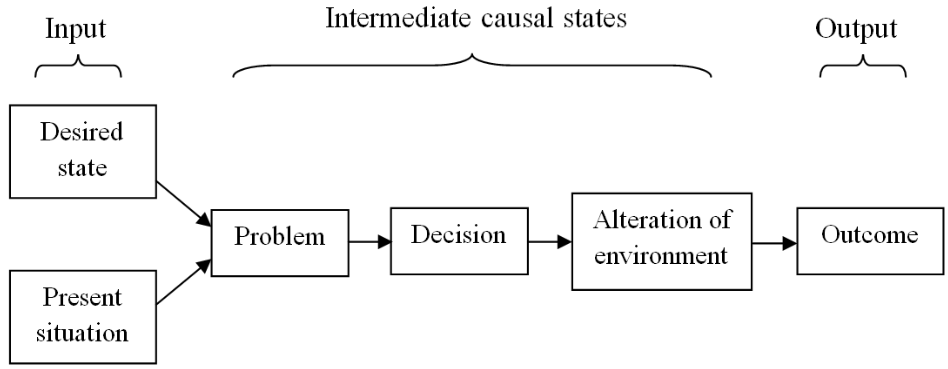


Figure 2.1. Linear causal thinking (adapted from Sterman, 2000).

Closed-loop or non-linear causal thinking enables analysts to consider important feedback loops and interconnections characterizing the system's structure, and to account for time delays, collectively shaping the behavior of complex systems (Richmond, 1993). This type of thinking is conceptually illustrated in Figure 2.2. The growing discrepancy between the existing and ideal state tends to generate a perception of problem, which often leads humans to alter the environment in hopes of reaching the desired state. Although the quick-fix solutions appear to alleviate the symptoms, which may be helpful when responding to emergencies, they often fail to address the problem appropriately and only result in its spatial and/or temporal translation (Richmond, 1993; Simonovic, 2009). The decisions to modify the environment may have unintended consequences, perhaps with time delays, which may aggravate the original problem or create even more challenging issues (Madani and Mariño, 2009). Unlike the quick-fix approach to planning and management of water resources, a non-linear thinking paradigm offers the holistic framework needed to promote sustainable development trajectories.

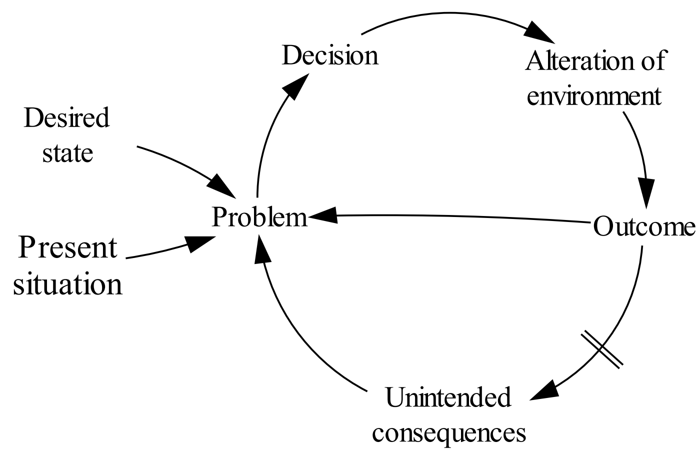


Figure 2.2. Non-linear causal thinking; causal states and causal relationships are denoted by words and arrows, respectively. Double bars indicate presence of time delay (Adapted from Sterman, 2000).

Systems thinking provides methods and techniques to apply non-linear causal thinking to planning and management problems. In essence, systems thinkers recognize the fact that while problematic systems are comprised of interrelated parts or subsystems, they function as a unit and should ultimately be treated as a whole (Simonovic, 2009). Simonovic and Fahmy (1999) consider the systems approach as a discipline for seeing wholes and for seeing structures that underlie complex domains. Further, they state that the systems approach is a framework for seeing patterns of change rather than static snapshots, and for seeing processes and interrelationships rather than objects. Thus, the principles of systems thinking are critical to solving problems in water resources systems which inevitably consist of interrelated subsystems.

System dynamics (Forrester, 1961 and 1969; Meadows, 1972; Richmond, 1993; Ford, 1999; Sterman, 2000) is one of the methods that facilitate recognition of interactions among disparate but interconnected subsystems driving the system's dynamic behavior. The method can thus help water resources analysts to identify problematic trends and comprehend their root causes in a holistic fashion. By identifying and capturing feedback loops between components, system dynamics models can provide insights into potential consequences of system perturbations, thereby serving as a suitable platform for sustainable water resources planning and management at the strategic level (Hjorth and Bagheri, 2006; Madani and Mariño, 2009; Simonovic, 2009). To this end, system dynamics offers several qualitative and quantitative tools to identify and explain system behavior over time.

System dynamics has not been used by most water resources scholars and practitioners to its full capacity. The majority of system dynamics applications in water resources have underutilized the method's qualitative modeling tools. The conceptualization or thinking phase of integrated water resources studies is of paramount importance as it provides fundamental understanding of leverage points for sustainable solutions. High level and qualitative models can be developed relatively quickly and affordably to facilitate trend identification, and to provide insights into root causes of multi-faceted water resources problems, facilitating formulation of preemptive and sustainable solution strategies. This chapter provides a synthesis of qualitative modeling techniques offered by system dynamics and argue that these techniques offer important insights and should not be overlooked by water resource modelers. To do this, the chapter first presents a synopsis of system dynamics applications in water resources. Then, the fundamentals of system dynamics and its qualitative modeling tools such as Causal Loop Diagrams (CLD) and Stock and Flow Diagrams (SFD) are discussed in detail, using tangible examples to illustrate why this approach is well suited for integrated water resources modeling, planning, and management. Furthermore, reference modes of dynamic behavior and merits of using system archetypes for qualitative modeling prior to quantitative analyses are illustrated. Finally, the method's benefits and *caveats*, stemming from application of the approach without proper regard for its philosophy, are discussed.

2.3. System dynamics and water resources

System dynamics, a sub-field of systems thinking (Richmond, 1994; Ford, 1999), originated in the 1960's when the concepts of feedback theory were applied by Forrester and his colleagues to understand the underlying structure and dynamics of industrial and urban systems (Forrester, 1961 and 1969). The method has since been widely used by analysts from various disciplines as a convenient tool to explore the causal relationships forming feedback loops between different components of large systems. In the past 50 years, system dynamics has become a well-established methodology that has been applied in many different practical and scientific fields, including management, ecology, economics, education, engineering, public health, and sociology (Sterman, 2000).

Application of system dynamics in water resources engineering and management has grown over the past two decades (Winz et al., 2009). Reviewing the literature, three general approaches to water resources system dynamics modeling can be identified: (i) predictive simulation models; (ii) descriptive integrated models; and (iii) participatory and shared vision models. In the first class of system dynamics models, modelers have successfully used the method as a tool to quantitatively simulate the processes governing particular subsystems within a broader water resources system. For example, Ahmad and Simonovic (2000) used system dynamics to model the interactive components of the hydrologic cycle to develop reservoir operation rules for flood mitigation. Ideally, this type of system dynamics model is developed to help predict the future behavior of the system accurately enough to provide a basis for tactical decisions. Table 2.1 presents some examples of water resources problems addressed using system dynamics as a convenient simulation tool for analyzing water resources problems and/or physical watershed processes.

In the second class of system dynamics models, analysts have adopted a more holistic approach, striving to identify and characterize the main feedback loops among two or more disparate subsystems, such as hydrological, ecological, environmental, socio-economic, and political subsystems. Typically, these integrated feedback models facilitate testing and selection of water resources management plans and policies at the strategic level. Table 2.2 summarizes example water resources studies, which have used system dynamics to describe and better understand the feedback structure and long-term behavioral patterns of interacting water resources subsystems.

Table 2.1. Example applications of system dynamics as a convenient simulation tool for modeling water resources problems and/or physical watershed processes.

Issue(s) addressed	**Modeling approach**	**Citation, Location**	**Authors' Remarks**
Freshwater eutrophication	Simulated direct discharge of nutrients from sewage and agriculture runoff on phosphorus and plankton dynamics	Vezjak et al. (1998), Slovenia	Facilitated setting standards for nutrient loading; suitable decision support tool for water quality management
Developing reservoir operation rules for flood damage mitigation	Simulated hydrologic behavior of the reservoir and upstream and downstream areas under major historical floods	Ahmad and Simonovic (2000), Canada	Ease of model modification and sensitivity analysis noted, suitable for participatory modeling and building trust into model results
Assessing climate change impacts on an urban flood protection system	Hydrologic processes and flood protection performance simulated under various climate scenarios	Simonovic and Li, (2003), Canada	Flexible model structure and ease of sensitivity analysis noted, suitable for flood management policy testing
Adaptive water quality management of an impaired stream	Total maximum daily load (TMDL) simulation	Tangirala et al. (2003), USA	Facilitated evaluation of alternative options for impairment mitigation
Flood damage estimation	Developed and applied a new methodology for spatiotemporal simulation of processes governing flood propagation	Ahmad and Simonovic (2004), Canada	Innovative generic approach for building distributed system dynamics models, capable of accounting for spatial variability and its impacts on feedbacks in multi-sectoral systems
Adaptive water resources planning and management	Basin-scale hydrologic simulation	Stewart et al. (2004), Mexico Sehlke and Jacobson (2005), USA	Integrated basin-scale watershed process model capable of incorporating policy, regulatory, and management criteria to form a decision support system
Thermal and mass balance of a spring	Simulated physical processes influencing the spring's geothermal characteristics	Leaver and Unsworth (2006), New Zealand	A lumped parameter model addressing hydrologic and geothermal processes
Salinity load forecast and removal from return flows	Simulated processes governing hydrology, water use, and water quality	Venkatesan et al. (2011a, 2011b), USA	Integrated simulation model providing insights into potential future water shortages, and cost-effective and energy-efficient water reuse plans

Table 2.2. Example applications of system dynamics in integrated or multi-subsystem feedback modeling of water resources systems for strategic policy testing and selection.

Issue(s) addressed	**Modeling approach**	**Citation, Location**	**Authors' Remarks**
Water resources policy analysis and decision making	Object-oriented modeling linking alternative socio-economic development plans with water availability at the national level	Simonovic and Fahmy (1999), Egypt Simonovic and Rajasekaram (2004), Canada	Flexible, transparent framework facilitating participatory modeling; complex due to accounting for several interconnected sectors
Water quality and environmental deterioration due to socio-economic growth	Various regional-scale physical and socio-economic subsystems linked to a water quality model	Guo et al. (2001), China Leal Neto et al. (2006), Brazil	Supported effective regional-scale environmental planning, management, and decision making
Sustainable water resources management in the face of growing demand	Various physical subsystems and water use sectors simulated under different scenarios (i.e., climate and management)	Xu et al. (2002), China Qaiser et al. (2011), USA	Captured main drivers of supply and demand, and provided insights for regional water management roadmap
Effective crisis management in response to flooding	Simulated human behavior during flood emergency evacuation	Simonovic and Ahmad (2005), Canada	Practical framework for monitoring and policy selection for emergency planning
Long-term impacts of interbasin water diversions into a water scarce area	Interactions among various drivers of water shortage analyzed, and sustainable water resources management strategies recommended	Madani and Mariño (2009), Iran	Provided insights for effective regional-scale water resources management and policy selection
Long-term water allocation among various stakeholder groups	Basin-scale hydrological, agricultural, economic, and ecological subsystems simulated	Gastélum et al. (2009), Mexico Ahmad and Prashar (2010), USA	Integrated watershed process model, supporting policy testing and formulating integrated management criteria
Post-disaster water resources management	Simulated post-earthquake changes in water consumption pattern, population, and water infrastructure development	Bagheri et al. (2010), Iran	Facilitates monitoring and policy selection for post-disaster water resources management to accommodate increased demand due to relief operation and reconstruction

Additionally, system dynamics models have been used as practical tools for promoting shared vision planning, participatory modeling, and shared learning opportunities for diverse groups of decision makers and stakeholder groups (Werick and Whipple, 1994; Lund and Palmer, 1997; Creighton and Langsdale, 2009). Stakeholders' participation in a group model building activity can increase understanding of the scope and complexity of the problem, increase trust in model results and, subsequently, increase support for the selected policy (Stave, 2003; Tidwell et al., 2004). Table 2.3 presents examples of participatory water resources modeling using system dynamics.

Table 2.3. Example applications of system dynamics in participatory water resources modeling for integrated policy assessment.

Issue(s) addressed	Modeling approach	Citation, Location	Author's Remarks
Over-appropriation of river flow to diverse stakeholder groups	A system dynamics model of processes governing annual river flow used by participants from agricultural and hydropower production sectors	Ford (1996), USA	Facilitated shared learning and useful participation of a diverse group of stakeholders, simulation results led to constructive discussions about complex water issues
Enhancing public understanding of water management options in a fast growing area	A strategic-level system dynamics model used in a public forum to illustrate the effectiveness of available and proposed management alternatives	Stave (2003), USA	Counterintuitive model results triggered informative discussions among participants, and effective management strategies were identified
The need for public participation in integrated water resources planning and management	Participatory system dynamics simulation of key hydrologic, social, and environmental drivers for quantitative comparison of alternative management options	Tidwell et al. (2004), USA	Facilitated public involvement in decision making, and increased public understanding of water management complexities; facilitated analysis of water supply, demand, and conservation
Incorporating the implications of climate change in integrated water resources planning and management	Participatory, scenario-based approach to build a watershed model to explore water resources futures and basin-scale policy options	Langsdale et al. (2007), Langsdale et al. (2009), Canada	Provided shared learning experience and increased the participants' appreciation of future water management challenges (reduced supply and increased demand)

2.4. Qualitative modeling tools in system dynamics

Qualitative modeling or conceptualization of systems' problematic behavior is useful for describing the problem, its possible root causes, and solutions. System dynamics depends heavily upon both quantitative and qualitative data to characterize feedback loops in complex systems (Forrester, 1975; Luna-Reyes and Andersen, 2003). In effect, a significant benefit of system dynamics stems from its ability to facilitate conceptualization of multi-disciplinary models by providing a number of qualitative tools to complement quantitative simulations (Wolstenholme, 1999; Coyle, 2000). Being accustomed to a tradition of developing highly quantified models, however, many water resources system dynamics modelers tend to overlook the approach's useful qualitative tools (Mirchi et al., 2010). Examples of these tools and ideas are CLDs, SFDs, reference modes of dynamic behavior, and system archetypes. This section provides an overview of the fundamental constructs and qualitative modeling techniques offered by system dynamics.

2.4.1. Causal relationships

At the core of system dynamics models are reinforcing (positive) and balancing (negative) causal relationships. A positive causal relationship means an increase/decrease in model Variable A would result in an increase/decrease in model Variable B, whereas a negative causal relationship signifies that an increase/decrease in model Variable A triggers a decrease/increase in model Variable B. For example, if the area of cultivated land in an agricultural district is increased, agricultural water demand will rise (positive causal relationship). Likewise, increase in hydraulic conductivity and temperature will increase groundwater recharge and evaporation, respectively. In contrast, as infiltration increases, the amount of surface runoff into a storage reservoir will decrease. Similarly, increased evaporation will cause the stored water in the reservoir to decrease. In another balancing relationship, as the groundwater table falls, the pumping cost will rise. A summary of the given examples, along with graphical notation of reinforcing and balancing causal relationships and their interpretation, is presented in Table 2.4.

Table 2.4. Graphical notation and polarity of causal relationships.

Connection	**Causal relationship**	**Mathematical definition**	**Examples**
$A \xrightarrow{+} B$	Any change in the state of A causes the state of B to change in the same direction; if A increases/decreases, B increases/decreases	$\frac{\partial B}{\partial A} > 0$	Cultivated land $\xrightarrow{+}$ Agricultural water demand Hydraulic conductivity $\xrightarrow{+}$ Groundwater recharge Temperature $\xrightarrow{+}$ Evaporation
$A \xrightarrow{-} B$	Any change in the state of A causes the state of B to change in the opposite direction; if A increases/decreases, B decreases/increases	$\frac{\partial B}{\partial A} < 0$	Groundwater table $\xrightarrow{-}$ Pumping cost Evaporation $\xrightarrow{-}$ Reservoir's stored water Infiltration $\xrightarrow{-}$ Runoff

2.4.2. Causal loop diagrams and basic feedback loops

Developing the system's CLD helps graphically capture the relationships between interactive subsystems, and can thus be considered as the conceptual modeling step. CLDs provide valuable information about the system including the presence of feedback loops, loop dominance, and presence of time delays. They are comprised of words and arrows with appropriate polarity, depicting combinations of positive and/or negative causal relationships. A causal relationship may exist between any two system variables, regardless of their type. In complex systems, combinations of positive and negative causal relationships may form feedback loops. There are two fundamental feedback loops--balancing (negative) and reinforcing (positive) loops. Typically, a balancing feedback loop comprises causal relationships which collectively attempt to reduce the discrepancy between the current state and a desired state. On the other hand, reinforcing feedback loops often characterize continuing trends of growth or decline. As a rule of thumb, a loop is reinforcing if the number of its negative causal links is even, and it is balancing otherwise, provided that the CLD appropriately represents the main drivers and causal relationships between them (Sterman, 2000). The ability to observe the structure of systems to identify dominant feedback loops in a representative CLD can provide qualitative information about their typical dynamic behavior. Therefore, when systems are not overly complex, it may be possible by looking at the CLD to determine the behavior of some variables even before quantitative modeling. A simple example is used to illustrate the behavior of reinforcing and balancing feedback loops, and the use of CLDs in qualitative modeling.

To better understand the behavior of a reinforcing feedback loop, consider a reservoir supplying water for a growing urban area. Net precipitation increases the inflow, raising the reservoir's stored water and increasing the potential for development (positive relationships). Subsequently, new opportunities for development lead to actual development, raising water demand, which then prompts the reservoir operators to allocate more space in the reservoir to storage. Allocating more space to storage would then create potential for more development which, in turn, would ultimately call for still more stored water. In the absence of other operating feedbacks (e.g., flooding, environmental flows, evaporation), the stored water in the system would grow exponentially until storage capacity has completely been used. Figure 2.3 shows a simple

CLD broadly illustrating the interrelationships within the “urban water supply loop” and corresponding behavior of the hypothetical system.

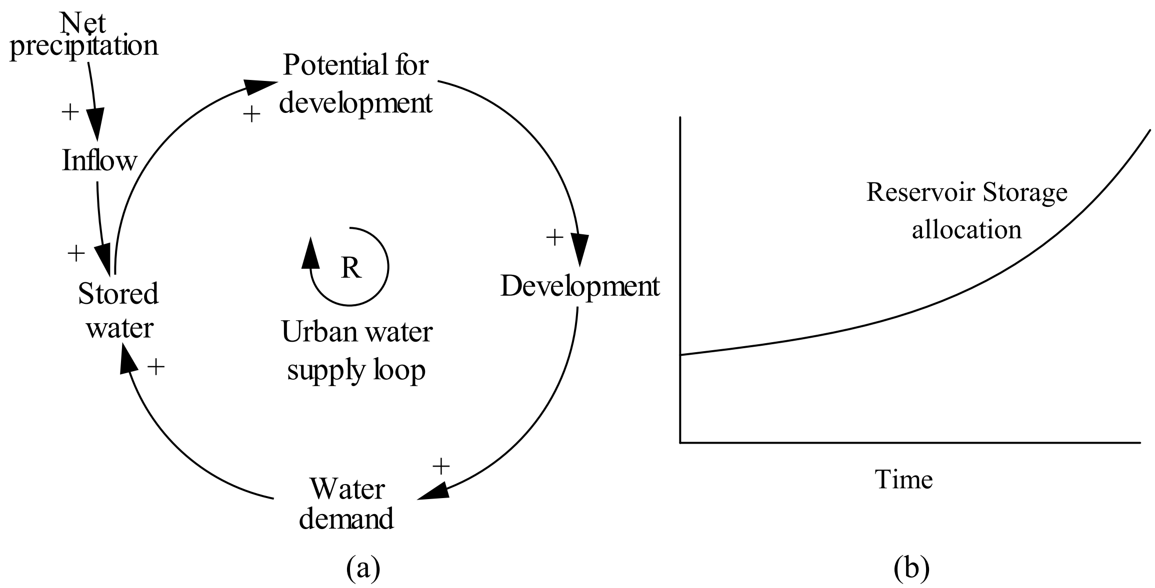


Figure 2.3. CLD of the “urban water supply loop” (a) and corresponding behavior of the hypothetical reservoir system (b). “R” denotes a reinforcing feedback loop.

In the “urban water supply loop” it was assumed that the reservoir is solely used for the purpose of water supply. Now suppose the reservoir functions only for flood control. In this case, high inflows raise the reservoir’s stored water, increasing flood risk. Consequently, the reservoir release is increased to reduce the stored water and accommodate the future inflows. This is an example of a balancing or negative feedback loop where the reservoir release helps maintain the reservoir’s water level below levels that would jeopardize the urban area. Neglecting all other operating loops, a CLD of the "flood control loop," along with behavioral graph of the reservoir system, is shown in Figure 2.4.

When studied separately, the "urban water supply loop" and "flood control loop" demonstrate distinctively different behavioral patterns (i.e., respectively, exponential growth and decline). However, when both feedback loops are present (Figure 2.5a), the system's long-term dynamic behavior may undergo variations depending on which loop is dominant. Figure 2.5b depicts potential long-term behavior of the system. Although this may seem like a trivial example, it represents the long-term behavior observed in a number of systems affected by development, where reservoir reallocation has been proposed (McMahon and Farmer, 2004).

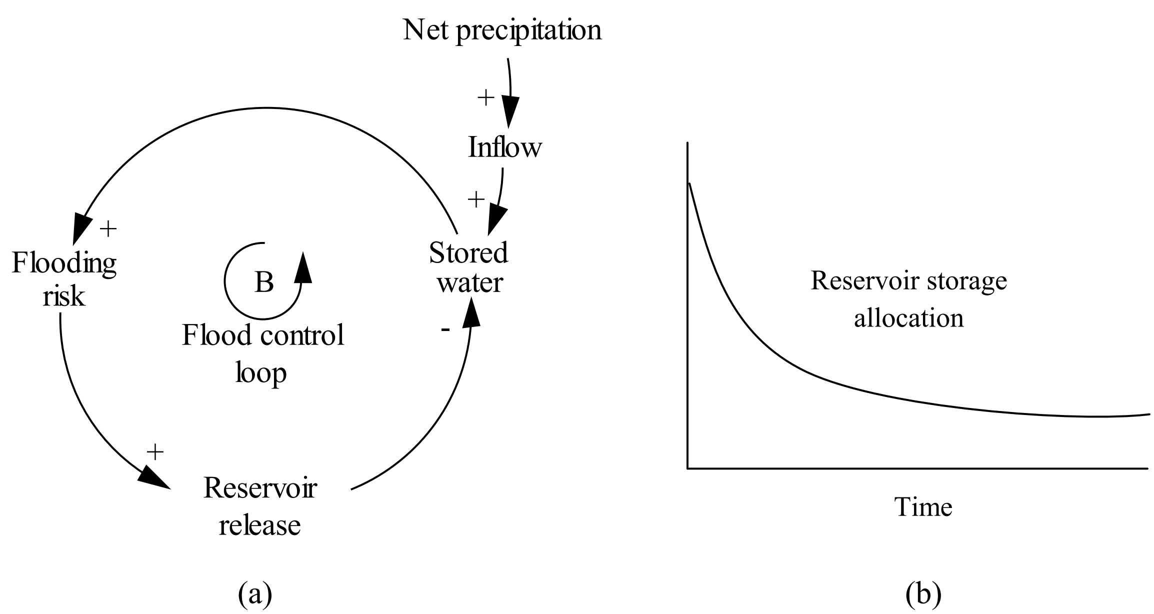


Figure 2.4. CLD of the "flood control loop" of the hypothetical reservoir system (a) and corresponding dynamic behavior (b). "B" denotes a balancing feedback loop.

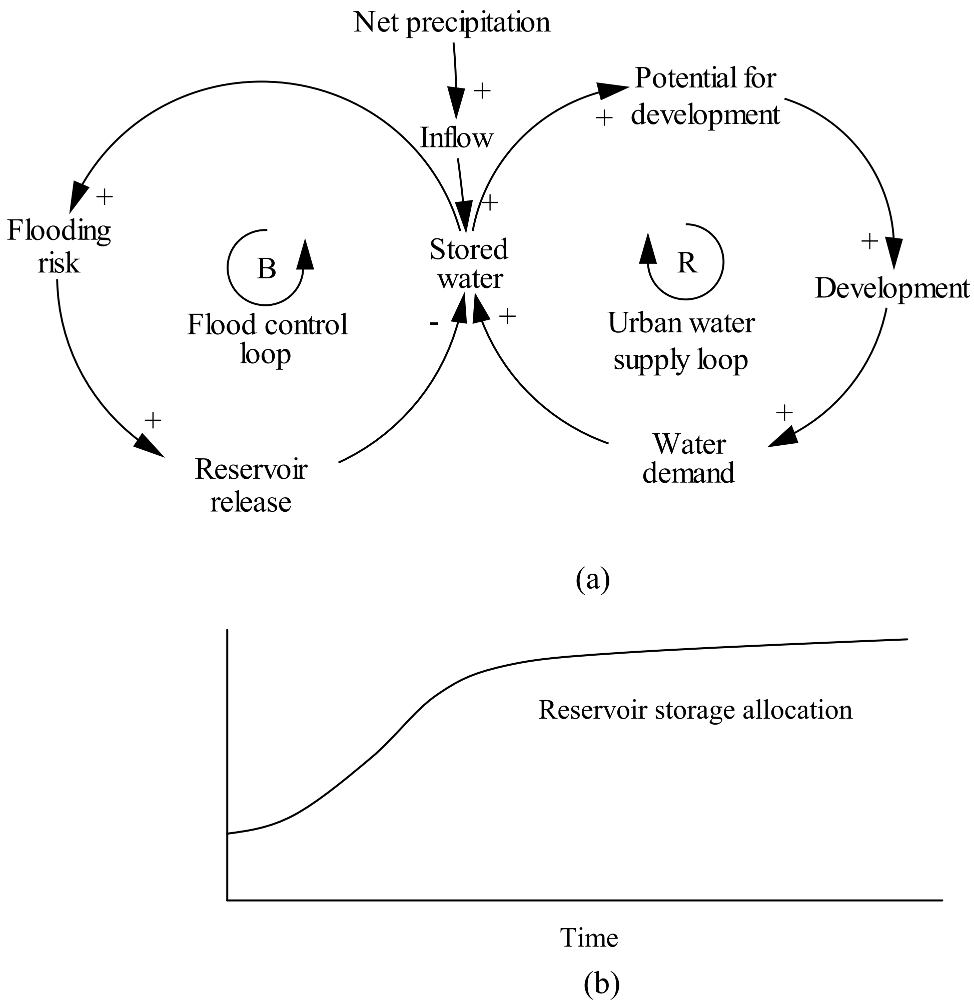


Figure 2.5. CLD of “water supply loop” and “flood control loop” (a), and corresponding long-term behavior of the hypothetical reservoir system (b). “R” and “B” denote reinforcing and balancing feedbacks, respectively.

2.4.3. Stock and flow diagrams

Based on the CLD of the problem, a Stock and Flow Diagram (SFD) can be developed to better characterize accumulation and/or depletion of stock(s) and flow of quantities in the system. General steps for translating CLD into SFD are summarized in

Table 2.5. Representing the system in terms of stocks and flows precedes quantification of the processes that have been accounted for in the CLD. Easy-to-learn software programs (e.g., STELLA (High Performance Systems 1992), Powersim (Powersim Corp. 1996), and Vensim (Ventana Systems 1996)) can be used to facilitate qualitative as well as quantitative system dynamics modeling. These simulation environments provide building blocks for developing quantitative models, obviating the need for learning complex programming languages, and allowing more people to gain hands-on experience with system dynamics modeling. Stocks (levels) are measured at one specific time and represent any variable that accumulates or depletes over time, while flows (rates) are measured over an interval of time and denote activities or variables causing the stock to change. For example, the stored water in a reservoir system can be modeled as a stock with inflow and release being its associated flows. Auxiliary variables, such as flood risk and potential for development, are functions of stocks or constants that help formulate and calibrate the model. Stocks, flows and auxiliary variables are connected by arrows (connectors), which are used to build relationships between the model variables by transferring information such as the value of parameters present in a particular model equation. Figure 2.6 shows a simple SFD of the reservoir example.

Table 2.5. Procedure for building SFD using CLD (Adapted from Wolstenholme and Coyle 1983).

Step	Purpose
Key variable recognition	Identify main drivers giving rise to problem symptoms
Stock identification	Identify system resources (stocks) associated with the main drivers
Flow module development	Provide rates of change and represent processes governing each stock
Qualitative analysis	Identify: (i) additional main drivers that may have been overlooked; (ii) causal relationships that require further analyzing by specific methods; (iii) controllable variables and their controllers; (iv) systemic impact of changes to controllable variables; (v) system's vulnerability to changes in uncontrollable variables

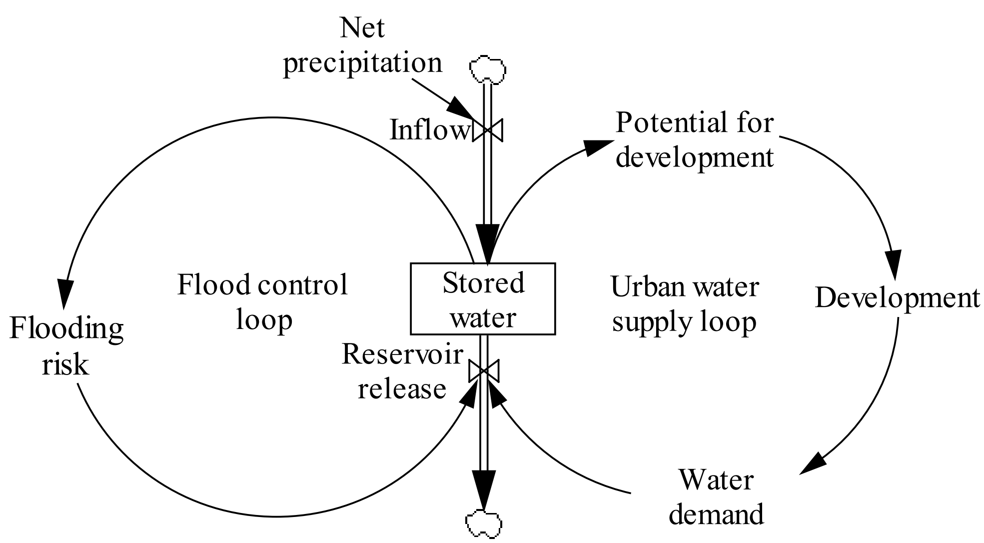


Figure 2.6. Stock Flow Diagram (SFD) of the reservoir problem.

2.4.4. Reference modes

A reference mode is an overall pattern of system's behavior over time as opposed to short historical time series which may be dominated by noise (Sterman, 2000; Ford, 1999; Simonovic, 2009). Saeed (1998) considers a reference mode as a qualitative and intuitive concept facilitating conceptualization processes, which does not represent the precise description/prediction of past/future events. Fundamental reference modes of dynamic behavior include exponential growth, goal seeking, and oscillation. Typically, reinforcing and balancing feedback loops demonstrate continuous growth and goal seeking behavior, respectively. Oscillation is generated by presence of delayed corrective components in balancing loops causing the system to constantly move above and then below its goal. Other common modes of dynamic behavior, which are caused by the fundamental modes, include S-shaped growth, oscillating overshoot, and overshoot and collapse. S-shaped growth is generated when the balancing feedbacks in a system dominate its behavior after it has, under impact of reinforcing loops, grown toward a limiting state (e.g., carrying capacity of an environmental system). When significant time delays hinder the balancing feedbacks to initiate the corrective action on time, the system will likely overshoot the limiting state, demonstrating an oscillatory behavior around the constraining limit (oscillating overshoot). In this situation, if the resource is non-renewable or carrying capacity is irreversibly exceeded, the system will collapse before the balancing feedbacks can salvage it (overshoot and collapse). Figure 2.7 presents common modes of dynamic behavior.

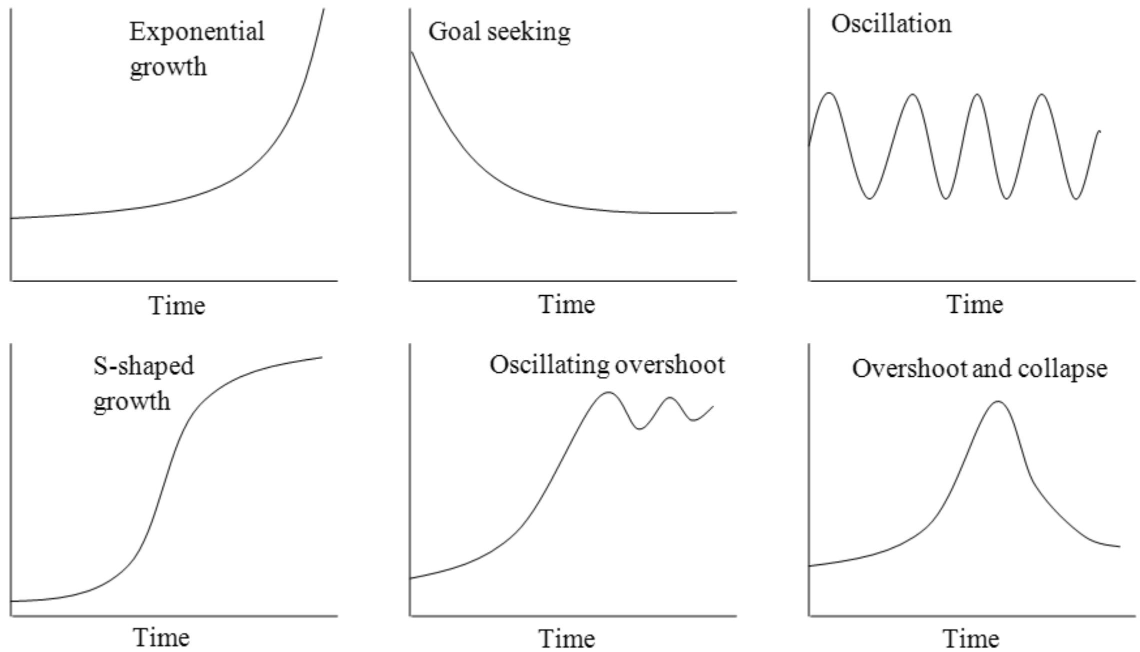


Figure 2.7. Common modes of dynamic behavior (Adapted from Sterman, 2000).

