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Development of an Urban Watershed Rehabilitation Method Using Stakeholder Feedback to Direct Investigation and Restoration Planning

October 1, 1998 – March 31, 2003

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RESEARCH FINDINGS & CONCLUSIONS

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Period Covered by the Report*:* October 1, 1998 – March 31, 2003

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OBJECTIVES OF THE RESEARCH PROJECT

We developed and tested a method for restoring the ecological integrity of urban watersheds that combined ecology, engineering and social sciences. We tested the following hypotheses:

1. Risk-based models developed for agricultural NPS pollution management can be applied to urban watersheds; and
2. a) Stakeholders' understanding of NPS pollution issues, b) use of systems thinking, c) ability to use scientific information about TMDLs/rehabilitation options, and d) communication competence will increase as a result of the Collaborative Learning (CL) intervention.

Baseline data on the San Antonio population were gathered through a survey of a random sample of that population. Stakeholder council members were recruited using both leads developed from the larger San Antonio survey and from face-to-face interviews with residents of the two watersheds, using a snowball sampling technique. Once councils were formed, the CL intervention used iterative input from stakeholders to guide risk-based research and restoration planning. Two integrated eco-indicators —periphyton and bluegill sunfish—were used to evaluate and communicate risk to the stakeholder groups. The effectiveness of the CL process was evaluated using both pretest and posttest surveys with council members and face-to-face interviews with council members at the conclusion of the CL process.

The project also integrated four modeling activities: A GIS-based riparian zone impact zone model, a BASINS-HSPF watershed model, an Ecophys.Fish model, and a STELLA watershed model linking watershed policy and management decisions to watershed hydrologic response. In addition, a simulation of citizen behavior was developed to integrate human behavioral models with watershed models.

SUMMARY OF FINDINGS: SOCIAL SCIENCE

Collaborative Learning Program Effectiveness. Our discussion of the results focuses on the four social science hypotheses tested in this project. First, with respect to learning/knowledge outcomes, we found support for Hypothesis 2a on one of the six objective knowledge survey questions relating to non-point source pollution. With respect to Hypothesis 2b, we found solid support: stakeholders reported greater familiarity and ability to use system thinking principles in ecosystem management at the conclusion of the CL meetings compared to the beginning of the project. Hypothesis 2c received strong support on the self-report learning/knowledge items: council members reported that their knowledge of and ability to use scientific information in water quality management improved over the course of the CL program. Hypothesis 2d, however, was not supported by the statistical analysis of the survey data. In terms of communication competence, stakeholders reported that the quality of communication at the group level (Salado Creek Restoration Council) decreased from pretest to posttest. At the individual level, we found either no change or change in the opposite direction from Hypothesis 2d.

With regard to interpersonal relationships, the results were clear: council members believed that relationship quality declined from pretest to posttest survey administration. There was also no change in trust in other council members over the course of the 14-month series of CL meetings.

Overall, we found that stakeholder members evaluated the CL format used in the meetings more positively compared to their previous experiences with other collaborative methods. Participants felt that their effectiveness in negotiating with other members of the Salado Creek Restoration Council declined from pretest to posttest.

Ancillary analyses of water quality beliefs revealed that, overall, participants changed their perceptions of water quality in Salado Creek in a positive direction: water was believed to be more safe for drinking, swimming, fishing, and livestock at posttest compared to the pretest. This pattern of results was unanticipated based on existing CL theory but is intriguing. It is possible that the increased knowledge about ecological principles and dynamics gained from the CL program may have stimulated stakeholders to revise their earlier, more negative perceptions of water quality in Salado Creek. Finally, analysis of personal efficacy beliefs showed no changes as a function of participation in the CL program.

In summary, the quantitative survey results suggest that learning about non-point source pollution, TMDLs, and rehabilitation options did occur over the course of the CL meetings. There is also clear evidence that participants improved their knowledge about systems thinking in the context of ecosystem management. Thus, in terms of learning/knowledge outcomes, we conclude that the CL intervention was successful in achieving its objectives. Survey data from council members provided little support for the prediction that communication competence would improve with the use of CL. There is also no evidence in the survey results to support the assumptions that interpersonal relationships among stakeholders and negotiation effectiveness improved due to the CL program. Finally, it appears that the CL intervention did change beliefs about water quality in the Salado Creek watershed toward greater confidence in the safety of the water for human and animal uses.

The qualitative data from personal interviews with council members suggested a more positive conclusion about the effectiveness of the CL program, compared to the quantitative survey results. Many comments in the post-interviews reinforced the survey-based indicators that significant learning had taken place about non-point source pollution, ecological principles and dynamics, and rehabilitation options. The interview data, however, also suggest that participants did improve their communication skills and develop new relationships with others over the course of the program. Agency representatives were also positively impressed with the quality of the CL meetings, and there is evidence from the interview data that the CL approach is now being used as a "model" for other watershed collaborative groups in San Antonio. Many interview comments focused on how the information gained during the project meetings helped the individual to become more active in other community groups and self-governance activities. We conclude that these qualitative results provide complementary, and in some respects, stronger support than the survey results for the original social science hypotheses.

Summary of Findings: Ecological Science

Ecological investigations included measuring ecological services changes in the San Antonio watersheds (Leon and Salado Creeks and the Upper San Antonio River) over the past 20 years, measuring the ecosystem health of Salado and Leon Creeks using periphyton, measuring the ecosystem health of Salado and Leon Creeks using bluegill sunfish, modeling fecal coliforms in Salado Creek using HSPF, assessing the impact of riparian zone loss on Leon Creek, and assessing the potential for ecosystem restoration using wastewater from the Dos Rios

Wastewater Treatment Plant for supplemental flow. The complex suite of measurements employed in this project were effective indicators of ecosystem health, and provided insight into potential remediation strategies.

Ecological Services Change. Based on the estimated size of six land cover categories and Costanza et al.'s ecosystem services values for related biomes, we determined that the total annual ecosystem service values in Bexar County declined from $21.94 to $21.16 million per year from 1976 to 1991.While our study showed that urban sprawl in Bexar county resulted in a decline in the value of ecosystem services delivered by the affected land, it also showed that changes in the value of ecosystem services over time depend on the interaction of changes in various land cover types. Our study suggests that urban spread may not necessarily lead to a large net decline in ecosystem services if there is a concomitant increase in size of other land cover types that provide a greater level of ecosystem services. This is not to say that urban sprawl is beneficial for the delivery of ecological services, but that the negative impacts of the spread of urban and suburban land uses can be potentially offset by other mitigating changes in land cover. This is important because it is unlikely that with increasing human population pressure, the conversion of land to urban and suburban land use will cease or even dissipate in the near future.

Measuring Ecosystem Health with Periphyton. Eutrophication of streams and rivers is recognized as a growing threat to the health of aquatic ecosystems throughout the United States. Two studies in this project aimed to make periphyton analysis a more effective tool for addressing issues of eutrophication. The first study showed that nutrient delivery via diffusion elicits a greater periphytic algae response than nutrient enrichment via whole-channel dosing. The second study demonstrated the importance of riparian shading on reducing periphytic growth through shading, illustrating the connectivity between stream riparian habitat and aquatic bioprocesses.

Measuring Ecosystem Health with Bluegill Sunfish. This study measured the effects of in-stream environmental factors on bluegill sunfish physiological health and weight gain over 14 days. Metabolic "health" of the fish at the end of the cage trial accounted for 31 % of variation in median weight change during the cage trial; physicochemical environment, independent of its effects on metabolic health, accounted for an additional 43 % (74 - 31 %); quantity or quality of feed accounted for yet another 15 % (89 - 74 %); leaving only 11 % (100 - 89 %) unaccounted for. Obviously, much of the variation both in median weight change and in median marginal metabolic scope was seasonal, and presumably temperature-related.

Modeling Fecal Coliforms in Salado Creek. The Salado Creek watershed in Bexar County, Texas was modeled using the HSPF model in BASINS. The model was calibrated for hydrology. A sensitivity analysis and first order approximation were performed to determine the parameters that most influence the in-stream fecal coliform concentration predictions. The specific findings from the study include:

1. The parameters that peak in-stream fecal coliform concentrations are most sensitive to are those that represent the maximum storage of fecal coliform bacteria over the pervious land segment (SQOLIM), and surface runoff that removes 90 percent of quality constituent from pervious land segment (WSQOP) of PERLND section (corresponding to pervious land segment) of HSPF model.
2. Other parameters in-stream fecal coliform concentration predictions are sensitive to are stream water temperature (TWAT), first-order decay rate of quality constituent (FSTDEC) and temperature correction coefficient for the first order decay rate (THFST) of RCHRES section.

These results point out the importance of parameterization in modeling with any complex, process-based watershed model. Small errors in assigning values to the maximum storage of fecal coliform over a given land use class may result in large errors in predicted coliform counts.

Modeling Leon Creek Riparian Zone Decline. Evaluations of the riparian vegetation of Leon Creek suggest a fragmentation trend from 1987 to 1999 indicated by the Image Difference Calculation, Percent Area Calculation, and the landscape indices. However, as with most landscape-level assessments, the results are not always consistent. There is an inconsistent result in the rural zone, with the percent area showing increases in dense vegetation by 1999, which would be inconsistent with increased fragmentation. The most probable explanation is the conversion of land cover from arid desert vegetation to rural landscaped/irrigated vegetation. Both the Percent Area Calculations and Landscape Indices are consistent in the urban zone. Both the Image Difference Calculation and Landscape Indices are consistent in both the rural and urban zone, with the Image Difference Calculation showing decreases in vegetation, which would be consistent with fragmentation shown by the landscape indices. When the Leon Watershed was divided into urban and rural zones this showed an increase in fragmentation level between 1987 and 1999, especially in the urban zone.

Ecosystem Restoration Using Wastewater to Restore Base Flow. This study investigated the effect of increasing dry-period base flow in streams with discharge from a wastewater treatment plant (with pre-defined water quality parameters) on dissolved oxygen concentration in Leon Creek. An increase in daily mean DO was observed for all treatment scenarios tested. With 95 % confidence level, a significant change was observed with scenario S3 (one times base flow), i.e. doubling the base flow during low-flow periods. This constitutes a risk-based design approach to remedy DO problems by low-flow augmentation.

MODELING ACTIVITIES

STELLA Watershed Model. A STELLA model was developed for the project as a tool for facilitating group learning among council members. The model was intended to illustrate how changes in watershed condition affect hydrologic response. While limited in its usefulness for hydrologic prediction or for modeling specific management plans, this model did allow stakeholders to view the relationship between selected strategies and runoff generation. It created an effective communication interface between scientists, citizens, and policy makers.

Simulation of Citizen Behavior. Using data from the survey of the general San Antonio population, we (1) estimated a set of models of watershed-impacting behavior and (2) produced simulations for the residents in areas adjacent to Salado and Leon Creeks based on what we found to be the most influential predictor variables from step (1). We then developed a statistical model to explain willingness to change that included such variables as environmental attitudes and assessment of local conditions. The simulations suggest that of the existing attitudes/characteristics of San Antonio citizens, environmentalist views and a sense of being locally empowered are important predictors of willingness to change behavior in ways that would reduce non-point source pollution. In terms of public information, the model indicates that if residents were convinced that changes in their behavior would actually improve the quality of the watersheds, they would be more likely to change.

SELECTED PUBLICATIONS AND PRESENTATIONS

Eight theses, one book chapter, one technical report, two journal articles, one patent application, and fourteen presentations have been generated out of this research. In addition, three book chapters and eight journal articles are currently in process. A full list of the research products can be found in the final project report and in the near future at the web site of the Institute for Science, Technology and Public Policy (ISTPP) at Texas A&M University. A sample of these products is listed below.

Cummins, K. (2000). The temporal mapping of riparian vegetation at Leon Creek in Bexar County, Texas from 1987-1999. MS Thesis, Texas A&M University.

Gholkar, T. (2000). Modeling the effects of low-flow augmentation by discharge from a wastewater treatment plant on dissolved oxygen concentration in Leon Creek, San Antonio, Texas. MS Thesis, Texas A&M University.

Gore, D.C. (2001). Adam Smith's rhetorical sympathy: A return of moral sentiments to public policy. Master of Science thesis, Department of Communication, Texas A&M University, College Station, TX.

Grant, W. E., Peterson, T. R., & Peterson, M. J. (2002). A quantitative representation and application of Niklas Luhmann's social theory: Simulation of communication as it relates to ecological systems. *Ecological Modelling*.

Harris, H. (2000). Changes in ecosystem services and runoff due to land use change in the watersheds of San Antonio, Texas. MS Thesis, Texas A&M University.

Matlock, M., & Murawski, M. (Manuscript submitted, 2003). The efficiency of nutrient delivery in Matlock periphytometers.

Murawski, M. (2001). Evaluation of Nutrient Diffusing Substrates and the Primary Importance of Light in Controlling Periphyton. MS Thesis, Texas A&M University.

Musacchio, L. R., Grant, W. E., & Peterson, T. R. (2003). Adaptive management of complex socio-environmental systems in the southwestern United States: Examples of urbanizing watersheds in Arizona and Texas. In S. Guhathakurta (Ed.), *Integrated land use and environmental models: A survey of current applications and research*. New York, NY: Springer-Verlag.

Neill, W. H., & co-authors. (Manuscript submitted, July 2003). Ecophys.Fish: A simulation model of fish growth in time-varying environmental regimes. *Ecological Modelling*.

Peterson, T. R., Kenimer, A. L., & Grant, W. E. (Projected Publication, 2004). Using mediated modeling to facilitate collaborative learning among residents of the San Antonio watershed, Texas, U.S.A. In M. v. d. Belt (Ed.), *Mediated modeling: A quantitative group modeling approach to build understanding and consensus in solving complex environmental problems*. Washington D.C.: Island Press.

Peterson, T. R., Samuelson, C.D., & Gilbertz, S. (Manuscript in process, 2003). A qualitative assessment of Collaborative Learning as a method for facilitating stakeholder communication in urban watershed restoration planning.

Rodriguez, A. (2001). Impact of Nutrient Loading from Point and Non-Point Sources on Water Quality and Lotic Ecosystem Health in the North-Bosque Watershed using a Bio-Indicator Response Approach. MS Thesis, Texas A&M University.

Sabatier, P., Focht, W., Lubell, M., Trachtenberg, Z., Vedlitz, A., & Matlock, M. (2003, Under contract). *Swimming upstream: Collaborative approaches to watershed management*. Cambridge, MA: MIT Press.

Samuelson, C. D., Peterson, T. R., & Putnam, L. (2003). Group identity and stakeholder conflict in water resource management. In S. Clayton & S. Opotow (Eds.), *Identity and the natural environment*. Cambridge, MA: MIT Press.

Samuelson, C.D., Peterson, T.R., & Whitten, G. (Manuscript in process, 2003). Evaluating Collaborative Learning as a method for facilitating urban watershed restoration planning: A quasi-experimental study.

Samuelson, C.D., Vedlitz, A., Whitten, G., Matlock, M., Alston, L., Peterson, T.R. & Gilbertz, S. (2003, Under contract). Citizen participation and representation in collaborative engagement processes. In P. Sabatier, W. Focht, M. Lubell, Z. Trachtenberg, A. Vedlitz & M. Matlock, *Swimming upstream: Collaborative approaches to watershed management* (Chapter 5). Cambridge, MA: MIT Press.

Patent

Osborn, G. S., Matlock, M. D., & Teltschik, S. S. (Patent pending with the Texas Agricultural Experiment Station, 2000). Portable system to enhance biological treatment processes for improving water quality: A portable water ecosystem oxygenator.

SUPPLEMENTAL KEYWORDS

Ecosystem, Stakeholder, Complex Systems, Restoration, Collaborative Learning

RELEVENT WEBSITES

The team assisted the local project partner, the San Antonio River Authority (SARA) in establishing and maintaining a web site <http://www.agen.tamu.edu/sara/>.