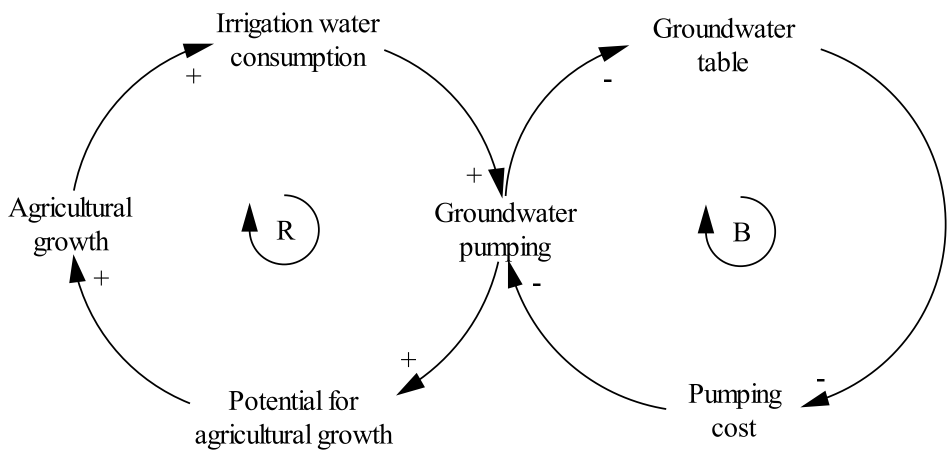
A water-stressed region's ongoing trend of development, facilitated by continuous supply of imported water, is an example of exponential growth driven by a reinforcing feedback structure (Madani and Mariño, 2009). In contrast, irrigation water withdrawal in an agricultural district where appropriate water pricing schemes are applied often follows a goal seeking dynamic behavior (Cai and Wang, 2006). An example of oscillatory dynamic behavior is a lake's water level which fluctuates seasonally due to seasonal precipitation patterns and water demands, and perhaps as mandated by lake operation rules (e.g., Watkins and Moser, 2006). When water resources limit a region's development (i.e., no water can be supplied from outside the watershed), the water consumption pattern will likely demonstrate S-shaped behavior (Bagheri and Hjorth, 2007). If long delays hinder timely response to warning signals (e.g., severe water stress), the system may overshoot its limit, causing social, economic, and/or environmental hardships. On the other hand, if the water resources are eventually replenished, it is likely that continued growth and consumption will exhaust the newly available water, generating alternating periods of replenishment and depletion, characterized by an oscillating overshoot mode of dynamic behavior. In an extreme case, extensive development in resource-stressed areas can completely exhaust the resources necessary for survival of the system causing it to collapse (Erickson and Gowdy, 2000).

2.4.5. System archetypes

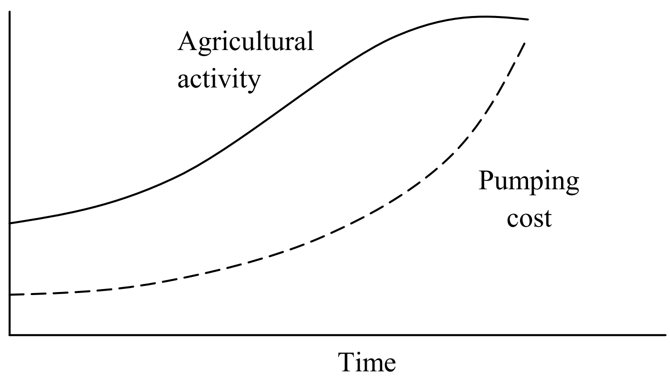
System archetypes are generic system structures showing common patterns of behavior (Senge, 1990; Wolstenholme, 2003). Reinforcing and balancing feedback loops are essentially the basic system archetypes. In real systems, however, a combination of reinforcing and balancing feedback structures can form more complex dynamic behaviors that can be characterized using more sophisticated system archetypes. Through closely studying the structure of many systems, a number of archetypes have been identified that can serve as diagnostic tools, describing or predicting the system's long-run behavior. Some common archetypes are Limits to Growth, Success to the Successful, Fixes that Backfire, and Tragedy of the Commons (Senge, 1990). Knowledge of the governing dynamics of water resources systems may help decision makers prognosticate problematic behavior and take appropriate corrective actions in a timely fashion, leading to more sustainable water resources planning and management. In this section, the applicability of some system archetypes to characterize water resources management problems is illustrated through a number of examples.

2.4.5.1. Limits to growth

The Limits to Growth archetype hypothesizes that continuous growth, driven by reinforcing feedback loops in natural systems, will eventually push the system toward its limit (e.g., carrying capacity). Once the system has grown beyond a critical level, balancing feedback loops take over and dominate the system's behavior, attempting to prevent its collapse (Meadows, 1972). Dynamic behavior of an agricultural system using groundwater as its source of irrigation water is an example of this archetype. A CLD of the Limits to Growth archetype for the given agricultural system is depicted in Figure 2.8a. In this system, agricultural growth raises the demand of irrigation water. Farmers may then develop additional groundwater resources and increase pumping, enhancing the potential for agricultural growth and an increase in cultivated land (reinforcing loop). However, pumping excessive amounts of groundwater will cause severe drawdown of the groundwater table, increasing the pumping cost which, in turn, reduces the demand for groundwater (balancing loop). As shown in Figure 2.8b, pumping cost increases with continuous agricultural growth until groundwater withdrawal is no longer economical, which then reduces the growth. In an extreme case, if a non-renewable groundwater resource is completely exhausted, the agricultural practice may cease altogether.



(a)



(b)

Figure 2.8. CLD (a) and behavioral graph of the Limit to Growth archetype (b) for the presented agricultural system.

2.4.5.2. Fixes that backfire

The Fixes that Backfire archetype characterizes quick-fix (short-sighted) solutions, stemming from linear causal thinking, which treat the symptoms of a problem rather than addressing its root causes. Interbasin water transfer with unintended consequences (e.g., false perception of water abundance, encouraging continued development and population growth (Madani and Mariño, 2009)), can be characterized by a Fixes that Backfire archetype. As shown in Figure 2.9a, intense water shortages prompt water managers to initiate water transfer projects to increase water supply, which will temporarily reduce the shortage. However, continuous supply of abundant water in a water-stressed region sends a false message to its current residents and inhabitants of neighboring areas about potential for development. Consequently, in the long run, while water resources are being depleted, increased development and immigration cause water shortages to grow more severe (the reinforcing loop dominates) (Figure 2.9b).

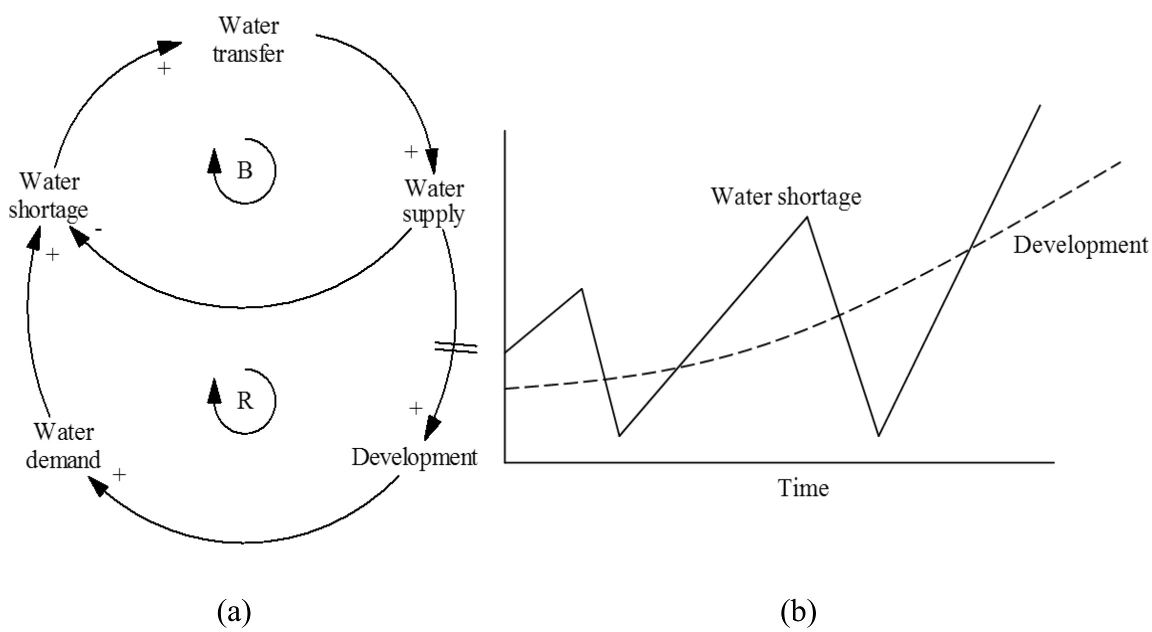
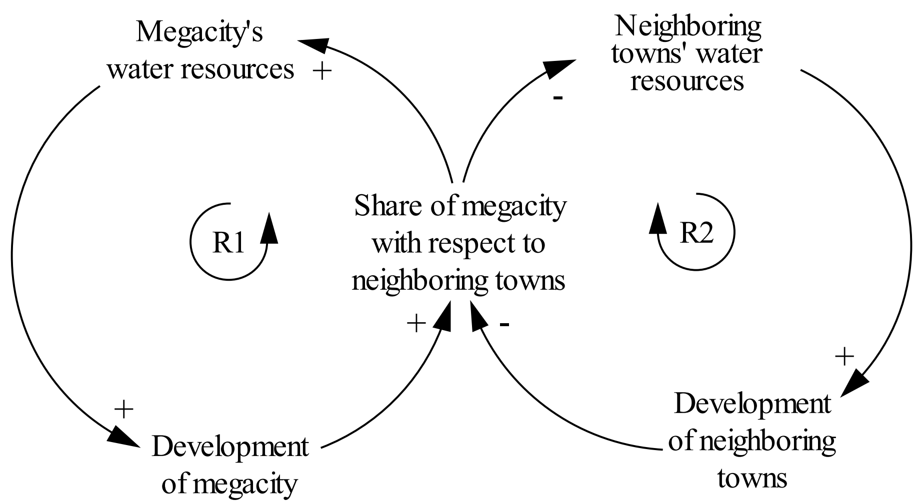


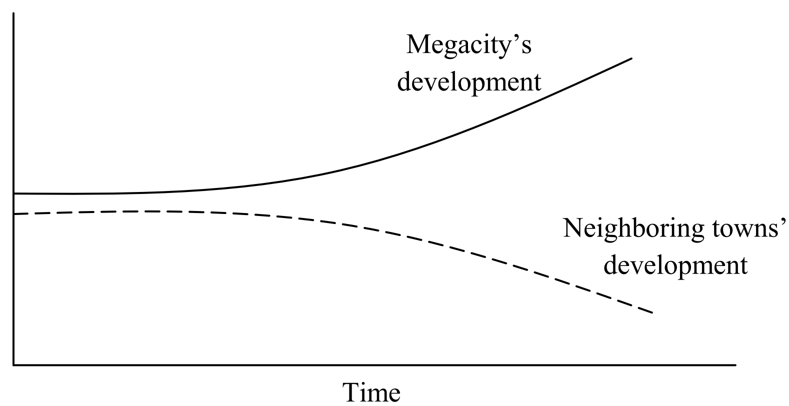
Figure 2.9. CLD (a) and long-term behavior of Fixes that Backfire archetype (b) for interbasin water transfer. Note the lag time until unintended consequences are observed, indicated by the double bars.

2.4.5.3. Success to the successful

The Success to the Successful archetype states that good performance will earn an entity more resources, making it possible for the entity to generate even better results and gain still more resources. Dominance of this archetype in a natural setting where resources are limited can deprive the weaker competitors of the resources they need to improve their condition and become more competitive. Consequently, the successful entity continuously grows while other entities gradually decline and possibly collapse. This archetype can ultimately result in considerable inequity and imbalance among entities (e.g., water resources stakeholders), threatening the system's sustainability. Supply-oriented water resources management in a large metropolitan area can be explained using the Success to the Successful archetype, illustrated in Figure 2.10a. Water scarcity in less-populated neighboring areas is secondary to the needs of a water-stressed megacity (Bagheri and Hjorth, 2007). As the megacity's share of water resources increases, so does potential for development, which in turn adds to the power of the megacity to gain more water resources. Simultaneously, the neighboring towns' share of water resources decreases, hindering their development and ability to gain necessary resources (Figure 2.10b). Another real example of a water resources problem based on the Success to the Successful archetype is floodplain development in California, which results in a continuous profit to the local developers (success) and continuous increase in risk of economic loss to the state of California as a result of development behind unreliable levees (Madani et al., 2007).



(a)

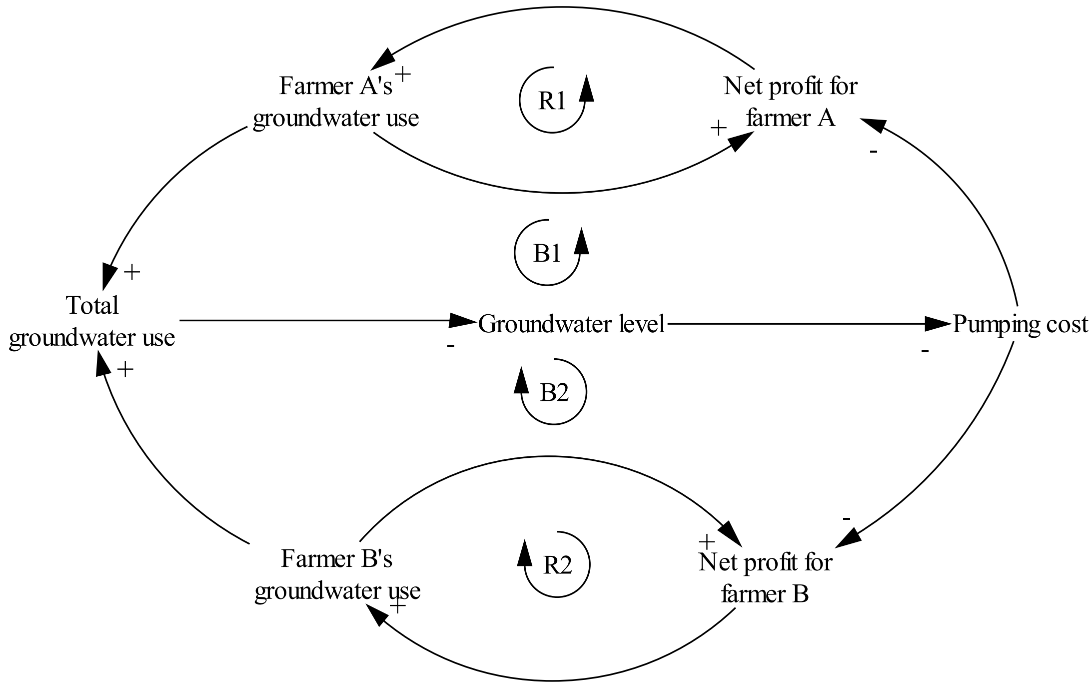


(b)

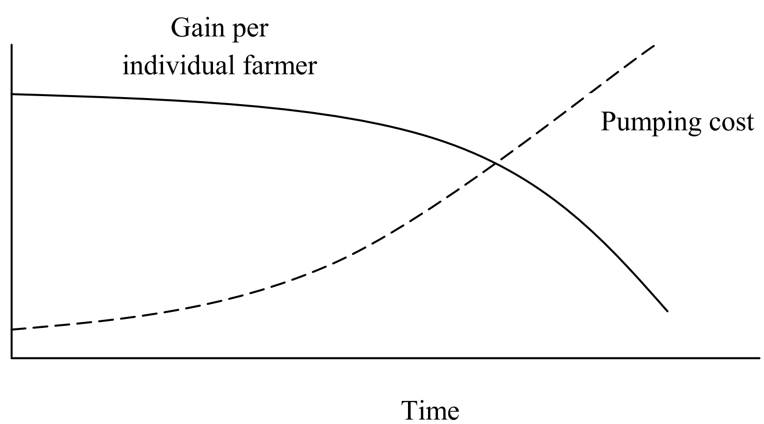
Figure 2.10. CLD (a) and long-term behavior of Success to the Successful archetype (b) for the urban water development problem.

2.4.5.4. Tragedy of the commons

This archetype is observed when multiple users exploit a shared water resource. Suppose two farmers use groundwater as the primary source for irrigating their crops. The shared resource can last longer under a regulated groundwater consumption scheme, maximizing the long-term profit of each individual stakeholder. However, in the absence of appropriate regulations, the farmers can pump as much as they want to maximize their profit. This situation is well represented by a Tragedy of the Commons (Hardin, 1968) archetype, whereby each party depletes the common pool resource solely based on their own self-interest (Loaiciga, 2004; Madani, 2010). Figure 2.11a shows a CLD of the Tragedy of the Commons archetype for the groundwater problem in which two farmers (A and B) compete to maximize their own net profit by exploiting the groundwater resource. Initially, the reinforcing loops R1 and R2 drive the system such that each farmer gains satisfactory profits. This situation holds until the groundwater table is excessively drawn down, at which point the balancing loops B1 and B2 dominate the system's dynamic behavior. Ultimately, increased pumping cost due to pumping from a lower groundwater table reduces the net profit for each individual farmer (Figure 2.11b). This archetype can be generalized to qualitatively analyze any common pool resource problem (Madani and Dinar, 2012), including transboundary water resources. In an extreme case, the competition between stakeholders can jeopardize local or regional sustainability and wellbeing of inhabitants, potentially initiating political conflicts (Lowi, 1993).



(a)



(b)

Figure 2.11. CLD (a) and the behavioral graph of Tragedy of the Commons archetype (b) for the groundwater problem when two users (farmers A and B) compete to maximize their own share of the resource.

2.5. Discussion

2.5.1. Qualitative versus quantitative modeling

Qualitative system dynamics modeling can be used at different levels for different purposes (Richmond, 1994). Randers (1980) believes that “most human knowledge takes a descriptive non-quantitative form”, and thus analysts should not restrict themselves to numerical data, which is a small fraction of knowledge fit for statistical analysis. However, developing qualitative models may not be enough for complete analysis of the problem. Proponents of quantitative modeling argue that numerical simulation nearly always adds value, even in the face of significant uncertainties about data and important qualitative information used in simulations (Forrester, 1975; Homer and Oliva, 2001; Dhawan et al., 2011). It is necessary to recognize the pitfall of oversimplifying a problem and neglecting the value of conducting detailed simulations, which may reveal complex system behaviors that could not be understood through simple diagramming (Homer and Oliva, 2001; Forrester, 2007). However, to accomplish a successful system dynamics application, extensive computer simulations should be performed only after a clear picture of the integrated water resources system has been established through reasonably simplified conceptual models. Contrary to conventional modeling which may fail to capture the big picture of the problem with important feedback loops, a thorough system dynamics study can provide reliable qualitative and quantitative bases for policy selection. In this way, instead of investing resources prematurely, analysts can prioritize what to study in more detail to ensure an in-depth understanding of the problem is obtained.

System dynamics' qualitative modeling tools, and the insights that they provide, make this approach accessible to a wide range of decision makers and stakeholders. The tools for visually exploring systems are a major distinguishing factor between this approach and traditional simulation methods. These qualitative analysis tools help generate a constructive medium for understanding a system's structure using an iterative approach best implemented through interaction with people who are familiar with the system at different levels (Randers, 1980). Therefore, attempts to reveal the main drivers of the problem using CLDs, SFDs, and archetypes are necessary. However, analysts should be aware of the general concerns about using qualitative modeling tools. In particular, problems might be encountered when translating CLDs into SFDs. Richardson (1986) argues that traditional definitions of the polarities of causal links and feedback loops are inadequate. In order to address this inadequacy he suggests that CLDs should account for the accumulating nature of the Flow-to-Stock links. As an alternative way for dealing with this problem, modelers can use CLDs along with reasonably representative SFDs to ensure qualitative insights are properly communicated.

System archetypes provide generic CLDs that reveal qualitative information about the underlying structure of the system, enabling water managers to detect current problematic trends and anticipate future problems. Thus, system archetypes are not meant to address any specific problem, but instead are applicable to classes of problems that share one or more modes of dynamic behavior. For various classes of problems, generic solution archetypes have been reported in the literature (Wolstenholme, 2004). As such, system archetypes can be used along with other system dynamics tools such as SFDs to generate a broad, holistic understanding of the system's state and its long-term behavioral pattern. Essentially, once analysts reach a consensus regarding the system archetype that governs their particular system of interest, they can obtain valuable insights into solution strategies, which can be further analyzed and tested using detailed simulations (Wolstenholme, 2003). This is a unique characteristic of system dynamics that facilitates conceptual or high-level strategic water resources modeling.

System dynamics models should have comprehensive, and yet simple structures, particularly when presented to non-technical audiences (Stave, 2003). In this context, simplicity is not equivalent to misrepresentation of the system's structure. Rather, it is more consistent with Albert Einstein's maxim that "a good explanation is one that is as simple as possible, but not simpler." Often, it is also important for system dynamics water resources models to have transparent structures that facilitate sensitivity analysis, which is critical for adaptive water resources management and scenario-based policy screening (Simonovic and Fahmy, 1999). If too much detail is included in the CLD, the structure of the system dynamics model may become overly difficult to understand for people who have not been involved in the model development. In addition to increased data requirements for complex, integrated models, to be able to provide meaningful interpretation of behavioral trends modelers must develop appropriate methods and protocols for quantifying socio-political subsystems--a task which remains a formidable challenge (Hellström et al., 2000; Luna-Reyes and Andersen, 2003). Furthermore, regardless of the scope of the problem, modelers need to apply appropriate aggregation and hierarchical decomposition principles to accomplish the modeling task, with the level of aggregation and decomposition varying according to the scale of problem, modeling objectives, and desired model sophistication.

2.5.2. Validation of system dynamics models

Model "verification" or "validation", i.e., testing of the model using an independent data set, is often problematic due to limited data and, in some cases, a lack of appropriate methods for quantifying particular (e.g., socio-political) subsystems. Sterman (2000), in fact, argues that no model can ever be verified or validated, for models are simplified representations of real processes and are thus different from reality in infinitely many ways. Nevertheless, in order for models to be useful as decision support tools for water resources planning and management, it is necessary to verify the model structure to ensure that mathematical equations and interrelationships between subsystems follow logical explanations and are not spurious or erroneous. Unlike purely data-driven black-box models, generating an "accurate" output behavior is not sufficient for validation of causal-descriptive white-box system dynamics models, which in addition to reproducing the system behavior, should explain how the behavior is generated (Barlas 1996). Thus, as Barlas (1996) explains, in the context of system dynamics, model validation is a semi-formal process consisting of a balanced mix of both quantitative tests and qualitative behavioral criteria targeting the system's internal structure. In participatory system dynamics modeling, validation can be done throughout model development by a range of experts and stakeholders, which may be much more reliable than an external review of the model at the end of the process. The verification phase of system dynamics models developed for water resources problems has not always been discussed in detail, but modelers have reported a variety of verification methods, including behavior replication, sensitivity analysis, dimensional consistency, and structure assessment (Ahmad and

Simonovic, 2000; Stave, 2003; Tangirala et al., 2003). Table 2.6 summarizes these methods of verification of water resources system dynamics models.

2.5.3. Strengths and limitations of system dynamics modeling

To summarize, Table 2.7 lists the major benefits and potential pitfalls of holistic water resources system dynamics models, including problems which might arise due to inappropriate application of the method without proper regard for its philosophy. Caution should be used when interpreting system dynamics models, for it is easy to formulate erroneous dynamic hypotheses due to inadequate information about a complex system, or due to lack of expertise. Biased simulation results may stem from faulty CLDs and SFDs. This *caveat* is particularly important when creating integrated models to simulate feedback relationships among socio-economic, political, natural, and technological subsystems. Tradeoffs among accuracy, breadth, and time must be considered in any modeling study. Nevertheless, although quantification of socio-economic and political components of water resources systems is challenging, and sometimes even speculative (Madani and Mariño, 2009), system dynamics modeling helps to prioritize information gathering and holistically investigate interactions and potential impacts of different drivers of the problem.

Table 2.6. Common methods for verification of water resources system dynamics models (Adapted from Sterman, 2000, revised for water resources applications)

Method	**Rationale**	**Procedure(s)**	**Citation**
Behavior replication	Reproduce the system's common modes of dynamic behavior both qualitatively and quantitatively	Perform statistical analyses of model results and observed data (e.g., R^2); qualitatively compare model results with data; investigate anomalies; change equilibrium conditions to disequilibrium conditions	Ahmad and Simonovic (2000), Guo et al. (2001), Stave (2003), Tangirala et al. (2003) , Tidwell et al. (2004), Madani and Mariño (2009), Bagheri et al. (2010), Venkatesan et al. (2011a), Qaiser et al. (2011)
Dimensional consistency	Ensure each model equation is dimensionally correct	Perform dimensional analyses; double check conversion factors; ensure correlation coefficients are dimensionally correct	Tangirala (2003)
Sensitivity analysis	Test numerical, behavioral, and policy sensitivity	Conduct univariate and multivariate sensitivity tests; simulate extreme conditions; change time step	Ahmad and Simonovic (2000), Tangirala et al. (2003), Bagheri et al. (2010), Venkatesan et al. (2011)
Structure assessment	Ensure model structure complies with natural laws (e.g., continuity) and represents description of the system, and appropriate aggregation and decision rules are applied	Develop CLDs and SFDs; delineate appropriate boundaries; test performance of each sub-model; change aggregation level and decision rules	Bagheri et al. (2010)Qaiser et al. (2011)

Table 2.7. Benefits and limitations of integrated water resources system dynamics models

Benefits	Limitations
• Provide tools for graphical representation of systems (CLDs and SFDs) promoting qualitative modeling • Facilitate flexible, transparent modeling • Facilitate holistic understanding of the problem • Capture long-run behavioral patterns and trends • Facilitate clear communication of model structure and results • Promote shared vision planning, participatory modeling, and shared learning experience • Facilitate sensitivity analysis • Suitable for policy assessment and/or selection	• Easy to conceptualize erroneous CLDs and SFDs • Easy to develop faulty models based on wrong CLDs and SFDs • Require experience and expertise to develop sufficiently detailed, insightful, and representative description of the system (dynamic hypothesis) • Require substantial interdisciplinary knowledge to generate meaningful quantitative predictions due to complexity and multitude of subsystems • Speculative quantification of some subsystems (e.g., socio-economic, and political subsystems).

2.6. Conclusions

The traditional linear thinking paradigm lacks the mental and organizational framework for sustainable development trajectories, and may lead to quick-fix solutions that fail to address key drivers of water resources problems. In contrast, systems thinking can help water resources decision makers comprehend the interactions among various interlinked subsystems of a water resources system which drive its long-run dynamic behavior. Applying a systems thinking paradigm to water resources modeling is thus critical in the thinking phase of formulating strategic-level water management policies and plans. System dynamics modeling facilitates the application of systems thinking and holistic conceptualization of water resources systems.

In recent decades, while system dynamics has been widely used by water resources scholars as a tool for quantitative water resources modeling, it has not typically been utilized to its full capacity for scrutinizing the system's structure to provide insights into potential reasons behind problematic behavioral trends. At the strategic level, emphasis should be placed on trend identification and pattern recognition rather than exact quantitative predictions of dynamic variables. Although the quantitative modeling phase using extensive computer simulations is still very important and needed for policy screening, especially when characterizing complex systems, qualitative system dynamics models can improve understanding of general trends and the root causes of problems, and thus promote sustainable water resources decision making.

In this chapter, tangible water resources examples were presented to illustrate the fundamentals of system dynamics, emphasizing that developing CLDs and SFDs is necessary for identifying causal relationships forming feedback loops within water resources systems. Furthermore, water managers should use the knowledge of reference modes and system archetypes (e.g., Limits to Growth, Fixes that Backfire, Success to the Successful, and Tragedy of the Commons) to gain insights into sustainable solution strategies by recognizing common patterns of dynamic behavior. Compared to other modeling approaches, perhaps the most significant advantage of system dynamics is that when systems are not too complicated, the qualitative modeling tools can help describe the behavior of many variables, even before quantitative (numerical) modeling begins. This characteristic facilitates conceptual or high-level strategic water resources modeling using multi-disciplinary, multi-sectoral, and participatory approaches critical to sustainable water resources planning and management.

2.7. References

Ahmad, S., Simonovic, S.P. (2000). System dynamics modeling of reservoir operations for flood management. *Journal of Computing in Civil Engineering* 14(3), 190–198.

Ahmad, S., Simonovic, S.P. (2004). Spatial system dynamics: a new approach for simulation of water resources systems. *Journal of Computing in Civil Engineering* 18(4), 331-340.

Ahmad, S., Prashar, D. (2010). Evaluating municipal water conservation policies using a dynamic simulation model. *Water Resources Management* 24(13), 3371-3395.

Bagheri, A., Hjorth, P. (2007). A framework for process indicators to monitor for sustainable development: practice to an urban water system. *Environmental Development and Sustainability* 9(2), 143–161.

Bagheri, A., Darijani, M., Asgary, A., Morid, S. (2010). Crisis in urban water systems during the reconstruction period: a system dynamics analysis of alternative policies after the 2003 earthquake in Bam-Iran. *Water Resources Management* 24(11), 2567-2596.

Barlas, Y. (1996). Formal aspects of model validity and validation in system dynamics. *System Dynamics Review* 12 (3), 183-210.

Cai, X., Wang, D. (2006). Calibrating holistic water resources-economic models. *Journal of Water Resources Planning and Management* 132(6), 414–423.

Coyle, G. (2000). Qualitative and quantitative modelling in system dynamics: Some research questions. *System Dynamics Review* 16(3), 225–244.

Creighton, J.L., Langsdale, S. (2009). Analysis of process issues in shared vision planning cases, IWR Report 09-R-05, Institute for Water Resources, U.S. Army Corps of Engineers, Alexandria, Virginia.

Davies, E.G.R., Simonovic, S.P. (2011). Global water resources modeling with an integrated model of the social–economic–environmental system. *Advances in Water Resources* 34, 684-700.

Dhawan, R., O'Connor, M., Borman, M. (2011). The effect of qualitative and quantitative system dynamics training: an experimental investigation. *System Dynamics Review* (2011).doi: 10.1002/sdr.455

Erickson, J.D., Gowdy, J.M. (2000). Resource use, institutions and sustainability: a tale of two Pacific island cultures. *Land Economics* 76(3), 345– 354.

Ford, A. (1996). Testing the Snake River Explorer. *System Dynamics Review* 12(4), 305-329.

Ford, A. (1999). Modeling the environment. Island Press, Washington.

Forrester, J. (1961). Industrial dynamics. MIT Press, Cambridge.

Forrester, J. (1975). Industrial dynamics—a response to Ansoff and Slevin. In: Collected Papers of Jay W. Forrester. Wright-Allen Press, Cambridge, Massachusetts, pp 151–165.

Forrester, J. (1969). Urban dynamics. MIT Press, Cambridge.

Forrester, J. (2007). System dynamics—the next fifty years. *System Dynamics Review* 23, 359-370.

Gastélum, J.R., Valdés, J.B., Stewart, S. (2009). A decision support system to improve water resources management in the Conchos Basin. *Water Resources Management* 23(8), 1519–1548.

Guo, H.C., Liu, L., Huang, G.H., Fuller, G.A., Zou, R., Yin, Y.Y. (2001). A system dynamics approach for regional environmental planning and management: a study for the Lake Erhai Basin. *Environmental Management* 61(1), 93–111.

Hardin, G. (1968). The tragedy of the commons. *Science* 162, 1243-1248.

Hellström, D., Jeppsson, U., Kärrman, E. (2000). A framework for systems analysis of sustainable urban water management. *Environmental Impact Assessment Review* 20(3), 311–321.

High Performance Systems (1992) Stella II: an introduction to systems thinking. High Performance Systems, Inc., Hanover, New Hampshire.

Hjorth, P., Bagheri, A. (2006). Navigating towards sustainable development: a system dynamics approach. *Futures* 38(1), 74–92.

Homer, J., Oliva, R. (2001). Maps and models in system dynamics: a response to Coyle. *System Dynamics Review* 17(4), 347-356.

Langsdale, S., Beall, A., Carmichael, J., Cohen, S., Forster, C. (2007). An exploration of water resources futures under climate change using system dynamics modeling. *Integrated Assessment* 7 (1), 51-79.

Langsdale, S., Beall, A., Carmichael, J., Cohen, S., Forster, C., Neale, T. (2009). Exploring the implications of climate change on water resources through participatory modeling: case study of the Okanagan Basin, British Columbia. *Journal of Water Resources Planning and Management* 135(5), 373–381.

Leal Neto, A.C., Legey, L.F.L., Gonzalez-Araya, M.C., Jablonski, S. (2006). A system dynamics model for the environmental management of the Sepetiba Bay watershed, Brazil. *Environmental Management* 38(5), 879–888.

Leaver, J.D., Unsworth, C.P. (2006). System dynamics modeling of spring behaviour in the Orakeikorako geothermal field, New Zealand. *Geothermics* 36(2), 101-114.

Loaiciga, H.A. (2004). Analytic game - theoretic approach to ground-water extraction. *Journal of Hydrology* 297, 22-33.

Lowi, M, (1993). Water and power – the politics of a scarce resource in the Jordan River Basin. Cambridge University Press, Cambridge.

Luna-Reyes, L.F., Andersen, D.L. (2003). Collecting and analyzing qualitative data for system dynamics: methods and models. *System Dynamics Review* 19(4), 271-296.

Lund, J.R., Palmer, R.N. (1997). Water resource system modeling for conflict resolution. Water Resources Update, Universities Council on Water Resources 108, 70–82.

Madani, K. (2010) Game theory and water resources. *Journal of Hydrology* 381 (3-4), 225-238.

Madani, K., Dinar, A. (2012). Non-cooperative institutions for sustainable common pool resource management: application to groundwater. *Ecological Economics* 74, 34-45.

Madani, K., Mariño, M.A. (2009). System dynamics analysis for managing Iran's Zayandeh-Rud river basin. *Water Resources Management* 23 (11), 2163-2187.

Madani, K., Rowan, D., Lund, J.R. (2007). The next step in Central Valley flood management: connecting costs and benefits. In: Proceedings of the University Council on Water Resources Annual Conference, Boise, Idaho.

McMahon, G.F., Farmer, M.C. (2004). Reallocation of federal multipurpose reservoirs: principles, policy, and practice. *Journal of Water Resources Planning and Management* 130(3), 187-197.

Meadows, D.H., Meadows, D.L., Randers, J., Behrens, W.W. (1972). The limits to growth. Universe Books, New York.

Mirchi, A., Watkins, D.W. Jr, Madani, K. (2010) Modeling for watershed planning, management and decision making. In: Vaughn JC (ed) Watersheds management, restoration and environmental impact. Nova Science Publishers, Hauppauge, New York.

Powersim Corporation (1996). Powersim 2.5 reference manual. Powersim Corporation Inc., Herndon, Virginia.

Qaiser, K., Ahmad, S., Johnson, W., Batista, J. (2011). Evaluating the impact of water conservation on fate of outdoor water use: a study in an arid region. *Journal of Environmental Management* 92(8), 2061-2068.

Randers, J. (1980). Guidelines for model conceptualization. In: Randers J (ed) Elements of the system dynamics method. MIT Press: Cambridge, Massachusetts, pp 117–139.

Richardson, G.P. (1986). Problems with causal-loop diagrams. *System Dynamics Review* 2, 158-170.

Richmond, B. (1993). Systems thinking: critical thinking skills for the 1990ies and beyond. *System Dynamics Review* 9(2), 113 – 133.

Richmond, B. (1994). Systems thinking/system dynamics: let's just get on with it. *System Dynamics Review* 10(2/3), 135-57.

Saeed, K. (1998). Defining a problem or constructing a reference mode. In: Proceedings of the 16th International Conference of the System Dynamics Society Quebec '98, Quebec City, Canada.

Sehlke, G., Jacobson, J. (2005). System dynamics modeling of transboundary systems: the Bear River Basin model. *Ground Water* 43(5), 722–730.

Senge, P. (1990). The Fifth Discipline: the art & practice of the learning organization. Doubleday/Currency, New York.

Simonovic, SP. (2009). Managing water resources: methods and tools for a systems approach. UNESCO, Paris and Earthscan James & James, London.

Simonovic, S.P., Ahmad, S. (2005) Computer-based model for flood evacuation emergency planning. *Natural Hazards* 34(1), 25-51.

Simonovic, S.P., Fahmy, H. (1999). A new modeling approach for water resources policy analysis. *Water Resources Research* 35(1), 295–304.

Simonovic, S.P., Li, L. (2003). Methodology for assessment of climate change impacts on large-scale flood protection system. *Journal of Water Resources Planning and Management* 129(5), 361–371.

Simonovic, S.P., Rajasekaram, V. (2004). Integrated analyses of Canada's water resources: a system dynamics approach. *Canadian Water Resources Journal* 29(4), 223–250.

Stave, K.A. (2003). A system dynamics model to facilitate public understanding of water management options in Las Vegas, Nevada. *Environmental Management* 67(4), 303–313.

Sterman, J.D. (2000). Business dynamics, systems thinking and modeling for a complex world. McGraw-Hill, Boston.

Stewart, S., Valdés, J., Gastélum, J., Brookshire, D., Aparicio, J., Hidalgo, J., Velazco, I. (2004). A decision support system for demand management in the Rio Conchos Basin, México. In: Proceedings of hydrology: science & practice for the 21st Century. British Hydrological Society II, pp 487–494.

Tangirala, A.K., Teegavarapu, R.S.V., Ormsbee, L. (2003). Modeling adaptive water quality management strategies using system dynamics simulation. *Environmental Informatics Archives* 1, 245-253.

Tidwell, V.C., Passell, H.D., Conrad, S.H., Thomas, R.P. (2004). System dynamics modeling for community-based water planning: application to the Middle Rio Grande. *Aquatic Sciences* 66, 357–372.

Venkatesan, A.K., Ahmad, S., Johnson, W., Batista, J.R. (2011a). Salinity reduction and energy conservation in direct and indirect potable water reuse. *Desalination* 272 (1-3),120-127.

Venkatesan, A.K., Ahmad, S., Johnson, W., Batista, J.R. (2011b). System dynamics model to forecast salinity load to the Colorado River due to urbanization within the Las Vegas Valley. *Science of the Total Environment* 409, 2616-2625.

Ventana Systems (1996) Vensim user's guide. Ventana Systems Inc., Belmont, Massachusetts.

Vezjak, M., Savsek, T., Stuhler, E.A. (1998). System dynamics of eutrophication processes in lakes. *European Journal of Operational Research* 109(2), 442-451.

Watkins, D.W. Jr, Moser, D.A. (2006). Economic-based optimization of Panama Canal system operations. *Journal of Water Resources Planning and Management* 132(6), 426-438.

Werick, W.J., Whipple, W. (1994). Managing Water for Drought, IWR Report 94-NDS-8, Institute for Water Resources, U.S. Army Corps of Engineers, Alexandria, Virginia.

Winz, I., Brierley, G., Trowsdale, S. (2009). The use of system dynamics simulation in water resource management. *Water Resources Management* 23(7), 1301–1323.

Wolstenholme, E.F. (1999). Qualitative vs. quantitative modelling: the evolving balance. In: Coyle RG, Morecroft JDW (eds) Special issue: system dynamics for policy, strategy and management education. *Journal of the Operational Research Society* 50(4), 354-361.

Wolstenholme, E.F. (2003). Towards the definition and use of a core set of archetypal structures in system dynamics. *System Dynamics Review* 19(1), 7–26.

Wolstenholme, E.F. (2004). Using generic system archetypes to support thinking and modelling. *System Dynamics Review* 20, 341–356.

Wolstenholme, E.F., Coyle, R.G. (1983). The development of system dynamics as a methodology for system description and qualitative analysis. *Journal of the Operational Research Society* 34(7), 569–581.

Xu, Z.X., Takeuchi, K., Ishidaira, H., Qhang, X.W. (2002). Sustainability analysis for Yellow River water resources using the system dynamics approach. *Water Resources Management* 16(3), 239–261.

Chapter 3 - A systems approach to holistic TMDL policy: Case of Lake Allegan, Michigan[3]

3.1. Abstract

Systems thinking can provide insights for developing effective plans to protect environmental integrity of natural systems impacted by human activities. In this study, a system dynamics archetype governing Lake Allegan's eutrophication problem is hypothesized to explain the system's problematic behavior and identify policy leverage points for mitigation. To operationalize the systems thinking concepts, an integrated system dynamics model is developed to simulate the interaction between key socio-economic subsystems and natural processes driving eutrophication. The model is applied to holistically characterize the lake's recovery from its hypereutrophic state and assess a number of proposed TMDL reduction policies, including phosphorus load reductions from point sources and non-point sources. It is shown that, for a TMDL plan to be effective, it should be considered a component of a continuous sustainability process, which considers the functionality of dynamic feedback relationships between socio-economic growth, land use change, and environmental conditions.

[3] This chapter is a reprint of the article: Mirchi, A., Watkins, D.W. Jr., 2012. A systems approach to holistic TMDL policy: Case of Lake Allegan, Michigan. Journal of Water Resources Planning and Management, doi: 10.1061/(ASCE)WR.1943-5452.0000292. Reprinted with permission from the American Society of Civil Engineers (ASCE).

3.2. Introduction

Extensive socio-economic development without proper regard for environmental integrity and ecosystem resilience can result in unsustainable development (Bishop, 1993; Arrow et al., 1995). For instance, nutrient enrichment of water bodies causing loss of biodiversity may be symptomatic of a human-induced environmental degradation known as cultural eutrophication (Cooke et al., 1993; Effler et al., 2002). A fundamental premise for this study is that population growth, affluence, and technology are three critical factors that drive anthropogenic environmental change (Ehrlich and Holdren, 1971). Important feedback relationships exist between socio-economic growth and environmental damage, although environmental degradation may not be severe enough to stop the growth process within the typical planning timeframes (Arrow et al., 1995). Ehrlich and Holdren's theorem provides a contextual foundation for understanding the potential impacts of the linkage between socio-economic dynamics and lake phosphorus (P) concentration on the success or failure of human intervention to maintain and/or improve environmental integrity.

The Federal Water Pollution Control Act Amendment of 1972, commonly known as the Clean Water Act (CWA), introduced the concept of Total Maximum Daily Load (TMDL) to guide water quality management in the US (FWPCA, 1972). TMDL is a written plan prescribing the maximum amount of pollutant loads that a water body can receive without violating predefined water quality standards. The CWA amendment of 2002 requires each state to identify the main pollutant(s) impacting the impaired water bodies, and to meet necessary water quality standards using TMDLs, i.e., quantifying allowable levels of load allocation and waste-load allocation (FWPCA, 2002). In this context, load allocation refers to pollutant loads from non-point sources (NPS) along with natural background inputs, whereas waste-load allocation is the allowable pollutant load from point sources (PS). Furthermore, a margin of safety is considered to compensate for lack of knowledge as to relationship between pollutant inputs and water quality (FWPCA, 2002). Comprehensive TMDL plans are very challenging due to economic, legal, and political aspects of TMDL implementation. Nevertheless, inclusion of both PSs and NPSs of pollution rather than only applying restrictions on end-of-pipe discharges, partly addresses the concern that TMDL plans overemphasize the contribution of PSs without adequately accounting for nutrient loads from NPSs (Boyd, 2000).

TMDL plans have been widely implemented in the US (e.g., Benham et al., 2008). However, many TMDL studies do not address the dynamic socio-economic setting of the problem, as this has neither been mandated by law nor adequately addressed in the TMDL literature. While many TMDL studies have identified NPS pollution to be a major driver of eutrophication, land use change driven by socio-economic development is often not accounted for when proposing nutrient reduction levels to reach the prescribed water quality targets. This shortcoming is particularly critical because the NPS component of TMDL plans is typically founded on average nutrient loads that are calculated for different land use categories by conducting limited monitoring programs and/or using literature values. Although physical aspects of water quality problems have been the subject of extensive research, including modeling and quantitative analyses addressing spatiotemporal variability and uncertainty (e.g., Doerr et al., 1996; Haven and Schelske, 2001; Vondracek et al., 2003; Shirmohammadi et al., 2006; Canale et al., 2010), little attention has been paid to the interplay of socio-economic and land use dynamics and their implications for water quality problems (Vergura and Jones, 2000).

Adopting a systems thinking approach and applying system dynamics (SD) modeling (Forrester, 1961 and 1969; Senge, 1990; Ford, 1999; Sterman, 2000; Simonovic, 2009) can facilitate holistic understanding of TMDL problems, and may thus guide the formulation of long-term TMDL plans. TMDL studies are inherently multidisciplinary as they should characterize a host of physical, ecological, and biogeochemical processes. Socio-economic characteristics of the study area add yet another piece to the puzzle. SD provides useful qualitative and quantitative tools for characterizing different feedback loops that govern water resources systems (Simonovic and Fahmy, 1999; Hjorth and Bagheri, 2006; Madani and Mariño, 2009). Winz et al. (2009) reviewed water resources SD literature to illustrate its use in integrated water resources management. Researchers have used SD for simulating biophysical processes to develop decision support tools for water quality management (Vezjak et al., 1998). Furthermore, SD has been used in integrated analysis of water quality problems and environmental deterioration by linking regional-scale physical and socio-economic subsystems to water quality models (Guo et al., 2001; Leal Neto et al., 2006).

Failing to recognize and appropriately account for the feedbacks between socio-economic and natural sub-systems may lead to inadequate nutrient reduction plans in the long-term. Understanding causal structures driving the system's long-run behavior, and using this information to formulate integrated TMDL plans provides opportunities for holistic policy making to direct water quality management (Boyd, 2000). Hence, there is a need for a framework to conceptualize the eutrophication problem, and quantify the processes to help describe the system's long-run behavior to decision makers and stakeholders, increasing support for the proposed nutrient reduction measures. The eutrophication problem of Lake Allegan, Michigan, is used as a case study to demonstrate the potential long-term role of socio-economic factors in nutrient loading of water bodies. The problem is explained using the "Growth and Underinvestment" system archetype (Senge, 1990), illustrating the need for making sufficient investment in maintaining environmental integrity to allow for sustained economic growth.

3.3. Lake Allegan's Eutrophication Problem

Lake Allegan is a 647.5-hectare impoundment formed by the Calkins Dam on the Kalamazoo River in Southwestern Michigan, USA (Figure 3.1). Located at the outlet of a 401,500-hectare drainage area within the Kalamazoo River watershed, the lake has a volume of approximately 21.22 million m^3 with a mean and maximum depth of about 3.35m and 6.1m, respectively, and a short mean hydraulic retention time of seven days (US EPA, 1975). The impoundment has altered the Kalamazoo River sediment transport pattern and water quality by trapping most sediment and associated pollutants (MDNR, 1987). Lake Allegan receives water from the Kalamazoo River with a mean flow of about 35.1 m^3/s, and from Dumont Creek and other minor tributaries, and direct drainage from surrounding areas, collectively increasing the lake's mean total inflow to about 36 m^3/s, which eventually discharges into Lake Michigan (US EPA, 1975). In 1978, agriculture was the dominant land use type covering about 50% of the lake's watershed area, followed by 34% forest/open land. The rest of the watershed land use was characterized by 8% urban and built-up and 7% water/wetland.

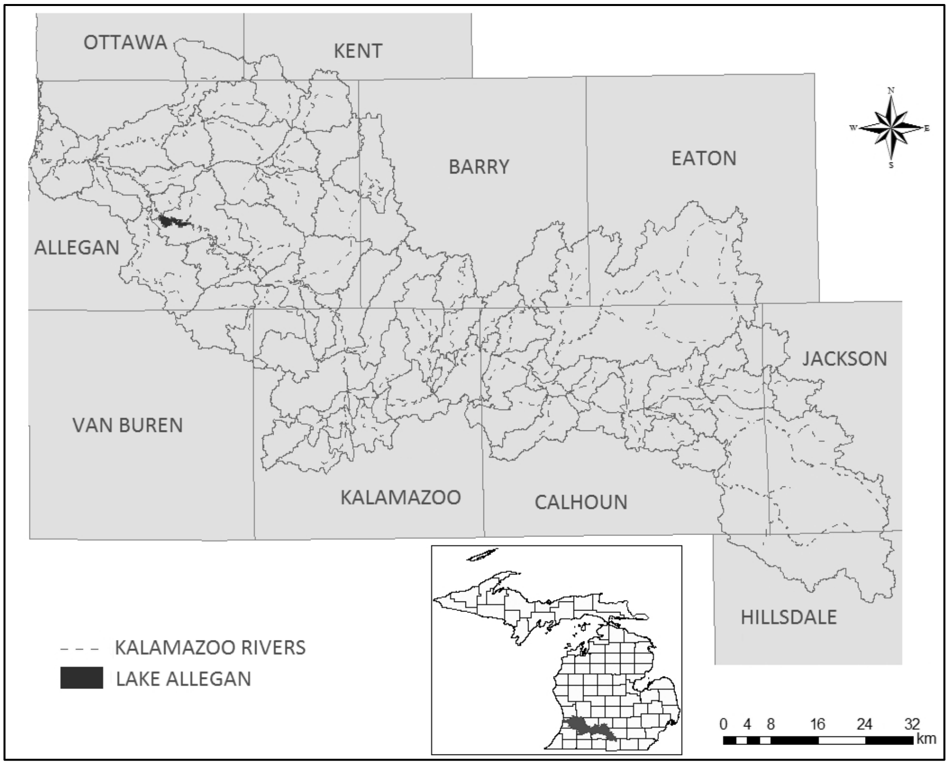


Figure 3.1. Lake Allegan's drainage area (Courtesy of Meredith Ballard Labeau, Source of data: Michigan Geospatial Data Library).

In the early 1970s Lake Allegan was classified as hypereutrophic due to high concentration of total phosphorus (TP) ranging from 92 µg/l to 180 µg/l, with a mean concentration of 123 µg/l and a median of 111 µg/l (US EPA, 1975). Periodic monitoring campaigns conducted by Michigan Department of Environmental Quality (MDEQ) suggest that the lake has somewhat recovered from its hypereutrophic state (MDEQ, 1999), likely due to decreased P-loading associated with the high-P detergent ban in 1977 (Hartig and Horvath, 1982). However, the lake's average TP concentration in the late 1990's was about 96 µg/l, which was still high enough for the hypereutrophic state to prevail (Wuycheck, 1998). As such, observed problems such as periodic algal blooms, excessive turbidity, low dissolved oxygen levels, and imbalance in fish populations are largely attributable to high TP concentration (Wuycheck, 1998).

Since 1998 P reduction plans have been implemented in the Lake Allegan watershed to reduce the growing season (April-September) TP concentration of the lake to 60 µg/l by 2015 (Heaton, 2001; KRLATIC, 2002). Kieser and Associates (2001) conducted a spatiotemporal analysis of the water quality throughout the Lake Allegan drainage area and recommended that PS loadings to the lake be cut back by 23% and that NPS loadings be reduced by 50% (Kieser and Associates, 2001). Table 3.1 presents the breakdown of the proposed TMDL components for Lake Allegan during growing season (April-September). While PS discharges have effectively complied with the prescribed waste-load allocation throughout the decade following the implementation of Lake Allegan's TMDL plan, the lake's recovery appears to be taking longer than anticipated.

Table 3.1.Breakdown of TMDL components for Lake Allegan (Kieser and Associates 2001).

TMDL	**1998**		**Goal**	
(kgs / month)	**Apr-Jun**	**Jul-Sep**	**Apr-Jun**	**Jul-Sep**
Waste load	3,946	3,946	3,946	3,039
Load allocation	7,810	3,690	4,445	1,854
Margin of safety	N/A		45	23
Total	11,756	7,636	8,437	4,916

3.4. System Dynamics and Archetypes

SD is a framework for exploring the behavior of complex systems (Forrester, 1961 and 1969). The approach enables analysts to use both quantitative and qualitative data to characterize feedback loops within large, multi-sector systems. Among the key qualitative modeling tools offered by SD are Causal Loop Diagrams (CLDs), which are a combination of words and arrows with appropriate polarity, showing positive and/or negative causalities. A combination of positive and negative causal relationships may form balancing or reinforcing loops. While balancing feedback loops attempt to reduce the discrepancy between the system's current and desired state (goal seeking behavior), reinforcing feedback loops often characterize continuous growth or decline. Using CLDs, Stock and Flow Diagrams (SFD) are developed to illustrate and simulate accumulation and/or depletion of stock(s) and flow of quantities in the system (Ford, 1999; Sterman, 2000; Simonovic, 2009). System archetypes are generic system structures that are part of

SD's suite of diagnostic and/or descriptive qualitative tools, applicable to classes of problems that share one or more modes of dynamic behavior (Senge, 1990). Some common archetypes include Limits to Growth, Success to the Successful, Growth and Underinvestment, Fixes that Backfire, Tragedy of the Commons, and Escalation (Senge, 1990; Braun, 2002).

Lake Allegan's eutrophication problem can be explained using the Growth and Underinvestment (G&U) archetype. Building on the Limits to Growth (Meadows, 1972), G&U addresses the system's need to continuously invest in factors that tend to limit its growth (e.g., environmental degradation). The archetype's CLD comprises three major feedback loops (Figure 3.2). The left reinforcing loop captures the system's socio-economic dynamics. Growing population creates new business opportunities, increasing the number of proprietors which leads to increased employment. More job opportunities triggers a growth in employment rate, increasing regional income and, in the case of Lake Allegan, income per capita. It should be noted that growth and/or decline of employment depends also on exogenous economic factors such as bank loans, interest rates, and economic upturns and downturns, among others. Furthermore, the magnitude of population growth/decline inversely affects employment rate. Overall, better employment and income opportunities make the region a more attractive place to live in, which with a time lag may result in further population growth (Figure 3.3).

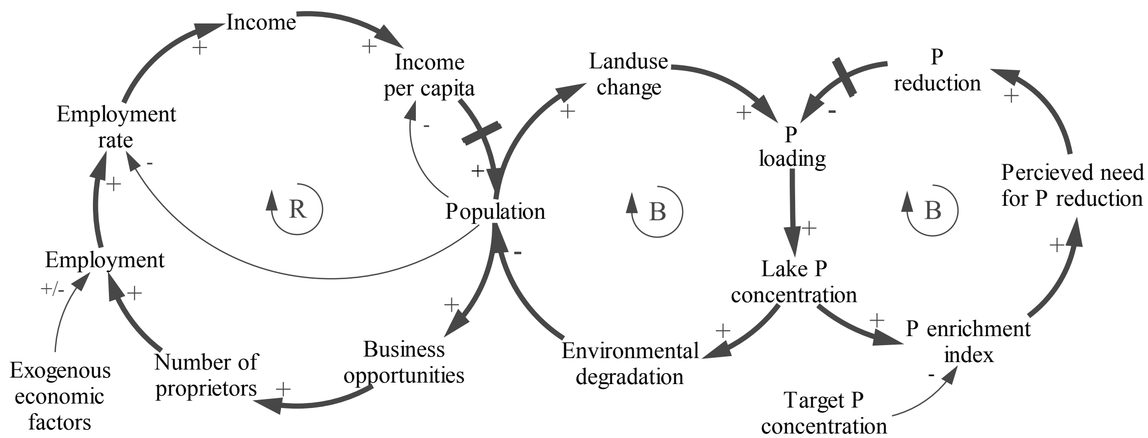


Figure 3.2. CLD of Lake Allegan's G&U archetype. Double bars indicate presence of potential delays.

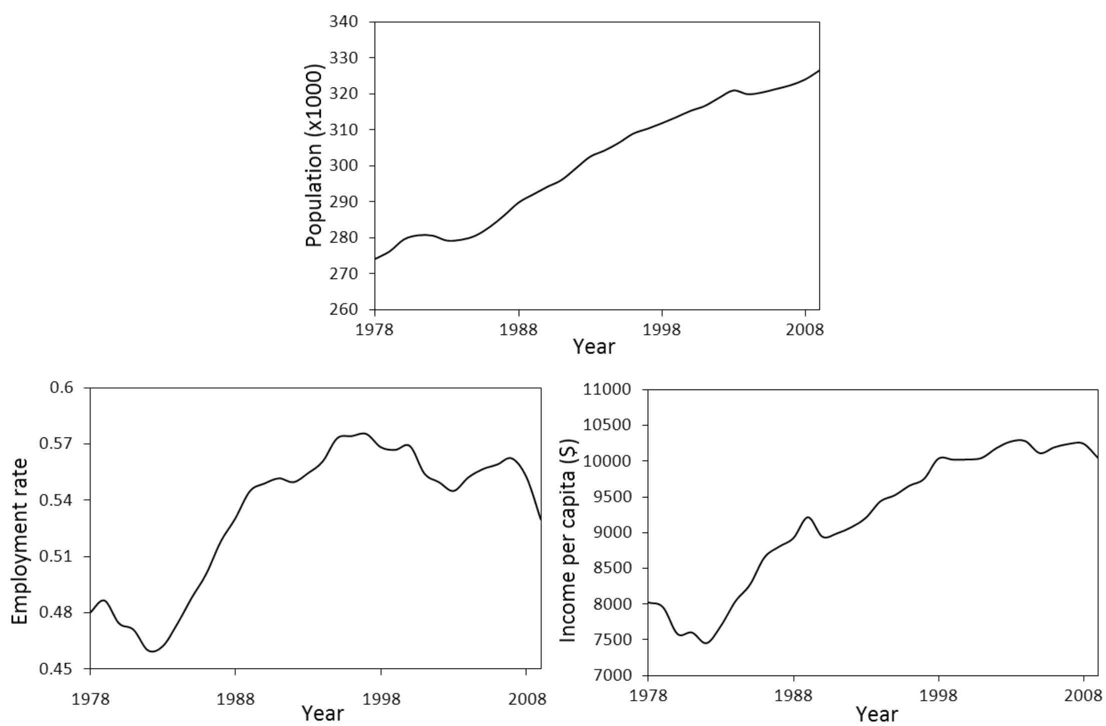


Figure 3.3. Employment rate, income per capita, and population in the Kalamazoo-Portage area (Source of data: US BEA, 2011).

The socio-economic loop drives the anthropogenic environmental degradation depicted in the middle loop. In this balancing loop, population increase brings about land use change, in turn affecting P loadings from different land use types. In the case of Lake Allegan, land use change will likely increase P input to the lake. Eventually, as nutrient enrichment of the lake continues, symptoms of environmental degradation emerge, forming a negative causal relationship with population. While eutrophication may not have an immediate impact on upstream land owners, it directly impacts property owners around the lake, which may lower the area's attractiveness. The second balancing loop (right) characterizes the measures taken to mitigate environmental degradation, representing investment in environment. As the lake's P concentration rises, the discrepancy between observed and target P concentrations leads to increased perception of the need for P reduction. P reduction plans such as TMDLs may be implemented to reduce P loading from PSs and NPSs, helping the lake meet the prescribed water quality target. However, in many cases, the P reduction measures will not immediately rehabilitate the water body (Hamilton, 2012).

3.5. Model and Data Inputs

Vensim Professional 5 (Ventana Systems, 2010), one of several software packages available for SD modeling, was used to develop and run the Lake Allegan model. A generalized SFD of the problem, including various components of the G&U archetype, is shown in Figure 3.4. Population and the lake TP concentration were modeled as stock variables, whereas economic drivers, land use change, and physical processes governing P inputs are characterized using auxiliary variables and/or equations. Nutrient reduction measures are shown to target PSs and NPSs, distinctively. When simulating existing conditions, population, income per capita, and employment were input to the model as time series data obtained for the period of 1978-2009 from the U.S. Bureau of Economic Analysis (US BEA, 2011). Additionally, Bureau of Labor Statistics' Customer Price Index calculator (US BLS, 2009) was used to obtain inflation-adjusted income in 1978 dollars. Explicit representation of the link between all different intermediate drivers introduces multi-colinearity problems. Thus, some socio-economic variables, such as business opportunities and number of proprietors, have been included to qualitatively capture intermediate drivers that lead to an increase in employment as a result of population growth. While it might be difficult to actually measure the potential increase in business opportunities associated with more population, there is a strong correlation between population and number of proprietors ($r^2 = 0.887$).

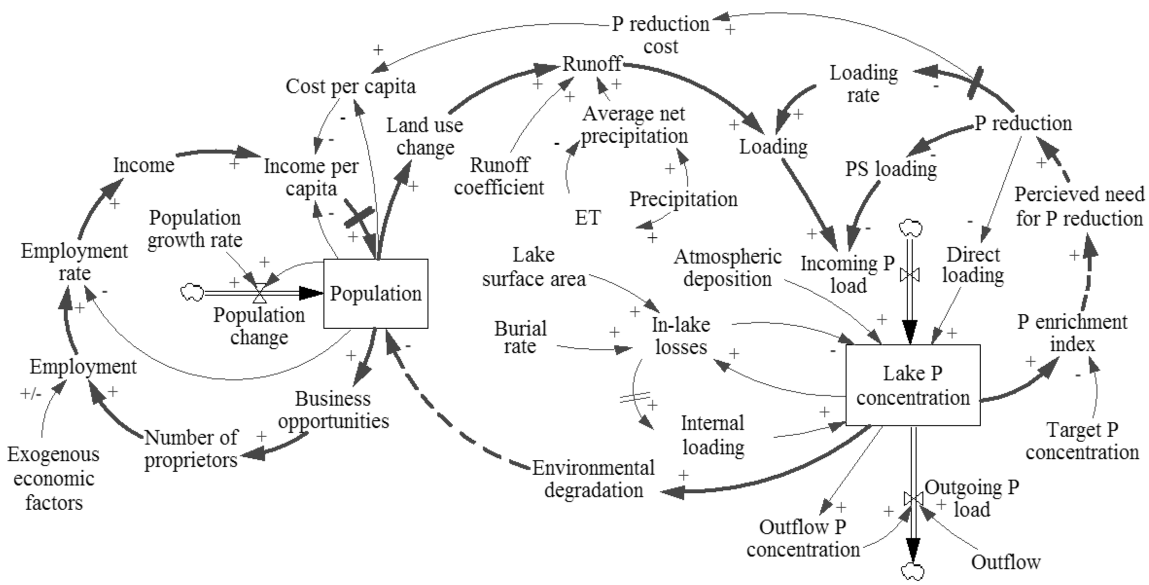


Figure 3.4. Generalized SFD of the Lake Allegan SD model. Signs denote if changing a variable will trigger a change in the other variable in the same (+) or opposite (-) direction. Dashed links have not been quantified. Double bars indicate potential delays.

Using the Anderson level I land use classification scheme (Anderson et al., 1976), four major land use types of urban/built up, agricultural, forest/open land, and water/wetland were identified in the Kalamazoo River watershed. In order to characterize the change between these land use types throughout the simulation period, land cover/use data for 1978 from the Michigan GIS data library were used along with land cover/use data for 1996 and 2001 from Coastal Change Analysis Program (C-CAP), and output layers from the Purdue Land Transformation Model (Pijanowski et al., 2002) for years 2005, 2010, and 2015. The C-CAP land cover/use data and LTM outputs were cross-walked to Anderson level I land use classes to provide a consistent basis for detecting changes in major land use types in the watershed. Unlike changes in forest/open land and water/wetland, changes over time in urban/built up (r_{ur}^2= 0.66) and agricultural land (r_{ag}^2= 0.88) were found to be statistically significant. The linear models best fitting the latter land use types were used to generate land use change time series, providing land use data for regression analyses. To satisfy land-accounting requirements, calculated areas for urban/built up and agricultural lands were summed, and the result was subtracted from total watershed area. The remaining area was divided between water/wetland and forest/open land based on their respective areas in 1978. Table 3.2 summarizes the results of linear regressions between various components of the socio-economic loop, as well as between population and urban/built up and agricultural land use types to quantitatively establish the links illustrated in the SFD.

Table 3.2. Summary of linear regression results.

Dependent Variable	Independent Variable	Equation (Y= aX+b)		P-value*		Adjusted r^2
		Coefficient	Intercept	Coefficient	Intercept	
Population	Income per capita	15.6	155472	6.15E-10	2.56E-9	0.88
Employment†	Population	1.38	-249471	3.12E-12	6.94E-9	0.94
Income per capita	Employment	0.04	2513	2.61E-11	2.10E-5	0.93
Urban/built up land	Population	0.38	-66060	3.94E-10	8.98E-7	0.90
Agricultural land	Population	-0.42	378742	4.40E-11	4.15E-19	0.92

* P-value represents the chance of observing the coefficient/intercept value when the actual value is zero (no effect).

† The equation is valid for the population range of 250,000 to 450,000 people.

Other data requirements of the Lake Allegan SD model include average hydrologic conditions, P loading characteristics of PSs and NPSs, and the lake's physiographic attributes that contribute to in-lake processes. Average quarterly precipitation data for Southwestern Michigan were obtained from National Oceanic and Atmospheric Administration (NOAA). Net average annual precipitation was calculated as the difference between precipitation and evapotranspiration, which was assumed to be 50% of precipitation (Webster et al., 1995). Surface runoff is assumed to be the key transport mechanism for P loads from land-based NPSs. Runoff coefficients for different land use types within Lake Allegan drainage area estimated from pervious and/or imperviousness of the terrain were applied to calculate the runoff (*eq.3.1*) (Kieser and Associates, 2001). Additionally, P loading rates estimated for the neighboring Rouge River watershed in Southeast, MI, were applied to the Lake Allegan SD model. These loading rates were determined using representative dry-weather field samples of storm water pollutants collected from over fifty sampling stations (Cave et al., 1996). Reported values of sediment burial rate in the Kalamazoo River system are in the range of $1.4x10^{-7}$ to $4.0x10^{-7}$ m/min (MDNR, 1987). The upper bound of this range was used in *eq.3.2* to characterize nutrient loss due to in-lake processes. NPS loads from immediate drainage areas and atmospheric deposition have collectively been estimated at 154 kg/month for April through September (Heaton, 2001). Finally, discharge data was obtained from Fennville USGS gauging station at Lake Allegan's outlet, and outflow P concentration was assumed to be equal to the P concentration of the completely mixed lake. Table 3.3

presents the runoff coefficients and P loading rates for different land cover/use types. Using the described input data, a P mass balance was formulated for Lake Allegan as shown in *eq.3.3*.