INTRODUCTION

The sources of pollutants in urban areas are diverse and complex. Rehabilitation of urban watersheds requires commitment and agreement among the varied stakeholders in them. Top-down autocratic resource management methods have not proven effective in protecting these valuable ecological services. A method for dealing more realistically with the varied perspectives and interests of stakeholders must be found.

This study was designed to develop and test improved methods for stakeholder-driven watershed restoration planning. Our research team formed two watershed restoration councils in San Antonio, Texas to develop long-term watershed restoration plans. The findings, based on surveys of citizens at large as well as participants in the watershed restoration councils, suggest that representation may be the most difficult legitimacy criterion to accomplish in the collaborative engagement process.

We selected two watersheds in San Antonio because the streams in these watersheds were slated for Total Maximum Daily Load (TMDL) due to low dissolved oxygen, and both watersheds were in predominantly urban landscapes with diverse constituency groups. Working with the San Antonio River Authority, a non-regulatory agency in San Antonio, we implemented a three-year program that included: 1) designing and implementing a large-scale, representative survey of residents in the San Antonio metropolitan community; 2) testing hypotheses that predict citizen willingness to participate in a collaborative engagement process dealing with watershed management; 3) recruiting citizens into long-term collaborative stakeholder forums; 4) evaluating stakeholder representation patterns and the stated goals for citizen participation; and 5) evaluating the Collaborative Learning (CL) method for facilitating communication during monthly workshop meetings with citizens.

Initial Objectives and Hypotheses

We developed and tested a method for restoring the ecological integrity of complex watersheds. This method incorporated iterative input from stakeholders to guide risk-based research and restoration planning. A risk-based watershed model was linked with two integrated eco-indicators to evaluate and communicate to the stakeholder groups the uncertainty associated with rehabilitation options. This regressive ecological risk assessment will quantify the uncertainty inherent in predicting complex ecological responses (Suter, 1993; Matlock et al., 1994).

We tested the following hypotheses:

1. Risk-based models developed for agricultural NPS pollution management can be applied to urban watersheds; and
2. a) Stakeholders' understanding of NPS pollution issues, b) use of systems thinking, c) ability to use scientific information about TMDLs/rehabilitation options, and d) communication competence will increase as a result of the Collaborative Learning (CL) intervention.

The approach we used to investigate these questions incorporated three innovative strategies. *First*, the in-stream model that was calibrated and tested on Salado and Leon Creeks addressed the problem of uncertainty in a new way. The factors that influence NPS loadings include land use, soil characteristics, climate topography, and land management practices (Beaulac & Reckhow, 1982). Estimates of nutrient loadings from specific land uses exhibit considerable uncertainty, making temporal and spatial modeling of water quality based solely on land use difficult (Beaulac & Reckhow, 1982). To the extent that decision alternatives and their consequences are stable and clear-cut, decision making is fairly straightforward. Environmental decisions, however, are rarely of this type, and decisions surrounding water quality improvement are no exception. For example, relatively simple, market-based approaches have been cited in the economics literature as a cost-effective means to improving environmental quality (Dales, 1968; Baumol & Oates, 1975; Hahn & Hester, 1989; Atkinson & Tietenberg, 1991). However, few have met with success, and Taff & Senjem (1996) suggest that uncertainty is a key to this failure. Uncertainty is a factor that we addressed in this research.

A *second* innovation in this project was the enhanced collaborative interaction between scientists and stakeholders. It is becoming increasingly apparent that the effectiveness of the TMDL process required by the U.S. Environmental Protection Agency (USEPA) under the Clean Water Act will be strongly affected by the nature of the interaction between the scientific and regulatory communities, and local community representatives and citizens who review and process this technical information. The effective functioning of a decision making group hinges on: 1) reliable information on watershed integrity and the efficacy of potential watershed rehabilitation strategies, 2) a method for communicating such information to members of the group in a climate of collaborative decision making, and 3) a common knowledge base on which to build a consensus over acceptable solutions to the problem.

The *third* innovation was the Collaborative Learning (CL) intervention. This method was designed for use in multi-party, public discussions of ecosystem management (Daniels & Walker, 2001). Our use of this method was grounded in the theoretical model of common-pool resource (CPR) management developed by Elinor Ostrom and her colleagues at Indiana University (Ostrom, 1990; Ostrom, Gardner & Walker, 1994). This model is known as the Institutional Analysis and Development (IAD) framework. Traditional economic analyses of CPR situations have been pessimistic about the likelihood of voluntary cooperation among actors making independent decisions about CPR use and/or preservation (see Hardin, 1968, for a classic example). Ostrom's (1990) research, however, has concluded that this standard economic analysis is incomplete. She has found substantial evidence (see Ostrom, 1990) from numerous case studies of local communities faced with CPR management dilemmas that institutional solutions (rules and norms) can be devised to manage CPRs successfully. One important key to these local success stories is communication among community members. This is the variable that is typically given insufficient weight in standard game theoretic analyses of CPR situations (Ostrom, 1998). Moreover, there is abundant evidence from experimental studies of social dilemmas (Gardner, Ostrom & Walker, 1994; Kerr & Kaufmann-Gilliland, 1994; Orbell, et al., 1988) that group discussion produces a robust, positive effect on the level of cooperation among group members. Thus, there are strong theoretical and empirical grounds for targeting the communication process among community stakeholders as a leverage point for interventions in managing watersheds.

Our version of the CL intervention targeted one aspect of stakeholder communication: the communication between scientist and non-scientist stakeholders. This communication was facilitated using both group process mechanisms and computer models designed to "visualize" information.

Environmental Science Hypotheses Tested

Two specific aspects of risk-based modeling were tested.

- The Lotic Ecosystem Trophic Status Index (LETSI) measures the nutrient integrity of aquatic ecosystems. This hypothesis will be tested by comparing monitored results with mesocosm growth in control and nutrient-enriched mesocosms to determine predictive accuracy of LETSI.
- MMS measurements in acclimated versus unacclimated fish provide a measure of the relative integrity of the aquatic ecosystem. This hypothesis will be tested by comparing the MMS of bluegill newly captured (by seining) at each site with that of bluegill caught at the site two weeks previously and subsequently held in the effluent streams from the control and pollution-simulated flow-through mesocosms (nutrient- and metal-enriched).

Social Science Hypotheses Tested

We tested the impact of Collaborative Learning on stakeholders' knowledge and skills as follows:

- Stakeholders' general understanding of NPS pollution issues in this watershed will increase as a result of the CL intervention.

- Stakeholders' understanding of and ability to use scientific information about TMDLs and rehabilitation options will increase as a result of the CL intervention.
- Stakeholders' understanding and ability to use systems thinking in ecosystem management will increase as a result of the CL intervention.
- Communication competence of stakeholders will increase as a result of the CL intervention.

The Context

The San Antonio metroplex, with approximately one million people, was one of the fastest growing metropolitan areas during the 1990's (San Antonio Economic Development Foundation (SAEDF), 1999). The metroplex and surrounding areas experienced a 25.2% increase in population from 1990 to 1998 and are expected to grow at an average annual rate of 1.9% through the year 2010 (SAEDF, 1999). Bexar County, which primarily comprises the San Antonio Metroplex, was 53.9% Hispanic, 37.5% Anglo, and 6.6% Black in 1995.

This complex urban system is located in the upper reaches of the San Antonio River Basin. Three watersheds drain the entire San Antonio Metroplex: Leon Creek, the Upper San Antonio River, and Salado Creek (see Figure 1). Water quality in the basin is declining due to non-point source pollution from urban runoff from the City of San Antonio and its surrounding areas (Texas Natural Resources Conservation Commission (TNRCC), 1996). Segments of both Leon and Salado Creeks failed to meet their designated use criteria under the Clean Water Act (CWA) because of low dissolved oxygen and high fecal coliform bacteria and thus were listed by the Texas Commission on Environmental Quality (TCEQ, formerly the Texas Natural Resources Conservation Commission) in the 2000 Report as non-compliant as required under the Clean Water Act (subsection 303(d)) (TCEQ, 2002). Sources of the water quality problems in both watersheds are complex and are not known with certainty. Urban runoff during rain events is likely a major problem, resulting in contaminants such as fertilizer and/or animal waste from farms, ranches, and residential areas entering the streams. In addition, it is believed that factory discharges, illegal dumping by citizens and/or commercial operations, sewage system leaks, and fluctuations in water flow levels may all contribute to the declining water quality in the creeks. Restoration of the water bodies under the Environmental Protection Agency's rules requires development of a Total Maximum Daily Load (TMDL) for each component not in compliance. Water quality in these river segments must be restored by 2003 under the TNRCC Statewide Basin Management Schedule (TNRCC, 1999).

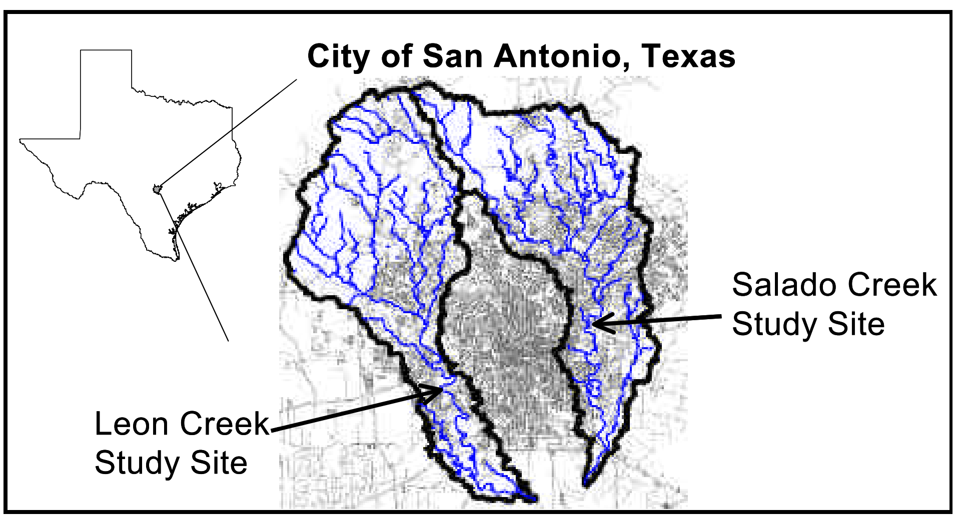


Figure 1. Location of Study Site

Quality Assurance

The quality assurance commitments made for this project were followed. The highest standards for data collection, analyses, and human subjects protection were maintained by the social scientists.

Data Collection Activities

Several types of data were collected in this investigation: physical/chemical water samples, biological samples, and geographic information data from a variety of sources. The data generated from each sampling event at each station (classified as *Critical Measurements*) included chlorophyll *a* measurements of each Matlock Periphytometer treatment replicate, conductivity of the nutrient solution, and the following stream water quality parameters: pH, conductivity, temperature, total phosphorus (T-P), ortho-phosphorus (O-P), total kheldal nitrogen (TKN), ammonia nitrogen (NH3-N), nitrate nitrogen (NO3-N), alkalinity, and total dissolved solids (TDS).

Sampling Experimental Design—Periphytometers

A randomized block design was used to sample each station using a series of replicated treatments per block. Ten replicates of each of the following treatments will be collected: 1) Control; 2) Nitrogen; 3) Phosphorus; and 4) Total Nutrients (2 and 3 combined).

Ten replicates of each treatment block were collected from two stations in the two basins. Ten replicates were an adequate number of replicates to insure that the variability among treatments was statistically different than the variability between treatments. The sites were sampled in the winter and summer of 1999 and 2000 for a total of 8 sample periods.

The chlorophyll *a* data from each sample station was standardized to the growth media surface area and analyzed to determine the statistical significance of the variance among and between treatments over time and space. The *control* treatment chl. *a* value for a sample station represents the primary productivity at a site. The *total nutrient* treatment chl. *a* represents the maximum potential productivity (MPP) at a site.

Procedures for Sample Handling

Chain-of-custody procedures were followed with all samples collected from the field. All water quality samples were collected in the appropriate type of container, stored in the dark at 4C, and preserved when appropriate. Samples were delivered to the appropriate laboratory for analysis within standard holding times.

Methods of Analysis

The stream water quality analyses were performed according to APHA Standard Methods. All data collection procedures conformed to the ANSI standards established for that particular type of data.

Calibration and Performance Evaluation of Sampling and Analytical Equipment

Field probes (pH, conductivity, D.O., and temperature) were calibrated in the laboratory and in the field prior to use, using commercial standards or other acceptable methods.

Data Reduction and Reporting

Chlorophyll a production for each Matlock Periphytometer treatment was compared with the control and total treatments using the Waller-Duncan Mean Comparison Test, a modified F-test.

Intended Use of the Data

Matlock Periphytometer data was used to estimate the mean primary productivity and LETSI of each site. Fish respirometry data was used to calculate the MMS at each site.

Procedures for Evaluating the Project Success

The success of this project was measured by the peer-review process through publication and professional oversight by the funding organization and collaborators.

Peer-Review Process for Study Methods

The procedures used in this investigation were reviewed by the collaborators, peers within State and Federal agencies, and through presentation at national and international meetings.

SOCIAL SCIENCE INVESTIGATION

The social science component of this project began with a large-scale survey of the San Antonio community. From March 13, 1999, to May 5, 1999, we conducted a telephone survey of Bexar County, Texas, adults. The two principal goals of the survey were to measure public opinion in Bexar County on a variety of issues relevant to watershed condition and management and to identify possible participants for our stakeholder process. The next step in this project was to recruit citizens into long-term stakeholder forums (watershed restoration councils). Survey respondents who reported interest in participation in discussions were one source of recruits. Toward the end of the phone survey, respondents were asked the following question: "Would you be willing to represent your community by participating in discussions on the topic of clean water in San Antonio?" Those who said "yes" were later contacted and invited to participate in the stakeholder forums. Participants for these forums were also recruited via a "snowball" technique that started with lists of community organizations. This technique is described in more detail later in this section.

In summary, the study included two distinct groups: A representative survey sample of San Antonio residents and a recruited group of citizens who participated in our stakeholder forums. A few of the participants in the stakeholder forums were interviewed in the original citizen survey, but the vast majority were recruited through the snowball technique. We first examined these groups in detail regarding the correlates of their willingness to participate in stakeholder forums, then we directly compare them later in this section.

Public Opinion Survey

Two hundred and forty-five (245) of the 1,017 adults interviewed in the phone survey reported that they would be willing to participate in discussions. To evaluate what factors influenced Bexar County residents' willingness to participate in discussions on the topic of clean water, we used existing theories to develop a set of hypotheses about what would make an individual more or less willing to participate. We then evaluated these hypotheses in a multivariate model of respondents' answers to a survey question asking them whether they would be willing to participate. If respondents said "yes", they were coded as willing to participate. Those who refused to answer this question or had dropped out of the survey by the time this question would have been asked were excluded from these analyses. Thus, all calculations are comparisons between those who answered "yes" and those who answered "no".

Using these respondents, we tested several hypotheses about democratic participation that emerge from the literature on individual political participation. This literature suggests at least three distinct sets of variables that may be appropriate in this environmental policy context—personal outlook attitudes, specific environmental attitudes, and demographic characteristics. We will look at each set separately and then in a multivariate context.

Willingness to Participate: General Outlook Attitudes

The literature discusses three general outlook concepts deemed important in predicting a citizen's willingness to participate in the political process: personal efficacy, trust, and financial optimism. It is almost an accepted truth in the participation literature that a sense of personal efficacy is strongly related to political participation (see Verba & Nie, 1972; Verba, Schlozman & Brady, 1995; Clarke, 1996; Verba, Nie & Kim, 1971; Smidt, 1971). Personal efficacy, or a sense that one has some power to influence events outside one's personal sphere, has been evaluated vis-à-vis all levels of government—national, state and local. For the purposes of this study, we focused on national and local contexts. The hypotheses regarding efficacy were:

H.1a. *The greater the personal efficacy vis-à-vis the local government context, the greater the willingness to participate in watershed stakeholder decision processes.*

H.1b. *The greater the personal efficacy vis-à-vis the national government context, the greater the willingness to participate in watershed stakeholder decision processes.*[1]

In the literature, trust, like efficacy, has a fairly consistent relationship to the two more traditional forms of participation, voting and contacting. Researchers as varied as Putnam (1993), Farrand (1983), Anderson et al. (1998) and Gandy (1991) report consistently higher levels of voting and contacting associated with higher levels of trust. However, the type of involved, long-term, direct participation we investigated in stakeholder decision processes may relate to trust very differently. We argue that due to the significant investment required in long-term stakeholder processes, those who trust other participants will not need to make the substantial investment required to protect their interests, while those who are more distrustful of other participants will participate to protect their interests. This is really a transaction costs argument—if I, a potential stakeholder participant, trust other actors to represent my interests, then it is not worth the transaction costs for me to participate. If, however, I do not trust other participants to represent my interests, then I will not think the transaction costs of my participation are too high. This analysis focuses on traditional measures examining levels of trust that respondents place in other people and government generally. Following the instructions and arguments of Sabatier et al. (chapter 4:2003), the hypotheses examined regarding trust were:

H.2a. *The lower the level of interpersonal trust held by a respondent, the greater the willingness to participate in watershed stakeholder decision processes.*[2]

H.2b. *The lower the level of trust in government held by a respondent, the greater the willingness to participate in watershed stakeholder decision processes.*

While efficacy is relatively consistently found associated with political participation, the literature on financial optimism shows, like trust, more ambiguous possibilities. For example, Putnam (2000) finds no relation between financial optimism and American civic engagement. But Farrand (1983) and Day et al. (1984) report that financial optimism strongly relates to political participation. Verba, Schlozman and Brady (1995) argue that, at least among high SES citizens, financial optimism is associated with less civic engagement in some of its forms. There appears, then, to be no consistent picture emerging from the literature about the direction of the relationship between financial optimism and political participation. For the purposes of directional clarity and comparative examination across all the hypotheses, we assumed a positive relationship between financial optimism and political participation. We examined two aspects of financial optimism here. The first relates to one's personal finances and the second relates to the national economy. Thus, the hypotheses tested regarding financial optimism were:

H.3a. *The greater the sense of personal financial optimism, the greater the willingness to participate in watershed stakeholder decision processes.*

H.3b. *The greater the sense of national financial optimism, the greater the willingness to participate in watershed stakeholder decision processes.*

Willingness to Participate: Specific Environmental Attitudes

People who are more interested in politics are more likely to participate in political activities like voting, contacting and campaign activities (Verba, Schlozman & Brady, 1995: Chapter 14). Studies of particular policies also indicate that people with a particular interest in a policy area are more likely to be knowledgeable about that area, to talk about that area to others, and to participate in activities related to that issue area (Verba et al., 1995). The question arises about directionality of this interest and its effects on participation. Are citizens who are pro- and anti-policy X equally likely to participate in political activity? Are pro's more likely than anti's? Or are anti's more likely than pro's to participate? The literature is unclear whether citizens with pro or anti- environmental protection attitudes will participate more. Both sides of this issue have participated in past conflicts, but no empirical evidence shows that one or the other subgroup dominates in this process.

In order to evaluate these distinctions, the study examined three specific categories of environmental attitudes. The first was general attitudes relating to national and international concerns like population growth, resource availability, role of animals, etc. Interviewers asked 12 environmental questions that gave respondents a chance to select a pro- or anti- environmental protection position on each (Dunlap & van Liere, 1978). We created an index of environmental beliefs from this battery of questions ranging from –2 for the most anti-environmental set of beliefs to +2 for the most pro-environmental set of beliefs.

The second environmental attitude area looks at local attitudes about San Antonio's most important problem (those respondents who mentioned in an open-ended question format "water" or "environment" vs. those who didn't). The question asked is whether or not those who mentioned water or a related environmental problem will participate more than their less concerned citizens. The literature suggests this greater salience should be reflected in a greater willingness to participate.

The third question examined ratings of local water quality. Respondents rated the quality of San Antonio's water on a scale of very good to very bad. Again, the literature gives us no indication of which responses are more or less likely to be associated with willingness to participate. Although the literature on salience does suggest that greater perceived importance should result in higher levels of participation, because of the uncertainty in directionality for two of these policy specific attitudes, we will present null hypotheses for them. The following hypotheses were tested:

H.4a. *There will be no difference in levels of willingness to participate in watershed stakeholder processes between pro- and anti- environmental protection policy attitude holders.*

H.4b. *There will be no difference in levels of willingness to participate in watershed stakeholder processes between those who give high and low ratings of local water quality.*

H.4c. *Those for whom water and environmental issues have higher salience as major local problems will be more willing to participate in watershed stakeholder decision processes than those who do not express this concern.*

In addition to these specific categories of variables, we included a number of demographic indicators frequently used to differentiate populations on attitudinal dimensions—sex, age, race, education, etc. A large number of political participation studies find that increases in socioeconomic status indicators like income, education and occupation are positively associated with higher levels of participation (see, for example, Verba and Nie, 1972). These variables were included in the overall model tested.

To test the hypotheses discussed above, we estimated a single multivariate model. The results of this model are presented in Table 1.[3] The dependent variable for this analysis was dichotomous, representing each respondent's "yes" or "no" answer to the question of whether or not they would be willing to participate. With such a dependent variable, a binary probit model is the most appropriate for simultaneously evaluating all the hypotheses. It is standard practice among social scientists to estimate a binomial logit or probit model when the variable being modeled can take on only two values (in this case "yes" or "no"). Aldrich and Nelson (1984) provide an excellent introduction to this type of statistical problem and the various approaches available.

Table 1. Multivariate Model of Respondents' Willingness to Participate in Discussions on Clean Water in San Antonio

Variable	Maximum Estimated Effect
Female	-12.2%
Afro-American	*
Asian	*
Latino	+12.1%
Native American	*
High School Drop Out	*
High School Graduate	*
Took Some College	*
College Graduate	*
Some Post Graduate Schooling	*
Household Income less than or equal to $25,000	*
Household Income greater than or equal to $40,000	*
Age	*
Working	*
Retired	*
Working Greater than 40 Hours per week	-3.9%
Number of Years Living in Greater San Antonio Area	*
Named a Water Problem as San Antonio's Greatest Problem	*
Rating of San Antonio's Water Quality	-3.0%
Self Efficacy-City Government	*
Self Efficacy-National Government	*
Believes most people can be trusted	*
Frequency with which government can be trusted	*
National Economy—Expectations	*
Personal Finances—Expectations	+4.5%
Environmental Beliefs Index	+5.5%

996 Observations; 76.2% Classified Correctly

Notes: Entries in the second column indicate the estimated maximum effect of the variable in the first column on the probability that the respondent will be willing to participate, holding all other variables constant. The ethnic categories should be interpreted as the predicted effect of each category relative to Anglos. The educational categories should be interpreted as the predicted effect of each category relative to people who had a ninth grade education or less. The household income categories should be interpreted as the predicted effect of each category relative to people whose reported household income was between $25,000 and $40,000. Almost all of the variables on which analyses are reported are dichotomous, indicating that the respondent either had the characteristic or not. Exceptions to this are:

- age, measured in years;
- individuals' rating of San Antonio's water quality, varying from 4 ("very good") to 0 ("very bad");
- Self Efficacy-City/National Government which were based on whether respondents strongly agreed (0) to strongly disagreed (4) with the following statement: "People like me don't have any say about what the city/national government does";
- Frequency with which government can be trusted, ranging from 1 ("almost all the time") to 4 ("almost never");

National Economy/Personal Finances—Expectations which ranged from +1 ("better") to –1 ("worse"); The variable "believes most people can be trusted" was based on respondent's agreement (coded as 1) or disagreement (coded as 0) with this statement;

Environmental Beliefs Index—an index constructed from responses to a set of 12 environmental statements from the New Environmental Paradigm (NEP) scale (Dunlap and Van Liere, 1978). We created this summary index ranging from –2 (anti-environment) to +2 (pro-environment).[4]

Summary of Telephone Survey Results

This section identifies respondent characteristics that predict willingness to participate in discussions on clean water in San Antonio. Only a handful of the variables expected to influence respondents' willingness to participate were statistically significant in this model. An examination of demographic characteristics of survey respondents indicated that women were less likely to express willingness than men. Among ethnic groups, we were mildly surprised to find evidence that Latinos were more likely to express a willingness to participate than were other ethnic groups. People working more than forty hours per week were slightly less willing to participate.

In terms of general outlook attitudes, only financial optimism toward personal finances was positively related to participation—more optimistic individuals reported greater willingness to participate than less optimistic residents (providing support for H.3a). For specific environmental attitudes, the rating of San Antonio's water quality was a significant predictor of willingness to participate in discussions on clean water in San Antonio with respondents who thought that water quality was worse being more willing to participate. The environmental beliefs index was also significant--respondents who had more pro-environmental set of beliefs were more likely to participate than those individuals who reported less supportive environmental positions. Trust and efficacy were not significant.

Stakeholder Participant Recruitment Process

Thus far, we have identified factors that predict the expressed willingness of citizens to participate in collaborative engagement processes using respondents from the large-sample phone survey. The next step in this research project was to recruit participants for two collaborative stakeholder forums. A major objective was to ensure the procedural legitimacy of the watershed policy-making process by meeting the autonomy criterion in recruitment efforts (Sabatier et al., chapter 3: 2003). We used alternative recruitment techniques to accomplish this task.

We recruited people willing to make a year-long commitment to serving on a watershed restoration council that brought together research scientists, agency personnel, and citizen groups to share watershed information and develop management recommendations. Two councils were organized in San Antonio, one for the Leon Creek watershed and one for the Salado Creek watershed. This section describes the watershed collaboration process employed to provide insight into the choice of such time and resource intensive recruitment methods. We then describe those recruitment methods and associated results.

We adopted two perspectives essential to understanding the recruitment of citizen stakeholders as Council members. First, we held the view that all who participate in a particular "cultural domain" are "experts" of that domain (Lindlof , 1995). This perspective allowed us to be highly inclusive; virtually everyone whose life experiences included the Leon Creek watershed and/or the Salado Creek watershed were seen as potential citizen stakeholders. With attention to the geographical character of the two watersheds, Council members were recruited from the various segments of the watershed, including those living in the most affluent and the most disadvantaged neighborhoods.

Second, we used the 'snowball technique' to recruit participants. Using this technique, we developed a set of initial community contacts that gave us access to potential Council members. Each contact was approached for two reasons: 1) to determine whether or not the individual would serve as a Council member, and 2) to generate additional referrals. The technique is referred to as the 'snowball technique' because contact with one individual leads to the identification of two or three additional participants, and so on.

Briefly, citizen stakeholders were recruited as follows:

- List of community groups and organizations was generated;
- San Antonio city officials were asked to identify organizations and leaders with an interest in watershed management issues;
- Community "gatekeepers" were contacted for further recommendations of potential participants;
- Citizens were recruited directly via telephone and personal interviews.

Our recruitment strategy showed community "gatekeepers" (Lindlof, 1995) that getting involved in a Watershed Council was consistent with their other commitments, such as service to the neighborhood association (O'Keefe, 1990). Several of these individuals realized that participation in the Councils allowed them to report important information back to their groups. Others felt they did not have time to serve on yet other committees; however, contacting them was useful in terms of generating referrals. Research shows that when some people are unable to grant a large request, they are "more likely to agree to a second, smaller request" (O'Keefe, 1990). People who were unable to serve as Council members were often assisted in helping locate people to serve as citizen representatives.

Some recruitment was done via telephone. In addition, many of the recruits were also interviewed by members of the research team in one-on-one settings prior to the first Council meeting. This is an important recruitment technique because people often need encouragement to follow through on their commitment to serve on such councils. Lindlof (1995) explains that researchers "negotiate access" primarily as a matter of goodwill and intrinsic rewards. Likewise, we assured the recruits that their input was important and meaningful. By conducting personal interviews with participants, their level of commitment increased as they saw that their input was valued. In this sense, a self-selection/self-perpetuating process takes over where willing participants become even more willing to participate. During these interviews, several recruits expressed gratitude that their input was being carefully documented.

We interviewed 51 potential participants prior to, or during the first 2 months of the meetings. We conducted 41 separate interviews, most with individuals, and some with small groups. The initial interviews served both as a recruiting tool and also provided baseline information regarding participants' relevant previous experience, motivations, and expectations. Of those interviewed, 39 became regular participants (attending over 50% of the meetings).

Personal interviews also work as part of a strategy used with people who say, "well, maybe." The persuasive strategy is known as "foot-in-the-door" (FITD), and research shows this to be a successful technique when people need coaxing (Petty and Cacioppo, 1996). We first asked individuals to spend a few moments on the phone, then asked for an interview at a time and place convenient for them, and finally, we asked them to attend one meeting. Clearly there are limits to the strategy; however, one should not discount the persuasive importance of having good attendance at the first Council meetings. Some Council members later admitted they planned to attend only one meeting, but once they saw so many other 'committed' people, they decided to continue attending meetings.

The goal of the recruitment activities was to achieve representativeness in public participation for an ethnically diverse population. We attempted to construct the Councils as representative of the variety of demographic categories, including race, age, sex, socioeconomic status, and education, that exists in these watersheds. The problem with drawing too heavily from "active" community organizations, however, is that traditionally under-represented groups remain under-represented. It became apparent early on that we had primarily tapped into two groups of people. One group was composed of middle-class citizens, mostly Anglo, who were active in local government and community functions. A second group appeared to be a group of retirees committed to using their time to making their communities better places to live. This second group was much more racially diverse than the first group, however, they also tended to represent the middle class.

To enhance diversity, we adjusted our recruitment efforts to target missing groups. One approach was to use a pool of names generated in conjunction with the telephone survey described earlier. Survey respondents, who indicated they would be willing to participate in discussions on clean water in San Antonio (N=245), were asked for their contact information. To maximize the representativeness of the stakeholder groups in terms of Bexar County's basic demographic profile, these 245 respondents were ranked in terms of the importance with which contact with them should be pursued. Specifically, each of these 245 individuals was assigned a weight. Beginning with the person with the highest weight, calls were made sequentially to each of the 245 individuals on the list. Individuals were informed that they would receive a follow-up call to invite them to participate in a group that would address water-related concerns in their watershed. Following presentation of information about Council meeting times and objectives, researchers asked if they would be interested in participating in the group and if they knew of anyone (friends, neighbors, etc.) who would be interested in participating as well. After each calling session, individuals with the highest weights who were not directly contacted were placed at the beginning of the next calling session schedule. Calls were cycled through in this manner until the lists had been exhausted. Repeated call attempts were made to all individuals on the list with concerted efforts to recruit those individuals with the highest weights. Finally, letters and telephone contacts were made a few days prior to meetings to encourage attendance.

Overall, the recruitment methods yielded initial agreements to participate from 26 agency/government officials, 40 community/civic organizational contacts, 22 personal referrals, and 5 phone survey respondents.