Table 3.3- Runoff coefficients and P loading rates.

Land use type	**Runoff coefficient***	**Loading rate (mg/L)****
Agriculture	0.042	0.37
Urban/Built up	0.232	0.45
Forest/Open land	0.042	0.11
Water/Wetland	0.465	0.08

* Source: Kieser & Associates (2001)

**Source: Cave et al. (1996)

$$RLU_i = RC_i \times NP \times ALU_i \qquad (eq.3.1)$$

$$IL = LP(t) \times LV \times BR \qquad (eq.3.2)$$

$$LP(t) = \int_{t_0}^{t_n} [(\sum_{i=1}^{4} LU_i\, P) + AD + DL - IL - OL]\, dt + LP(t_0) \qquad (eq.3.3)$$

where RLU_i = runoff from land use i (4 different land use types); RC_i = runoff coefficient for land use i; NP = net monthly precipitation; ALU_i = area of land use i; IL = in-lake loss; $LP(t)$ = lake's P concentration at time t; LV= lake's volume; BR= burial rate; $LU_i\,P$ = P loading from land use i; AD = atmospheric deposition; DL = loading from direct drainage; OL = outflow loss.

3.6. Model Verification

The Lake Allegan model's performance was evaluated using two sets of tests available for verification of SD models, namely model structure and behavior tests. First, tests of model structure were performed including structure verification, parameter verification, extreme conditions, and dimensional consistency (Forrester and Senge, 1980; Sterman, 2000). In the structure verification test, the developed CLD and SFD were analyzed to ensure that the proposed underlying structure (i.e., interactions between the socio-economic loop, land use change, and the lake's P input and output) accounts for the main processes driving the lake's TP concentration. The purpose of parameter verification test is to verify that parameter values are consistent with observations of the real system. The extreme conditions test was performed to investigate the impact of extremely large or small nutrient inputs on the TP concentration. Finally, dimensional consistency of the model's auxiliary equations and stocks was examined.

In the second step of evaluating model performance, tests of model behavior were carried out. Figure 3.5 shows the plot of simulated growing season TP concentration, along with the 95% confidence bound and the lake's average growing season TP concentration (dots) measured during the period of 1998 through 2008 (Kieser and Associates, 2011). This figure has been produced based on a Monte Carlo simulation approach for accommodating the effects of parameter variability in TMDL studies (Gelda et al., 2001). The most critical parameters impacting Lake Allegan's P concentration are runoff coefficients and P loadings from different land use types, especially from agricultural and urban and built up areas. Therefore, a Monte Carlo simulation was performed by running thousand simulations to sample runoff coefficients between -10%

and +10% of the applied coefficients, and P loads between -20% and +20% of the best-estimate average loads.

A series of behavior reproduction tests including symptom generation, frequency generation, behavior characteristic, and behavior sensitivity tests were done to evaluate how well the model generates observed behavior (Forrester and Senge, 1980; Sterman, 2000; Simonovic, 2009). The model generates the primary symptom of eutrophication reasonably well by showing that the peak of TP concentration occurs around mid-growing season. Likewise, the model passes the frequency generation test by capturing the seasonality of fluctuations in the TP concentration. The cyclical behavior of the simulated TP concentration can be explained by considering the cyclical pattern of precipitation and outflow time series (Figure 3.6). In addition, the variability and uncertainties due to timing and frequency of TP measurements are a source of mismatch between simulated and observed values. As illustrated in Figures 3.5 and 3.6, the simulated TP mimics the local hydrologic behavior (i.e., peaks of precipitation and outflow occurring during the growing season). Finally, sensitivity analysis was performed by applying plausible shifts to model parameters such as runoff coefficients and nutrient loading rates for different land use types to ensure the generated behavior does not change dramatically.

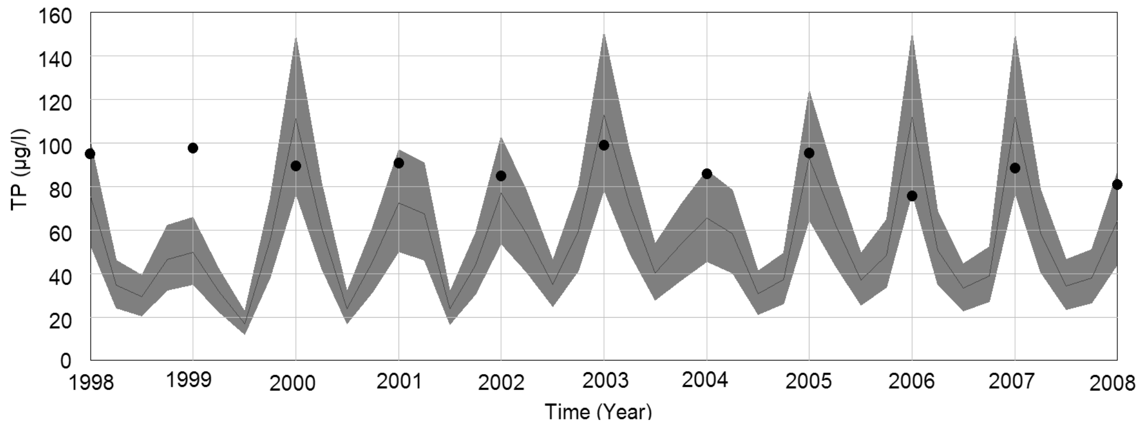


Figure 3.5. Simulation of Lake Allegan's average seasonal TP with effects of parameter variability (mean and 95% confidence interval), and measured growing season TP.

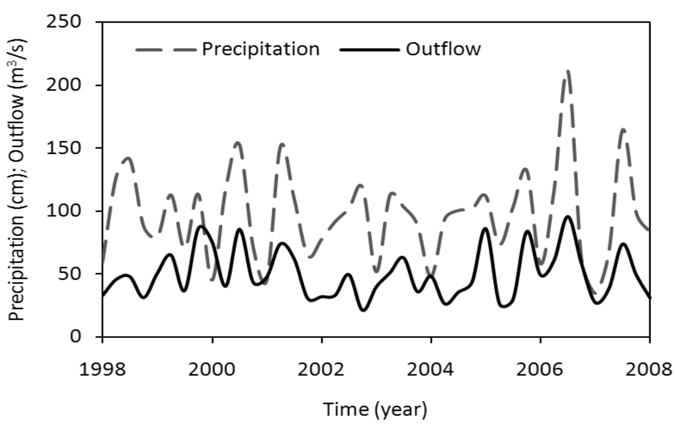


Figure 3.6. Precipitation and outflow time series.

3.7. Scenario Simulation Results

Ten different scenarios were simulated (Table 3.4). The first four scenarios illustrate the effect of socioeconomic dynamics on the success of Lake Allegan's TMDL program until 2008. First, a static condition was simulated in which different land use types were kept constant at 1978 levels and the TMDL program was absent. In the second scenario, a static condition was simulated assuming that TMDL plan is implemented. Likewise, the third and fourth scenarios represent dynamic conditions with or without the TMDL plan assuming that different land use types change in response to socio-economic growth. Furthermore, using best estimates of historical drivers, the model was run under six additional scenarios of socioeconomic dynamics and TMDL implementation to project the lake's trophic state beyond 2008. The scenarios include no population growth with TMDL plan; moderate socioeconomic growth without the TMDL plan; slow, moderate, and moderate socioeconomic growth with the TMDL plan; and moderate socioeconomic growth with the TMDL plan and internal P loading. For the no population growth scenario, the population was kept constant at the 2009 level. A population growth rate of 1% was used to characterize rapid socioeconomic growth. Slow socioeconomic growth was simulated using a population growth rate of 0.1%. The average population growth rate for the Kalamazoo-Portage area (0.56%) was used for simulating moderate socioeconomic dynamics. Finally, due to strong indications of internal P loading from sediments in mid growing season, which may delay the success of the TMDL plan (KRLATIC, 2002) an internal loading scenario was simulated.

Table 3.4. Simulation scenarios.

Scenario	Description	Abbreviation
S1: Static condition without TMDL	Land use fixed at 1978 levels	SC without TMDL
S2: Static condition with TMDL	Land use fixed at 1978 levels	SC with TMDL
S3: Dynamic condition without TMDL	Land use changed due to socioeconomic growth	DC without TMDL
S4: Dynamic condition with TMDL	Land use changed due to socioeconomic growth	DC with TMDL
S5: No population growth without TMDL	Population fixed at 2008 level	NPG without TMDL
S6: Moderate population growth without TMDL	Socioeconomic loop driven by 0.56% population growth	MPG without TMDL
S7: Slow population growth with TMDL	Socioeconomic loop driven by 0.1% population growth	SPG with TMDL
S8: Rapid population growth with TMDL	Socioeconomic loop driven by 1% population growth	RPG with TMDL
S9: Moderate population growth with TMDL	Socioeconomic loop driven by 0.56% population growth	MPG with TMDL
S10: Moderate population growth with TMDL and internal P loading	Same as scenario 9, additional linearly decreasing load applied to the period of 2010-2020	MPG with TMDL & IL

Simulation results for the first four scenarios (S1 through S4) suggest that land use change associated with socioeconomic growth leads to greater lake P concentrations, due to higher incoming loads from former agricultural lands that are transformed to urban and built up (Figure 3.7). Thus, failing to incorporate the noted long-term P load dynamics in the TMDL plan may cause an overestimation of reduction in lake P concentration. For example, a thirty-year simulation period (1978-2008) for Lake Allegan shows that lake P concentrations for the case of dynamic conditions with the TMDL plan may be more than 10% greater than under static conditions. Furthermore, reducing P input from NPSs is a major contributor to effectiveness of Lake Allegan TMDL plan, given the achieved success of PSs in controlling P loading, and the large share of NPS pollution. Implementation of the TMDL plan assumes full PS and NPS compliance with specified load allocation and waste load allocation. However, if P reduction from NPSs is not achieved, the variation of lake's P concentration will be similar to the case of dynamic conditions without the TMDL plan, in which case it is not surprising to see a discrepancy of over 20% between expected and observed P concentrations.

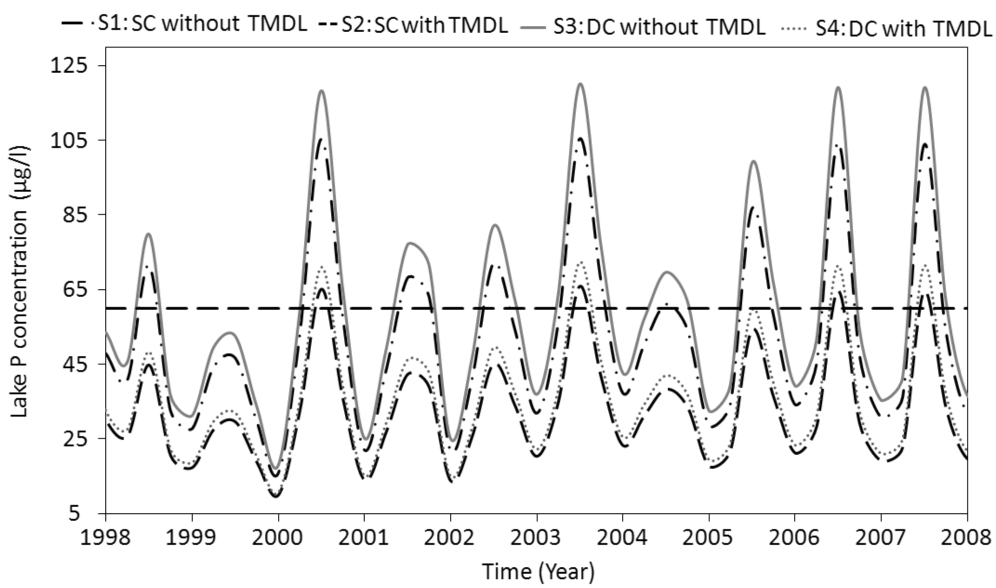


Figure 3.7. Simulation results for static and dynamic conditions with/without the TMDL plan (S1 through S4). Note the target TP concentration shown with horizontal dashed line.

The results for scenarios S5 through S8 demonstrate the importance of a continuous TMDL implementation program, and potentially the need for more aggressive plans. The year-to-year variability results from applying historical hydrology to the future period. As shown in Figure 3.8, under scenario S6 the system frequently violates the specified growing season TP target (dashed line). In contrast, the target TP concentration is met under scenario S5, which restricts socioeconomic growth (Figure 3.8). Therefore, the TMDL plan can support socioeconomic growth by ensuring that environmental degradation is managed adequately. Although simulated growing season TP concentrations for scenarios S7 and S8 occasionally violate the criterion, depending on the amount of precipitation, they portray significant reduction in TP concentrations as compared to scenario S6, indicating the general effectiveness of the TMDL plan (Figure 3.9). Additionally, the results for scenarios S7 and S8 illustrate the need for adapting the long-term TMDL plan to local socioeconomic growth, as rapid growth may require implementation of more aggressive TMDL plans. This is illustrated in Figure 3.9 which shows larger projected growing season TP concentrations for rapid socioeconomic growth as compared to slow growth (S7 and S8) during the period of 2019-2028.

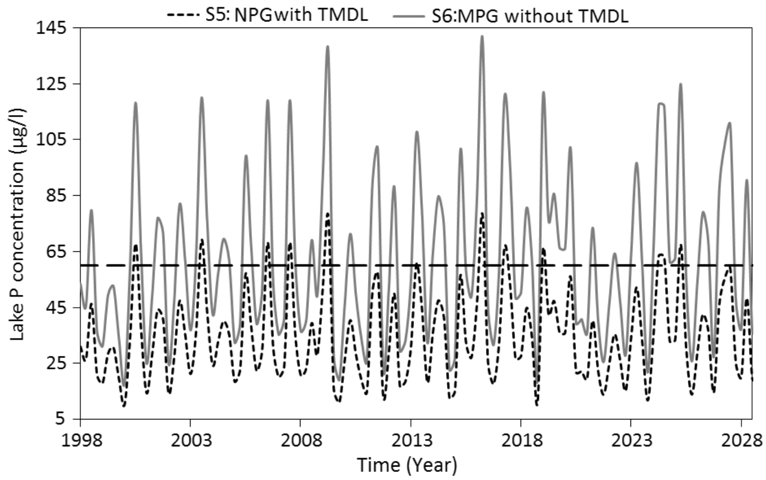


Figure 3.8. Simulation results for scenarios S5 and S6. Note the target TP concentration shown with horizontal dashed line.

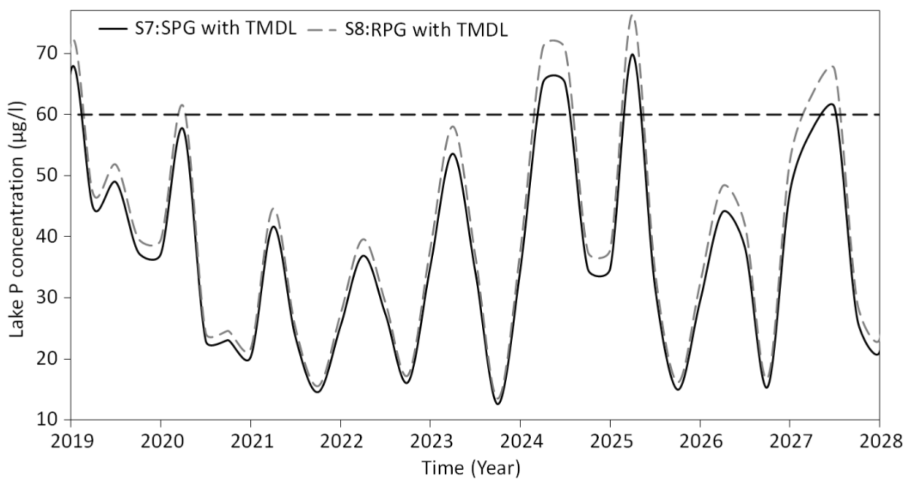


Figure 3.9. Simulation results for scenarios S7 and S8. Note the target TP concentration shown with horizontal dashed line.

Figure 3.10 presents the results of scenarios 9 and 10. While estimating how long it will take for the lake to recover from its hypereutrophic state is beyond the scope of this chapter, a simple scenario-based simulation of the lake's internal P loading illustrates how this phenomenon may cause delayed recovery. For this scenario, a load equivalent to 10μg/l (KRLATIC, 2002) was applied to first half of the growing season in 2010, decreasing by 1 μg/l until 2020. It is assumed that continued TMDL implementation can gradually reduce internal P loading as the P in the sediment pool depletes over time due to flushing of the lake.

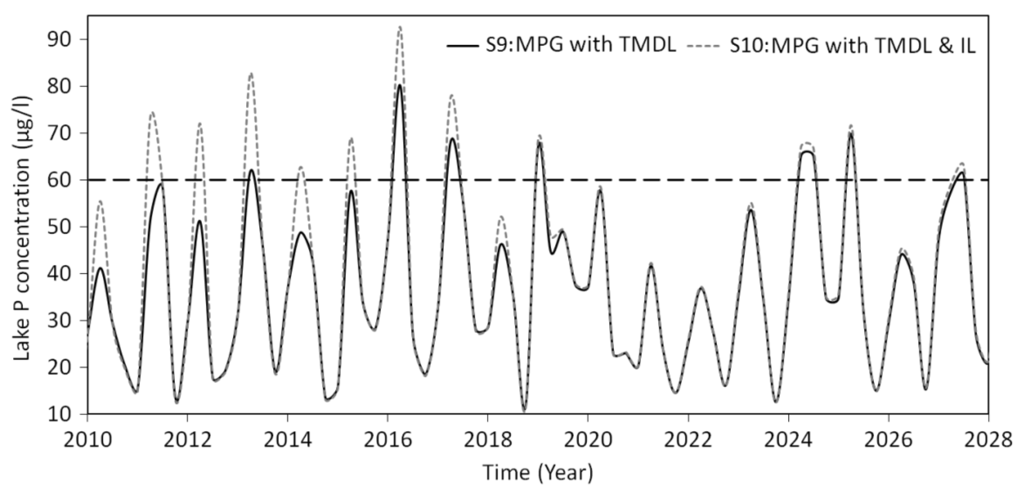


Figure 3.10. Simulation results for scenarios S9 and S10. Note the target TP concentration shown with horizontal dashed line.

To demonstrate the need for continued investment in the TMDL program, the average cost of implementing some typical stormwater management best management practices (BMPs) for reducing TP from urban and agricultural areas was projected.

Significant variability in the cost of BMPs and their effectiveness in removing TP from urban rainwater and farmland runoff has been discussed in the literature (e.g., Withers and Jarvis 1998; Weiss et al. 2007). Ryding and Rast (1989) provide an estimate of cost-effectiveness of BMPs for removing NPS P loads from agricultural areas in the Lake Erie Basin. They have reported that increased winter crop cover, spring tilling, improved pasture management, critical source area protection, gradient terracing, and grass waterways may provide 40% P removal at an average cost of 174 $/kg P saved. This value was used as the cost of TMDL implementation in agricultural areas. Furthermore, Kieser and Associates (2005) report BMP costs and effectiveness for implementation in Southwestern Michigan. The cost of a combination of wet retention ponds, vegetated swales, and bioretention basins to provide 50% P removal in urban areas was estimated at 316 $/kg P saved. Using these estimates, the cost of implementing BMPs to meet water quality standards was projected under slow, moderate, and rapid socioeconomic growth scenarios. Figure 3.11 shows the cumulative required investment in P removal, showing the sensitivity of investment to the level of socioeconomic growth, which in the long-run may contribute to lack of funds for implementing an effective TMDL program.

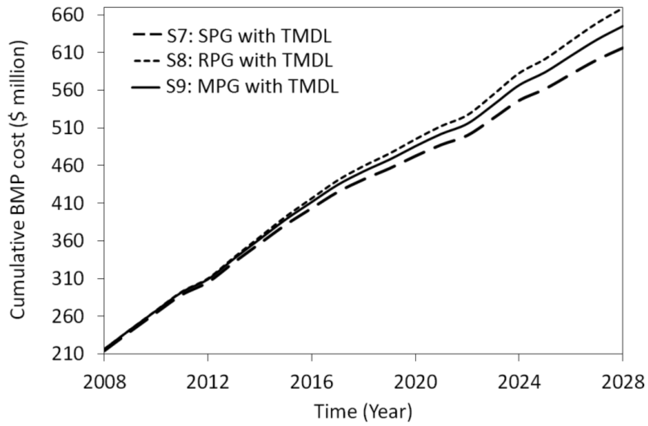


Figure 3.11. Projected cumulative investment in P removal under slow, moderate, and rapid population growth (S7, S8, and S9).

3.8. Discussion

Bagheri and Hjorth (2007) view sustainable development as a process involving evolutionary changes, with no definable end point, in which essential feedback loops in a given system are kept healthy and functional. Using this notion of sustainable development, as illustrated by the G&U archetype, implementation of environmental protection and/or restoration plans should be an ongoing, evolving process within the broader sustainable development scheme. As the archetype shows, continued investment is necessary to restrain nutrient loads to a level commensurate with the environmental capacity for pollutant assimilation. Therefore, in order for a TMDL plan to be effective, it should be considered a necessary component of the sustainability process, which helps maintain health and functionality of feedback relationships between socioeconomic growth, land use change, and environmental condition. As such, it is prudent to monitor local socio-economic changes and associated land use transformations to apply necessary adjustments to the allowable load and waste load allocations, as well as a TMDL's margin of safety.

The presented results are based on the fundamental assumption that the proposed TMDL plan for Lake Allegan is fully implemented by PSs and NPSs. In actuality, however, it has proven difficult for NPSs to comply with the designated load allocations. In the Lake Allegan's TMDL plan, P reduction from NPSs is to be achieved through voluntary cutbacks by participants, mainly from the agriculture sector (KRLATIC, 2002). While some level of public participation and voluntary cooperation are important for the success of management plans, applying appropriate regulations such as banning high-P agricultural fertilizers may be necessary to avoid Tragedy of the Commons (Hardin, 1968) and free-riding (Madani and Lund, 2011). Lack of political will, leadership, funds, and authority to enforce compliance with water quality standards are among key factors complicating TMDL implementation. Additionally, on-going monitoring to evaluate the effectiveness of the TMDL plan, and inform decision makers and stakeholders as to timely progress towards water quality targets, is essential for establishing adaptive plans.

It is critical to identify funding sources for long-term TMDL programs, and evaluate potential socio-economic impacts of TMDL implementation to guide environmental decision making. For example, applying an environmental tax may negatively affect economic growth, leading to reduced income per capita. However, as shown in the CLD of the G&U archetype for Lake Allegan, improvement in environmental conditions may compensate for reduced utility, attracting population from areas with inferior environmental quality, and creating opportunities for socioeconomic growth (Rephann, 2010). Therefore, effective policy for reducing the lake's P concentration in the long-run should aim to internalize environmental externalities associated with socioeconomic growth, increase voluntary public participation in the TMDL program, and regulate P load input to the lake from PSs and NPSs.

3.9. Conclusions

Using the Lake Allegan eutrophication problem as a case study, it is shown that simple SD models can be developed and verified to facilitate qualitative and strategic-level quantitative analysis of interlinked socioeconomic and biophysical subsystems. Obtaining an accurate match between simulated and observed values is not sufficient for verification of causal-descriptive SD models, which should help explain how the behavior is generated. Thus, the Lake Allegan model was verified using is a semi-formal approach involving a mix of qualitative criteria and quantitative tests focusing on the system's structure. The model facilitates trend identification and pattern recognition, guiding holistic TMDL policy and long-term adaptive management in which potential impacts of socio-economic dynamics may be partially addressed using an appropriate margin of safety in the proposed TMDL plan. On-going monitoring campaigns can help determine whether the proposed TMDL plan is adequate for the extant socio-economic condition, or if there is a need for a more aggressive plan.

The principles and tools of systems thinking can improve holistic understanding of the underlying system structure driving water quality problems. Herein the G&U system archetype is used to explain Lake Allegan's eutrophication problem, demonstrating the need for continued investment to limit environmental degradation, essential for balanced socio-economic growth. This need stems from the process-based nature of sustainable development which calls for managing feedback relationships between socioeconomic dynamics, land use change, and environmental integrity. Without appropriately capturing these important feedback loops, TMDL plans may set overambitious water quality targets to be achieved within an unrealistic timeframe.

3.10. References

Anderson, J. R., Hardy, E.E., Roach J.T., Witmer, R.E. (1976). A land use and land cover classification system for use with remote sensor data. U.S. Geological Survey Professional Paper 964, U.S. Geological Survey, Reston, Virginia.

Arrow, K., Bolin, B., Costanza, R., Dasgupta, P., Folke, C., Holling, C.S., Jansson, B.O., Levin, S., Mäler, K.G., Perrings, C.A., Pimentel, D. (1995). Economic growth, carrying capacity, and the environment. *Science* 268, 520–521.

Bagheri, A., Hjorth, P. (2007). Planning for sustainable development: a paradigm shift towards a process-based approach. *Sustainable Development* 15, 83–96.

Benham, B., Zeckoski, R., Yagow, G. (2008). Lessons learned from TMDL implementation case studies. *Water Practice* 2(1), 1-13.

Bishop, R. (1993). Economic efficiency, sustainability and biodiversity. Ambio 22 (2-3), 69-73.

Boyd, J. (2000). Unleashing the clean water act, the promise and challenge of the TMDL approach to water quality. *Resources* 139, 7-10.

Braun, W., 2002. The system archetypes. <http://wwwu.uni-klu.ac.at/gossimit/pap/sd/wb_sysarch.pdf> (accessed on Sep. 21, 2011).

Canale, R.P., Redder, T., Swiecki, W., Whelan, G. (2010). Phosphorus budget and remediation plan for Big Platte Lake, Michigan. *Journal of Water Resources Planning and Management* 136(5), 576-586.

Cave, K., Harold, E., Quasebarth, T. (1996). Preliminary pollution loading projections for Rouge River watershed and interim nonpoint source pollution control plan, Technical Report RPO-NPS-TR07.00, Rouge River National Wet Weather Demonstration Project, Wayne County, Michigan.

Cooke, G.D., Welch, E.B., Peterson, S.A., Newroth, P.R. (1993). Restoration and Management of Lakes and Reservoirs. Boca Raton, Florida.

Doerr, S.M., Canale, R.P., Effler, S.W. (1996). Development and testing of a total phosphorus model for Onondaga Lake. *Lake and Reservoir Management* 12(1), 141-150.

Effler, S.W., O'Donnell, S.M., Matthews, D.A., Matthews, C.M., O'Donnell, D.M., Auer, M.T., Owens, E.M. (2002). Limnological and loading information and a phosphorus total maximum daily load (TMDL) analysis for Onondaga Lake. *Lake and Reservoir Management* 18(2), 87-108.

Ehrlich, P.R., Holdrens, J.P. (1971). The impact of population growth. *Science* 171, 1212–1217.

Ford, A. (1999). Modeling the environment. Island Press, Washington.

Forrester, J. (1961) Industrial dynamics. MIT Press, Cambridge.

Forrester, J. (1969) Urban dynamics. MIT Press, Cambridge.

Forrester, J.W., Senge, P.M. (1980). Tests for building confidence in system dynamics models. In: System dynamics (Ed.). Legasto, A.A., Forrester, J.W., and Lyneis, J. M., Amsterdam.

FWPCA (1972). Federal water pollution control act, as amended (33 U.S.C. 1151 et seq.).

FWPCA (2002). Federal water pollution control act, as amended (33 U.S.C. 1251 et seq.).

Gelda, R.K., Effler, S.W., and O'Donnell, S.M. (2001). Probabilistic model of ammonia and toxicity status for urban lake. *Journal of Water Resources Planning and Management* 127, 337–347.

Guo, H.C., Liu, L., Huang, G.H., Fuller, G.A., Zou, R., Yin, Y.Y. (2001). A system dynamics approach for regional environmental planning and management: a study for the Lake Erhai Basin. *Environmental Management* 61(1), 93–111.

Hamilton, S.K. (2012). Biogeochemical time lags may delay responses of streams to ecological restoration. Fresh Water Biology 57, 43–57.

Hardin, G. (1968). The tragedy of the commons. *Science* 162, 1243-1248.

Hartig, J.H., Horvath, F.J. (1982). A preliminary assessment of Michigan's phosphorus detergent ban. *Journal Water Pollution Control Federation* 54, 193-197.

Havens, K.E. Schelske, C. L. (2001). The importance of considering biological processes when setting total maximum daily loads (TMDL) for phosphorus in shallow lakes and reservoirs. *Environmental Pollution* 113, 1-9.

Heaton, S. (2001). Total maximum daily load (TMDL) for total phosphorus in Lake Allegan. Surface Water Quality Division, Michigan Department of Environmental Quality. Lansing, Michigan.

Hjorth, P., and Bagheri, A. (2006). Navigating towards sustainable development: a system dynamics approach. *Futures* 38(1), 74–92.

Kieser and Associates (2001). Non-point source modeling of phosphorus loads in the Kalamazoo River/Lake Allegan watershed for a total maximum daily load. Kieser and Associates Environmental Science and Engineering, Kalamazoo, Michigan.

Kieser and Associates (2005). Analysis of urban stormwater best management practice options for the St. Joseph River watershed. Prepared for: Friends of the St. Joseph River, Kieser and Associates Environmental Science and Engineering, Kalamazoo, Michigan.

Kieser and Associates (2011). 2010 TMDL evaluation: a look at how we are doing on TMDL goals. Kieser and Associates Environmental Science and Engineering, <http://kalamazooriver .net/> (accessed on Oct. 3, 2011).

KRLATIC (2002). Water quality improvement (implementation) plan for the Kalamazoo River watershed and Lake Allegan through a phosphorus total maximum daily load (TMDL) process. The Kalamazoo River/Lake Allegan TMDL Implementation Committee, Kalamazoo, Michigan.

Leal Neto, A.C., Legey, L.F.L., Gonzalez-Araya, M.C., Jablonski, S. (2006). A system dynamics model for the environmental management of the Sepetiba Bay watershed, Brazil. *Environmental Management* 38(5), 879–888.

Madani, K., Lund, J.R. (2011). California's Sacramento-San Joaquin Delta Conflict: from Cooperation to Chicken. *Journal of Water Resources Planning and Management* 138(2), 90-99.

Madani, K., Mariño, M.A. (2009). System dynamics analysis for managing Iran's Zayandeh-Rud river basin. *Water Resources Management* 23(11), 2163-2187.

MDEQ (1999). Loading assessments of phosphorus inputs to Lake Allegan. Staff Report, Surface Water Quality Division, Michigan Department of Environmental Quality, Lansing, Michigan.

MDNR (1987). Remedial action plan for the Kalamazoo River. Michigan Department of Natural Resources, Bay City, Michigan.

Meadows, D.H., Meadows, D.L., Randers, J., Behrens, W.W. (1972). The limits to growth. Universe Books, New York.

MWQS (2006). Michigan water quality standard. Water Resources Protection, Water Bureau, Department of Environmental Quality, Lansing, Michigan.

Pijanowski, B.C., Brown, D.G., Shellito, B.A., Manik, G.A. (2002). Using neural networks and GIS to forecast land use changes: a land transformation model. *Computers, Environment and Urban Systems* 26(6), 553–575.

Rephann, T.J. (2010). Economic impacts of implementing agricultural best management practices to achieve goals outlined in Virginia's tributary strategy, Center for Public Service University of Virginia, Charlottesville, Virginia.

Ryding, S.O., Rast, W. (1989). The control of eutrophication of lakes and reservoirs. UNESCO, Paris, France.

Senge, P. (1990). The fifth discipline: the art & practice of the learning organization. Doubleday/Currency, New York.

Shirmohammadi, A., Chaubey, I., Harmel, R.D., Bosch, D.D., Muñoz-Carpena, R., Dharmasri, C., Sexton, A., Arabi, M., Wolfe, M.L., Frankenberger, J., Graff, C., and Sohrabi., T.M. (2006). Uncertainty in TMDL models. *Transaction of The American Society of Agricultural and Biological Engineers* 49(4), 1033−1049.

Simonovic, S.P. (2009). Managing water resources: methods and tools for a systems approach. UNESCO, Paris and Earthscan James & James, London.

Simonovic, S.P., Fahmy, H. (1999). A new modeling approach for water resources policy analysis. *Water Resources Research* 35(1), 295–304.

Sterman, J.D. (2000). Business dynamics, systems thinking and modeling for a complex world. McGraw-Hill, Boston.

US BEA (2011). U.S. economic accounts. U.S. Bureau of Economic Analysis, <http://www. bea.gov/index.htm> (accessed on Aug.3, 2011).

US BLS (2009). Customer price index. U.S. Bureau of Labor Statistics, < http://www. bls.gov/cpi/cpifact8.htm> (accessed on Aug. 5, 2011).

US EPA (1975). Report on Lake Allegan, Allegan County, Michigan. Working Paper 182. U.S. Environmental Protection Agency Region V, Chicago, Illinois.

Ventana Systems (2010) User's Guide Version 5. 2010. Ventana Systems, Inc., <http://www. vensim.com> (accessed on Aug.13, 2011).

Vergura, J., Jones, R. (2001). The TMDL program: land use and other implications. *Drake Journal of Agricultural Law* 6(2), 317.

Vezjak, M., Savsek, T., Stuhler, E.A. (1998). System dynamics of eutrophication processes in lakes. *European Journal of Operations Research,* 109(2), 442-451.