Stakeholder Pretest Survey

Representation: Comparison of Collaborative Learning Stakeholder Groups with Phone Survey Samples

The criterion of procedural legitimacy (Sabatier et al., chapter 3: 2003) can be satisfied in two distinct ways. One approach is based on statistical representation—that is, to what extent does the composition of the stakeholder groups represent the larger populations from which these participants were recruited? Survey data collected in this project allowed us to address this question quantitatively. This section presents survey results using two subsamples from the Bexar County population (Salado and Leon Creek watersheds) as well as the corresponding CL stakeholder groups from these watersheds. The phone survey of the general Bexar County population and surveys administered to Council members at the first CL meeting (November 1999) contained a series of overlapping questions with identical content. The final section provides a direct comparison of the survey results from these groups of respondents with respect to the following variables: demographic characteristics, general political beliefs/ideology, environmental attitudes, and knowledge/beliefs of respondents about water quality in the Salado and Leon Creek watersheds of San Antonio.

Concerning demographic attributes, inspection of Table 2 reveals a general picture of the similarities and differences between the stakeholder councils and the corresponding watershed subsamples of the Bexar County population. For comparison purposes, the second column contains demographic data from the 1990 Census. The third column presents the weighted frequencies for the overall phone survey sample of Bexar County residents. The fourth column of Table 2 provides the percent of respondents (or median values) in the Salado Creek watershed subsample phone survey (N= 194) for each demographic characteristic listed. The fifth column in Table 2 presents the comparable values for the Salado Creek stakeholder council (N = 60). The sixth and seventh columns present the Leon Creek watershed subsample (N= 218) from the phone survey and the Leon Creek stakeholder council (N= 35), respectively.[5]

On balance, Table 2 reveals that the composition of the two stakeholder groups was reasonably similar to the statistical profile of residents in the Salado and Leon Creek watersheds for the demographic attributes of age, income, and median hours worked. However, for the characteristics of sex, race, educational level, marital status, children in household, and employment status, there were some apparent differences. Women in both councils were under-represented relative to men. Afro-Americans were over-represented to some degree in the Salado Council. However the percentage of Anglos in the Leon Council was substantially higher than in the Leon watershed subsample proportion. One striking finding in Table 2 was that Latino participants were substantially under-represented in both council groups (particularly the Leon Creek Council) relative to the proportions in the corresponding watershed subsamples. Both college-level and graduate education was over-represented in both stakeholder councils, relative to the watershed subsample populations.[6] For marital status, the results were mixed: Married individuals were somewhat over-represented in the Salado Council, but not in the Leon Council. In addition, individuals with children (under 18) in the household were somewhat over-represented in the Salado Council, but this pattern was reversed in the Leon Council. Finally, members of both councils were employed in higher proportions than in the relevant watershed subsample.

General political beliefs and ideology (see Table 3) were assessed with the same set of four questions in the phone survey and in the stakeholder council surveys. One item asked respondents to indicate which of the following two statements was closer to their own views: (1) “The less government, the better”, or (2) “There are more things that government should be doing”. (A third option offered was “Indifferent”.) The next two questions were identical to the national government and city government self-efficacy items described earlier in this section. The final question asked, "Generally, when you think about politics, which of the following best describes you?" (Strong Republican; Weak Republican; Independent, but lean Republican; Pure Independent; Independent, but lean Democrat; Weak Democrat; and Strong Democrat).

Table 2. Demographic Characteristics

Demographic Characteristic	Bexar County 1990 Census	Total Bexar County Sample (N= 1017)	Salado Creek Watershed Subsample (N= 194)	Salado Creek Stakeholder Council (N = 60)	Leon Creek Watershed Subsample (N= 218)	Leon Creek Stakeholder Council (N = 35)
% Female	51.4	52.6	55.7	41.7	54.1	42.9
% Married	53.9	59.9	52.1	76.3	60.6	57.6
% Children 18 /Under (in Household)	---	62.2	34.0	41.7	47.3	36.4
Median Age	35-39	47	49	48	45	51
Median Income	25-29K	50-59K	50-59K	60-69K	50-59K	60-69K
% HS Graduate	72.5	90.5	25.8	5.0	17.4	3.0
% Some College/ College Graduate	41.1	68.1	52.6	45.0	62.4	42.4
% Some Graduate/ Masters/Doctorate	5.9	15.4	12.4	46.7	15.1	54.5
% Afro-American	4.7	6.9	11.9	20.7	3.7	0.0
% Anglo	49.7	58.7	62.4	67.2	59.2	78.1
% Latino	33.0	27.4	19.1	6.9	30.3	3.1
% Employed	---	60.6	55.7	67.8	65.6	81.3
Median Hours Worked (Weekly)	---	40	40	40	40	48

Table 3 presents the results on these four items for the Salado Creek and Leon Creek stakeholder groups using a similar format to Table 2. On the role of government question, there appear to be no differences between the two stakeholder groups and their respective watershed subsamples in terms of support for statement that "government should do more." However, we did observe less frequent endorsement of the "less government, the better" ideology in the stakeholder council groups relative to the corresponding watershed subsamples. Another consistent result across both the Salado and Leon watersheds was that the stakeholder councils reported a higher sense of self-efficacy in influencing national and city government policies. In terms of political affiliation, the patterns varied depending on the particular Council group: the Salado Council is typical with respect to percentage of Independents, although Democratic representation was somewhat higher and Republican representation lower relative to the Salado watershed subsample; for the Leon Creek Council, however, Independents were over-represented compared to the statistical profile in the Leon watershed subsample. Democrats and Republicans were under-represented in the Leon Council.

Environmental attitudes were measured in the stakeholder surveys using the same 12-item New Environmental Paradigm (NEP) scale (Dunlap & van Liere, 1978) described earlier for the telephone survey. Table 4 lists the 12 items in the NEP scale and the corresponding percentage of respondents who indicated agreement with each statement.[7]

Table 4 shows that, in general, the Salado Council members were representative of the watershed in terms of their support for the New Environmental Paradigm. The only three items where there was stronger endorsement of the NEP in this Council were statements dealing with limits to growth (items highlighted in **boldface** in Table 4)

Table 3. General Political Beliefs and Ideology

General Political Beliefs/Ideology	Total Bexar County Sample	Salado Creek Watershed Subsample	Salado Creek Stakeholder Council	Leon Creek Watershed Subsample	Leon Creek Stakeholder Council
Less Gov't, The Better	41.6	44.3	31.6	42.7	31.3
Gov't Should Do More	54.2	53.6	54.4	53.2	56.3
Indifferent	2.8	1.0	14.0	2.8	12.5
Self-Efficacy/National Gov't (% Disagree)	56.7	54.1	71.7	59.7	69.7
Self-Efficacy/City Gov't (% Disagree)	62.3	64.4	80.0	64.3	81.8
Democrat	26.1	30.6	39.2	26.3	16.2
Independent	43.1	38.3	42.9	38.3	58.1
Republican	30.7	31.0	17.9	35.5	25.9

Table 4 shows that the Leon Council was considerably stronger in its support for the NEP compared to the corresponding Leon watershed subsample. In addition to the limits to growth items, the Leon Council also rejected the notion of man ruling over nature to a greater extent than the broader Leon watershed population. Endorsement for the NEP was stronger (i.e., 10% difference or higher) on 9 of the 12 scale items (see percentage values for LCRC highlighted in **boldface** in Table 4). Thus, the level of support for the NEP appears to be substantially higher in the Leon Council compared to the Salado Council.

Table 4. New Environmental Paradigm (NEP) Scale Items

New Environmental Paradigm (NEP) Scale Item	Total Bexar County Sample (% Agree)	Salado Creek Watershed Subsample (% Agree)	Salado Creek Stakeholder Council (% Agree)	Leon Creek Watershed Subsample (% Agree)	Leon Creek Stakeholder Council (% Agree)
We are approaching the limit of the number of people the earth can support.	**46.4**	**45.3**	**62.7**	**40.8**	**60.6**
The balance of nature is very delicate and easily upset.	76.2	76.8	81.7	73.4	68.8
Humans have the right to modify the natural environment to suit their needs.	42.1	30.9	35.6	**46.8**	**35.5**
Mankind was created to rule over the rest of nature.	41.7	36.6	35.0	**46.3**	**31.0**
When humans interfere with nature, it often produces disastrous consequences.	75.8	75.3	83.1	**68.8**	**86.7**
Plants and animals exist primarily for human use.	36.2	34.5	28.8	**39.5**	**17.2**
To maintain a healthy environment we will have to develop a steady-state economy where industrial growth is controlled.	75.2	80.9	71.4	74.8	82.8
Humans must live in harmony with nature in order to survive.	90.9	87.1	95.0	89.5	90.9
The earth is like a spaceship with only limited room and resources.	**69.1**	**68.6**	**83.1**	**69.7**	**84.4**
Humans need not adapt to the natural environment because they can remake it to suit their needs.	24.2	20.6	16.7	**23.4**	**9.4**
There are limits on growth beyond which our industrialized society cannot expand.	**59.2**	**57.2**	**77.6**	**55.1**	**77.4**
Mankind is severely abusing the environment.	77.3	77.8	84.7	**73.4**	**84.4**

Finally, Table 5 presents the results from five survey questions dealing specifically with water quality in the Salado and Leon Creek watersheds of San Antonio. We wanted to assess how familiar respondents were with Salado and Leon Creeks and how safe they considered various activities that required contact with the water. In general, Salado Creek was quite familiar to both stakeholders and watershed subsample respondents alike. With respect to water quality assessments, the overall picture in Table 5 was that Salado Council members were consistently less convinced of the safety of Salado Creek water for human consumption, swimming, eating fish, and for livestock consumption than were residents in the broader Salado watershed sample.

Table 5 also presents the comparable results for Leon Creek. In general, Leon Creek was not as familiar to watershed subsample respondents as to Leon Council members. More striking, however, was the large disparity in the water safety beliefs between the Leon Council and the broader Leon watershed population. This pattern of results suggests that knowledge of water quality in Leon Creek may be less widely distributed among residents living in this watershed compared to Salado Creek. In sum, Table 5 demonstrates that Leon Council members were both more familiar with the creek and rated the water as less safe for all uses compared to the corresponding Leon watershed subsample. These effects appear to be more pronounced than for the Salado Council (see Table 5). The differences between the watershed subsamples and the council groups in perceived seriousness of water quality problems may be related to the demographic differences in education level and ethnic status (% Hispanic) documented in Table 2 (see also Endnote 6).

Summary of Recruiting Results in Achieving Representative Participation

This section addresses the following basic research question: "To what extent did the combined effect of the snowball, interview, and survey recruitment techniques achieve statistical representation of the relevant watershed populations in our watershed councils?" We explored this question along four dimensions: demographics, general political beliefs/ideology, environmental attitudes, and evaluations of water quality. The council groups were representative of their respective watershed populations on 3 of 9 demographic characteristics: age, income, and number of hours worked. Despite dedicated efforts to achieve adequate representation of the watershed communities at large, the recruitment methods did not fare as well for two specific subgroups of citizens: women and Latinos. However, our recruitment efforts were effective with the Afro-American community in the Salado Creek watershed.

Table 5. Knowledge and Beliefs About San Antonio Water Quality

Knowledge/Beliefs About SA Water Quality	Total Bexar County Sample—Salado Creek	Salado Creek Watershed Subsample	Salado Creek Stakeholder Council	Leon Creek Watershed Subsample	Leon Creek Stakeholder Council
Familiar with Creek (% Yes)	73.0	86.1	85.0	60.1	91.4
Safety of Water—Drinking (% Safe)	11.0	8.4	3.4	22.1	0.0
Safety of Water—Swimming (% Safe)	16.3	10.2	1.7	34.4	5.9
Safety of Eating Fish (% Safe)	20.0	16.8	6.9	29.0	8.8
Safety of Water—Livestock Drinking (% Safe)	46.3	45.5	27.6	62.5	29.4

With respect to political beliefs/ideology, we succeeded in representing residents holding the belief that "government should do more", but under-represented residents with the belief "the less government, the better". However, our methods did not reproduce distributions of political affiliation mirroring the respective watershed populations. For environmental attitudes, the council groups demonstrated higher environmental concern (i.e., support for NEP) than their corresponding watershed samples. Finally, the council groups were composed of residents who reported both greater familiarity with the creeks and more negative beliefs about water quality than the larger Salado and Leon watershed populations.

Stakeholder Collaborative Learning (CL) Meetings

Design, Methods, and Goals

Members of the social science research team (Peterson, Gilbertz, Samuelson, Whitten), working closely with the ecological science research team (Matlock, Neill, Kenimer), facilitated a total of 26 stakeholder meetings over a period of 14 months (from November 1999 until December 2000) in the two watersheds (Salado Creek and Leon Creek). Each meeting took the form of a 4-hour workshop and was both audio- and video-recorded for later review by the research team. The meetings were designed to enable stakeholders to work together to develop an acceptable, workable plan for watershed development and protection.

The research team employed the Collaborative Learning (CL) process as the mechanism to facilitate analysis and deliberation in the watershed restoration council meetings (Daniels & Walker, 2001). Collaborative Learning is a relatively new method for structuring and facilitating communication among stakeholders in the context of ecosystem management. The CL process intervention (Daniels & Walker, 2001) is grounded in theoretical work on soft systems methodology (Checkland & Scholes, 1990; Wilson & Morren, 1990) and alternative dispute resolution (Fisher, Ury, & Patton, 1991; Gray, 1989). The concept of "soft systems" represents an extension of theoretical work on systems analysis (Senge, 1990) and experiential learning (Kolb, 1986). The basic assumption in soft systems methodology is that management of complex problem situations demands a different approach than the typical "hard" systems method used in engineering. Such hard systems methods focus on <u>outcomes</u> instead of processes, and consequently do not attach importance to learning. Management of an ecosystem is a situation where problems are often characterized by a high degree of uncertainty and equivocality. The soft systems approach focuses on "situation improvements" that can result from active learning and debate. The CL process uses information as the foundation of communication between conflicting parties, with a clear objective that participants are not expected to achieve consensus, but rather improved understanding of the watershed processes being managed and the basis for each other's perspectives.

The CL process also promotes development and identification of stakeholder concerns and underlying interests. Recent theory and research on negotiation and mediation has adopted an interests-based approach. One viewpoint (e.g., Bazerman & Neale, 1992; Fisher et al., 1991) maintains that traditional positional approaches to negotiation may be inefficient because positions taken by parties in conflict are often extreme and obscure the underlying interests (i.e., needs, concerns, values) that parties seek to advance through negotiation. The structured set of CL activities is designed to move participants away from positional strategies and toward the identification of mutual interests and joint gains from collaboration. Moreover, the presence of outside facilitators during CL sessions permits use of effective principles from mediation theory and research (Gray, 1989).

The CL meetings were designed to enable stakeholders to move through four phases of application. First, participants were informed about the CL process and received training in systemic thinking and collaborative discussion. Second, formal presentations by technical experts (including members of the ecological science research team), panel question/answer sessions, and informal small group discussions provided a common knowledge base regarding the major issues affecting the watershed. Participants began to generate specific suggestions for improvement of the current situation. Third, a series of active learning exercises helped participants think systemically about the watershed ecosystem and enable them to identify key issues, concerns, and interrelationships between variables affecting water quality. Participants were encouraged to refine their ideas by sharing them with other participants through structured activities that offered opportunities to articulate ideas within a modeling framework. Fourth, through collaborative debate with other stakeholders, a final system of improvement suggestions was organized, discussed, and refined. Although we have listed the layers of application numerically for ease of explanation, the description of specific group activities below should demonstrate that the process did not occur in such a linear manner, but iteratively.

When CL operates as it should, all participants (including scientists and technicians, decision-makers, residents) become stakeholders who learn from each other. In the beginning of the relationship, the technical experts tend to act as teachers; but, as all stakeholders become better informed and gain confidence in their understanding of new perspectives, the roles of teacher and student become blurred.

Collaborative Learning Meeting Activities

Thirteen CL meetings were held over a 14-month period (November 1999 until December 2000) with each of the two watershed restoration councils (Salado and Leon Creek). Over this period, we designed a variety of activities to encourage the development of systemic understanding of the watersheds. Stakeholders used these exercises to determine what material and information flows were most critical to include in a model of their watersheds. Highlights from some of these activities illustrate a sense of how the watershed as a system emerged.