Vondracek, B., Zimmerman, J.K.H., Westra, J.V. (2003). Setting an effective TMDL: sediment loading and effects of suspended sediment on fish. *Journal of American Water Resources Association* 39(5), 1005–1015.

Webster, J.R., Wallace, J.B., Benfield, E.F. (1995). Organic processes in streams of the eastern United States. In: Cushing, C.E., Cummins, K.W., and Minshall, G.W. (eds), Ecosystems of the world, 22, River and stream ecosystems. Elsevier, Amsterdam, 117-187.

Weiss, P.T., Gulliver, J.S., Erickson, A.J. (2007). Cost and pollutant removal of storm-water treatment practices. *Journal of Water Resources Planning and Management* 133, 218–229.

Winz, I., Brierley, G., Trowsdale, S. (2009). The use of system dynamics simulation in water resource management. *Water Resources Management* 23(7), 1301–1323.

Withers, P.J.A., Jarvis, G.C. (1998). Mitigation options for diffuse P loss to water. *Soil Use and Management* 14, 186-192.

Wuycheck, J. (1998). Water quality and pollution control in Michigan. Surface Water Quality Division, Report No. MI/DEQ/SWQ-98/030, Michigan 305(b) Report, Michigan Department of Environmental Quality, Lansing, Michigan.

Chapter 4 - A high-level simulation-optimization framework for non-point source phosphorus load reduction in the Kalamazoo River watershed (Michigan, USA)[4]

4.1. Abstract

The US Clean Water Act has been reasonably successful in point source (PS) pollution abatement through technology-based end-of-pipe regulation. However, total maximum daily load (TMDL) programs stipulated under the Clean Water Act fail to provide guidance for systematic approaches to evaluating the tradeoffs associated with alternative non-point source (NPS) pollutant abatement strategies. Furthermore, TMDL programs do not typically account for socioeconomic and biophysical feedbacks and uncertainties in future conditions affecting long-term attainability of the desired water quality target. Using the eutrophic Lake Allegan in Michigan as a case study, this chapter presents a high-level simulation-optimization framework, comprising system dynamics simulation and a best management practice (BMP) optimization model, for NPS total phosphorus (TP) reduction in the Kalamazoo River watershed. Long-term adequacy of the watershed's TMDL plan and tradeoffs between BMP implementation cost and reliability of meeting the lake's TP concentration target are investigated. The results suggest that NPS TP load reduction through agricultural BMPs such as buffer strips and conservation tillage should be given priority, supplemented by urban grassed swale with buffer strips and storage practices (detention and retention basins). The NPS pollution abatement required for achieving the lake's TP concentration target with high reliability

[4] This chapter is being considered for publication as Mirchi, A., Watkins, D.W. Jr. A high-level simulation-optimization framework for non-point source phosphorus load reduction in the Kalamazoo River watershed. *Science of the Total Environment*.

using agricultural BMPs can be prohibitively costly to agricultural producers, indicating the potential need for government intervention, as well as potential for environmental offset programs. Furthermore, although the lake's TP concentration is primarily governed by intra- and inter-annual hydrologic variability, the projected socioeconomic growth in the watershed is expected to negatively affect the reliability of meeting the water quality goal. Periodic updating of the TMDL plan can increase reliability, and result in potentially significant cost savings by improving the timing of the required BMP investments.

4.2. Introduction

Human intervention is necessary for maintaining environmental integrity by mitigating anthropogenic environmental degradation associated with socioeconomic growth. The need for the human intervention can be explained by the Growth and Underinvestment (G&U) system archetype (Senge, 1990), which builds on the well-known Limits to Growth archetype (Meadows et al., 1972). In an environmental context, the G&U archetype portrays how inadequate investment in maintaining environmental quality may activate socio-ecological feedbacks that, ultimately, will limit the system's growth. One example of the feedbacks threatening to limit the growth of socio-ecological systems is the widespread algae blooms in eutrophic lakes (e.g., Smith, 2003; Paerl et al., 2011; Michalak et al., 2013), indicating unsustainable development. There is an evident need for systematic initiatives and aggressive policies to mitigate degradation of aquatic systems (Booth and Jackson, 1997; Carpenter et al., 1998).

In the US, the Clean Water Act (CWA) of 1972 (FWPCA, 1972) and the subsequent amendments (e.g., the CWA amendment of 2002 (FWPCA, 2002)) provide a legal framework for human intervention to maintain environmental integrity of the nation's water bodies. The CWA triggered efforts to determine impairment of water bodies and recommend ways to address the issue(s), although, in some cases, lack of clear impairment listing and delisting methodologies has caused inconsistent impairment determination in different states (Keller and Cavallaro, 2008). The states must quantify the total maximum daily load (TMDL) of a pollutant that a water body can receive without violating specified water quality standards (FWPCA, 2002). Point sources (PS), non-point sources (NPS) and natural background nutrient loads (e.g., mineralization of native organic matter) are characterized, and a factor of safety is warranted to account for potential lack of knowledge. The statute has brought about significant environmental improvements in the riverine areas (Boyd, 2000). However, more than four decades after its passage, the US is far from having healthy aquatic communities, as 55% of the country's rivers and streams are in poor biological condition, mostly due to excessive amounts of phosphorus and nitrogen (US EPA 2013, P11). Likewise, of 39% of the assessed lakes and reservoirs, excluding the Great Lakes, 64% were "impaired or not clean enough to support their designated uses, such as fishing and swimming" (US EPA 2009, P16).

The biophysical aspects of watershed processes and NPS pollutants driving anthropogenic eutrophication have been the subject of extensive research (e.g., Carpenter et al., 1998; Gburek et al., 2000; Haygarth et al., 2005; Heathwaite et al., 2005; Rao et al., 2009), providing valuable insights for implementation of best management practices

(BMPs) for pollutant reduction (e.g., Veith et al., 2003; Gitau et al., 2004; Zhen et al., 2004; Hsieh and Yang, 2007; Panagopoulos et al., 2011; Lee et al., 2012; Giri et al., 2012). In the US, PS pollution abatement has been reasonably successful due to the relative ease of command-and-control, technology-based end-of-pipe regulation, whereas urban (e.g., combined sewer overflows, runoff from lawns, and septic tanks) and agricultural (e.g., excess fertilizer) NPS pollutant load reduction is much more challenging (Sharpley et al., 1994; Boyd, 2000). A variety of BMPs have been used for TMDL implementation in urban settings (Tsihrintzis and Hamid, 1997; Sample et al., 2003; Weiss et al., 2007) and agricultural areas (Sharpley et al., 1994; Bottcher et al., 1995; Withers and Jarvis, 1998; Kleinman et al., 2011). BMPs that reduce total phosphorous (TP) loads (e.g., buffer strips, tillage practices, and basin practices) have received particular attention, as this nutrient is usually the limiting factor for eutrophication of fresh water aquatic systems (Lee et al., 1978; Sharpley et al., 1994; Correll, 1998; Mainstone and Parr, 2002).

A systematic approach to TMDL implementation, including cost-effective NPS pollution reduction, needs further investigation. The approach should consider the potential impacts of socioeconomic growth, associated biophysical changes, and hydrologic variability on the tradeoffs between TP abatement cost and reliability of meeting the desired water quality goal. Many researchers have focused on watershed processes to demonstrate the tradeoffs between BMP cost and pollutant load reduction at the watershed scale (e.g., Milon, 1987; Arabi et al., 2006; Maringanti et al., 2009; Rodriguez et al., 2011, Panagopoulos et al., 2012), as well as the farm scale (Gitau et al., 2004; Gooday et al., in press). Fewer studies have recommended optimal eutrophication control policies using macroeconomic and ecologic approaches (e.g., Hein, 2006; Deng et al., 2011). Hein (2006) developed an ecological-economic model to study cost-benefit implications of optimal eutrophication control policies for improving ecosystem functions of a shallow lake. Deng et al. (2011) presented a macroeconomic model of cost-effective policies to balance regional economic growth with pollutant reduction to address the eutrophication of Poyang Lake in China. Accounting for the socioeconomic and biophysical feedbacks driving eutrophication will help TMDL planners to avoid the pitfall of setting overly ambitious water quality targets or unrealistic time frames to achieve those targets (Mirchi and Watkins, in press).

Furthermore, TMDL programs have not typically provided systematic guidance on alternative mitigation methods and long-term attainability of the target. Often, the scope of the programs is practically limited to identifying the main pollutant of concern, quantifying existing pollutant loads, and specifying target water quality standards based on the available guidelines (e.g., US EPA, 1991). However, a comprehensive TMDL process to address the NPS pollution requires a number of other important components to facilitate effective policies. The Chesapeake Bay TMDL program has applied a more holistic environmental systems analysis approach to TMDL planning as compared with typical programs (Schwartz, 2010; US EPA, 2012). Environmental systems analysis techniques should be systematically incorporated in TMDL studies in order to explore a potential set of “optimal” strategies for meeting the target. Moreover, the long-term attainability of the target using the desired mitigation method in the presence of dynamic system-wide drivers (e.g., socioeconomic) and associated feedbacks (e.g., land use change) should be investigated. The insights obtained from these steps will provide a good basis for identifying effective mitigation policies and practices whose success can be evaluated through adequate monitoring campaigns.

The objective of this chapter is to investigate the noted components of the proposed TMDL planning paradigm using a high-level simulation-optimization framework for watershed-scale NPS TP reduction in the Kalamazoo River watershed, Michigan. A system dynamics (SD) simulation model is used in conjunction with a screening-level optimization model to find a set of least-cost BMPs for TP load reduction. Using the eutrophic Lake Allegan as a case study, this chapter draws insights into 1) the long-term (30 years) adequacy of the proposed TMDL; 2) the tradeoffs between NPS BMP implementation cost and reliability of meeting water quality target; and 3) the potential impacts of the watershed's socioeconomic growth on attainment of the target. The next section presents the methodology, including the SD model and the optimization model for finding the least-cost BMP set. Section 3 provides the results, a discussion of the existing challenges of TMDL programs, and the implications of potential policies for BMP implementation. Section 4 concludes the chapter.

4.3. Data and methods

4.3.1. Study area

Lake Allegan is a small run-of-the-river impoundment (outflow roughly equals inflow) with an area of 642 ha, volume of 21.2 million m^3, mean depth of 3.3 m, and residence time of less than 12 days (Reid and Hamilton, 2007). It is located at the outlet of a 401,500-ha drainage area with a mean annual precipitation of about 864 mm in the Kalamazoo River watershed, Michigan (Figure 4.1). The lake receives a mean annual flow of about 38 m^3/s, mostly from the Kalamazoo River (Wesley, 2005). The watershed is covered predominately with agricultural (47%) and forested/open land (34%), while developed areas and water/wetland cover 9% and 7% of the watershed, respectively. The share of the watershed's developed area has increased over the last few decades due to urbanization of agricultural lands.

The lake was hypereutrophic in the late 1990's because of high (~96 µg/l) TP concentrations (Wuycheck 1998), causing an undesirable fish community dominated (>80%) by carp and catfish (Heaton, 2001). Since 1998, a TMDL program has been underway in the watershed, under the direction of the Michigan Department of Environmental Quality (MDEQ), to attain an average growing season TP concentration of 60 µg/l in the lake (Heaton, 2001), by reducing the PS and NPS TP loads by 23% and up to 50%, respectively (Kieser and Associates, 2001; KRLATIC, 2002). The goal is to meet the water quality target by 2015 (KRLATIC, 2002). The proposed NPS component of the TMDL is being implemented using a participatory approach, whereas PSs have effectively complied with the recommended load reduction target (Kieser and Associates, 2011).

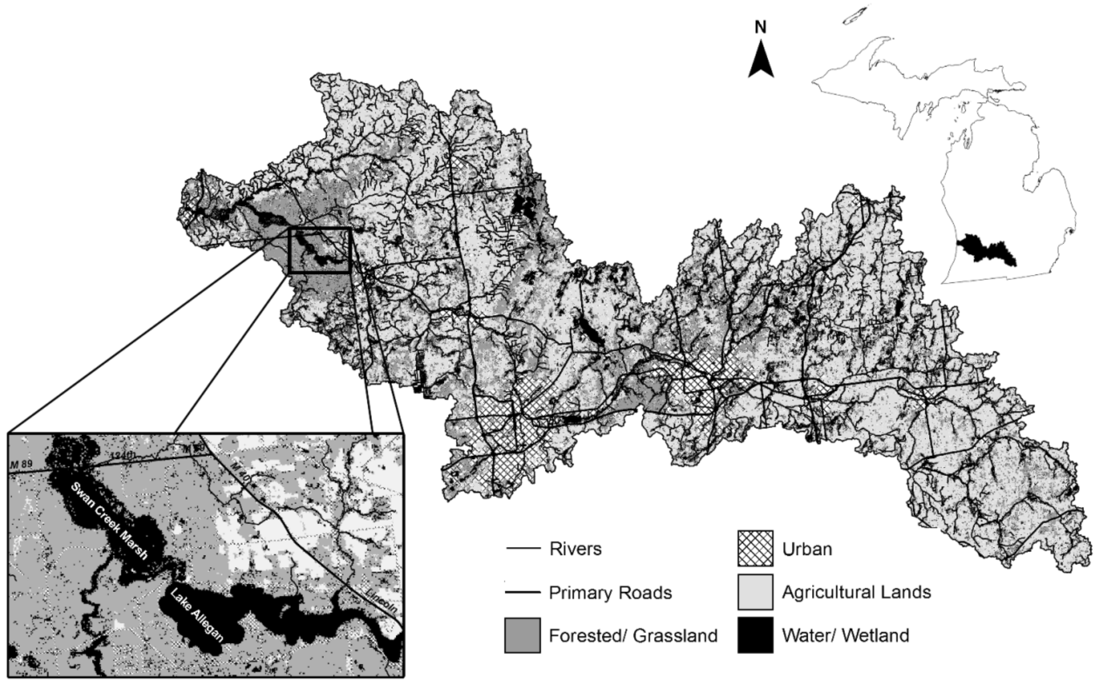


Figure 4.1. Location of Lake Allegan in the Kalamazoo River watershed (Source of data: Michigan Geographic Data Library (MiGDL), http://www.mcgi.state.mi.us/mgdl/).

4.3.2. Simulation-optimization framework

A schematic of the simulation-optimization framework, comprising SD and BMP optimization models, is shown in Figure 4.2. TP loads from different land use types are given by the SD model, which simulates the lake's TP loading condition and projects the water quality trend into the future (1998-2028). A reliability index (RI) is used for assessing the long-term attainability of the specified water quality target (eq. 4.1).

$$RI = \frac{n}{N} \qquad (eq. 4.1)$$

where n= the number of times that the target TP concentration was met; and N=length of simulation period.

The TP loads obtained from the SD model are input to the optimization model to identify the least-cost BMP set. The outcomes of the optimization model are evaluated using the SD model to produce tradeoff curves with consideration of feedbacks under different socioeconomic growth scenarios. This is done by using the post-BMP TP loads in the SD model in order to examine the effectiveness of the least-cost BMP set. Higher values of RI may be obtained through considering more stringent mean TP concentration requirements (<60µg/l) in the deterministic optimization model. Comparison of the projected water quality trends for the two cases of with and without BMPs provides insights as to adequacy of the BMP set for long-term attainment of the water quality target.

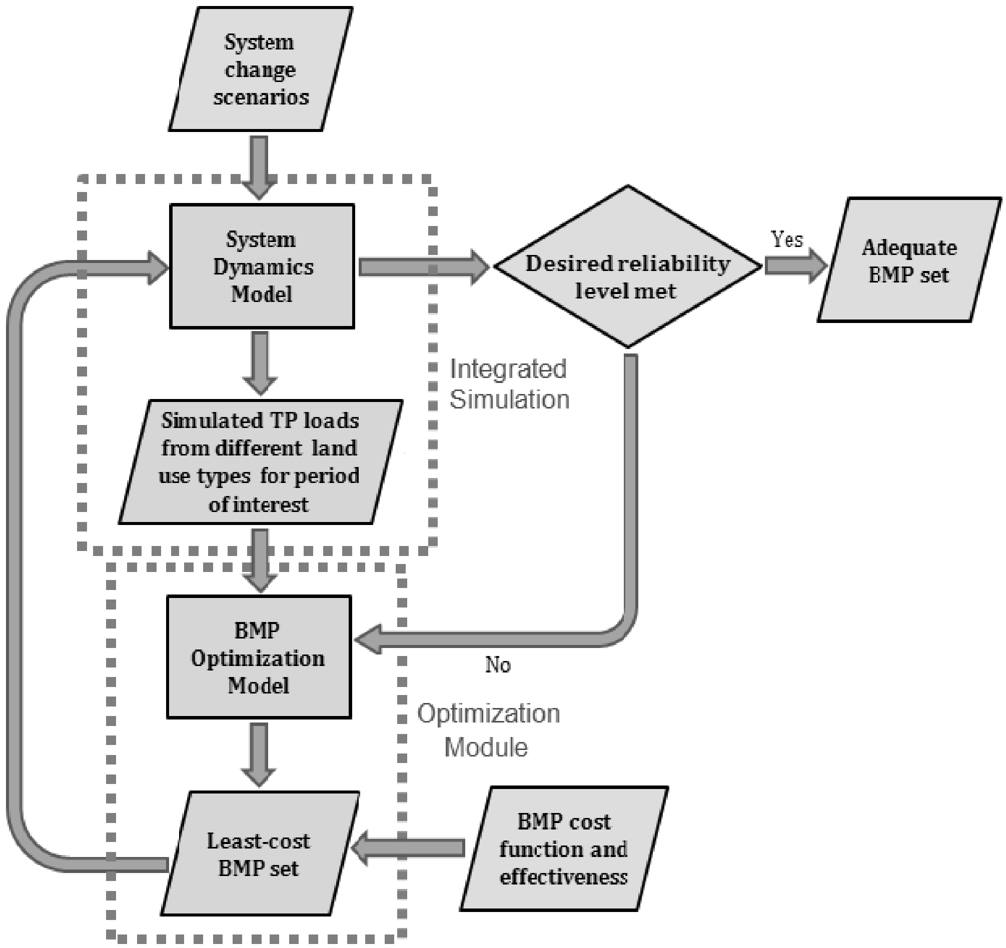


Figure 4.2. Schematic of simulation-optimization framework.

4.3.3. System dynamics simulation

System dynamics simulation (Forrester 1961 and 1969; Ford 1999; Sterman 2000) is a method for operationalizing systems thinking (Senge 1990), facilitating holistic water resources modeling (Simonovic 2009; Mirchi et al., 2012). Winz et al. (2009) and Mirchi et al. (2012) reviewed system dynamics applications to water resources problems, as well as the pros and cons of the approach. Mirchi and Watkins (in press) used the G&U archetype with three main feedback loops, i.e., socioeconomic, biophysical, and human intervention to explain system-wide processes governing the anthropogenic eutrophication of Lake Allegan. A high-level system dynamics model was developed to simulate the changes over time in the socioeconomic loop, driving the land use change, and TP loading of the lake. A generalized stock-and-flow diagram of the model was presented in chapter three (Figure 3.4). The model is used in this study to simulate the effect of NPS TP load reduction using least-cost urban and agricultural BMPs. The model and data inputs, provided in greater detail in Mirchi and Watkins (in press), are described here.

Time series data (1978-2009) for population, income per capita, and employment were obtained from the U.S. Bureau of Economic Analysis (US BEA 2011). Additionally, Bureau of Labor Statistics' Customer Price Index calculator (US BLS 2009) was used to obtain inflation-adjusted income. Using available land use data from Michigan GIS data library for 1975, Coastal Change Analysis Program (C-CAP) for 1996 and 2001, and Purdue Land Transformation Model (Pijanowski et al. 2002) for 2005, 2010, 2015, linear regression models best fitting the trend of change for urban/built up and agricultural lands were developed to generate a land use time series. Statistically significant linear regression equations (p-value: <0.001, adjusted r^2: 0.88-0.94) were used to establish the relationships between socioeconomic variables (e.g., population, employment, and income) and urban and agricultural land use change.

A seasonal (3-month) time-scale was used to capture intra-annual variability. Precipitation data from the National Oceanic and Atmospheric Administration (NOAA) was used to calculate net precipitation--the difference between precipitation and evapotranspiration--assuming 50% of precipitation is lost to evapotranspiration (Webster et al., 1995). Land use-specific runoff coefficients estimated from fraction of pervious and/or impervious cover were applied to calculate runoff. The TP loading from atmospheric deposition and direct drainage to the lake was estimated at 154 kg/month for April through September (Heaton 2001). A sediment burial rate of $4.0\text{x}10^{-7}$ m/min was used (MDNR 1987) to characterize nutrient loss due to in-lake processes. The flow data for Lake Allegan outlet was obtained from the Fennville USGS gauging station. TP export coefficients for different land uses estimated for the neighboring Rouge River watershed, southwestern Michigan, were applied to calculate the lake's TP load assuming steady-state mass balance (eq.4.2).

$$LP(t) = \int_{t_0}^{t_n} [\sum_{i=1}^{i=4} LU_i\, P + AD + DL - IL - OL]\, dt + LP(t_0) \qquad (eq. 4.2)$$

where *LP(t)* = *P* loading at time t; *i*= land use type (e.g., agriculture, urban/built-up, forested, and water/wetland (i=4)); $LU_i\,P$ = P loading from land use *i*; *AD* = atmospheric deposition; *DL* = loading from direct drainage; *IL* = in-lake loss; *OL* = outflow loss.

The historical hydrology was used to simulate future variability. In addition, a Monte Carlo simulation approach accounting for the effects of parameter variability in

TMDL studies (Gelda et al. 2001) was applied to evaluate model performance. Lake Allegan's P concentration is most sensitive to estimated P loadings for agricultural and urban areas, and runoff. One thousand simulations were run to sample P loads between -20% and +20% of the best-estimate average loads, and between -10% and +10% of the average runoff, assuming uniform distributions. The model satisfactorily reproduces the historical TP concentration within the specified ranges of variability (Mirchi and Watkins, in press).

4.3.4. Optimization model

The mathematical formulation of the BMP optimization problem is shown by equations 4.3 through 4.5. Following the work of Hsieh and Yang (2007), the objective function in this formulation (eq.4.3) is to minimize the cost (C) of BMP implementation in urban and agricultural land uses as a function of the BMP-specific decision variables (*X*) such as volume, area, or length. This cost minimization should be done subject to the lake's target TP concentration constraint (eq.4.4), set by the existing TMDL plan at 60μg/l.

$$Min \sum_{i=1}^{i=I} \sum_{j=1}^{j=J} C_{i,j}(X_{ij}) \qquad (eq. 4.3)$$

$$Conc_{TP} \leq Conc_{targ} \qquad (eq. 4.4)$$

where C_{ij}(-) = BMP implementation cost($); i = land use type (urban and agriculture; I = 2); j = BMP type (J_{urban} = 4, J_{ag} = 4); X_{ij} = BMP-specific decision variable; $Conc_{TP}$ = post-BMP TP concentration (μg/l); $Conc_{targ}$ = the lake's target TP concentration (60 μg/l).

The maximum allowable level of BMP implementation due to land availability creates another set of constraints (eq. 4.5).

$$X_{ij} \leq X_{ij}^{UB} \qquad (eq. 4.5)$$

where X^{UB}_{ij} = upper bound on BMP implementation.

Equations 4.6 through 4.8 are used to calculate the TP load reduction, and the resulting TP concentration in the lake, after BMP implementation. The TP load reduction (Lr) in urban and agricultural land uses is defined as the product of the existing load (e) emitted from an areal unit of land (in ha) and the collective TP reduction effect of the BMP set (eq.4.6). The latter is a function of TP removal effectiveness (R) of each unit of individual BMPs, the amount of each BMP, and the BMP-specific area treatment coefficient, defined as the areal units of land treated by a unit of BMP. The post-BMP TP load is the sum of TP loads from sources not impacted by BMPs, including atmosphere, direct drainage, PS, forests, and natural background load, and post-BMP NPS loads from urban and agricultural lands as given by eq.4.7. The post-BMP TP concentration in the lake is calculated using the Vollenweider model (Vollenweider, 1976), assuming complete mixing and steady state conditions (eq.4.8).

$$Lr_i = e_i \cdot \sum_{j=1}^{j=J} R_j \cdot X_{ij} \cdot a_{ij} \quad for\ all\ i = 1,2 \qquad (eq. 4.6)$$

$$L_{TP} = \sum_{k=1}^{k=K} Le_k + \sum_{i=1}^{i=I} (Le_i - Lr_{i)} \quad for\ all\ k = 1, \dots, 5; i = 1,2 \qquad (eq. 4.7)$$

$$Conc_{TP} = \left(\frac{1}{Q_L + V_s A_L}\right) \cdot L_{TP} \qquad (eq. 4.8)$$

where Lr_{ij} = reduced load (kg/yr); e_i = TP export coefficient (kg/ha/yr); R_j = removal effectiveness coefficient (dimensionless); a_{ij} = area treatment coefficient (ha/m^3, ha/m, or ha/ha); $L_{TP,i}$ = post-BMP TP load (kg/yr); Le_i = existing TP load (kg/yr); k = TP sources not affected by BMPs (K = 5); Q_L= lake's average discharge rate (L/yr); A_L = lake's surface area (m^2); V_s = in-lake settling velocity of TP (m/yr).

The feasible least-cost BMP set may be found using either linear programming (LP) or a genetic algorithm (GA).

4.4. Best management practices

A suite of conventional structural and non-structural BMPs for reducing TP load from urban and agricultural lands is considered, including basin practices (e.g., retention, detention, and constructed wetland), grassed swales, tillage practices, and buffer strips. Table 1 presents these BMPs, providing average cost functions, average annual operation and maintenance cost, and average TP removal effectiveness coefficients, which have been compiled from the literature (e.g., Bottcher et al., 1995; US EPA, 1999 and 2003; Sample et al., 2003; Weiss et al., 2007; Simpson and Weammert, 2009; Stagge et al., 2012). For buffer strips, a cost function was developed using the average of the cost range from US EPA (1999), assuming a width of 10 m. Net present value of the BMP costs was calculated for a service life of 30 years using a discount rate of 3% (eq. 4.9).

$$NPV(i,T) = \sum_{t=0}^{t=T} \frac{C_t}{(1+i)^t} \qquad for\ all\ t = 0,1,2,\ldots,30 \qquad (eq. 4.9)$$

where *NPV* = net present value; *i* = discount rate (3%); *t* = time (year); *C* = annual cost, which includes both investment cost and annual operations and maintenance costs.

A procedure using geographic information system (GIS) and available BMP guidelines (e.g., US EPA 1996, 2003, and 2010) was developed to identify potential locations that are suitable for BMP implementation in order to estimate the upper bound for BMP implementation. Up to 2% of low-lying lands with moderately and poorly drained hydric soils, i.e., hydrologic group B, C, and D (Cronshey, 1986), in urban and agricultural areas are considered for the storage BMPs (US EPA, 1996 and 2010). Urban grassed swale with and without buffer strip is considered for construction along moderately sloped (1-4%) main roads in highly developed areas with non-hydric soils of type A and B (US EPA, 1996). Well- and moderately-well drained soils are suitable for conservation tillage for corn production (DeJong-Hughes and Vetsch, 2007), which is the main agricultural crop in the watershed, whereas cover cropping can be used in all agricultural lands where conventional tillage is applied (Simpson and Weammert, 2009). All the streams running through agricultural lands are deemed suitable for implementation of agricultural buffer strips. These siting criteria are summarized in Table 4.2. Urban and agricultural TP load export coefficients and BMP-specific attributes such as upper bound and area treatment coefficients are provided in Table 4.3.

Table 4.1. BMP cost functions and TP removal effectiveness.

Land use	BMP	Cost*($)	Annual O&M cost	Decision variable	TP removal effectiveness	Reference(s)
Urban	Retention basin	$123.9V^{0.75}$	4.5%	Volume of storage (m^3)	52%	Cost: Sample et al. (2003); O&M and effectiveness: Weiss et al. (2007)
	Grassed swale	30.53L	91%	Length of swale (m)	43%	Cost: Sample et al. (2003); O&M: Weiss et al. (2007); Effectiveness: Stagge et al. (2012)
	Grassed swale and buffer strip	41.8L	69%	Length of swale and buffer strip (m)	63%	Cost: Sample et al. (2003), US EPA (1999); O&M: US EPA (1999); Effectiveness: Stagge et al. (2012)
	Detention basin	$120.96V^{0.69}$	2.25%	Volume of storage (m^3)	25%	Cost: Sample et al. (2003); O&M and effectiveness: Weiss et al. (2007)
Agricultural	Constructed wetland	$758V^{0.565}$	9%	Volume of storage (m^3)	42%	Cost, O&M, and effectiveness: Weiss et al. (2007)
	Conservation tillage	6.59A	100%	Area (ha)	22%	Cost: US EPA (2003); Effectiveness: Simpson and Weammert (2009)
	Cover crop	16.48A	100%	Area (ha)	7%	Cost: US EPA (2003); Effectiveness: Simpson and Weammert (2009)
	Buffer strip	15.78L	129.5 ($/ha)	Length of buffer strip (m)	40%	Cost: US EPA (1999), O&M: Weiss et al. (2007), Effectiveness: Bottcher et al. (1995)

*Costs are in 1998 US dollars, and do not include cost of land acquisition.

Table 4.2. Criteria for potential BMP locations.

BMP	Siting criteria
Retention basins	Low lands with hydric soils of hydrologic group B, C, and D [a,b] in medium intensity urban areas
Grassed swale with and without buffer strip	Along main roads in highly developed areas with non-hydric soils of type A and B with slopes between 1-4% [b]
Detention basin	Low lands with hydric soils of hydrologic group B, C, and D [a,b] in highly developed urban areas
Constructed wetland	Low lands with hydric soils of hydrologic group B, C, and D [a,b] in agricultural areas
Conservation tillage	Well- and moderately well-drained soils (e.g., hydrologic group A and B) [c]
Cover crop	All agricultural lands [d]
Buffer strip	10m buffer around streams that run through agricultural land

References: [a] Young et al. (1996) and US EPA (2010); [b] US EPA (2010); [c] DeJong-Hughes and Vetsch (2007); [d] Simpson and Weammert (2009)

Table 4.3. Urban and agricultural TP load and BMP-specific upper bound and area treatment coefficients.

Land use (Load)	Urban* (e=0.380 kg/ha)				Agricultural* (e=0.128 kg/ha)			
	Retention basin	Grassed swale	Grassed swale & buffer strip	Detention basin	Constructed wetland	Conservation tillage	Cover crop	Buffer strip
Upper bound (X^{UB})	12.85 (10^6 m^3)	127 (10^3 m)	127 (10^3 m)	1.07 (10^6 m^3)	80.30 (10^6 m^3)	251.33 (10^3 ha)	251.33 (10^3 ha)	729.4 (10^3 m)
Area treatment coefficient	0.0025 (ha/m^3)	0.25 (ha/m)	0.25 (ha/m)	0.0025 (ha/m^3)	0.005 (ha/m^3)	1 (ha/ha)	1 (ha/ha)	0.55 (ha/m)

*Estimated using TP concentration values from the neighboring Rouge River watershed (Cave et al., 1996).

4.5. Results and Discussion

4.5.1. Least-cost BMP set

Model results indicate that NPS TP load reduction through agricultural BMPs (buffer strips and conservation tillage) should be given priority in the Kalamazoo River watershed, as they are more cost-effective as compared with urban BMPs. Using the average cost functions and TP removal effectiveness available from the literature (Table 4.1), agricultural buffer strip is the most cost-effective BMP, followed by conservation tillage. The agricultural BMPs dominate the selected BMP set when the target TP concentration constraint is relaxed. More stringent TP targets (lower concentrations) will require BMP implementation in urban areas due to significant increase in the marginal cost of TP reduction through wetland construction in agricultural lands. The most cost-effective urban BMPs were found to be grassed swale with buffer strip, detention basin, and retention basin, while grassed swale alone was not selected as it was outperformed by grassed swale with buffer strip. Likewise, cover cropping was not selected as it is inferior in performance as compared with conservation tillage. Figure 4.3 shows the tradeoff between the cost of agricultural and urban BMPs and the target TP concentration.

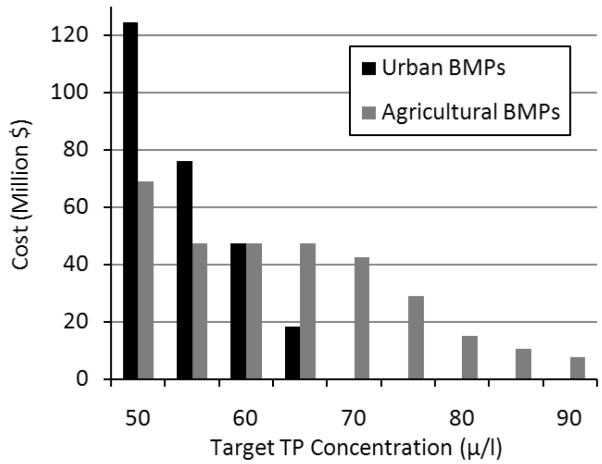


Figure 4.3. Cost of selected agricultural and urban BMP sets for meeting different target TP concentrations. Estimated using the period-of-simulation average growing season TP loads assuming moderate socio-economic growth.