Introductory Activity

At the beginning of the first council meeting, facilitators wound a length of blue fabric across the floor of the meeting room. We asked members to imagine the fabric was their creek, and to position themselves at an appropriate point along its banks. Members came forward tentatively, and by using the locator markings on the fabric (such as points where major highways crossed the creek, parks, and schools), chose a position. Facilitators gave each member a lunch bag filled with objects from the creek, then pulled an empty red wagon to the furthest upstream point. They offered the handle of the children's toy to the person standing furthest upstream, and invited him to introduce himself, dump the contents of his lunch sack into the creek, then pass the wagon to the person directly downstream from him. Each member followed this procedure. As the wagon filled up with objects ranging from leaves and twigs to chunks of old tires, it began to overflow, and once tipped over, spilling its contents all over one stakeholder's "property". Introductions were punctuated by laughter, shouts, and groans, depending on what happened with the wagon and its contents. A sampling of the self-introductions by SCRC members indicates the diverse identities of group members (pseudonyms are used to protect confidentiality), as well as a tendency to think only of one aspect of the watershed.

Because he had positioned himself at the uppermost point along the creek, Calvin was the first participant to introduce himself. He began: "I'm with the San Antonio Water System, and I didn't really know where to stand along the creek. I'm actually concerned with all the water quality along Salado Creek. My department is concerned, obviously with water quality. We don't really care where we go; if it's dirty we help clean it up." Calvin illustrates the attitude shared by most agency participants. Although they did not identify with a particular location along the creek, they identified strongly with the aspect of the creek for which they held technical responsibility. Ramona, who lived near the creek and was interested in the aesthetic quality of the city, said, "I am seriously, anxiously, interested in seeing this creek reach the beautification that I know it can. It can be beautified." Hank, a farmer whose family members have lived on the creek since they immigrated from Italy in the 1890s, told participants, "We live up the road there on the ---- Farm. It's our livelihood. We farm there; a family farm. We are concerned about the quantity and quality of the water so we can keep the crops growing." Madeline, a science teacher, was "concerned about upstream development, non-point source pollution, and about truly educating our children about the tremendous value of the creek." She compared the creek to "an emerald necklace," and said she was participating because she wanted "to preserve the treasure." Daniel, a retired military officer, spoke about his concern with flooding: "I'm with the

---- Home Owners Improvement Association. . . . and when Salado Creek flooded it came right into my house and so I'm definitely interested in what slows down Salado Creek, and I am hoping that it won't happen again."

These individuals represent the primary motives that brought people to council meetings. People came because they had professional responsibility for the creek, wanted the creek to enhance San Antonio's aesthetic appeal and recreation opportunities, wanted to use the creek's economic potential, wanted to restore ecological integrity to the watershed, or wanted to prevent damage caused by flooding. In addition to having one of these interests, several participants also claimed a personal connection to the creek.

Following the self-introductions, the Mayor expressed his enthusiastic support for the council's work, and his hope that it would become a positive force for change in how the city managed creeks. Referring to the exercise, he encouraged members to think systemically about the watershed, including concerns from all reaches of the watershed in their recommendations. When he told them about a political initiative he was developing that could fund some of their recommendations for creek restoration. Daniel immediately raised his hand and requested that the funds be spent to clean out the creek bed immediately upstream of, and throughout his neighborhood. He asked that all brush and trees be removed from the creek banks, and that the creek bed be lined with concrete. The Mayor responded by saying he hoped Daniel would think about the entire watershed, and would consider the needs of other council members before deciding on such a drastic course. Daniel just shook his head in frustration at this answer.

When participants first began attending meetings, each had a specific point from which they identified with the watershed. As the process unfolded over time, individual identities became enmeshed in the system represented by the entire group, which was mutually shaped by all participants. Responses of those who developed strong systemic identity through their participation indicate that, while this more systemic perspective did not minimize individual interests, it enabled members to position themselves more productively in relationship with the identities of other members.

These stakeholders had not previously imagined the possibility that concerns so diverse as theirs could form any unifying motive to act. As they developed a more systemic understanding of the watershed, however, unifying motives emerged. They eventually developed a simulation model that centered on alternative flooding possibilities associated with different channel conditions, and land development patterns. They used this model internally to help refine a set of recommendations for improving their watersheds, and externally to help explain the logic of that recommendation set to city officials.

Meeting Plan and Agendas

General Meeting Format

Meetings generally followed a format that began with the introduction of new information that stakeholders had requested. Members then broke into small groups where they discussed how this information could be applied to improving the current situation. Finally, all groups reported back, and facilitators led a plenary discussion. Appendix A contains the formal agendas distributed at each of the Council meetings.

At first, individuals simply reiterated their own interests, often failing to respond to information presented by other stakeholders. Our facilitation approach in these discussions was grounded in CL's emphasis on mutual learning and systemic thinking. Most of the time we allowed stakeholders to express themselves, even when the extent of their remarks threatened the accomplishment of the meeting's pre-determined goals. So long as other participants appeared relatively attentive to someone's statement, we did not cut it off. We focused, rather, on guiding participants toward discussion of how individual interests and claims complemented each other, and how together, they defined the watershed.

Over time, participants developed an increasing desire and competence to engage in systemically grounded environmental management decisions, as they began to integrate their own needs into what they perceived to be the needs of fellow council members. At the eighth meeting, subgroups were attempting to use their situation maps to develop the essential elements for building a model of the watershed, and for determining the recommendation set they would ultimately present. One subgroup, including members representing all the interests mentioned during the first meeting, reported its discussion as follows:

> "Well, we wanted to protect the ecology and water quality. We felt like work that needed to be done, in this area that was flooded, that we have to consider protection of property and life and that would be the driving force in that area. And what we would say is that would be the driving force, but also keep in mind protecting ecology, protecting water quality and protecting wildlife in the area. And at the same time develop recreation facilities to serve people in that area and also all citizens so that the whole city would benefit which would encourage spending money on the creeks. So that is what we were looking at, that protection of life and property would be in the forefront, but flooding improvements should be done in such a way to protect the wildlife and also provide recreation opportunities and other economic opportunities."

Participants were beginning to use their situation maps to specify what they wanted to model, how the components of their model related to each other, and what they hoped to learn from their model. Some aspects of these situation maps were incorporated into a version of BASINS-HSPF. More importantly, they provided the basis for developing a more simplistic, but accessible model that responded directly to stakeholder concerns. From this point, council members spent an increasing portion of each meeting developing and refining a simulation model that would enable them to explain the recommendations they would present to the city as a unified system for improvement. They also began discussing how to encourage implementation and enforcement of their recommendations.

Simulation Model Development and Use

During the eleventh meeting, which was facilitated by Dr. William Grant (an expert in systems modeling at Texas A&M University), and Dr. Ann Kenimer (a member of the ecological science team), participants piloted a simulation model they had assisted in developing. We loaded the model onto several laptops, so council members could run the simulations they found most interesting, and even alter basic constructs if they so desired.

The model provided several features beneficial for use as a stakeholder education tool. First, watershed management options were presented to the stakeholders using an easy-to-use user interface. The stakeholders identified model input parameters based in different land uses (development patterns), retention structures, and channel conditions they had previously determined were possible. Kenimer and Grant had used these parameters to develop a menu of watershed characteristic options, from which users could choose. Although the model was severely oversimplified, it was still a useful learning tool. Model predictions were also easy to interpret. Each model run generated a single hydrograph showing the resulting watershed runoff as a function of time. By changing watershed characteristics, stakeholders could generate several runoff hydrographs. Hydrographs for multiple scenarios could be plotted on a single graph to facilitate comparison.

The four-hour workshop began with a brief verbal description of the model, given by a council member. We then asked stakeholders to divide into groups of two or three people, with each group having at least one person who had used a computer, and no group having more than one new member. Grant then showed the group what he and Kenimer had done with their concept. Using a copy of the same model they were working with, he walked members through the model basics, beginning with the user interface mentioned above. Members could either look at what he was projecting on the screen, or at their own screens. While Grant spoke, other research team members circulated among the groups, ensuring that all groups were able to manipulate the model as directed. When all the groups seemed comfortable working on their own, we invited them to experiment with various scenarios. Grant, joined the other research team members in roaming the room to assist when requested.

Most groups spent the remaining time running various combinations of the alternatives offered in the menu. After a short time of experimentation with the predetermined options, other groups became dissatisfied with the model's simplicity, and wanted to complexify it. For example, the model allowed only three pre-determined channel conditions, and some groups wanted to change that to a continuum that ranged from concrete to full brushy understory. Grant or Kenimer assisted groups who wanted to make changes of this nature.

While all stakeholders participated (to some degree) in this evaluation activity, their participation varied. A few had never touched a computer before their participation in the councils. These members usually contented themselves with asking another group member to manipulate the model. For example, one of them asked her partner, "let's see what happens in the hundred-year flood when we put in a concrete channel in the central segment only." Although she kept her hands in her lap throughout most of the workshop, she leaned forward, critically examining model output for each combination her partner punched in. On the other extreme, some groups wanted to make the model itself more complex. One group determined the data used to simulate the influence of retention structures were insufficient, so they sent one of their team members to his office, where additional data were available.

After allowing participants to experiment with their model for approximately two hours, facilitators asked the groups to join in a plenary discussion of what they had learned, and how this could be applied to the recommendations they had prepared. Most members were eager to share their insights with others. The discussion evolved into a lively debate regarding what combination of management options the council wanted to recommend. Sometimes when teams found it difficult to explain their rationale for a particular combination of management options, they turned to the simulation model. One team sent a member to the front of the room, and had him show the others what they meant by running a simulation they had found especially intriguing on the computer that was hooked up to the projector.

As the workshop drew to a close, stakeholders determined they would use their simulation models next month when they presented recommendations to the Mayor. They claimed it might help him understand why they preferred certain combinations over others. The final two meetings provided opportunities for ecological science team members (Matlock, Kenimer, Neill) to share preliminary data analyses, and for council members to craft recommendations for members of city government and relevant agencies.

One example of mutual learning that was facilitated by participation in this modeling activity occurred at the end of a formal presentation in the twelfth meeting of the Leon Creek Restoration Committee (LCRC). The LCRC members, who had by this time conceptualized, evaluated, and used a simple model of the flow in Leon Creek, listened attentively as William Neill, an aquatic ecologist, presented some rather complicated results that compared observed and modeled responses of fish-performance bioassays. The ecologist used a simple simulation model to interpret the significance of his results. Stakeholders began to interrupt (which they had been encouraged to do) with questions. They were familiar with STELLA, the program that Neill had used to develop his simulation model, and they suggested an alternative interpretation for some of his results. The questions and related discussion caused Neill to re-evaluate his previous interpretations. He offered to try some additional model simulations that more directly responded to the issues raised by council members. When they responded affirmatively to his offer, he constructed these alternative simulations on the spot. As he worked, he continued to project the visual image from the computer onto the large screen, explaining each step, and requesting continual feedback. In this way, both the research team and members of the LCRC developed an increasingly sophisticated sense of the simulation model's capacity and significance within the watershed.

Role of Ecological Scientists

The ecological science team (Matlock, Kenimer, Neill) played an integral role in the planning and implementation of the CL meetings with the councils. From the start of the project, it was decided that we would design the activities of these workshops to facilitate communication between the ecological scientists on the research team and the citizen participants. Formal presentations were frequent, followed by direct question-and-answer exchanges between the audience and the scientists. The ecological science team attended virtually all of the CL meetings and were available to answer questions from council members during the workshop sessions and also during breaks.

The method of Collaborative Learning used by the social science team provided an effective framework for integrating the results from the quantitative models described in the Ecological Investigation section of this report. A common knowledge base on basic hydrological principles, flood control strategies, and specific quantitative models such as BASINS-HSPF and ECOPHYS.FISH was achieved through formal presentations by Drs. Kenimer, Matlock, and Neill at CL workshop meetings beginning in April 2000. The iterative feedback cycle was initiated during this phase of the CL process by allowing stakeholders to question the ecological science team following their presentations and offer specific feedback about the results of their simulations. Discussion between members of the stakeholder groups and the ecological scientists ensued and alternative scenarios were suggested for simulation by these quantitative models. Drs. Matlock and Neill actually ran these suggested simulations and reported the results back to the stakeholder groups either at the same meeting, or at a subsequent stakeholder meeting. In October 2000, Dr. Kenimer, with the assistance of Dr. William Grant of the Wildlife and Fisheries Science Dept. at Texas A&M University, developed the STELLA Watershed model based on stakeholder feedback at previous CL meetings and the entire workshop agenda that month was devoted to allowing stakeholders to test various flood control options using various assumptions about storm severity and duration (see description under STELLA Watershed model). The "hands-on" experience gained by the stakeholder groups with this modeling activity had a significant impact on the thinking of participants, particularly about the issue of flood control in Salado and Leon Creeks. This impact was illustrated dramatically at the November 2000 meeting of the Leon Creek Restoration Council, in which the spokesperson presented the group's action plan to the Mayor of San Antonio, Howard Peak, and explicitly used results from their STELLA Watershed simulation runs to support their recommendations for watershed rehabilitation.

The following is an example of how one of our ecological scientists evaluated his experience in working with the council groups. Referring to the complex modeling of fish placed in cages in Salado and Leon Creek, William Neill observed: "These are difficult relationships to grasp, even for trained scientists and engineers. Perhaps, the most important lesson our ecological research team learned in this project, was that our stakeholder groups could and would engage the important technical aspects our findings—if we worked to make the communication process meaningful and efficient. The modeling approach that we used to facilitate our research team's own understanding and interpretation of experimental results turned out to be a natural tool for really substantive communication within the larger team. And, by showing the models in action (please see http://wfscdisted.tamu.edu/wfscdisted/courses/wfsc616/EPAdemo.exe)—as opposed to only showing modeled outcomes—we developed the kind of reciprocal exchange of ideas within the larger team that we researchers found not only rewarding and heartening but, at moments, actually astonishing. We were on the verge, by project's end, of capturing the vision stated in our title: *Development Of An Urban Watershed Rehabilitation Method Using Stakeholder Feedback To Direct Investigation And Restoration Planning*. It was a privilege to have been a part of the effort."

ECOLOGICAL SCIENCE INVESTIGATION

Water quality at the watershed or basin outlet is indicative of the degree to which the homeostatic processes within an ecosystem have been disturbed (Cairns & Pratt, 1990; Barbour & Stribling, 1991). Bio-indicators of water quality (quantifiable biological parameters of an ecosystem that are responsive to a pollutant or stressor of concern) within the watershed should provide valuable information on the ecological integrity of the watershed ecosystem (Cairns & McCormick, 1991). The benefit of using bio-indicators over standard water quality measurements is their ability to integrate over time the impact of water quality and flow and the interactions of these variables on the biological health and response of the ecosystem (Barbour & Stribling, 1991). Standard chemical sampling only indicates the water quality status at discrete points in time and does not reflect the over-all health or resiliency of the ecosystem. We investigated landscape-level ecosystem impacts on aquatic ecosystem processes. We evaluated land use change in the watersheds, and used periphyton and fish as bio-indicators of ecosystem integrity to compare these watersheds.

Watershed Land Use Change

Between 1982 and 1997, the amount of urbanized land in the USA increased by 47% to 30.75 million ha, while the population grew by 17% (Fulton et al., 2001). During the same time period, the conversion of land for development was estimated to have increased from about 500,000 hectares per year between 1982 and 1992 to 1.3 million ha per year between 1992 and 1997 (NRCS, 1999). In general, urban sprawl in the south has been exacerbated by a decline in population density in urban centers, though to a lesser extent in Texas where urban population densities have decreased less than in other southern metropolitan areas (Fulton et al., 2001).