Coincidentally, the costs of agricultural and urban BMPs, shown in Figure 4.3, are almost exactly the same for reducing the period-of-simulation average growing season TP loads to meet the 60 µg/l target, assuming moderate socio-economic growth. However, the amount of TP load reduction in agricultural and urban areas is disproportionate, i.e., 80.7% and 19.3%, respectively, which may create conflict among the agricultural and urban stakeholders as urban areas discharge the most TP load in the Kalamazoo River watershed. While a flexible TMDL plan in terms of land use can facilitate least-cost TP load abatement, in the absence of appropriate support policies, reducing the load mostly through agricultural BMPs may impose hardship on the agricultural sector. For this reason, a more balanced combination of urban and agricultural BMPs may be necessary to address the potential equity problem. In other words, a certain level of urban TP load reduction may be required in order to develop a fairer TMDL plan, although this may cause divergence from the least-cost BMP set. Figure 4.4 illustrates how the BMP implementation at the watershed scale becomes suboptimal under different scenarios of urban TP load reduction as compared with a flexible TMDL plan where the required urban TP load reduction is 0%. The figure indicates that there is potential for policies that allow for optimal abatement of TP loads at the watershed scale through environmental offset programs between urban and agricultural areas, whereby urban areas share the cost of agricultural BMP implementation.

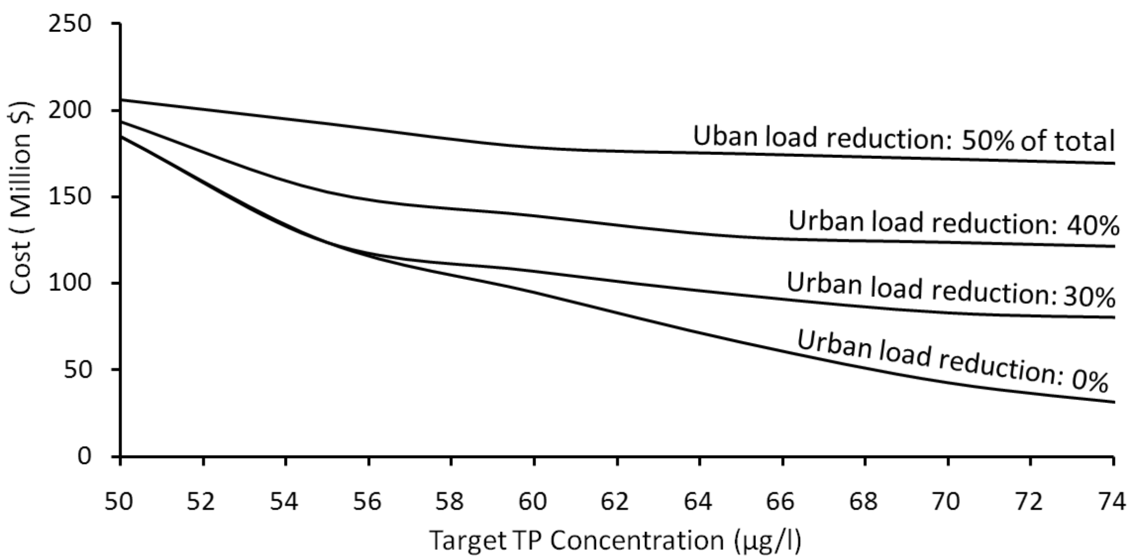


Figure 4.4. Tradeoff contours for different urban load reduction requirements.

4.5.2. Effect of socioeconomic growth

If the reported land use-based TP export coefficients (urban: 0.380 kg/ha, agricultural: 0.128 kg/ha) for the Kalamazoo River watershed hold into the future, it can be expected that TP load to the lake will increase over time as a consequence of urbanization driven by socioeconomic growth (Mirchi and Watkins, in press). Four socioeconomic growth scenarios were considered in this study based on historical trends of population growth, employment, and income. These scenarios are static conditions, slow growth, moderate growth, and rapid growth, which are equivalent to annual population growth rates of 0%, 0.1%, 0.56%, and 1%, respectively. Figure 4.5 shows the TP concentration exceedence curves for static conditions, along with moderate and rapid economic growth scenarios for the two cases of without and with BMPs. The concentration exceedence curves illustrate the effect of the least-cost BMP set obtained using period-of-simulation average growing season TP loads. The BMPs increase nutrient assimilation capacity of the area, which supports economic growth while maintaining the violation of the water quality goal at a minimal level. The probability of exceedence of the target concentration (60 μg/l) is reduced from ~40% when BMPs are not implemented to ~20% after BMP implementation. Table 4.4 summarizes the required TP load reduction and reliability of the least-cost BMP set for meeting the concentration target under different economic growth and BMP implementation scenarios.

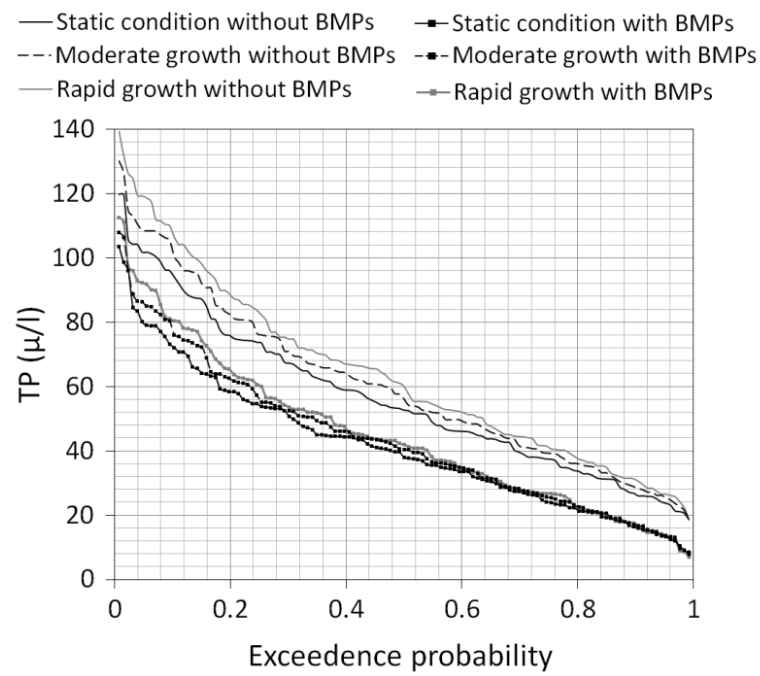


Figure 4.5. Concentration exceedence curves under different economic growth scenarios for the two cases of without (a) and with BMPs (b).

Table 4.4. Required load reduction and reliability of least-cost BMP set under different economic growth and BMP implementation scenarios.

Economic growth scenario	BMPs implemented	TP Load reduction (kg/yr)	Total cost (million $)*	Reliability index (%)	Selected BMP set
Static condition	No	0	0	63.92	N/A
	Yes	20896	61.13	87.37	Buffer strip, conservation tillage, and grassed swale with buffer strip
Slow growth	No	0	0	62.13	N/A
	Yes	21460	66.85	86.20	Buffer strip, conservation tillage, and grassed swale with buffer strip
Moderate growth	No	0	0	57.70	N/A
	Yes	24210	94.75	82.46	Buffer strip, conservation tillage, and grassed swale with buffer strip
Rapid growth	No	0	0	52.79	N/A
	Yes	27153	124.68	76.89	Buffer strip, conservation tillage, grassed swale with buffer strip, and retention basin

*Costs are in 1998 US dollars, and do not include cost of land acquisition.

4.5.3. Uncertainty in TMDL planning

There is a multitude of sources of uncertainty that complicate the TMDL planning for NPS pollutant abatement, including imperfect information, natural variability, and knowledge-based inconsistency (Shirmohammadi et al., 2006; Chen et al., 2007; Keller and Cavallaro, 2008). Uncertainty is inherent in the process of identifying the sources and magnitude of NPS pollution due to paucity of field data because of infrequent and limited monitoring campaigns, and inconsistent assessment methods (Keller and Cavallaro, 2008). There is significant uncertainty about the BMPs' pollutant removal effectiveness due to lack of a systematic approach for data collection and comparative assessment of stormwater pollutant removal (Strecker et al., 2001; Scholes et al., 2008), as well as site-to-site variability. Furthermore, determining the reference hydrologic conditions for developing the TMDL to adequately capture natural variability is a major source of uncertainty. Consequently, TMDL planners are faced with a wide range of potential strategies (e.g., combination of BMPs) for addressing the NPS pollution, which bears significant cost implications.

To illustrate the potential effect of uncertainties on the projected trend of TP concentration, a Monte Carlo analysis was performed using two thousand simulations between static and rapid growth conditions. Furthermore, +/- 10% variability in runoff was considered, and TP loads were sampled from a uniform distribution between -20% and +20% of the best-estimate average loads. The results of these simulations are shown in Figure 4.6 for the two cases of without and with BMPs. The figure illustrates that meeting the Lake Allegan's TP concentration target in any given year is primarily governed by intra- and inter-annual hydrologic variability, making the impacts of socioeconomic growth secondary in importance. In this situation, the lake's hydraulic flushing during the growing season is a significant determinant of eutrophication because the amount of TP in the Kalamazoo River is generally high (Reid and Hamilton, 2007). Therefore, NPS TP loads may have to be greatly reduced before the lake's water quality target can be met with high reliability. Furthermore, the long-term attainability of the target TP concentration may be contingent on updating of the NPS load reduction requirement to account for the effect of economically-driven urbanization.

The tradeoffs between TP load abatement cost and the reliability of attaining the water quality target are substantial. To illustrate this effect, the BMP optimization model was run under stochastic hydrologic (and resulting TP load) conditions using the Monte Carlo technique, assuming moderate socioeconomic growth. The least-cost BMP set from the optimization model with mean TP load estimates was used as a good initial guess in order to increase computational efficiency. Two thousand random samples of TP load were taken for different NPSs from a uniform distribution within two standard deviations of the average growing season TP load. The outcome of this process is two thousand least-cost BMP sets corresponding to different load conditions. Figure 4.7 shows the cost implication of this uncertainty due to TP load variability associated with inter-annual hydrologic variability. The long right-tail of the histogram denotes greater TP load reduction through more extensive BMP implementation using basin BMPs, providing a higher reliability but at significantly higher costs as compared with the average condition (~$95 million).

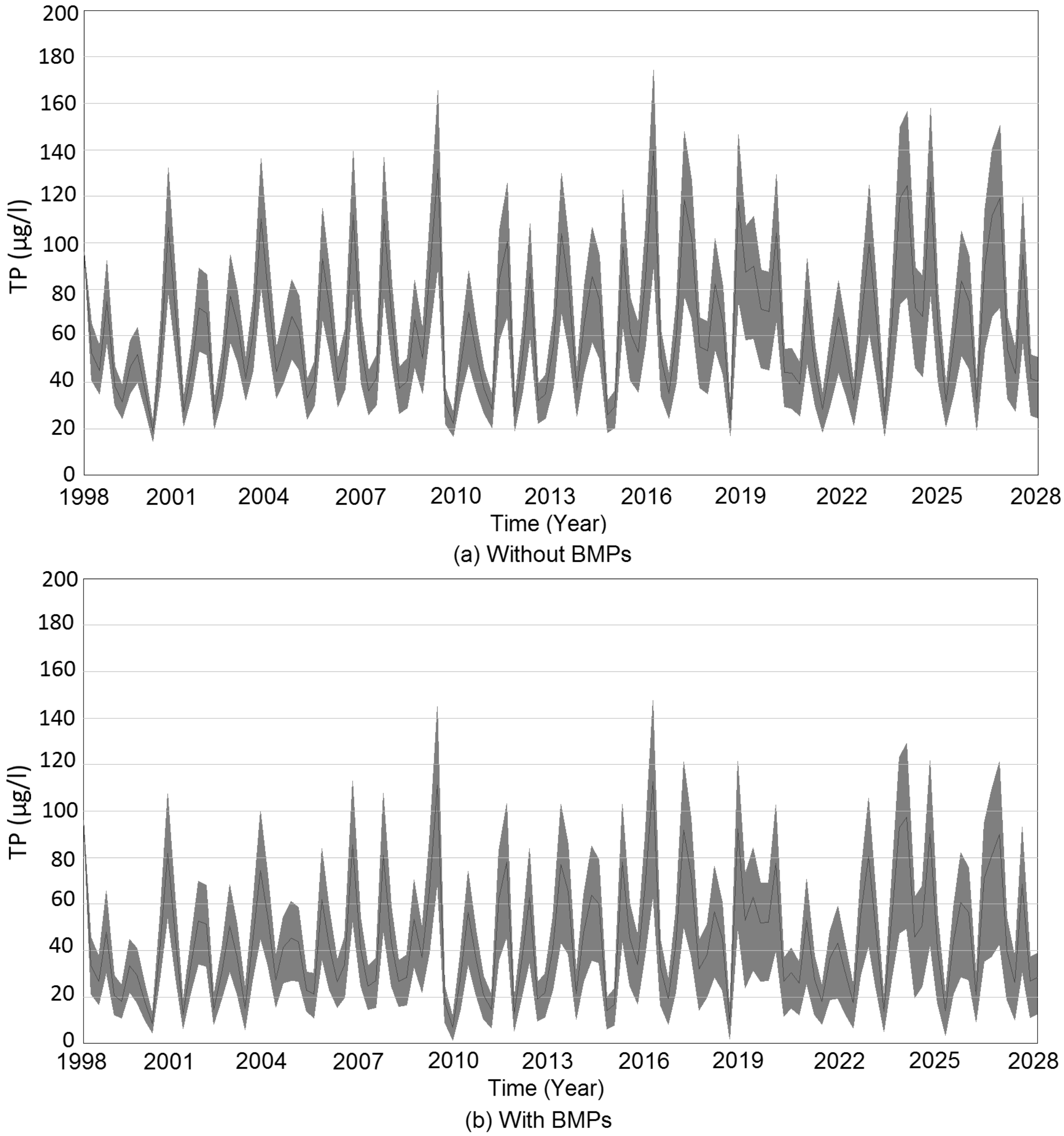


Figure 4.6. Simulated trend of TP concentration with potential effect of socioeconomic change for the two cases of without (a) and with BMPs (b).

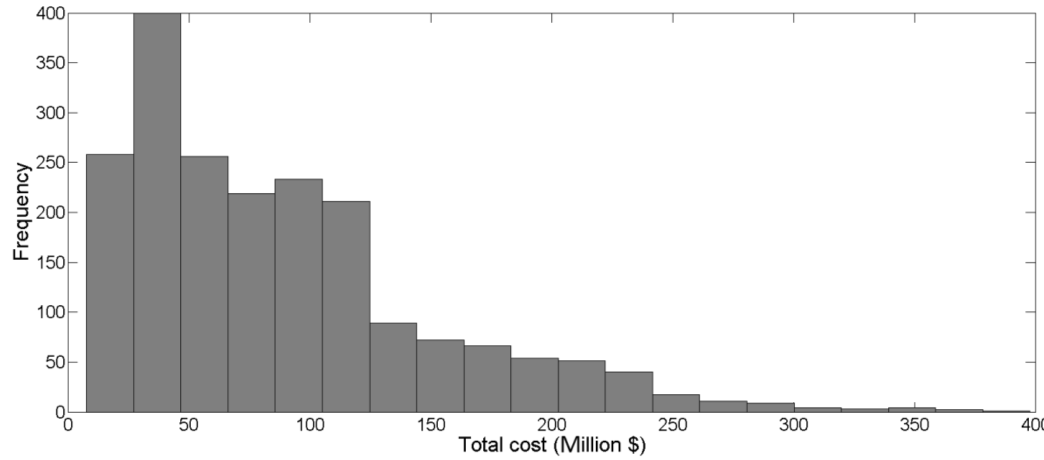


Figure 4.7. Cost implication of the average growing season TP load uncertainty associated with NPS TP load variability.

In the face of the vast uncertainty, adapting the TMDL to changing conditions based on routine monitoring campaigns and periodic assessment of the NPS pollutant loads can help increase cost-effectiveness and reliability of the plan. Periodic updating of the TMDL plan can result in potentially significant cost savings by improving the timing of the required BMP investments. Figure 4.8 illustrates an adaptive least-cost BMP plan for the Kalamazoo River watershed using the projected average growing season TP loading of the Lake Allegan for three planning periods of 1998-2008, 2008-2018, and 2018-2028, assuming rapid socioeconomic growth. For comparison, the corresponding plan based on the lake's period-of-simulation (1998-2028) projected average growing season TP has been provided as well. As shown in the figure, a combination of maximum amount of agricultural buffer strip and conservation tillage, along with about 45% of maximum possible amount of grassed swale with buffer strip, is selected for TP load reduction during the period 1998-2008. In the absence of policies limiting high-impact urbanization, the increasing trend of TP load will require implementation of additional BMPs (e.g., more grassed swale with buffer strip, retention basin, and detention basin) that are not part of the BMP plan for 1998-2028, which is comprised of agricultural buffer strip, conservation tillage, and grassed swale with buffer strip.

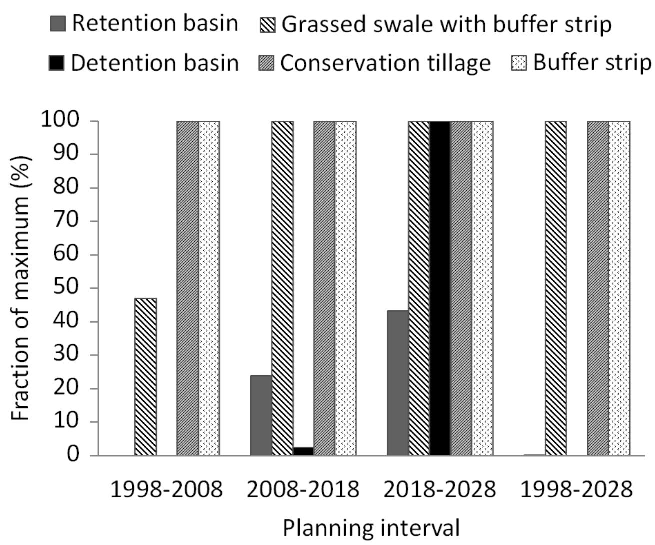


Figure 4.8. Required BMPs (fraction of maximum amount) for meeting the target TP concentration (60μg/l) under rapid socioeconomic growth

4.5.4. The Kalamazoo River watershed's TMDL

The Lake Allegan's TMDL program (Kieser and Associates, 2001) is a good example of thorough implementation of the present water quality-based approach to pollution reduction (US EPA, 1991), as it closely follows all the legal requirements for approval by the US EPA. These requirements include impairment designation, identification of PSs and NPSs discharging the pollutant of concern, description of the water quality target, determination of loading capacity and allowable loads from PSs and NPSs considering seasonal variations, estimation of an appropriate margin of safety, and adequate monitoring (US EPA, 1991). Although not a requirement for approval, TMDL planners are also strongly encouraged to provide an implementation plan with reasonable assurances for NPS load reduction on a voluntary basis (US EPA, 1991). The TMDL development process for Lake Allegan was completed through close stakeholder involvement within a participatory framework, facilitating a “cooperative agreement” between PS and NPS polluters whereby PSs (e.g., wastewater and industrial dischargers) assist NPSs (e.g., farmers) to reduce TP loads (KRLATIC, 2002).

Clearly, the voluntary NPS TP reduction in the Kalamazoo River watershed has not been on par with that of PS reductions (Kieser and Associates, 2011), which may be part of the reason why the lake’s recovery appears to be slower than anticipated. There is a growing need for more aggressive NPS TP load reduction policies because of the significant increase in the marginal cost of additional PS abatement. However, the command and control approaches to environmental management are particularly difficult to apply to “wicked” problems such as NPS pollution due to potentially substantial socioeconomic implications. Centner et al. (1999) investigated the challenges of BMP implementation from a legal institution perspective, and concluded that NPS pollution abatement may be prohibitively costly to agricultural producers, indicating that government intervention will be necessary for nutrient load reduction. The results of the developed BMP optimization model of the Kalamazoo River watershed corroborate this notion. Using average growing season TP loads and assuming a moderate socioeconomic growth scenario, the implementation cost of agricultural BMPs, excluding the cost of land acquisition and opportunity cost of land, is ~$50 million, providing a reliability index of slightly greater than 80%. Model results indicate that more stringent NPS TP reduction with significantly higher costs would be necessary for achieving higher reliability.

The presented simulation-optimization framework can facilitate application of systems thinking and environmental systems analysis techniques to the Lake Allegan TMDL program. The costs associated with BMP implementation should be viewed as a necessary environmental investment in order to maintain beneficial uses of the lake, avoiding the negative impact of environmental deterioration on socioeconomic growth. A potential area of improvement for the current TMDL program is projection of future socioeconomic growth and its potential implications for attainability of the water quality target. Furthermore, while the Lake Allegan's TMDL implementation plan has recognized the importance of identifying cost-effective abatement strategies and funding policies, systematic methods have not been applied. The full financial burden of BMP implementation may threaten to slow down the local socioeconomic growth. Systematic BMP optimization using sufficient site-specific water quality and BMP performance information can facilitate securing of adequate external funds through federal and state sources (e.g., cost shares for agricultural BMPs), which can balance the negative impact on the area's socioeconomic growth (e.g., US EPA, 2003).

4.5.5. Limitations

Simplifications are inherent in any modeling effort (Box and Draper, 1987), causing models to be different from reality in infinitely many ways (Sterman, 2000). In addition to the significant uncertainties involved in the TMDL studies, the limitations of the presented quantitative-qualitative methodology for finding least-cost BMPs need to be recognized in order to provide realistic insights for TMDL policy. The Lake Allegan SD model is a high-level simulation tool. Significantly more socioeconomic and biophysical details could be used for representing different feedback loops subject to data availability and the desired level of sophistication for characterizing the system-wide processes driving the degradation of aquatic systems. Socioeconomic growth, land use change over time, and hydrologic conditions have been simulated in a lumped fashion at the watershed-scale, and potential changes in climatic conditions have not been considered. Some of the links in the integrated SD simulation models are difficult to quantify, e.g., the negative link between environmental degradation and socioeconomic growth, which creates a balancing biophysical feedback loop.

Another limitation is the use of literature-reported cost functions and the BMP-specific TP removal effectiveness coefficients in the optimization model. Wide ranges of variability have been reported in the literature for TP removal effectiveness coefficient of different BMPs (e.g., Bottcher et al., 1995). The uncertainty associated with lack of information about the actual performance of different BMPs can influence the choice of the least-cost BMP set, affecting the reliability of meeting the target. Site-specific analyses should be conducted for actual BMP implementation, accounting for the local variability of BMP implementation cost, cost of land acquisition, and opportunity cost of land.

Despite the existing limitations, the methodology facilitates investigation of the tradeoffs between BMP implementation cost and attainable water quality targets. The use of GA allows for exploration of a wide range of possible BMP sets with non-linear cost-functions that capture economies of scale, whereas linear watershed scale cost functions should be used when applying a LP model. Stochastic elements can be incorporated in the optimization framework using site-specific ranges to characterize TMDL uncertainty and associated cost implications, although stochastic optimization using GA is computationally intensive. The screening level optimization model provides insights to least-cost BMPs and the cost of NPS TP load reduction, guiding cooperative agreements between individual PS and NPS polluters, as well as more extensive environmental offset programs facilitating TMDL implementation. Projection of the long-term adequacy of the proposed load reductions and potential impacts of socioeconomic growth on attainment of the target helps strategize BMP implementation policies.

4.6. Conclusions

The simulation-optimization framework presented herein facilitates application of systems thinking and environmental systems analysis to TMDL programs. A screening-level deterministic optimization model was used in conjunction with a system dynamics simulation model to investigate TP load reduction in the Kalamazoo River watershed. The results suggest that there is significant potential for cost-effective TP load abatement mostly through agricultural BMPs, as policies requiring substantial (>20%) urban TP load reduction are sub-optimal compared with greater abatement of agricultural loads. However, without sufficient support policies, the NPS pollution abatement required for achieving the Lake Allegan's TP concentration target with high reliability can be prohibitively costly to agricultural producers, indicating the need for government intervention, as well as potential for environmental offset programs between urban and agricultural areas. Furthermore, while the lake's TP concentration is primarily governed by intra- and inter-annual hydrologic variability, the socioeconomic growth of the watershed negatively affects the reliability of meeting the water quality goal, if the target were to be achieved using TMDLs designated assuming static conditions.

Implementation of TMDLs as required by the US CWA is a good opportunity for application of a long-term systems approach for reducing NPS pollution. Based on the US EPA guidelines for a water quality-based approach to pollution abatement, the Kalamazoo River watershed's TMDL program has been a successful initiative. PS polluters have been reasonably successful in reducing TP emissions. However, in spite of a collective watershed-scale effort using a participatory and cooperative framework, the voluntary NPS TP load abatement has only been partially successful, possibly delaying the attainment of the water quality goal. More aggressive TMDL implementation policies are needed to address the NPS TP emissions. The Lake Allegan program can be improved by accounting for future socioeconomic growth trends and its potential implications for attainability of the water quality target. Furthermore, the program should be equipped with environmental systems analysis methods in order to identify cost-effective abatement strategies, as well as equitable funding policies.

4.7. References

Arabi, M., Govindaraju, R. S., Hantush, M. M. (2006). Costeffective allocation of watershed management practices using a genetic algorithm. *Water Resources Research* 42, W10429.

Booth, D.B., Jackson, C.R. (1997). Urbanization of aquatic systems: degradation thresholds, stormwater detection, and the limits of mitigation. *Journal of American Water Resources Association* 33, 311–23.

Bottcher, A.B., Tremwell, T., Campbell, K.L. (1995). Best management practices for water quality improvement in the Lake Okeechobee watershed. *Ecological Engineering* 5, 341-356.

Boyd, J. (2000). The new faces of the Clean Water Act: A critical review of the EPA's proposed TMDL rules (discussion paper 00-12). Resources for the Future, Washington, DC.

Box, G.E.P., Draper, N.R. (1987). Empirical Model-building and Response Surfaces. John Wiley, New York.

Carpenter, S.R., Caraco, N.F., Correll, D.L., Howarth, R.W., Sharpley, A.N., Smith, V.H. (1998). Nonpoint pollution of surface waters with phosphorus and nitrogen. *Ecological Applications* 8, 559–68.

Cave, K., Harold, E., Quasebarth, T. (1996). Preliminary pollution loading projections for Rouge River watershed and interim nonpoint source pollution control plan, Technical Report RPO-NPS-TR07.00, Rouge River National Wet Weather Demonstration Project, Wayne County, Michigan.

Centner, T.J. , Houston, J.E., Keeler, A.G., Fuchs, C. (1999). The Adoption of Best Management Practices to Reduce Agricultural Water Contamination Limnologica 29, 366–373.

Chen, C.F., Ma, H.W., Reckhow, K.H. (2007). Assessment of water quality management with a systematic qualitative uncertainty analysis. *Science of the Total Environment* 374 (1), 13–25.

Correll, D.L. (1998). The role of phosphorus in the eutrophication of receiving waters: a review. *Journal of Environmental Quality* 27, 261-266.

DeJong-Hughes, J. and Vetsch, J. (2007). On-farm comparison of conservation tillage systems for corn following soybeans. University of Minnesota Extension Bulletin 08483.

Deng, X., Zhao, Y., Wu, F., Lin, Y., Lu, Q., Dai, J. (2011). Analysis of the trade-off between economic growth and the reduction of nitrogen and phosphorus emissions in the Poyang Lake Watershed, China. *Ecological Modeling* 222, 330–336.

Forrester, J. (1961) Industrial dynamics. MIT Press, Cambridge.

Forrester, J. (1969) Urban dynamics. MIT Press, Cambridge.

FWPCA (1972). Federal water pollution control act, as amended (33 U.S.C. 1151 et seq.).

FWPCA (2002). Federal water pollution control act, as amended (33 U.S.C. 1251 et seq.).

Gburek, W.J., Sharpley, A.N., Heathwaite, L., Folmar, G.J. (2000). Phosphorus management at the watershed scale: A modification of the phosphorus index. *Journal of Environmental Quality* 29, 130–144.

Gelda, R. K., Effler, S. W., O'Donnell, S. M. (2001). Probabilistic model of ammonia and toxicity status for urban lake. *Journal of Water Resources Planning Management* 127(5), 337–347.

Giri, S., Nejadhashemi, A.P., Woznicki, S.A. (2012). Evaluation of targeting methods for implementation of best management practices in the Saginaw River Watershed. *Journal of Environmental Management* 103, 24-40.

Gitau, M.W., Veith, T.L., Gburek, W.J. (2004). Farm-level optimization of BMP placement for cost-effective pollution reduction. *Transaction of The American Society of Agricultural and Biological Engineers* 47 (6), 1923–1931.

Gooday, R. D., Anthony, S. G., Chadwick, D. R., Newell-Price, P., Harris, D., Duethmann, D., Fish, R., Collins, A. L., Winter, M. (in press). Modelling the cost-effectiveness of mitigation methods for multiple pollutants at farm scale. *Science of The Total Environment*, doi: 10.1016/j.scitotenv.2013.04.078.

Haygarth, P. M., Condron, L. M., Heathwaite, A. L., Turner, B. L., Harris, G. P. (2005). The phosphorous transfer continuum: linking source to impact with an interdisciplinary and multiscaled approach. *Science of the Total Environment* 344, 5-14.

Heaton, S. (2001). Total maximum daily load (TMDL) for total phosphorus in Lake Allegan. Surface Water Quality Division, Michigan Department of Environmental Quality. Lansing, Michigan.

Heathwaite, A.L., Quinn, P.F., Hewett, C.J.M. (2005). Modelling and managing critical source areas of diffuse pollution from agricultural land by simulating hillslope flow connectivity. *Journal of Hydrology* 304(1– 4), 446– 61.

Hein, L.G. (2006). Cost-efficient eutrophication control in a shallow lake ecosystem subject to two steady states. *Ecological Economics* 59 (4), 429–439.

Hsieh, C.D., Yang, W.F. (2007). Optimal nonpoint source pollution control strategies for a reservoir watershed in Taiwan. *Journal of Environmental Management* 85, 908–917.

Keller, A. A., Cavallaro, L. (2008). Assessing the US Clean Water Act 303(d) listing process for determining impairment of a waterbody. *Journal of Environmental Management* 86, 699–711.

Kieser and Associates (2001). Non-point source modeling of phosphorus loads in the Kalamazoo River/Lake Allegan watershed for a total maximum daily load. Kieser and Associates Environmental Science and Engineering, Kalamazoo, Michigan.

Kieser and Associates (2011). Summary of 2010 total phosphorus loading to the Kalamazoo River and Lake Allegan with comparisons to historic data. Kieser and Associates Environmental Science and Engineering, (http://kalamazooriver.net/) (Oct. 3, 2011).

Kleinman, P.J.A., Sharpley, A.N., McDowell, R.W., Flaten, D.N., Buda, A.R., Tao, L., Bergstrom, L., Zhu, Q. (2011). Managing agricultural phosphorus for water quality protection: principles for progress. Plant and Soil 349,169-182.

Kalamazoo River/Lake Allegan TMDL Implementation Committee (KRLATIC) (2002). Water quality improvement (implementation) plan for the Kalamazoo River watershed and Lake Allegan through a phosphorus total maximum daily load (TMDL) process, The Kalamazoo River/Lake Allegan TMDL Implementation Committee, Kalamazoo, MI.

Lee, G. F., Rast, W., Jones, R. A. (1978). Eutrophication of water bodies: insights for an age-old problem. *Environmental Science and Technology* 12, 900–908.

Lee, G.J., Selvakumar, A., Alvi, K., Riverson, J., Zhen, X.J., Shoemaker, L., Lai, F. (2012). A watershed-scale design optimization model for stormwater best management practices. *Environmental Modelling & Software* 37, 6-18.

Mainstone, C.P., Parr, W. (2002). Phosphorus in rivers — ecology and management. *Science of the Total Environment* 282–283, 25–47.

Maringanti, C., Chaubey, I., Popp, J. (2009). Development of a multiobjective optimization tool for the selection and placement of best management practices for nonpoint source pollution control. *Water Resources Research* 45 (6), W06406.

Meadows, D.H., Meadows, D.L., Randers, J., Behrens, W.W. (1972). The limits to growth. Universe Books, New York.

MDNR (1987). Remedial action plan for the Kalamazoo River. Michigan Department of Natural Resources, Bay City, Michigan.

Michalak, A.M., et al. (2013). Record-setting algal bloom in Lake Erie caused by agricultural and meteorological trends consistent with expected future conditions. *Proceedings of the National Academy of Sciences of the United States of America* 110(16), 6448-6452.

Milon, J.W. (1987). Optimising NPS controls in water quality regulation. *Journal of the American Water Resources Association* (3), 387–396.

Mirchi, A., Madani, K., Watkins D, Ahmad, S. (2012). Synthesis of system dynamics tools for holistic conceptualization of water resources problems. *Water Resources Management* 26(9), 2421-2442.

Mirchi, A., Watkins, D.W. Jr. (in press). A systems approach to holistic TMDL policy: Case of Lake Allegan, Michigan. *Journal of Water Resources Planning and Management*, doi: 10.1061/(ASCE)WR.1943-5452.0000292.

Paerl, H.W., Hall, N.S., Calandrino, E.S. (2011). Controlling harmful cyanobacterial blooms in a world experiencing anthropogenic and climatic-induced change. *Science of the Total Environment* 40, 1739-1745.

Panagopoulos, Y., Makropoulos, C., Mimikou, M. (2011). Diffuse surface water pollution: driving factors for different geoclimatic regions. Water *Resources Management* 25 (14), 3635-3660.

Panagopoulos, Y., Makropoulos, C., Mimikou, M. (2012). Decision support for diffuse pollution management. *Environmental Modelling & Software* 30, 57-70.

Pijanowski, B.C., Brown, D.G., Shellito, B.A., Manik, G.A. (2002). Using neural networks and GIS to forecast land use changes: a land transformation model. Computers. *Environment and Urban Systems* 26 (6), 553–575.