Population growth has been especially rapid in the states along the USA-Mexico border (U.S. Census Bureau, 1993). In Texas, a border state, the human population is projected to increase from 19 million to 33 million by 2030, with over 70% of the growth expected to occur along the central and southern portions of the I -35 highway corridor and in the Lower Rio Grande Valley (Conner & James, 1996). As a result of this growth, San Antonio has become one of the fastest growing metropolitan areas in the USA experiencing a 25.2% increase in population from 1990 to 1998, reaching approximately one million in 1996, and now being the eighth largest city in the country (SAEDF, 1999). This growth can be largely attributed to a steady growth in employment in the San Antonio area during the latter half of the 1990s when several large manufacturers moved into the area in response to the North American Free Trade Agreement (Rylander, 1997).

This population growth is increasingly impacting rural areas, especially those close to major urban centers in the southern part of Texas, by accelerating land subdivision and reducing the average size of land parcels (Conner & James, 1996). In addition, increase in urban sprawl generally leads to greater traffic volumes, increased pressure on local resources, less open space (Holtzclaw, 1999), and such land-use changes often have a significant negative impact on the affected ecosystems and the goods and services that they provide. Ecosystem services represent the benefits that living organisms derive from ecosystem functions that maintain the Earth's life support system, and include nutrient cycling, carbon sequestration, air and water filtration, and flood amelioration, to name a few (Costanza et al., 1997).

While changes in land-use may significantly affect ecosystem processes and services, monitoring and projecting the impacts of such land-use changes are difficult for several reasons. Monitoring changes at the regional scale (where the impact of land-use changes on ecosystems often become noticeable) is difficult because of the large volume of data and interpretation required. In addition, accurately quantifying the impacts of urban sprawl on changes in ecosystem services is difficult because of the lack of information about the contribution of alternate landscapes to these services. Finally, in order to facilitate informed decision-making by comparing the impact of anthropogenic land-use changes with the effect of "natural" ecosystem changes requires more explicit measures than simple value indices.

The objectives of this study were: (1) to evaluate the efficacy of using LANDSAT multispectral scanner (MSS) data to quantify land-use change in Bexar County, Texas from 1976 through 1991, and (2) to determine if generalized coefficients can be used to evaluate changes in ecosystem services at the watershed scale.

Using LANDSAT MSS Data to Measure Land-Use Change

Potentially adverse ecological impacts of urban sprawl have increasingly prompted attempts to map and characterize urban and suburban growth. The U.S. Geological Survey is developing a geo-referenced database of urban land-use change in selected metropolitan regions by merging information from historical maps, census statistics, commerce records, remotely sensed data, and digital land-use data (Acevedo et al, 1997), but this database is incomplete. As historical satellite imagery has become more readily available and less expensive, LANDSAT imagery has become an important tool for acquiring environmental data at spatial, temporal, and spectral resolutions appropriate for assessing broad land-use changes (Verstraete, et al., 1996).

While the relatively low 80x80m spatial resolution of the LANDSAT MSS data limits the detail that can be extracted from these data, ancillary data, such as maps reflecting land-use at the time that a satellite image was taken, can facilitate classification of coarse-resolution images. If coarse-resolution data and classification levels provide sufficient explanatory power for a given purpose, their use may be advantageous because they are less data intensive and provide better broad-scale uniformity than finer resolution data and classification levels (Bourgeron, et al., 1999). Moreover, because LANDSAT MSS images were initially produced as early as 1972, MSS data represent the most comprehensive data set available for analyzing large-scale land-use changes during the last 25 years.

While in some instances it is desirable to use high-resolution data to conduct detailed land-use analyses, such data cannot be used to quantify long-term land-use changes. Aerial photographs have been used since the 1940s and thus predate LANDSAT MSS, but such images are generally not available for a specified area at regular intervals. In order to study temporal changes, a time series of images for the location in question must be available. Satellite based imaging (e.g., LANDSAT MSS, LANDSAT TM, etc.) was the first technology to routinely produce images at regular intervals. Digital land-use maps (based on a wide variety of data including LANDSAT images) can also facilitate analyses of land-use patterns. However, because they are composed of data averaged over some time period, such maps do not represent time-specific data and, therefore, cannot be used for time-series analyses of land-use change. We used LANDSAT MSS data to classify land-use during a 15-year period in Bexar County because: (1) They provided readily available and affordable time-specific digital data obtained at regular intervals since the early 1970s; (2) an objective of this study was to quantify long-term changes in land-use; and (3) the resolution of the data was sufficient for classifying land-use patterns at the watershed scale.

Estimating the Value of Ecosystem Services

Abramovitz (1998) pointed out that ecosystem services have extensive economic value but that they are not credited for the non-market values they provide until they become depleted. While economic tools can be used to identify trade-offs between known ecological values, it remains challenging to link technical measures of ecosystem services to attributes that can be effectively evaluated by untrained individuals (Schaberg et al. 1999). Despite this and other challenges, several attempts have been made to estimate the worth of natural resources. Most notably, Costanza et al. (1997) presented a model for placing an economic value on different biomes and the services that they provided. Based on their model, they estimated that the global biospheric value of 17 identifiable ecosystem services provided by the 16 dominant global biomes is $33 trillion per year, most of which is outside the market. However, because of uncertainties, they stated that this should be considered to be a minimum estimate.

While Costanza et al.'s article did focus debate on the importance of ecosystem services that are generally undervalued in standard economic analyses, their cross-sectional estimate based on average, often local, per-unit values, was widely criticized by economists for both theoretical and empirical reasons (Pimm, 1997; Towman, 1998; Masood & Garwin, 1998; Noorgaard et al., 1998; Pearce, 1998). For example, because the last hectare of an ecosystem to disappear is likely to be worth much more than the first, simple multiplication of selected average values by all the units in the biosphere underestimated a potentially infinite social value of ecosystem services. Pearce's (1998) greatest concern was that Costanza et al.'s estimated $33 trillion "value of everything" is larger than the world GNP which is around $18 trillion per year. Since 1997, additional studies conducted to quantify the value of ecosystem services have produced lower estimates. For example, Alexander et al. (1998) estimated that ecosystem services are 44 to 88% of global GNP and concluded that while this estimate is lower than Costanza et al.'s estimate, it nevertheless indicates that accounting for ecosystem service values would greatly alter current GNP estimates. In a regional study using locally derived data, Seidl and Moraes (2000) re-estimated the ecosystem contribution of the Pantanal sub-region Nhecolandia to global production and derived a value of $15.5 billion per year, approximately 50% of Costanza et al.'s corresponding estimate.

Although Costanza et al.'s estimates of the value of ecosystem services are imperfect, and we lay no claim to their veracity, they do represent the most comprehensive set of first-approximations available for quantifying the change in the value of services provided by a wide array of ecosystems. Since one objective of our study was to determine the effectiveness of using generalized value coefficients to estimate watershed-level changes in ecosystem services, and because the scope of our project did not allow us to obtain area specific value coefficients, we used Costanza et al's 1997 estimates in our study despite their limitations.

Study Area and Estimation Approach

San Antonio is centered in Bexar County (29°27´ N, 98°31´ W) near the head of the San Antonio River Basin, which traverses the Edwards Plateau, the Texas Blackland Prairies, and the Western Gulf Coastal Plain eco-regions (Fig 1). Bexar County was chosen for the study in order to maximize the probability of detecting changes in ecosystem services due to urban sprawl. The three major streams running through the county are Salado Creek, the Upper San Antonio River, and Leon Creek. Because watersheds often represent the minimum ecological management unit and the purpose of our study was to estimate changes in the value of ecosystem services in the wake of land use conversion, the watersheds associated with the three streams were used as the analytical units in Bexar County. The size of the three watersheds comprising the study area was 141,671 ha.

Satellite imagery and remote sensing analytical software were used to determine the area of six land-use classes described below in each of the three watersheds in Bexar County. These estimates were then incorporated into an economic valuation model to quantify changes in the value of ecosystem services in each watershed over time. In addition, a sensitivity analysis was conducted to determine how sensitive the estimates of the value of ecosystem services were to the applied valuation coefficients.

Land-Use classification

Data sets incorporating Bexar County during relatively cloud-free days in August 1976 (the earliest available MSS data set of the study area), August 1985 (the next available cloud free MSS data set in a year with rainfall similar to 1976), and June 1991 (the most recent available cloud free MSS data set in a similar rainfall year) were obtained from the US Geological Survey EROS Data Center. The data sets were geo-referenced and Bexar County was extracted as a separate data set using the ERDAS Imagine® software (ERDAS, 1997), which incorporates functions for both image processing and the use of geographic information systems (GIS). Groupings of the spectral properties of the pixels in each of the three data sets of Bexar County were obtained through unsupervised classification using the Iterative Self-Organizing Data Analysis Technique (ISODATA) (Jensen, 1996) that is available in ERDAS Imagine®.

After the unsupervised classifications of the 1976, 1985, and 1991 data sets were completed, the resulting 100 spectral signature classes in each image were grouped into six land cover categories. This was accomplished using the US Geological Survey Land-use/Land Cover Classification System for Use with Remote Sensor Data, Level I (Jensen, 1996) in combination with a 1991 ground-truthed land-use map of Bexar County, developed for zoning purposes by the City of San Antonio Planning Department. The six land cover categories were: rangeland, woodland, bare soil, residential, commercial, and transportation. In Bexar County, the rangeland category corresponds to relatively open grasslands, while *Juniper*, *Quercus* (oak) and *Ulmus* (elm) species dominate the woodlands. During the time period covered by the LANDSAT MSS data used in this study, cultivated areas would have been largely bare and thus bare soils were assumed to reflect mainly post-harvest croplands. Residential (mainly suburban) and commercial (mainly urban) areas were differentiated by assuming that commercial areas have a reflective value more closely resembling paving materials, which occurs extensively in urban areas, while residential areas were assumed to be intermediate to the commercial and bare soil spectral signatures.

Once images of the six land categories were derived for each of the three years, vector files delineating the Leon Creek, Salado Creek, and the Upper San Antonio River Watersheds were projected onto these images, in order to estimate the area of each of the six land cover type within each watershed. Because no true reference data were available for the 1976-1991 time periods of investigation, uncertainty associated with land-use classification change was not measured (see Congalton & Green, 1999).

Assignment of Ecosystem Service Values

In order to obtain ecosystem services values for various ground cover types, the six land cover categories used to classify the LANDSAT MSS data sets were compared with the 16 biomes identified in Costanza et al.'s (1997) ecosystem services valuation model. The most representative biome was used as a proxy for each land cover category, including grass/rangelands for rangelands, temperate/boreal forest for woodland, cropland for bare soil, and urban for the commercial, residential, and transportation categories (which were not differentiated by Costanza et al.). The total value of terrestrial ecosystem services in San Antonio in 1976, 1985, and 1991 were obtained as follows:

$$ESV = ?\ (A_k \times VC_k) \quad \text{(Equation 1)}$$

where ESV is the estimated ecosystem service value, A_k is the area (ha) and VC_k the value coefficient ($\$\ ha^{-1}\ yr^{-1}$) for land use category "k". The change in ecosystem service values was estimated by calculating the difference between the estimated values for each land cover category in 1976, 1985 and 1991.

The biomes used as proxies for the land cover categories are clearly not perfect matches in every case. Specifically, the juniper/mesquite/elm dominated woodlands in the study area may not be well represented by Costanza et al.'s temperate/boreal forest biome because of the different climatic conditions under which they occur, but in terms of the ecosystem services that they provide the woodlands of Texas and temperate/boreal forests do have some similarities. For example, Texas woodlands can increase soil nutrient concentrations and contribute to gas regulation through their roles as carbon sinks (e.g., Boutton et al., 1999), and they can provide recreation opportunities. Because of the uncertainties about the representativeness of the proxies used for each land cover category as well as the veracity of Costanza et al's value coefficients, sensitivity analyses were conducted to determine the dependence of temporal changes in ecosystem service values on the applied valuation coefficients. The ecosystem value coefficients for rangeland, woodlands, and bare soil categories were each adjusted by 50%. Pervasive woody plant invasion throughout much of Texas including Bexar County resulted in a large portion of the area identified as rangeland in 1976 being classified as woodland in 1991. However, because of the uncertainty about the representatives of Costanza et al's temperate/boreal forest biome for woodlands, one additional sensitivity analysis was conducted by reducing the ecosystem value coefficient for woodlands ($\$302\ ha^{-1}\ yr^{-1}$) by 30% to that of rangelands ($\$232\ ha^{-1}\ yr^{-1}$), thereby assuming that woodlands are a variant of rangelands.

In each analysis, the coefficient of sensitivity (CS) was calculated using the standard economic concept of elasticity, i.e., the percentage change in the output for a given percentage change in an input (Mansfield, 1985):

$$CS = \frac{(ESV_j - ESV_i) / ESV_i}{(VC_{jk} - VC_{ik}) / VC_{ik}} \quad \text{(Equation 2)}$$

where ESV is the estimated ecosystem service value, VC is the value coefficient, "i" and "j" represent the initial and adjusted values, respectively, and "k" represents the land use category.

If the ratio of the percentage change in the estimated total ecosystem value (ESV) and the percentage change in the adjusted valuation coefficient (VC) is greater than unity, then the estimated ecosystem value is elastic with respect to that coefficient, but if the ratio is less than one, then the estimated ecosystem value is considered to be inelastic. The greater the proportional change in the ecosystem service value relative to the proportional change in the valuation coefficient, the more critical is the use of accurate ecosystem value coefficients.

Periphyton as Bio-Indicators

A Comparison of Periphyton Growth with Different Nutrient Delivery Mechanisms: Diffusing Substrates Versus Whole-Channel Enrichment

Artificial substrates are commonly used in studies of periphyton growth in streams. The practice potentially reduces the heterogeneity of naturally occurring substrate, allows for comparisons between sites with different natural substrate, reduces the cost of sampling, decrease habitat disruption, and speeds quantification of periphytic response when the surface area of the artificial substrate is known (Aloi, 1990). The most common artificial substrates are glass slides (APHA, 1989), clay tiles (Boston & Hill, 1991; DeNicola & Hoagland, 1996; Dodds et al., 1999; Hepinstall & Fuller, 1994; Hetrick et al., 1998), styrofoam (Bothwell, 1983; 1988), and glass fiber filters (Matlock et al., 1998; 1999).

There has been in recent years an increasing interest in studying the effect of nutrients on periphyton. Fairchild and Lowe (1984) described a method in which clay flower pots or saucers are filled with an agar and nutrient solution and inverted in a lake or stream. The nutrients diffuse through the agar and clay pot providing elevated nutrients at the exterior surface of the pot. Munn et al. (1989), Mosisch et al. (1999), and Pringle and Bowers (1984) have since used similar methods. Matlock (1998) developed a nutrient diffusion system, known as the Matlock Periphytometer, in which nutrients dissolved in water diffuse out of a submerged plastic bottle through a nylon membrane and glass fiber filter. Algae colonize the filter of each bottle. All these methods of diffusive nutrient loading provide attractive alternatives to whole-stream nutrient dosing since they allow multiple nutrient treatments to be evaluated simultaneously at a common location, and avoid possible downstream impacts from whole-stream dosing.

Pringle (1987) asserted that nutrient-diffusing substrates are functionally different than whole-channel enrichment assays, in part because the rate of nutrient input declines with time in the diffusion study while the rate is constant for the whole-channel dosing method. Pringle (1987) also reported differences in taxa grown on glass slides compared to a sand and nutrient-enriched agar solution, but the difference may have been due to the substrate and not the nutrient delivery method. Grimm and Fisher (1986) studied algal growth on clay pots in both a nutrient-enriched stream and on clay pots filled with a diffusing nutrient solution. Unfortunately, the studies were not simultaneous so no comparison was possible. Evidence that the mechanism of nutrient delivery affects periphyton growth is lacking from the literature.