Rao, N.S., Easton, Z.M., Schneiderman, E.M., Zion, M.S., Lee, D.R., Steenhuis, T.S. (2009). Modeling Watershed-scale effectiveness of agricultural best management practices to reduce phosphorus loading. *Journal of Environmental Management* 90, 1385–1395.

Reid, N.J., Hamilton, S.K. (2007). Controls on algal abundance in a eutrophic river with varying degrees of impoundment (Kalamazoo River, Michigan, USA). *Lake and Reservoir Management* 23, 219–230.

Rodriguez, H.G., Popp, J., Maringanti, C., Chaubey, I. (2011). Selection and placement of best management practices used to reduce water quality degradation in Lincoln Lake watershed. *Water Resources Research* 47, 1-13.

Sample, D.J., Heaney, J.P., Wright, L.T., Fan, C.Y., Lai, F.H., Field, R. (2003). Cost of best management practices and associated land for urban stormwater control. *Journal of Water Resources Planning and Management* 129(1), 59-68.

Scholes, L., Revitt, D.M., Ellis, J.B. (2008). A systematic approach for the comparative assessment of stormwater pollutant removal potentials. *Journal of Environmental Management* 88(3), 467-478.

Schwartz, S. S. (2010). Optimization and decision heuristics for Chesapeake Bay nutrient reduction strategies. *Environmental Modeling and Assessment* 15(5), 345–359.

Senge, P. (1990). The fifth discipline: the art & practice of the learning organization. Doubleday/Currency, New York.

Sharpley, A.N. (1994). Managing agricultural phosphorus for protection of surface waters: Issues and options. *Journal of Environmental Quality* 23, 437–451.

Shirmohammadi, A., Chaubey, I., Harmel, R.D., Bosch, D.D., Muñoz-Carpena, R., Dharmasri, C., Sexton, A., Arabi, M., Wolfe, M.L., Frankenberger, J., Graff, C., and Sohrabi., T.M. (2006). Uncertainty in TMDL models. *Transaction of The American Society of Agricultural and Biological Engineers* 49(4), 1033−1049.

Simonovic, S.P. (2009). Managing water resources: methods and tools for a systems approach. UNESCO, Paris and Earthscan James & James, London.

Simpson, T.W., Weammert, S.E. (2009). Developing nitrogen, phosphorus, and sediment reduction efficiencies for tributary strategy practices. BMP Assessment: Final Report. < http://archive.chesapeakebay.net/pubs/bmp/BMP_ASSESSMENT_FINAL_ REPORT.pdf> (accessed on May 10, 2013).

Smith, V.H. (2003). Eutrophication of freshwater and marine ecosystems: a global problem. *Environmental Science and Pollution Research*10, 126–139.

Stagge, J.H., Davis, A.P., Jamil, E., Kim, H. (2012). Performance of grass swales for improving water quality from highway runoff. *Water Research* 46(20), 6731–42.

Sterman, J.D. (2000). Business dynamics, systems thinking and modeling for a complex world. McGraw-Hill, Boston.

Strecker, E.W., Quigley, M.M., Urbonas, B.R., Jones, J.E., Clary, J.K. (2001). Determining urban storm water BMP effectiveness. *Journal of Water Resources Planning and Management* 127 (3), 144–149.

Tsihrintzis, V. A., Hamid, R. (1997). Modeling and management of urban stormwater runoff quality: A review. *Water Resources Management* 11(2), 137–164.

US BEA (2011). U.S. economic accounts. U.S. Bureau of Economic Analysis, (http://www. bea.gov/index.htm) (Aug.3, 2011).

US BLS (2009). Customer price index. U.S. Bureau of Labor Statistics, (http://www. bls.gov/cpi/cpifact8.htm) (Aug. 5, 2011).

US EPA (1991). Guidance for water quality-based decisions: TMDL process. Rep. No. EPA-440-4-91-001, Office of Water, Washington, D.C.

U.S. EPA (2003). Economic analyses of nutrient and sediment reduction actions to restore Chesapeake Bay water quality. U.S. EPA Region III- Chesapeake Bay Program Office, Annapolis, MD.

US EPA (2009). National water quality inventory report to congress: 2004 reporting cycle. EPA 841-R-08-001. US EPA Office of Water Washington, DC 20460.

U.S. EPA (2010). Chesapeake Bay phase 5.3 community watershed model EPA 903S10002 - CBP/TRS-303-10. Chesapeake Bay Program Office, Annapolis MD.

U.S. EPA (2012). An optimization approach to evaluate the role of ecosystem services in Chesapeake Bay restoration strategies. U.S. EPA/600/R-11/001

US EPA (2013). National rivers and streams assessment 2008–2009: a collaborative survey, EPA/841/D-13/001, Office of Wetlands, Oceans and Watersheds. Washington, DC 20460.

Vollenweider R.A. (1976) Advances in defining critical loading levels for phosphorus in lake eutrophication. Memorie dell'Istituto Italiano di Idrobiologia 33, 53–83.

Veith, T. L., Wolfe, M. L., Heatwole, C. D. (2003). Development of optimization procedure for cost-effective BMP placement. *Journal of American Water Resources Association* 39(6), 1331−1343.

Webster, J.R., Wallace, J.B., Benfield, E.F. (1995). Organic processes in streams of the eastern United States. In: Cushing, C.E., Cummins, K.W., and Minshall, G.W. (eds), Ecosystems of the world, 22, River and stream ecosystems. Elsevier, Amsterdam, 117-187.

Weiss, P., Gulliver, J. S., Erickson, A. J. (2007). Cost and pollutant removal of storm-water treatment practices. *Journal of Water Resources Planning and Management* 133(3), 218–229.

Wesley, J.K. (2005). Kalamazoo River assessment. Michigan Department of Natural Resources, Fisheries Division. Special Report 35, Ann Arbor, Michigan.

Winz, I., Brierley, G., and Trowsdale, S. (2009). The use of system dynamics simulation in water resource management. *Water Resources Management* 23(7), 1301–1323.

Withers, P.J.A., Jarvis, G.C. (1998). Mitigation options for di€use P loss to water. *Soil Use and Management* 14, 186-192.

Wuycheck, J. (1998). Water quality and pollution control in Michigan. Surface Water Quality Division, Rep. No. MI/DEQ/SWQ-98/030, Michigan 305(b) Rep., Michigan Dept. of Environmental Quality, Lansing, MI.

Young, G. K., Stein, S., Cole, P., Kammer, T., Graziano, F., Bank, F. (1996). Evaluation and Management of Highway Runoff Water Quality. Office of Environment and Planning, FHWA, Washington, D.C.

Zhen, X.Y., Yu, S.L., Lin, J.Y. (2004). Optimal location and sizing of stormwater basins at watershed scale. *Journal of Water Resources Planning and Management* 130 (4), 339-347.

Chapter 5 - Market-based policy instruments for mitigating agricultural phosphorus loads in the Maumee Basin[5]

5.1. Introduction

The implementation of pollutant reduction programs (e.g., TMDLs) has been particularly difficult for agricultural NPS pollution due to formidable practical and institutional limitations. At the same time, the marginal cost for achieving more stringent PS pollution abatement is increasing significantly, and the nation's aquatic systems continue to be challenged by anthropogenic impairment. Thus, NPS pollution, especially from agricultural lands, is becoming the focus of great scrutiny due to its heightened relative importance as the driver of eutrophication. However, the NPS requirements of the Clean Water Act have been described in the literature as "not enough carrot, not enough stick" because adequate funding and other economic incentives, as well as noncompliance penalties, are conspicuously lacking (Zaring, 1996).

While any success in reducing nutrient loads from agricultural lands will depend on collective support of individual farmers, the voluntary pollution reduction initiatives will most likely remain ineffective except for the cases where farmers are environmentally aware and value environmental stewardship more than economic profit. On the other hand, the traditional command and control approaches to environmental management are particularly difficult for "wicked" problems such as agricultural NPS pollution, which may have extensive socioeconomic implications. Nonetheless, given the extent of

[5] This chapter is being considered for publication as Mirchi, A., Watkins, D.W., Meredith Ballard-LaBeau, Daya Muralidharan, and Alex Mayer. Market-based policy instruments for mitigating agricultural phosphorus loads in the Maumee Basin . *Environmental Management*.

impairment of the water bodies, devising mechanisms to address NPS pollution from agricultural sources is becoming inevitable.

Anthropogenic eutrophication is a market failure or policy failure (Turner et al. (1999). A clean, well-functioning aquatic system is beneficial to all for use, but not directly profitable for any given interest group to bring about positive change in times of degradation. Thus, using appropriate policy instruments to manage agricultural NPS pollution, including standard-setting and market-based instruments, is expected to be more effective than voluntary programs (Shortle and Horan, 2001). Standard-setting is a command and control approach that refers to enforcing stringent NPS control regulations by applying minimum required management measures. This approach may result in economic inefficiency of pollution abatement due to geographic variability of impact, coupled with the inflexibility and static nature of the regulatory regime. Market-based instruments, in theory, will encourage agricultural NPS polluters to reduce their impact either through a "polluter pays" tax-based approach for internalizing environmental externalities, or by providing other economic incentives such as environmental subsidies (Shortle and Horan, 2001). Moreover, flexible mechanisms such as water quality trading and environmental offset programs that may allow stakeholders within the agricultural sector to reduce their environmental impact without considerable loss of utility, can be considered along with standard-setting to further increase the efficiency of NPS pollution reduction.

Environmental taxes have long intrigued economists as a powerful potential means for addressing the anthropogenic environmental degradation. The primary rationale for a tax- and penalty-based environmental policy is the need for internalizing environmental externalities associated with a production process (Ekins, 1999). The inefficient nutrient management in the agricultural sector is a classic economic externality whereby farmers maximize their utility by applying abundant amounts of fertilizers, while the total cost of mitigating environmental consequences such as off-farm eutrophication is borne by others (e.g., downstream users). Ekins (1999) classifies environmental taxes into three categories that are not necessarily separable, including cost-covering charges, incentive taxes, and revenue-raising taxes. He splits the cost-covering charges into two types, i.e. user charges such as wastewater treatment charges, and earmarked charges, where generated revenues are spent on general environmental purposes such as mitigating environmental degradation. The incentive taxes are applied merely to change an environmentally damaging behavior. The revenue-raising taxes are those that generate revenues in excess of cost-covering charges, which can be allocated towards purposes other than environmental services (e.g., creating job opportunities). Different categories of environmental taxes have received attention in the European OECD countries (Ekins, 1999).

Watershed-based management approaches, and capabilities for characterizing biophysical aspects of pollutant loading and transport at the watershed scale, bear promise for advancing towards pollution taxes. The ultimate goal is reduce to the total load of a given pollutant below a certain allowable amount that the downstream water body can receive without violating stipulated standards. Quantifying and monitoring the impact of each individual farm is at this point infeasible, hindering the use of the "polluter pays" approach. On the other hand, application of input-based taxes as the "second-best" policy option should be grounded on some sort of biophysical and agro-economic analysis to avoid over- or under-charging. In large watersheds, redefining the scale of the problem to an appropriate aggregate level, i.e., sub-watersheds whose impact on the nutrient load from the large watershed can be quantified, can help overcome some of the technical challenges of implementing an efficient tax-based policy. The pollution tax for the sub-watershed can be obtained based on the required investment in BMP implementation for that sub-watershed such that, collectively, the large watershed complies with TMDL requirements of the downstream water body. This information can be used to obtain reasonable input-based taxes, such as fertilizer use tax which can generate revenues for application of BMPs where they are most effective. This chapter provides insights into the implications of the scale of TMDL programs on cost efficiency of TP load reduction, discussing the potential for imposition of input-based agricultural tax as a market-based policy instrument for reducing NPS TP loads.

5.2. Method

Market-based instruments as a potential eutrophication management policy should be analyzed in conjunction with a biophysical model that, despite inherent modeling limitations due to uncertainty and abstraction of natural processes, can adequately characterize the transport of pollutants in the aquatic system. This chapter presents the results of an agro-economic BMP optimization model developed for the Maumee River watershed. The study area and various components of the policy analysis framework are discussed in this section.

5.2.1. Problem definition and study area

Accelerating re-eutrophication of Lake Erie over the last decade has raised concern as to adequacy of water quality management programs in this basin. Lake Erie is the shallowest, warmest, and the most biologically productive of the Laurentian Great Lakes. It covers an area of about 25,700 km^2, and has a volume of about 1,640 km^3. The lake was hypereutrophic in the 1960s and 1970s due to excessive amounts of TP, the main cause of its environmental degradation (e.g., harmful algal blooms, beach closings, and drinking water contamination). Lake Erie's trophic turnaround boosted optimism about the success of phosphorus control programs (Makarewicz and Bertram, 1991; Ludsin et al., 2001), when nutrient load reduction and erosion control plans implemented by PSs and NPSs, respectively, decreased the lake's phosphorus load from 25,000 metric tonnes per year in the 1960s to the target load of 11,000 metric tonnes per year in 1995 (Ohio EPA, 2010). However, the eutrophication problem has reappeared since the mid 1990s, likely due to increased concentrations of dissolved reactive phosphorus in the runoff from agricultural nutrient applications, as well as changes in runoff patterns, i.e., increased fall and winter runoff (Ohio EPA, 2010).

Proper management of agricultural NPS phosphorus in the Maumee River Basin (Figure 5.1), the largest drainage basin (over 17,000 km^2) in the Great Lakes region, is critical for effective mitigation of environmental degradation in the western Lake Erie Basin (Ohio EPA, 2010; Lake Erie LaMP, 2009). Lake Erie is comprised of three basins with distinct physical characteristics, i.e., the shallow western basin (average depth ~7.4 m), the central basin (average depth ~18.3 m), and the eastern basin (average depth ~24 m). As of 2008, the Maumee River Basin was the largest contributor of TP to Lake Erie, discharging about 1,800 metric tonnes of TP (Lake Erie LaMP, 2009). Consequently, the western basin at the outlet of Maumee River, which is the most degraded and most vulnerable portion of the lake, has had episodes of harmful algal blooms in the 2000s. The extent of environmental degradation is comparable to the era before adoption of TP control programs. While TP loads from Maumee River have been decreased by upgrading wastewater treatment plants and implementation of agricultural BMPs (Chapra and Dolan, 2012), the TP concentration in the mid 2000s was about 300 μg/L (Lake Erie LaMP, 2009). This is an order of magnitude higher than 32 μg/L, which is the target average annual TP concentration for desired ecological conditions at the mouth of Lake Erie tributaries (Lake Erie LaMP, 2009). Likewise, the concentration exceeded the TP concentration target of 170 μg/L recommended by the State of Ohio for rivers with a drainage area ranging between 500-1600 km^2 (Ohio EPA, 1999).

The phosphorus load transported through agricultural runoff from the Maumee River Basin is suspected to be the most significant driver of the recent harmful algal blooms (Ohio EPA, 2010). Based on land use and land cover classification of the basin using geospatial data for 2010 (USDA NASS, 2010), the Maumee River Basin consisted of over 76% agricultural row crops, of which about 85% is for production of corn, soybeans, and wheat. The rest of the watershed was urban and developed open spaces (12.3%), forest and shrubland/grassland (9%), and water and wetland (2.4%) (Figure 5.1). While the initial increasing trend of phosphate use for corn production in Ohio has leveled off since the late 1970s, occasional spikes are observed in the historical time series of phosphate use (Figure 5.2). The amount of phosphate use per acre of corn field has plateaued at a level significantly higher than that of the 1960s. As for soybeans and wheat, however, an increasing trend of phosphate use per acre of land has persisted until today.

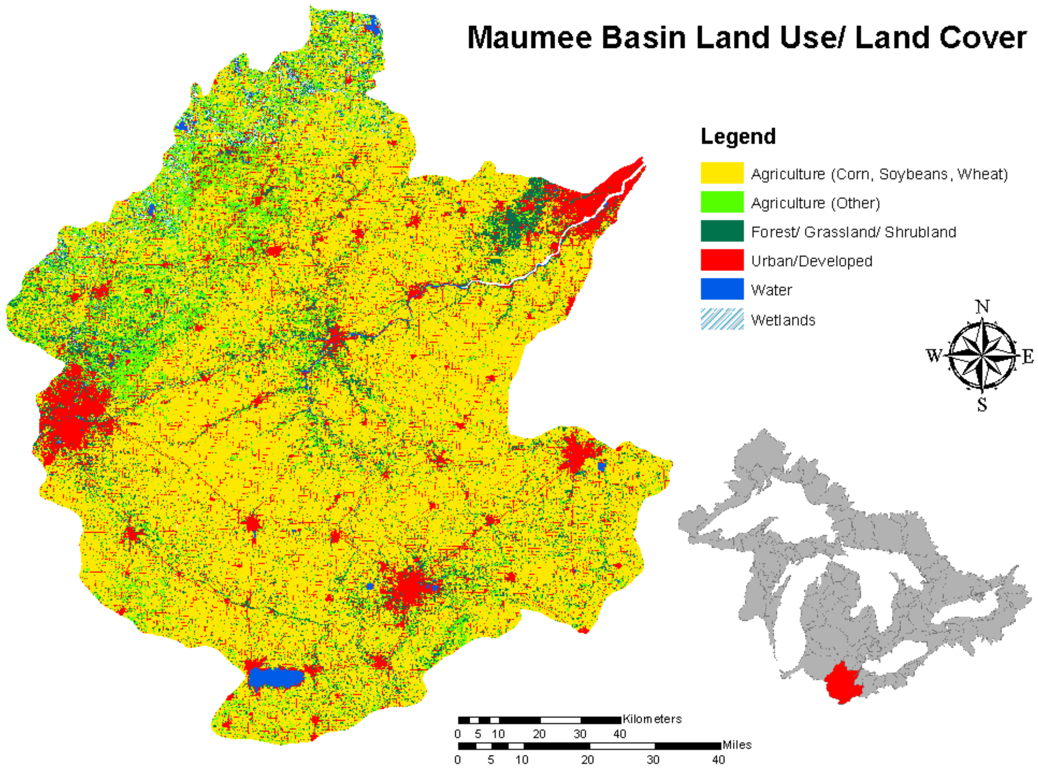


Figure 5.1.Maumee River Basin and its major land use and land cover (Source of data: USDA NASS, 2010).

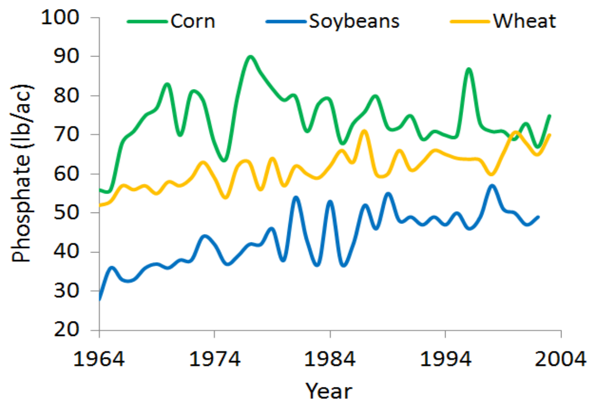


Figure 5.2. Historical trends of phosphate application for production of corn, soybean, and wheat in Ohio (Source of data: USDA ERS).

5.2.2. Biophysical model

Robertson and Saad (2011) estimated phosphorus (P) and nitrogen (N) loads, as well as sources of these pollutants, in the U.S. portion of the Great Lakes. They developed SPAtially Referenced Regressions On Watershed attributes (SPARROW) models for the Upper Midwest, reporting that about 50% of the Lake Erie's TP load in 2002 was contributed by the Maumee Basin with its intense agriculture. The SPARROW model predicts annual TP loads at the outlet of watersheds using transport process and mass balance relationships within a GIS-based watershed model. The sources of TP quantified at the catchment (watershed) level include point sources, confined manure, unconfined manure, farm fertilizers, and undefined inputs from urban/developed open lands and forested areas. Table 5.1 summarizes the 2002 TP loads for the Maumee Basin, as well as its seven HUC-8 sub-watersheds.

Table 5.1. TP loads for the Maumee Basin, as well as its seven HUC-8 sub-watersheds estimated for 2002 (Robertson and Saad, 2011).

Watershed	Area (km^2)	Mean annual flow at outlet (million m^3/yr)*	TP load (kg)	Contribution by source (%)				Delivery ratio
				Urban	Agricultural	Forest	PS	
St. Joseph	2,780	792.48	138,424	11.1	68.7	5.4	14.9	0.97
St. Marys	2,133	608.04	278,478	5.9	85.3	0.7	8.1	0.99
Upper Maumee	991	1,683.02	97,959	7.9	33.4	0.9	57.7	0.99
Tiffin	2,026	577.54	83,910	10.7	75.3	4.2	9.9	0.99
Auglaize	4,314	1,810.16	309,622	8.1	71.3	1.1	19.6	0.99
Blanchard	2,036	580.39	114,672	10.2	71.2	1.3	16.5	0.96
Lower Maumee	2,778	4,862.63	269,222	9.3	28.0	0.6	62.2	1.00
Maumee	17,058	4,862.63	1,265,023	8.5	62.4	1.5	27.6	1.00

*Estimated using area weighted method based on annual flow data for the period 1930-2012 for Waterville station (USGS 04193500) which measures flow from over 95% (16,395 km^2) of the Maumee basin.

5.2.3. BMP and agro-economic optimization model

The biophysical model provides TP load data at different scales (e.g., sub-watersheds and watershed), allowing for investigation of watershed scale programs as compared with zone-based sub-watershed scale programs. The BMP optimization identifies the least cost BMP set considering urban and agricultural BMPs, using the model discussed in Chapter 4. The agricultural BMPs considered include constructed wetlands (CW), conservation tillage (CT), cover crops (CC), and buffer strips (BS) constructed on privately owned agricultural land. Furthermore, retention basins (RB), grassed swales and buffer strips (GSB), and detention basins (DB) are considered for urban areas.

The BMP cost functions given in Table 4.1 were linearized for use in the optimization model. Furthermore, the model uses TP load data and export coefficients obtained from the biophysical model (Table 5.1), as well as BMP-specific upper bounds to provide the minimum-cost BMP set subject to allowable TP load constraints at the watershed and sub-watershed scale. Tables 5.2 and 5.3 provide BMP specific upper bounds and area treatment coefficients estimated for the Maumee Basin and its HUC-8 sub-watersheds using the procedures described in Chapter 4.

Table 5.2. BMP specific upper bounds (XUB) estimated for the Maumee Basin and its HUC-8 sub-watersheds. Values estimated using the procedures described in Chapter 4.

Watershed	Upper bounds for Urban BMPs			Upper bounds for Agricultural BMPs			
	RB (10^3 m^3)	GSB (km)	DB (10^3 m^3)	CW (10^3 m^3)	CT (ha)	CC (ha)	BS (km)
St. Joseph	880	69	360	8,680	39,800	149,700	1,129
St. Marys	1,200	87	608	1,622	18,800	140,800	1,126
Upper Maumee	560	37	296	9,360	8,700	66,400	648
Tiffin	368	54	208	12,540	38,100	117,800	1,291
Auglaize	112	174	600	50,160	69,400	272,800	3,062
Blanchard	604	53	280	15,200	69,500	89,900	1,173
Lower Maumee	1,840	199	840	30,660	57,100	145,400	2,325
Maumee	7,120	673	3,400	148,600	311,200	1,022,400	10,887

Table 5.3. Area treatment coefficients (a_{ij}) estimated/assumed for the Maumee Basin and its HUC-8 sub-watersheds.

Watershed	Area treatment coefficient for Urban BMPs			Area treatment coefficient for Agricultural BMPs			
	RB (ha/m^3)	GSB (ha/m)	DB (ha/m^3)	CW (ha/m^3)	CT (ha)	CC (ha)	BS (ha/m)
St. Joseph	0.0025	0.40	0.0025	0.022	1	1	0.168
St. Marys	0.0025	0.16	0.0025	0.012	1	1	0.142
Upper Maumee	0.0025	0.39	0.0025	0.008	1	1	0.116
Tiffin	0.0025	0.29	0.0025	0.012	1	1	0.121
Auglaize	0.0025	0.26	0.0025	0.007	1	1	0.322
Blanchard	0.0025	0.38	0.0025	0.012	1	1	0.136
Lower Maumee	0.0025	0.38	0.0025	0.007	1	1	0.087
Maumee	0.0025	0.31	0.0025	0.01	1	1	0.118

Implementation of agricultural BMPs may affect the profitability of agricultural practices, which can be characterized using an agro-economic BMP optimization model. The agro-economic BMP optimization model uses crop production for major crop types in the Maumee River Basin (e.g., corn, soybean, and wheat) to estimate agricultural income. The objective function is to maximize utility while complying with environmental constraints on nutrient loads from the watershed and limits to available land for agricultural BMPs. The target TP load is calculated by multiplying the allowable TP concentration at the watershed outlet (i.e., point of discharge to Lake Erie) by average annual flow. The general mathematical formulation of the agro-economic model is as follows:

$$Max \sum_{i=1}^{i=I} U_{ic} \qquad (eq. 5.1)$$

subject to

$$L_{TP,BMP} \leq L_{TP,target} \qquad (eq. 5.2)$$

$$LA_c + LA_{BMP} \leq LA_{ag} \qquad (eq. 5.3)$$

where U = utility, $L_{TP,BMP}$ = TP load after BMP implementation, $L_{TP,target}$ = target TP load, LA_c = cultivated land area, LA_{BMP} land area allotted to BMP, LA_{ag} = agricultural land area.

Layard et al. (2006) demonstrated that utility can be modeled by a logarithmic function of income. Thus, the agro-economic model for TMDL implementation will be developed assuming that net income is a proxy for utility of farmers (eq. 5.4). Net income is calculated as total annual revenue less total production cost (eq. 5.5). The annual yield per hectare for each crop is assumed to be a function of fertilizer use (eq. 5.6). Irrigation water will not be considered in the crop production function because the production of major crops in the basin is rain-fed (Antosch, 2006). Similar to the BMP model, equations 5.7 and 5.8 give the post-BMP total phosphorus load.

$$U_c = f(I) \qquad (eq. 5.4)$$

$$I_c = \sum_{c=1}^{c=C} (P_c Y_c - C_c) \, . LA_c \qquad (eq. 5.5)$$

$$Y_c = f(F) \;\; = aF + b \qquad (eq. 5.6)$$

$$L_{TP,BMP} = Le - \sum_{c=1}^{c=C} Lr_c \qquad (eq. 5.7)$$

$$Lr_c = e_c \cdot \sum_{j=1}^{j=J} R_j \cdot X_{cj} \cdot a_{cj} \quad for\ c = 1,2,3 \qquad (eq. 5.8)$$

where c is the index for major crop types, i.e., corn, soybean, and wheat, I = Income, P_c = unit price (\$), Y_c = yield (Bu/ha), C_c = crop production cost (\$/ha), F = fertilizer (kg/ha), Le = existing TP load (kg/yr), Lr = reduced load (kg/yr), e_c = TP export coefficient, R_j = removal effectiveness coefficient of BMP j (dimensionless), X_{cj} = BMP-specific decision variable (e.g., length (m) or land area (ha)), and a_{cj} = area treatment coefficient (ha treated per unit of BMP installed).

It is assumed that agricultural TP load can be abated by reducing the amount of fertilizer applied on agricultural lands. In other words, fertilizer use has a direct impact on TP export coefficients. Fertilizer use for different crops for the year 2002 is considered as the reference for adjusting the export coefficients (eq. 5.9).

$$e_c = (F_c/F_{c,2002}) \times e_{c,2002} \qquad (eq. 5.9)$$

The total production cost comprises the regular production and operating cost, as well as the earmarked fertilizer tax to cover BMP cost. Assuming the funds for BMP implementation will be generated through imposition of a cost covering input (i.e., fertilizer use) tax, the cost function can be written as follows:

$$C_c = Cr_c + T_c \qquad (eq. 5.10)$$

$$T_c = \frac{C_{BMP,c}}{F} \qquad (eq. 5.11)$$

$$C_{BMP,c} = \sum_{j=1}^{j=J} C_{BMP,j}.\left(X_j\right) \qquad (eq. 5.12)$$

where Cr_c = regular production and operating costs (\$/ha-yr) including the cost of seed, labor, and interest on operating capital, T_c = fertilizer use tax for crop *c (\$/kg-ha)*, C_{BMP} = cost of BMP (\$), *j* = BMP type, including constructed wetland, conservation tillage, cover crop, buffer strip (*J* = 4).

The agronomic data for deriving the crop production functions were obtained from the United States Department of Agriculture (USDA ERS, 2012). The data sets include time series of corn, soybeans, and wheat yield (Bu/ha), acreage, fertilizer use (kg/ha), total economic costs of production (\$/ha), price of the crop (\$/Bu), and fertilizer price (\$/kg-ha) was estimated using the cost of fertilizer (\$) and amount of fertilizer applied per hectare of agricultural land. Precipitation data was obtained from the online portal of National Oceanic and Atmospheric Administration's National Climatic Data Center (NOAA NCDC, 2012). Based on the regression analyses, precipitation is not a significant predictor variable for crop production in the study area. These analyses suggest that production levels of major crops are linearly related to the amount of total fertilizer applied (r^2_{corn} = 0.46, $r^2_{soybean}$ = 0.46, r^2_{wheat} = 0.75). Figure 5.3 shows time series of observed and estimated yields for corn, soybean, and wheat in the state of Ohio.

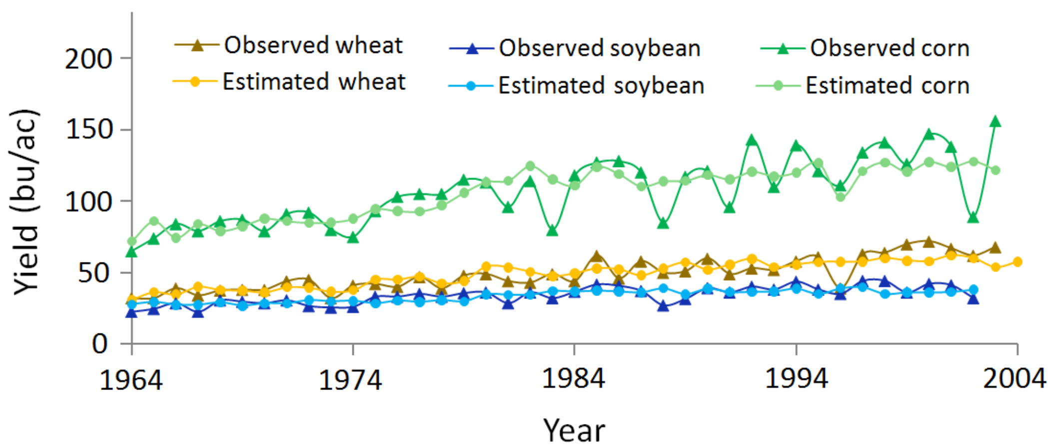


Figure 5.3.Observed and estimated yield for corn, soybean, and wheat in the state of Ohio (Source of data: USDA ERS, 2012).

5.3. Results and discussion

5.3.1. Scale dependence of BMP implementation efficiency

Applying the BMP optimization model at the watershed and sub-watershed scales provides insights for improving the cost efficiency of a TMDL program. Two implementation scenarios have been considered. In the first scenario, sub-watershed scale BMP implementation across individual sub-watersheds, the least-cost BMP set was obtained assuming that there is no interaction between sub-watersheds in terms of BMP implementation. However, the effect of the applied BMP set in an upstream sub-watershed was propagated downstream. This scenario represents lack of coordination in terms of BMP implementation across sub-watersheds. In the second scenario, a more flexible sub-watershed scale BMP implementation was modeled to meet the TP target concentration at the outlet of each sub-watershed while meeting the Maumee Basin's TP concentration target at the point of discharge to Lake Erie. In other words, this scenario allows for application of additional BMPs in an upstream sub-watershed to facilitate attainment of the water quality target at the outlet of a downstream sub-watershed. This scenario was named sub-watershed scale BMP implementation across the Maumee Basin, representing coordinated BMP implementation. Figure 5.4 shows the seven sub-watersheds in the Maumee Basin, and Table 5.4 provides land use characteristics of each sub-watershed and their water quality target.

Figure 5.4. Maumee Basin's sub-watersheds.

Table 5.4. Land use and load characteristics of the sub-watersheds.

Sub-watershed	Land use (km^2)				Urban export coefficient (kg/ha)	Agricultural export coefficient (kg/ha)
	Urban	Agricultural	Forested	Water/wetland		
St. Joseph	275	1898	348	241	0.56	0.5
St. Marys	279	1596	209	41	0.59	1.49
Upper Maumee	144	751	73	23	0.54	0.44
Tiffin	157	1,559	172	115	0.57	0.41
Auglaize	446	3,421	406	35	0.58	0.52
Blanchard	202	1,594	189	13	0.56	0.65
Lower Maumee	393	2,025	278	25	0.64	0.37
Maumee	1,896	12,844	1,675	493	0.57	0.61

Figure 5.5 illustrates the difference in cost efficiency of TP abatement under the two scenarios. The highest TP abatement cost efficiency (i.e., 1.5 kg TP reduced/ 1000$ spent) was achieved under coordinated watershed-scale BMP implementation across the Maumee Basin (scenario 2) while meeting the TP concentration target of 170 µg/L. In contrast, the basin-scale average of cost efficiencies for scenario 1 provided a low cost efficiency of 0.33 kg TP reduced/ 1000$ spent. An interesting observation when comparing scenarios 1 and 2 is that the cost efficiency of sub-watershed scale BMP implementation varies over a wider range under scenario 1 (0.08-3.55 kg TP reduced/ 1000$ spent) as compared with scenario 2 (0.87-3.55 kg TP reduced/ 1000$ spent). The amount of TP reduced in the St. Marys and Upper Maumee sub-watersheds increases under coordinated BMP implementation (scenario 2) because of opportunities for cost-efficient TP abatement in these sub-watersheds. In the downstream watersheds (e.g., Upper Maumee and Lower Maumee) TP abatement in the absence of coordination among sub-watersheds may be cost inefficient. Coordination with upstream sub-watersheds may facilitate target attainment in a downstream sub-watershed through additional BMP implementation in areas where TP load abatement may be achieved relatively inexpensively.

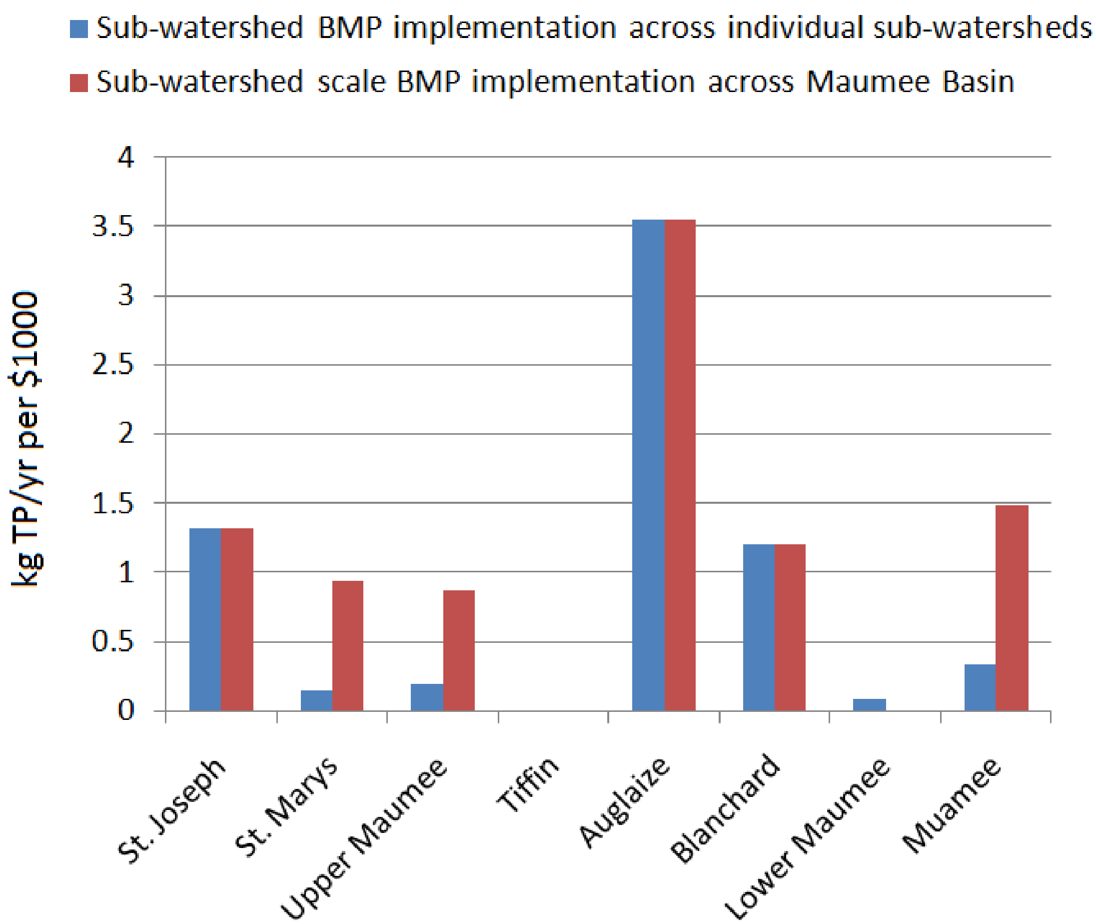


Figure 5.5. Comparison of cost efficiency between sub-watershed scale and watershed scale implementation of least-cost BMP set.

Agricultural BMPs dominate the least-cost BMP set for different sub-watersheds, as may be expected because agriculture is the primary largest land use in the study area. Agricultural buffer strips and constructed wetlands, as well as tillage practices, can result in significant TP load abatement in each of the Maumee Basin's sub-watersheds. In particular, development and proper maintenance of agricultural buffer strips should be considered as an important agricultural BMP. Potential TP load abatements resulting from BMPs implemented on sub-watershed scales across individual sub-watersheds and across the entire Maumee Basin, scenarios 1 and 2, are shown in Figure 5.6. The smallest TP load abatements occur in upstream sub-watersheds such as St. Joseph and Blanchard. At the current scale of analysis, Tiffin meets the TP concentration target without BMPs. In contrast, large TP export coefficients for St. Marys and Auglaize result in intensive BMP implementation in these sub-watersheds. Similarly, although BMP implementation may significantly reduce TP emission throughout the basin, the Lower Maumee sub-watershed will require extensive BMP implementation, in part because it receives a large TP load (~339,000 kg/yr) from upstream sub-watersheds. As shown in Figure 5.5b, a more flexible BMP implementation scheme can facilitate additional TP load abatement in areas with higher cost-efficiency (e.g., St. Joseph and Auglaize), denoting the importance of a basin approach to BMP implementation. Under scenario 2, the load abatement in upstream watersheds increased, facilitating the attainment of the TP concentration target of 170 μg/L in downstream sub-watersheds.

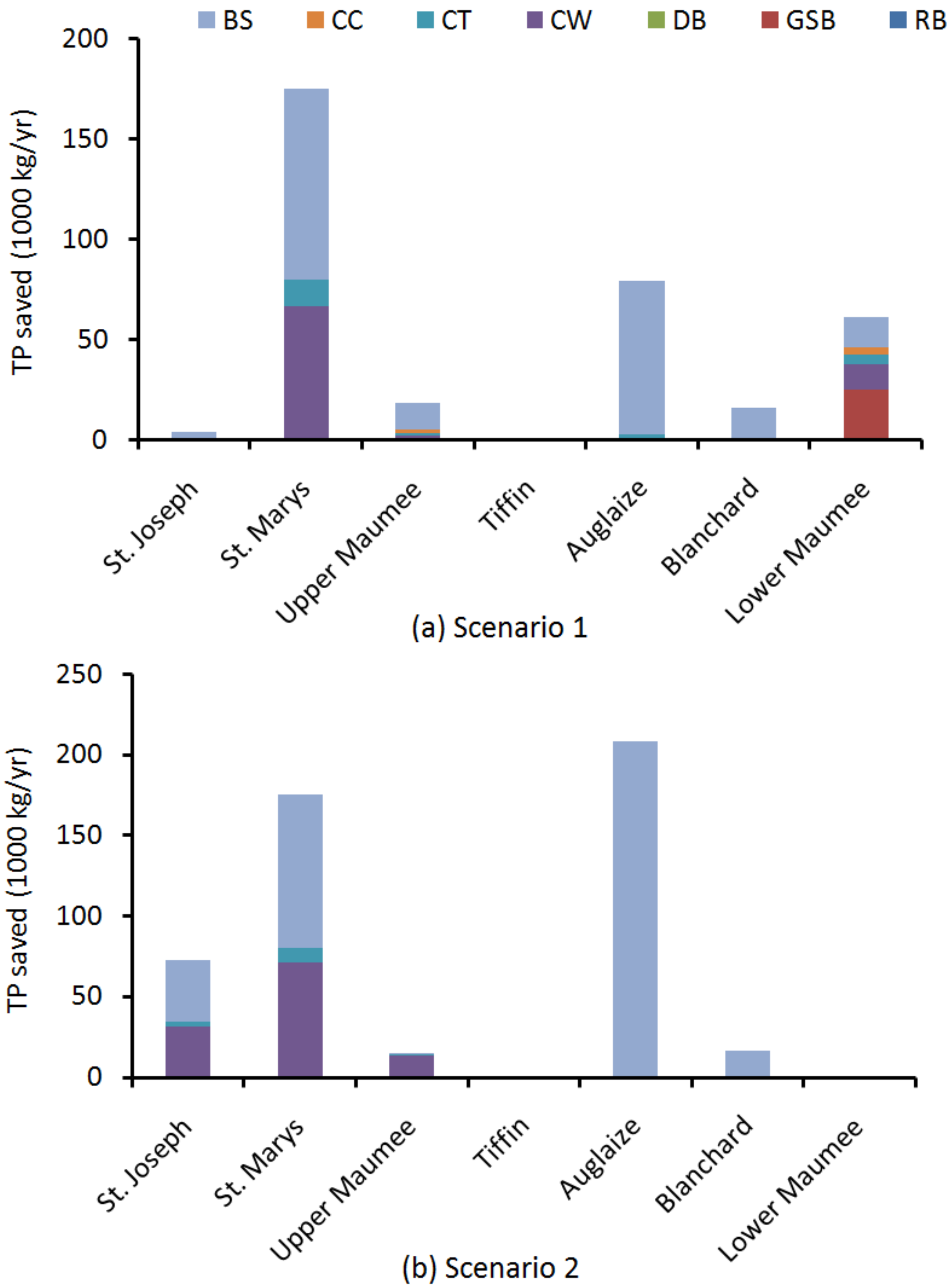


Figure 5.6. Potential TP load reduction resulting from BMPs implemented on sub-watershed scale across individual sub-watersheds (a) and across Maumee Basin (b).

5.3.2. Fertilizer tax as a TP abatement policy

The use of fertilizer tax as a potential TP abatement policy was investigated through maximization of net basin scale agricultural income under three scenarios: (1) fertilizer tax imposition when agricultural land may be partially cultivated (land under cultivation is a decision variable); (2) fertilizer tax imposition when at least 50% of agricultural land is cultivated; and (3) BMP cost is fully covered by external funds. Scenarios 1 and 2 capture the effects of fully internalizing the cost of TP load emissions considering the effect of land constraint, whereas scenario 3 represents a case where TP load abatement is achieved through government subsidy and without significant impacts on agricultural sector profits. The analysis was conducted at the Maumee Basin scale assuming that all the agricultural lands will be allotted to cultivation of corn, soybeans, and wheat. The land under cultivation of corn, soybeans, and wheat was scaled up to cover the entire agricultural area based on the 2010 cultivation area available from Crop Data Layer (CDL) (USDA NASS, 2010). As assumption was made that agricultural TP export coefficient comprises contributions from the three major crops. Average TP export coefficients for each of the three different crops were estimated based on fertilizer use and cultivation area such that the collective effect of the export coefficients would give the agricultural TP load estimated by SPARROW. A linear relationship was assumed relating the export coefficients to total fertilizer use for each crop. Table 5.4 provides the basin scale data for Maumee Basin's major crops including available land area for cultivation, agro-economic data, and average TP export coefficients.

The analysis results for the three noted policies are summarized in Table 5.5. Internalization of the cost of TP load abatement has a significant direct impact on agricultural production through decisions about cultivated land area and fertilizer use. The agricultural sector will strive to maintain the highest possible yield while meeting the constraints of the TP load abatement policy. Under the extreme policy of full internalization of TP load abatement cost through fertilizer tax imposition using land as a decision variable (policy scenario 1), only a very small proportion of the available land is cultivated for soybeans (1.45%) and wheat (1.39%), while most available land was used for corn production (82.22%). When an additional constraint is imposed requiring the use of at least 50% of available agricultural lands for different crops (policy scenario 2), results indicate a reduction of fertilizer use for corn while fertilizer use for soybeans and wheat was at the lower bound, thus reducing the crop yields. Upper and lower bounds of fertilizer use for different crops were estimated with reference to historical fertilizer use using data from the United States Department of Agriculture's Economic Research Service (USDA ERS, 2012).

In contrast to abatement cost internalization policies, TP load abatement through external funds for BMP implementation (policy scenario 3) will have a minimal impact on the agricultural production (Table 5.5). Under this extreme scenario, all the available lands for the three crops were cultivated, and maximum yield was achieved through application of fertilizer at the upper bound of fertilizer use. A significant amount of TP load from different crop lands was reduced through implementation of agricultural BMPs, namely agricultural buffer strips and conservation tillage, and cover crops.

Table 5.5. The Maumee Basin's major crops.

Crop	Available [a] land (10^3 ha)	Price [b] (\$/Bu)	Operating cost [b] (\$/ha-yr)	Average yield [b] (Bu/ha-yr)	Average fertilizer use [b]	Export coefficient [c]
Corn	526.4	7.6	507.4	282.07	335.40	0.898
Soybeans	663.6	17.2	233.0	88.17	144.06	0.385
Wheat	94.9	10.2	237.8	127.52	216.01	0.578

[a] Land areas under cultivation of the major crops were scaled up to cover the entire agricultural land in the basin using data from CDL 2010 (USDA NASS, 2010).

[b] Available from the United States Department of Agriculture's Economic Research Service (USDA ERS, 2010). Price and operating cost are in 2010 dollars.

[c] Estimated based on the cultivation area and fertilizer use based on basin-scale aggregate agricultural TP export coefficient estimated by SPARROW.

Table 5.6. Model results for different TP abatement policy scenarios.

Abatement policy	Crop	Cultivated land (ha)	Fertilizer use (kg/ha-yr)	Yield (Bu/ha-yr)	TP export coefficient (kg/ha-yr)	TP load reduction from BMPs (kg/yr)
1. Fertilizer tax imposition (partial cultivation)	Corn	432,823	400	308.8	1.056	0
	Soybeans	9,605	300	93.6	1.604	15,409
	Wheat	1,319	300	176.1	1.167	1,540
2. Fertilizer tax imposition (at least 50% cultivation)	Corn	263,200	376	292.2	0.99	0
	Soybeans	331,800	100	44.1	0.535	0
	Wheat	47,450	100	48.3	0.389	0
3. BMP cost covered by external funds	Corn	526,400	400	308.8	1.056	555,710
	Soybeans	663,600	300	93.6	1.604	1,064,600
	Wheat	94,900	300	176.1	1.167	39,170

Figure 5.7 shows the basin scale net economic benefit (profit) to the agricultural sector as a surrogate for utility. The highest profit is achieved under the extreme policy of TP load abatement through external funds (policy scenario 3), whereas internalization of TP load abatement cost with the constraint requiring use of at least 50% of available lands (policy scenario 2) resulted in the lowest utility. This is because application of upper bounds of fertilizer under policy scenario 3 provides the highest yield and the cost of TP abatement through agricultural BMP implementation is not incurred by the agricultural sector. The requirement for use of at least 50% of available agricultural lands creates a high fertilizer cost component coupled with low yields, indicating sub-optimality of inflexible policies with respect to cultivate land area. Full internalization of TP abatement cost allowing partial cultivation of crops (policy scenario 1) resulted in about 50% reduction in profit compared with scenario 3 because the optimal solution is to produce a high yield crop (i.e., corn) using upper bound of fertilizer use. Essentially, under this policy scenario, the agricultural sector opts to meet the TP load requirement through significant curtailment of cultivation of soybeans and wheat than extensive BMP implementation.

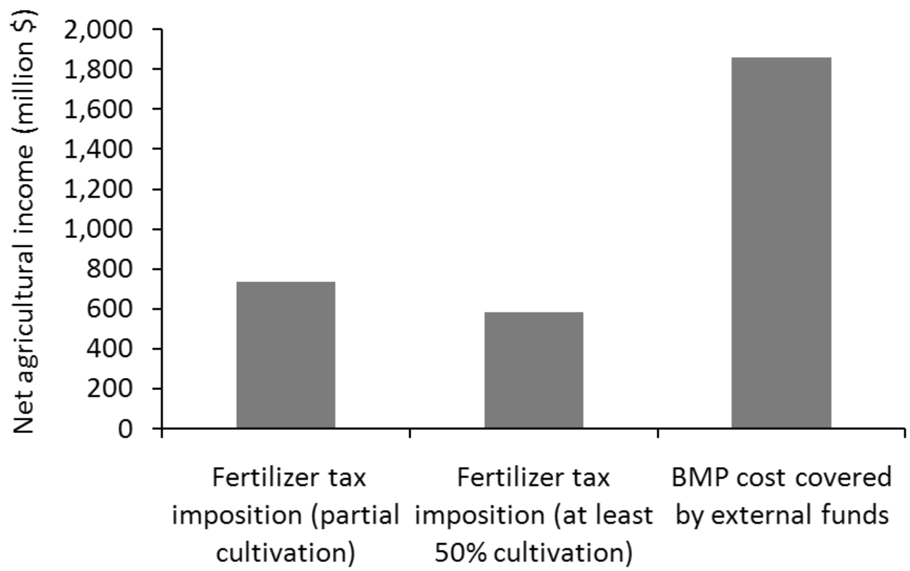


Figure 5.7. Net agricultural income when (1) agricultural land may be cultivated partially and BMP cost is fully internalized through fertilizer tax imposition; (2) at least 50% of agricultural land is cultivated and BMP cost is fully internalized through fertilizer tax imposition; and (3) BMP cost is covered by external funds.

Maintaining the balance between food production and environmental integrity poses a formidable environmental policy challenge. The results of this simple agro-economic BMP model indicate that while extreme TP abatement cost internalization scenarios may have severe negative impacts on the agricultural sector, external funding of costs will encourage free riding, i.e., maximum net economic profit to the agricultural sector at government expense. In a sense, the food produced by the agricultural sector is already highly subsidized because the society as a whole is getting a free ride by not covering the environmental costs of food production. The extreme policy of fully internalizing the environmental externalities associated with agricultural production will impact the cost and abundance of food. However, as the world prepares to double food production by 2050 (Tilman et al., 2002), partial internalization of environmental externalities is necessary because it can reduce the environmental impacts of agricultural production. Alleviating the impact of the environmental cost internalization on agricultural production will require government intervention in the form of subsidies and incentive programs, as well contribution from society as a whole through paying a higher price for food in order to partially cover the environmental costs.

Although TP load abatement through external funds will be the most desirable policy to the agricultural sector and consumers of agricultural products, the cost of implementing this policy on the national level may be overwhelming. Recognizing over-application of fertilizers as one of the root causes of eutrophication, agricultural BMP implementation may only provide a short-term solution by temporarily capturing the nutrients. The sustainable solution to agricultural nutrient loads is to adapt fertilizer application to the carrying capacity of the water bodies and in amounts that can be taken up by plants and assimilated by BMPs.

5.4. Future work

Future research investigating the application of a fertilizer tax as a TP abatement policy will consider analysis at the scale of seven individual sub-watersheds that comprise the Maumee River watershed. At the basin scale, the collective impact of different sub-watersheds should be considered when optimizing the fertilizer tax to cover the cost of an agricultural BMP plan for compliance with target total TP loads.

Application of a zone-based or sub-watershed scale fertilizer tax will be contrasted with imposition of a uniform fertilizer tax across the watershed.

Finally, a policy scenario may be considered whereby a zone-based BMP implementation policy is adopted along with an environmental offset program allowing water quality trading between sub-watersheds. Using the presented agronomic-economic framework, the trade between buyers and sellers of TP load credits can be investigated as a policy to meet the specified water quality target at the mouth of the Maumee River while maximizing the utility of individual agricultural zones. Sub-watersheds in which TP load abatement may be achieved more cost efficiently can be modeled as sellers of TP load credits, i.e., recipients of BMP implementation funds provided by buyers from areas of low cost efficiency for load TP load abatement. To maintain a utility-maximizing level of TP emissions, it is assumed that the buyers will be willing to pay sellers to cover the cost of BMP implementation and utility loss. This case can be analyzed by modifying the buyers' and sellers' corresponding income functions as follows:

$$I_b = \sum_{c=1}^{c=C} (P_c Y_c - C_{r,c}) \,.\, LA_c - C_t \qquad (eq. 5.13)$$

$$I_s = \sum_{c=1}^{c=C} (P_c Y_c - C_{r,c}) \,.\, LA_c - C_{BMP} + I_t \qquad (eq. 5.14)$$

where indices *b* and *s* denote buyers and sellers of TP load credits, C_t = total cost of trade (including transaction cost), and I_t = net income from trade.

5.5. Conclusions

A BMP optimization model was developed and applied to the Maumee Basin to provide insights for NPS TP load reduction in the Maumee Basin. A coordinated basin scale BMP implementation whereby the target TP concentration of 170 µg/L at the outlet of the Maumee Basin can be achieved by coordinated BMP implementation across the watershed is found to be cost-efficient. In terms of cost-efficiency and feasibility of meeting the target, this BMP implementation scheme is superior to a case where individual sub-watersheds attempt to meet the outlet TP concentration target without coordination among other sub-watersheds. However, individual sub-watersheds, as is the case in the Maumee Basin, may need to reduce their TP load due to local water quality concerns. If this is the case, a coordinated sub-watershed scale BMP implementation may be recommended whereby meeting the TP concentration target at the outlet of a downstream watershed may be facilitated through implementation of additional BMPs in upstream watersheds.

Implementation of agricultural BMPs may be unaffordable to the agricultural sector because it significantly reduces the utility of the agricultural production. However, reemergence of severe nuisance algal blooms, is an indication of the need for more aggressive TMDL implementation policies. Market-based policy instruments may provide a more flexible means of addressing NPS pollution as compared with command-and-control approaches. Future research should explore the potential for imposition of a zone-based fertilizer tax, as well as environmental offset programs to facilitate NPS TP abatement.

5.6. References

Antosch, L.M. (2006). Ohio irrigation assessment. Ohio Farm Bureau. <ftp://ftp.dnr.state.oh.us/Soil_&_Water_Conservation/Bleim/Irrigation/Irrigation-Assessment.pdf> (accessed on Oct. 21, 2012).

Boyd, J. (2003). Water pollution taxes: a good idea doomed to failure?. *Public Finance Management* 3(1), 34–66.

Chapra, S., Dolan, D.M. (2012). Great Lakes total phosphorus revisited: 2. Mass balance modeling. *Journal of Great Lakes Research* 38, 741-754.

Ekins, R. (1999). European environmental taxes and charges: recent experience, issues and trends. *Ecological Economics* 31, 39– 62.

Hsieh, C.D., Yang, W.F. (2007). Optimal nonpoint source pollution control strategies for a reservoir watershed in Taiwan. *Journal of Environmental Management* 85, 908–917.

Lake Erie LaMP (2009). Status of nutrients in the Lake Erie Basin. Prepared for the Lake Erie LaMP Management Committee. <http://www.epa.gov/lakeerie/erie_nutrient_2010.pdf > (accessed on Oct. 10, 2012)

Layard, R., Mayaraz, G., Nickell, S. (2006). The Marginal Utility of Income. Mimeo, London School of Economics.

Ludsin, S.A., Kershner, M.W., Blocksom, K.A., Knight, R.L., Stein, R.A., (2001). Life after death in Lake Erie: nutrient controls drive fish species richness, rehabilitation. *Ecological Applications* 11, 731-46.

Makarewicz, J. C., Bertram, P. (1991). Evidence for the restoration of the Lake Erie ecosystem. *BioScience* 41,216-223.

National Oceanic and Atmospheric Administration National Climatic Data Center (2012). < http://www.ncdc.noaa.gov/oa/climate/climatedata.html> (accessed on Oct. 5, 2012).

Ohio EPA (1999), Association between nutrients, habitat, and the aquatic biota in Ohio rivers and streams, Ohio EPA Technical Bulletin MAS/1999-1-1, Ohio EPA, Division of Surface Water, Monitoring and Assessment Section, Columbus, Ohio.

Ohio EPA. (2010). Ohio Lake Erie phosphorus task force final report. <http://www.epa.ohio.gov/portals/35/lakeerie/ptaskforce/Task_Force_Final_Report_April_2010.pdf> (accessed on Oct. 5, 2012).

Plourde, J. D. (2012). Assessing changes and drivers of crop rotation patterns and their implications for nitrogen loading in the United States. MS Thesis, Purdue University, West Lafayette, IN.

Robertson, D. M.; Saad, D. A. (2011). Nutrient inputs to the Laurentian Great Lakes by source and watershed estimated using SPARROW watershed models. *Journal of American Water Resources Association* 47(5), 1011-1033.

Selman, M., Greenhalgh, S., Branosky, E., Jones, C., Guiling, J. (2009). Water quality trading programs: An international overview. WRI Issue Brief, World Resources Institute, Washington, D.C.

Shortle, J., Horan, R. (2001). The Economics of Nonpoint Pollution Control. *Journal of Economic Surveys* 15(3), 255–289.

Tilman, D., Cassman, K.G., Matson, P.A., Naylor, R., Polasky, S. (2002). Agricultural sustainability and intensive production practices. *Nature* 418, 671–677.

Turner, R.K., Georgiou, S., Gren, I., Wulff, F., Barrett, S., Söderqvist, T., Bateman, I.J., Folke, C., Langaas, S., Zylicz, T., Karl-Goran Mäler, K., Markowska, A. (1999). Managing nutrient fluxes and pollution in the Baltic: an interdisciplinary simulation study. *Ecological Economics* 30, 333–352.

United States Department of Agriculture Economic Research Service (2012). Available online: http://www.ers.usda.gov/data-products.aspx (Accessed in October 2012).

United States Department of Agriculture National Agricultural Statistics Service (2012). < http://www.nass.usda.gov/research/Cropland/SARS1a.htm> (accessed on October 10, 2012).

Zaring, D. (1996). Agriculture, nonpoint source pollution, and regulatory control: The clean water act's bleak present and future. Harvard Environmental Law Review 20, 515-545.

Chapter 6 – Conclusions and Future Research

6.1. Need for systems thinking

Water resources systems involve natural and anthropogenic processes that are complex, dynamic, and spatially variable. Previous experiences of unsuccessful or unsustainable watershed planning and management practices manifest how a lack of understanding of water resources subsystems can cause environmental disasters as well as socioeconomic problems affecting humans' wellbeing. Water resources modeling has become a commonplace tool for water resources system design, planning, and management at an affordable cost and within a reasonable timeframe. Over the past decades, water resources systems models have evolved from describing only physical processes to describing the interaction of social, economic, and environmental systems objectives in support of decision making. The gradual shift from merely employing engineering-based simulation models to applying integrated hydroeconomic models, and more recently multi-criteria/multi-objective decision making and conflict resolution models, is an indicator of promising changes in the traditional paradigm for the application of water resources models. More holistic understanding of water resources systems and improved abilities to predict and plan for future impacts are likely to lead to more sustainable watershed planning and management decisions.

6.2. System dynamics and water resources modeling

This dissertation illustrated the role of the systems thinking paradigm in water resources planning and decision making, demonstrating qualitative, as well as quantitative capabilities of system dynamics modeling in facilitating holistic water resources modeling and policy making. Using tangible water resources examples, the fundamentals of system dynamics, including causal relationships, CLDs, SFDs, and water resources system archetypes, were illustrated. Applying a systems thinking paradigm to water resources modeling is critical when formulating strategic-level water management policies and plans, considering that a traditional linear thinking paradigm may lead to quick-fix solutions that fail to address key drivers of the problem. Systems thinking and system dynamics modeling can help water resources decision makers comprehend the interactions among various interlinked sub-systems of a water resources system which drive its long-run dynamic behavior.

A wave of water resources modeling efforts using system dynamics has emerged in the past two decades as modelers strive to capture the main drivers of water resources problems, including interrelationships between disparate natural, technological, and socio-economic subsystems. The approach has proven useful for providing valuable insights into problems and systems' long-run behavior at the strategic level. In addition, system dynamics models' transparent structure and convenient sensitivity analysis make them practical tools for participatory modeling, policy screening, and high-level adaptive management. Object-oriented modeling tools enable transparent system dynamics modeling by providing generic building blocks to capture systems' nonlinear behavior.

However, the current versions of these modeling tools tend to be limited in terms of flexibility of programming.

Compared to other modeling approaches, a significant advantage of system dynamics is that when systems are not too complicated, the CLD and SFD of the system can help determine the qualitative behavior of many variables, even before quantitative (numerical) modeling begins. Furthermore, water managers can use knowledge of archetypal behavior (e.g., Limits to Growth, Fixes that Backfire, Success to the Successful, Tragedy of the Commons, and Growth and Underinvestment) to recognize common patterns of dynamic behavior in water resources systems. Thus, understanding the underlying structure of water resources systems can help avoid unintended consequences and unsustainable development trajectories by detecting the root causes of problematic trends and identifying potential corrective measures.

Perhaps the most significant aspect of system dynamics is its ability to facilitate multi-disciplinary, multi-sectoral, and participatory modeling of integrated systems. At the strategic level, when studying water resources systems with disparate dynamic variables, emphasis should be placed on trend identification and pattern recognition rather than exact quantitative predictions. In this way, decision makers can learn about the potential impacts of their decisions on different natural and socio-economic subsystems using "what-if" analyses. Optimization methods may also be applied to develop prescriptive plans and facilitate trade-off analysis. Thus, system dynamics models are developed to promote understanding of general trends and the reasons behind them. Nonetheless, from the standpoint of integrated water resources planning and management, system dynamics models can improve understanding of the big picture, while specific plans and designs should be studied in more detail using hydroeconomic models and engineering-based watershed process models to ensure informed decisions.

6.3. A systems approach to water quality management

The United States Clean Water Act provides a good opportunity for applying the systems approach to water quality management. The CWA requires each state to identify the main pollutant(s) of concern impairing water bodies. The states should meet water quality standards using TMDLs, i.e., a written plan quantifying allowable levels of load allocation (NPS pollution) and waste-load allocation (PS pollution). Furthermore, a margin of safety is warranted to compensate for lack of knowledge as to relationship between pollutant inputs and water quality.

Within this framework, system dynamics modeling was applied to identify and simulate the system structure driving the long-term eutrophication-recovery trend of Lake Allegan, Michigan to provide insights into policies for mitigating impairment. Once a potential strategy (e.g., TP load mitigation in agricultural and urban areas) has been identified, it can be investigated in more detail using optimization modeling in order to find cost-effective policies to address the problem. The Lake Allegan case study illustrated how simple system dynamics models can facilitate qualitative and strategic-level quantitative analysis of interlinked socioeconomic and biophysical subsystems. The Growth and Underinvestment system archetype was used to illustrate that the lake's eutrophication problem is partially due to lack of investments in reducing the TP loads to levels that can be assimilated without side-effects. Continuous investments should be made to mitigate eutrophication, a natural feedback between socioeconomic growth, land use change, and environmental integrity.

Furthermore, a screening-level BMP optimization model was developed and used in conjunction with the system dynamics model, creating a simulation-optimization framework to find least-cost TP BMPs. The Kalamazoo Watershed has significant potential for reducing Lake Allegan's TP load mostly through mitigating agricultural NPS loads. Policies requiring substantial (>20%) abatement of TP loads from urban areas may be suboptimal as compared with implementation of agricultural BMPS. It was shown that inter- and intra-annual hydrologic variability is the primary factor governing the lake's TP concentration. However, meeting the TP concentration target of 60 μg/L with high reliability will also be contingent on adapting the TMDL plan to land use change associated with socioeconomic growth.

Although reducing TP loads mostly through agricultural BMPs may be given priority over urban BMPs, intensive BMP implementation in agricultural lands without adequate policy support may severely impact agricultural producers. In larger watersheds such as the Maumee Basin, a coordinated watershed scale effort may provide opportunities for cost-effective BMP implementation in areas where TP loads can be reduced cost-efficiently. Market-based policy instruments such as a fertilizer tax and environmental offset programs generating funds for BMP implementation may provide a means for more aggressive TMDL policy than voluntary participation in TMDL programs.

6.4. Future research

The current research can be expanded in a number of key areas including system dynamics modeling, biophysical aspects of BMP implementation, and policy instruments for NPS pollution abatement. Causal loop diagrams developed within the system dynamics modeling framework provide opportunities for identifying problematic behaviors in subsystems that can be studied in greater detail using sophisticated quantitative models (e.g., hydrologic models and regional economic models). Ideally, systems dynamics modeling should be conducted using a participatory modeling approach involving stakeholders and experts from different academic and professional disciplines. Communication with stakeholders (e.g., farmers and wastewater treatment managers) will help identify the practical challenges of TP load abatement and BMP implementation.

Potential areas of future research to better understand biophysical aspects of BMP implementation include characterization of uncertainty due to climate change, BMP implementation at smaller scales (e.g., catchment or farm scale), and accounting for uncertainty associated with BMP pollutant removal effectiveness. Accounting for the impact of climate change will be important for long-term BMP implementation plans because in many cases (e.g., TP loading of the Lake Allegan) hydrologic variability is the primary governing factor for the transport of pollutants to the water bodies. Small scale studies of BMP implementation facilitate the estimation of the upper bounds and area treatment coefficients for the BMPs. Further research is needed to obtain reliable estimates of pollutant removal coefficients because the wide range of removal effectiveness for different BMPs available in the literature creates obstacles for BMP planning.

Future research should investigate in more detail different policy instruments (e.g., command-and-control, market-based instruments, and coordinated BMP implementation across watersheds) that may facilitate NPS pollution abatement. Unlike the point sources, the NPS of pollution have not been mandated to reduce pollution through command-and-control approaches. However, as the environmental problems associated with NPS pollution (e.g., Lake Erie's nuisance algal bloom) continue to intensify, effective policies to mitigate NPS pollution are increasingly needed. Market-based policy instruments for NPS pollution abatement (e.g., input-based taxation and environmental offset programs) may provide an alternative to command-and-control and voluntary BMP implementation approaches. The potential for adoption of market-based policy instruments should be investigated in conjunction with robust biophysical and economic models. For example, zone-based fertilizer taxation may create an opportunity for implementation of the "polluter pays" approach for internalizing the environmental externalities. The use of fertilizer tax as a TP abatement policy in the Maumee Basin may consider sub-watershed scale analysis of TP concentration target attainment and economic benefits to the agroeconomic sector. Similarly, the environmental offset trade between buyers and sellers of TP load credits can be investigated to provide a mechanism for generating funds for BMP implementation in areas where TP load abatement can be achieved with relatively high cost-effectiveness.