The usefulness of any study employing nutrient-diffusing artificial substrates would be enhanced if it could be shown that the diffusion method of nutrient delivery elicits the same response as whole stream nutrient enrichment. Such a demonstration would allow researchers to more reasonably extrapolate the results obtained with nutrient diffusing substrates to natural instream substrates.

The purpose of this study was to determine what role, if any, the method of nutrient delivery plays in the growth of periphyton. Specifically, the experiment was designed to test the following hypothesis: Nutrient enrichment via diffusion does not elicit the same algal growth response as whole-channel nutrient enrichment.

Methods Overview

Four artificial channels were constructed at two locations in San Antonio, Texas during July and August 2000. At each location, nutrients were delivered via diffusion to artificial growth surfaces in two of the channels using modified Matlock Periphytometers. In the remaining two channels at each site, nutrients were delivered to the growth surfaces by dosing water in the channel. The periphyton growth surfaces were collected after two weeks, and algal response under the two methods of nutrient delivery was compared.

Site Description

The study was conducted on Leon and Salado Creeks in San Antonio, Texas (Figure 1). Leon and Salado Creeks originate in from springs in the limestone formations north of San Antonio, and flow along the western and eastern sides, respectively, of the metropolitan area. At the study sites the streams were fourth-order, perennial waterways. The artificial channels were constructed on the banks of the creeks approximately four meters from the water's edge.

Artificial Channel Design

The artificial channels were constructed of 0.6 mm plexiglass measuring 0.61 m high, 0.61 m wide, and 2.44 m long (Figure 2). The sides and bottom were connected by lengths of aluminum angle secured to the plexiglass with machine screws, and the inside joints were caulked with aquarium-grade silicon. Each channel was filled to a depth of 5 cm with bed material taken from the adjacent stream. A drain hole was drilled just above the bed material on the downslope end of each channel, and a bulkhead fitting with a 90-degree PVC elbow was placed in the hole. A length of plastic hose was attached to the elbow and routed upward along the outside of the channel, ending 2.5 cm below the top of the artificial channel. Water exited the channel through this hose, and thus the top elevation of this hose dictated the depth of water in the channel.

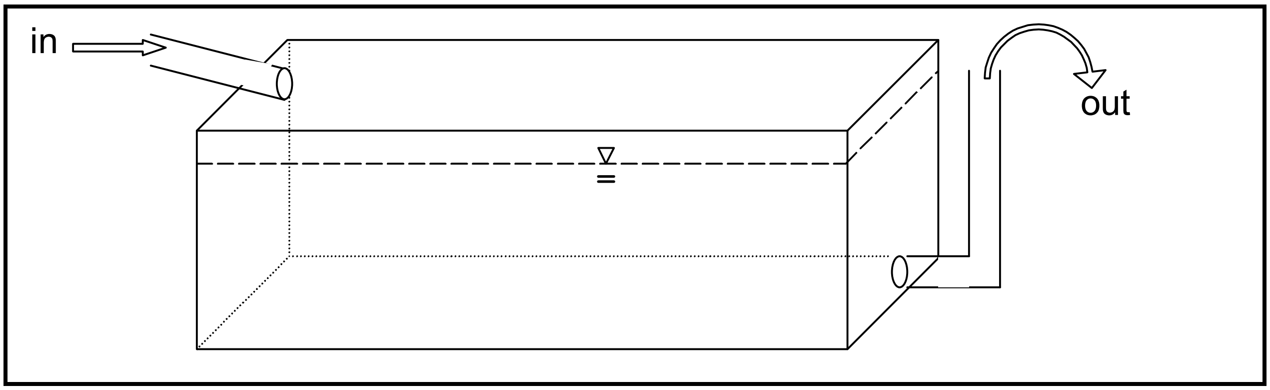


Figure 2. Diagram of an Artificial Channel

Flow Control

A 12V DC submersible pump delivered water from the adjacent stream. Three 12V marine batteries were connected in parallel to increase the time between charges, and the batteries were replaced each day with a fully recharged set. Stream water was pumped into a 190-liter barrel situated 0.5 m above the channels (Figure 3). The barrel had one outlet at the bottom to supply water to the four channels and a second outlet at the top for overflow. The rate of water flow to the reservoir averaged 11.4 liters per minute (L/min), and flow to each channel was adjusted (using four separate ball valves) to 1.9 L/min, or 7.6 L/min for the four channels combined. Thus, the rate of water into the barrel (11.4 L/min) exceeded output to the channels (7.6 L/min) ensuring that water consistently reached the level of the overflow outlet, and a constant elevation head was maintained above the lower outlet. The 1.9 L/min flow into each channel equated to an average retention time of approximately eight hours and an average water velocity of 0.3 m/hour.

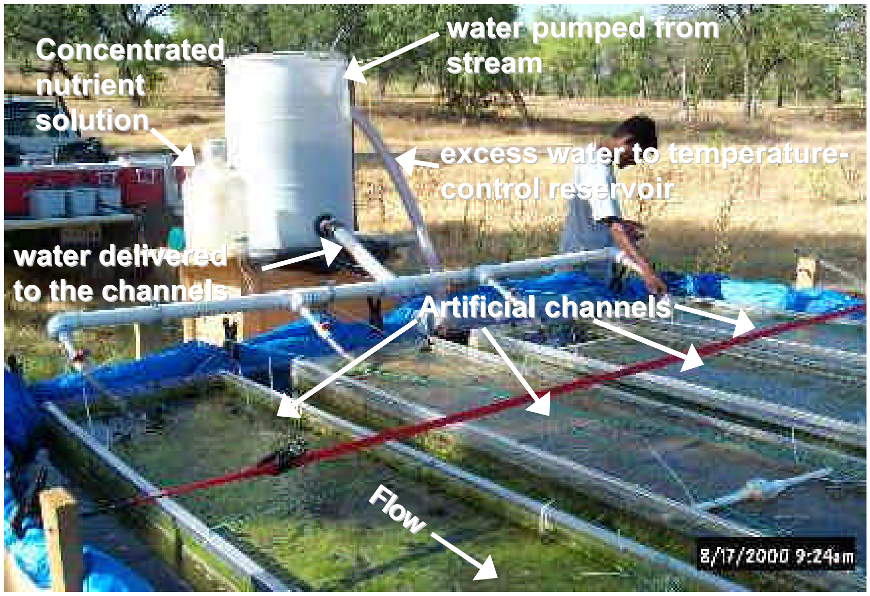


Figure 3. Artificial Channel Setup

Temperature Control

The four artificial channels were positioned side-by-side inside a large temperature control reservoir measuring 3 x 4 x 0.58 meters (Figure 3). The reservoir was made of wood and lined with a plastic pond liner. Water in excess of the 7.6 L/min needed for the four channels was routed through the barrel overflow and into this reservoir. Water exited the reservoir and returned to the stream through a drain on the down slope side of the structure. The top of the temperature control reservoir was approximately 2.5 cm below the top of the artificial channels so cooling water would spill over the outer walls of the reservoir before overtopping the channels. Temperature was monitored and recorded every fifteen minutes in the actual stream channel and in two of the four channels with YSI6500xlm Data Sondes.

Nutrient Delivery via Diffusing Matlock Periphytometers

Modified Matlock Periphytometers (Matlock, 1998) were used to deliver nutrients via diffusion to an artificial growth surface. In this method, nutrients dissolved in water diffuse out of a submerged plastic bottle through a nylon membrane and glass fiber filter (Figure 4). Algae colonize the filter of each bottle. The Matlock Periphytometer consists of forty bottles (10 control replicates and 10 replicates of three different nutrient treatments) are attached to a rigid galvanized steel frame. PVC pontoons on two ends allow the rack to float with the bottles submerged several inches below the water surface (Figure 5).

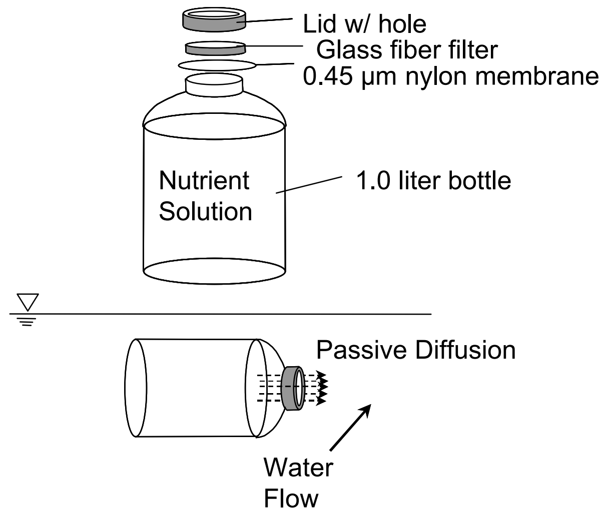


Figure 4. Matlock Periphytometer Passive Diffusion Bottle

Figure 5. Matlock Periphytometer Treatment Array

For this study, the 1.0 liter bottles of the original Matlock Periphytometer were replaced with 0.25 liter bottles, and the square metal rack was replaced with a 0.61 meter by 2.4 meter rack that fit into the artificial channel. Twenty bottles were attached to each side of the rack, and the rack was then secured 5 cm below the water surface in two of the four channels at each site (Figure 6). The forty bottles were divided equally as follows:

1. Control (C): water treated by reverse osmosis (RO).
2. Nitrate (N): 26.1 mg/L NO_3 -N in the form of $NaNO_3$ in RO water.
3. Phosphate (P): 86.7 mg/L PO_4-P in the form of $Na_2HPO_4*7H_2O$ in RO water.
4. Combined Nitrate and Phosphate (NP): 26.1 mg/L NO_3 -N and 86.7 mg/L PO_4-P in the forms listed above in RO water.

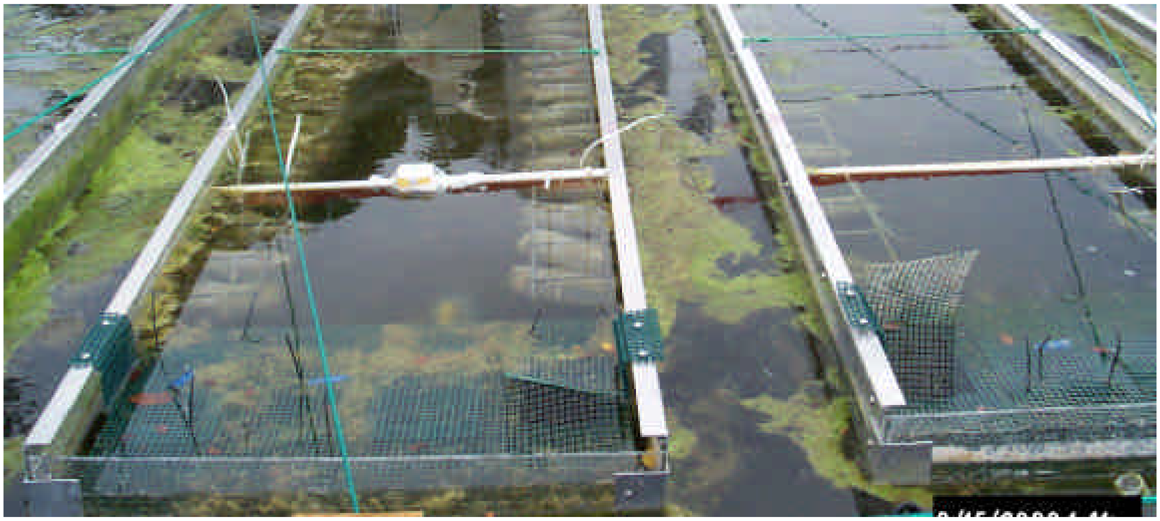


Figure 6. Modified Matlock Periphytometers for the Artificial Channels

The Matlock Periphytometers were retrieved after fourteen days. Each glass fiber filter growth surfaces was removed and placed in 7 ml of 90% acetone solution saturated with magnesium carbonate, wrapped in aluminum foil, and placed on ice for transport back to the laboratory. Chlorophyll-a analysis was done in the laboratory using the tri-chromatic method (APHA, 1989). The results were expressed as mass chlorophyll-a per unit surface area of the exposed filter. Mean chlorophyll-a concentrations for the different treatments were compared using the Student-Newman-Keul's (SNK) Test (a=0.05) in SAS/STAT© (SAS Institute Inc., 1990).

Nutrient Delivery via Whole-Channel Dosing

Two of the four channels were dosed continuously with nitrate and phosphate. The aim was to maintain sufficiently high concentrations of the nutrients that they would not limit algal growth. The delivery system consisted of a 20 L carboy containing a highly concentrated solution of nitrate (98.9 g/L NO_3-N) and phosphate (29.7 g/L PO_4-P) in R.O. water positioned 0.5 m above the channels. The nutrient solution was siphoned through two 0.4 mm inner diameter plastic tubes from the bottom of the carboy to each channel (Figure 3). The flow rate was a function of the length of the tubing and the height of the nutrient solution above the channels. The tubing length was adjusted to allow approximately 60 drips per minute into two of the artificial channels. With water delivered to each channel at 1.9 L/min, the resulting nutrient concentrations in the tank were designed to be 50.0 mg/L NO_3–N and 15.0 mg/L PO_4-P phosphate. Samples of the nutrient-enriched channel water were analyzed on three occasions during the study to determine the actual concentrations in the dosed channels.

Modified Matlock Periphytometers were placed in these dosed channels in the same manner as the diffusion channels. A hole was cut in the bottom of each bottle so that the channel's nutrient-dosed water was present on both sides of the periphyton growth surface. Unlike the bottles in the diffusion channels, there was no concentration gradient across the growth surface, and thus there was no diffusion. The growth surfaces from the dosed channels were collected and analyzed after fourteen days using the same procedures described for the diffusion channels.

Results and Discussion

Diffusion Rates of Matlock Periphytometers

The concentration of nitrate and phosphate in the MP bottles was measured before and after deployment for two of the ten replicates of each treatment. The ending concentrations reported in Table 6 are the average of two replicates taken from each site on day 14. The ambient concentrations reported in the table are the time-weighted averages of three samples taken during the study. The data demonstrate that the ending concentration of nutrients is still well above the ambient concentration, which implies that nutrients continued to diffuse through the membranes throughout the 14-day deployment.

Table 6. Nutrient Diffusion Rates for Matlock Periphytometers

Location	Nutrient	Mean Ambient Concentration (mg/L)	Mean Starting Concentration (mg/L)	Mean Ending Concentration (mg/L)	Mean Diffusion Rate (mg/day)
Leon Creek Site	Nitrate	0.18	84.7	27.6	1.02
	Phosphate	0.00	26.1	10.4	0.28
Salado Creek Site	Nitrate	0.13	84.7	8.5	1.36
	Phosphate	0.00	26.1	3.8	0.40

The mean diffusion rates at the Leon Creek site over the two-week deployment are approximately 30% lower than at the Salado site. There are several possible explanations. The most immediate is a difference in nutrient gradient at the two sites, but this is probably not important. Molecular diffusion across the semi-permeable membrane is linearly proportional to the concentration gradient across the membrane (Fick's Law), and while the ambient concentration of nitrate at the Salado site is lower than at the Leon site, the difference is far too small to explain the difference in diffusion rates. Further, there is no difference in the phosphate gradients at the two sites yet the phosphate diffusion rate differs for this nutrient as well. A more probable cause of the different diffusion rates is that the significantly higher algal growth at the Leon Creek site clogged the pores of the glass fiber filter through which the nutrients were diffusing, as described later in this section.

Nutrient Concentration in Dosed Channels

The concentration of nutrients in the stream and artificial channels was measured three times over fourteen days. The values reported in Table 7 are the time-weighted averages of the three samples. The "loaded" concentrations for the dosed tanks were calculated using measurements (taken six times over the two weeks) of water and concentrated nutrient solution flow rates into the dosed channels.

Table 7. Nutrient Concentration in Dosed Channels

Location	Nutrient	Loaded (mg/L)			Measured (mg/L)		
		Tank 1	Tank 3	Mean	Tank 1	Tank 3	Mean
Leon Creek Site	Nitrate	31.66	32.77	32.21	27.87	26.14	27.01
	Phosphate	9.50	9.83	9.66	8.25	7.67	7.96
Salado Creek Site	Nitrate	38.20	41.07	39.64	23.20	22.11	22.66
	Phosphate	11.46	12.32	11.89	6.83	6.32	6.58

The measured concentrations in the dosed channels at both sites are lower than the "loaded" concentrations, which suggests that the algae are removing nutrients from the water column. Further evidence of algal uptake of nutrients is provided by the nutrient concentration in Salado Creek and in the two channels that were not dosed. The phosphate concentration in the stream was 0.19 mg/L PO_4-P while in the artificial channels into which this same water was pumped, no phosphate was detected at all. The effect is present for nitrogen, though less dramatic; the instream concentration was measured as 0.16 mg/L NO_3 -N compared to 0.13 mg/L NO_3 -N in the artificial channels. At the Leon Creek site, where only nitrate was detected in the stream, algae in the artificial channel did not reduce the concentration. This may indicate that the periphyton community at the Salado site is phosphorous limited, and with the relatively long residence time in the artificial channels relative to the actual stream channel, the algae are able to scrub the available phosphorous from the water.

Algal Growth Response

The chlorophyll-a data were analyzed first with the results of each channel considered separately (Table 8) and then a second time with the two dosed channels pooled and the two diffusion channels pooled (Table 9). In Tables 8 and 9, treatments having the same letter in the SNK grouping column are not significantly different at the a = 0.05 level. In both analyses (separate and pooled), the Leon Creek and Salado Creek data were analyzed separately. Therefore, the Leon Creek treatments grouped as "A" are not the same as the "A" group from Salado Creek. Separating the two sites was necessary to prevent environmental differences between the sites from masking differences in periphyton growth due to the nutrient delivery mechanism.

With each channel treated separately (Table 8), significantly less periphyton grew in the dosed environment than in the diffusion environment. The one exception is Channel 3 at the Salado Creek site, which showed no significant difference from the two diffusion channels. All the treatments at this site showed relatively little periphyton growth, which was likely due to the riparian shading. In the case of Channel 3, the low periphytic responses may have allowed for analytic and natural variability to obscure differences between treatments.

Table 8. Comparison of Algal Growth: Diffusion vs. Channel Dosing

Location	Nutrient Delivery Method	SNK Grouping	Mean Chl-a (mg/m^2)	Stnd Dev (mg/m^2)	Sample Size
Leon Creek Site	Diffusion (Channel 2)	A	86.36	28.78	10
	Diffusion (Channel 4)	A	85.46	28.45	10
	Dosing (Channel 3)	B	56.64	17.33	18
	Dosing (Channel 1)	C	32.53	12.18	20
Salado Creek Site	Diffusion (Channel 2)	A	6.92	1.98	10
	Diffusion (Channel 4)	A	6.87	1.45	10
	Dosing (Channel 3)	A	6.74	1.27	20
	Dosing (Channel 1)	B	5.54	0.80	19

The second statistical approach (Table 9) pooled the two dosed channels (1 and 3) together and the two diffusion channels (2 and 4) together. This pooling of data gives us a better estimate of the intra-treatment variance by virtue of a larger sample size. The pooled approach changes only the results for dosed Channel 3 at the Salado Creek site. Before pooling the data, the results indicated that the periphyton growth in dosed Channel 3 was no different than the growth due to diffusion in Channels 2 and 4, suggesting that the method of nutrient delivery does not matter. When the data were pooled and treatment effects became more apparent, we found that dosing consistently elicited significantly lower periphyton growth compared to diffusion at both locations.

Table 9. Comparison of Algal Growth: Diffusion vs. Channel Dosing with Pooled Data

Location	Nutrient Delivery Method	SNK Grouping	Mean Chl-a (mg/m^2)	Stnd Dev (mg/m^2)	Sample Size
Leon Creek Site	Diffusion (2 and 4)	A	85.91	27.85	20
	Dosing (1 and 3)	B	43.96	19.05	38
Salado Creek Site	Diffusion (2 and 4)	A	6.90	1.62	20
	Dosing (1 and 3)	B	6.15	1.31	39

Conclusion

Four artificial channels were constructed at two locations in San Antonio, Texas. At each location, nutrients were delivered via diffusion to artificial periphyton growth surfaces in two of the channels using modified Matlock Periphytometers. In the remaining two channels at each site, nutrients were delivered to the growth surfaces by dosing the water in the channel. After two weeks, the growth surfaces were collected and the algal response under the two methods of nutrient delivery was compared to test the null hypothesis that the two methods of nutrient delivery—diffusion and whole-channel dosing—elicit different algal responses.

Based on the results observed, we failed to reject our null hypothesis. There was evidence that delivery of nutrients to a substrate by diffusion and by whole-channel dosing does elicit significantly different periphytic responses. Use of diffusing substrates remains a legitimate technique to compare the relative influence of different nutrient treatments and to determine limiting nutrients, but the results of periphyton studies employing diffusing substrates should be extrapolated to natural conditions with caution.

Further work is needed to determine the taxonomic composition of the periphyton grown in the different treatments to see if observed differences in chlorophyll-a are due to differences in biomass production, algal composition, or a combination of both.

Fish (Bluegill) as Eco-Integrators

Hypothesis: "MMS measurements in acclimated versus unacclimated fish provide a measure of the relative integrity of the aquatic ecosystem (Neill)." This, the original hypothesis, was restated to make it more tractable, and, at the same time, more meaningful: Ecoassay of fish performance, using an ecophysiological model to integrate results of in-stream cage trials and subsequent respirometry in the laboratory, permits a highly informative assessment of variation in aquatic ecosystem integrity, from the perspective of representative fish. This assessment approach not only allows a partition of fish-performance variation attributable to the various components of environment, but also provides a convenient and effective basis for substantive communication with stakeholders.

Beginning in late summer 1999, a project subteam led by William Neill performed a series of ecoassays using caged bluegill (the sunfish *Lepomis macrochirus*) to assess impacts of urbanization on streams of Texas' upper San Antonio River basin. The resulting data on stream environment, bluegill growth and mortality, and bluegill metabolic performance (Fontaine, 2002) provided opportunity–and, owing to some uncertainty about the meaning of the data, the motivation–for application of Ecophys.Fish, a STELLA model developed by Neill et al. (2003 manuscript) for simulation of fish growth and mortality in response to environmental variation.

Ecophys.Fish (Figure 7) has two functional sectors, metabolism and bioenergetics. The metabolism sector has a STELLA space compression object (SCO) for each of Fry's (1947, 1971) five classes of environmental factors. At present, the directive-factor SCO is empty, and the lethal-factor SCO contains a computation of mortality rate associated only with exposure to low concentrations of dissolved oxygen (DO). In the loading-factor SCO, the salinity subroutine computes and returns a standard-metabolism intercept, which is multiplied by a function returned by the temperature subroutine in the controlling-factor SCO, and by a power function of fish weight, to give standard metabolic rate. Temperature's controlling effect on standard metabolism is modeled as the product of steady-state and transient-state components, reflecting both the Ahrennius effect and thermal acclimation. The temperature and pH subroutines in the controlling-factor SCO produce outputs that control weight-dependent active metabolism. These controlling effects are modeled as interactions with the limiting effect of DO, in the limiting-factor SCO. Responses to DO, like those to temperature, are subject to metabolic acclimation. The residual intercept of active metabolism is MMSO, which we interpret to represent inherent metabolic efficiency of the fish-environment system, after the cumulative and integrated effects of temperature, pH, DO, salinity, and fish size have been taken into account. MMSO can be estimated (Figure 8) by coupling the simulation model with routine respirometry trials that have as their end-point the determination of marginal metabolic scope, MMS (= routine metabolic rate, divided by DO limiting for that rate). The model's bioenergetics module reflects conventional generalizations, with the addition of a novel component for metabolic limitation of food intake. Consumed energy is converted to new fish biomass as a residual, after deduction of expenditures for standard and routine-activity components of metabolism, food-processing costs, and wastes. Our modeled fish conserves body form, by reducing and increasing caloric density of its tissues (within species-specific ranges) during times of food-energy limitation and surplus, respectively. The model's inputs are MMSO, initial size of fish, and time series (hourly basis) of temperature, pH, DO, salinity, food availability and energy content. Outputs are food consumption, oxygen consumption, waste production, energy content of fish biomass, and growth. Indirectly, the output is a measure of relative fitness of the fish-environment system to support fish growth. In field tests of Ecophys.Fish with the euryhaline red drum (*Sciaenops ocellatus*, the fish species for which the model was developed), it was able to account for about 80% of growth variation.

We exposed bluegill (initial weight, 7 - 20 g), each confined to a 30x30x30 cm^3 cell within a floating cage of 0.8-cm plastic-coated-wire mesh, to in-stream conditions at two sites in each of the two urbanizing streams–Leon and Salado Creeks–during summer 1999, summer 2000, and winter 2000 (the "winter 2000" ecoassay extended from December 2000 into January 2001). Water temperature, dissolved-oxygen concentration, pH, and conductivity were recorded at 15-min intervals via one or more environmental dataloggers at each cage site. For each of the 12 stream-site-season/year combinations, we successfully measured MMS (marginal metabolic scope) via respirometry for 4 to 9 of the 12 individual fish, at termination of their 10-23 day exposure periods; we then compared the 12 median values of MMS and estimated MS (metabolic scope), with the corresponding medians of Wchg (weight-change rate, %/day). Subsequently, we adapted Ecophys.Fish for application to bluegill and used the modified model to perform additional analysis of observed relationships.

Salado Creek Fecal Coliform Modeling

The transmission of human pathogenic agents via source water and treated water has been reported extensively in the literature (Barwick et al, 2000; Cruz et al., 1990; Carter et al., 1987). It is reasonable to believe that human population growth and anthropogenic activities such as intensive animal rearing and feeding operations are partially responsible for the contamination of water bodies. Given that a variety of microbial pathogen contamination sources do occur and will probably continue to occur in the future, modern society has to identify those water bodies that are contaminated or potentially vulnerable to contamination, and develop management and remediation strategies for those systems. However, evidence is mounting that these microbial organisms survive and proliferate in some ecosystems under specific climatic conditions. A geographic information system (GIS)-based geophysical model of the range of conditions and rates of proliferation of *E. coli* and coliphages would be a valuable tool in water quality management and human health protection. The objective of this project was to evaluate the effectiveness of the Hydrological Simulation Program-FORTRAN (HSPF), a GIS-based model, as a prediction tool for fecal coliform bacterial concentration, and to determine the sensitivity of the model to parameters associated with bacterial persistence and growth simulation.

The U.S. Environmental Protection Agency (USEPA), under its Clean Water Action Plan of 1996, emphasizes the need for State, local and tribal authorities to carry out a watershed level study and management approach in order to address the issues of nonpoint source runoff and pollution and restore the health of impaired waters. The restoration of water quality of the impaired streams starts with acquiring knowledge about the system, such as the amount and sources of pollutant loading and the sources that need to be controlled.

Bacteria and Water Quality

The primary sources of pollution to the waters of the US are urban and agricultural runoff (USEPA, 1998). The most common pollutants from these non-point sources are nutrients, bacteria, and silt (USEPA, 1998). Thus, the persistence of potential microbial pathogens from wastes in soil and water is a constant concern. The Texas Natural Resource Conservation Commission (TNRCC) listed 148 stream segments in Texas as not meeting their designated use under the Clean Water Act, subsection 303(d) (TNRCC, 1998).

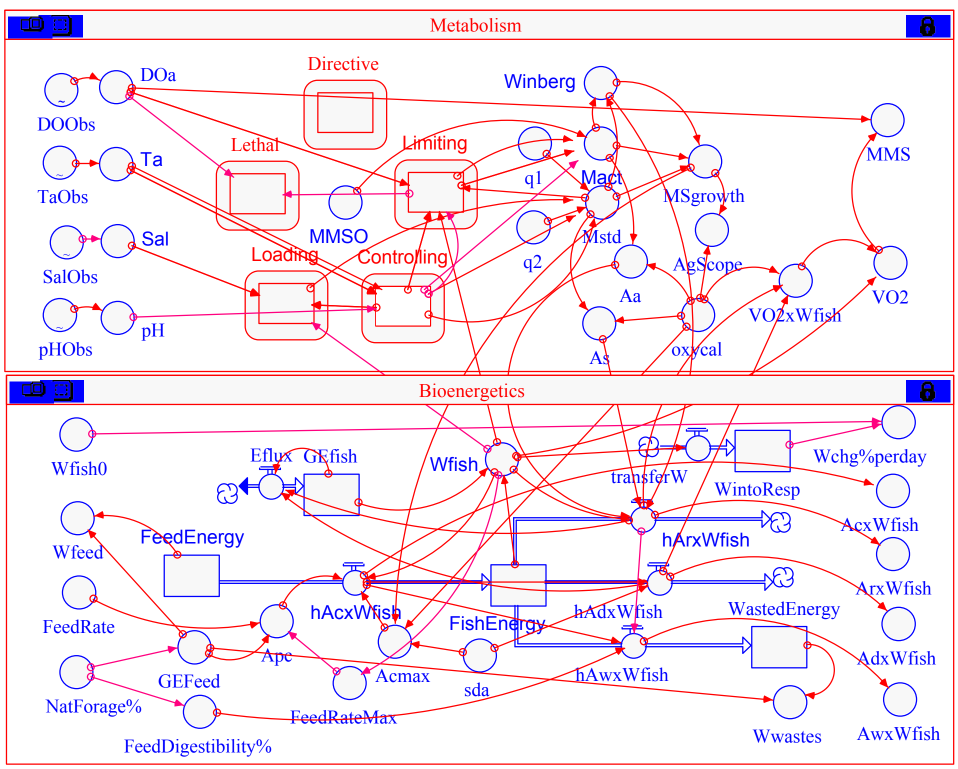
Designated uses of streams include fishing, swimming, boating, wildlife habitat, agriculture, and industry. Bacterial contamination was listed as a pollutant of concern in 65 of these segments. Salado Creek in Bexar County, Texas was one of the streams found out to be unfit for human and wildlife consumption due to elevated concentrations of nutrients, fecal coliform bacteria and violation of dissolved oxygen standards (TNRCC, 1998). Subsequently, these water bodies have been included in the Federal Clean Water Act Section 303 (d) listing of impaired water bodies for Texas.

Fecal coliforms are a group of bacteria that primarily live in the lower intestines of warm-blooded animals, including humans. Many water-borne diseases, like dysentery and cholera, are associated with certain strains of E. Coli, which is but one category of fecal coliforms. Because of the serious potential health threat associated with certain strains of this general type of bacteria, the fecal coliform test is very important as an indicator of the health risk associated with human contact with a body of water. Fecal coliforms are rare or absent in unpolluted waters because they are associated with warm-blooded mammals.

Coliform Terminology and Measurement

Coliform bacteria are a collection of relatively harmless microorganisms that live in large numbers in the intestines of animals. Fecal coliforms, a subgroup of these bacteria, can grow at elevated temperature, and include bacteria such as *E. coli*. US EPA and TNRCC use the presence of fecal coliform bacteria in aquatic environments as an indicator that water bodies have been contaminated with fecal material and other pathogens. This contamination may result in exposure of humans and wildlife to harmful pathogens causing

Figure 7. Ecophys.Fish: STELLA Model of Fish Growth in Response to Ecophysiological Factors

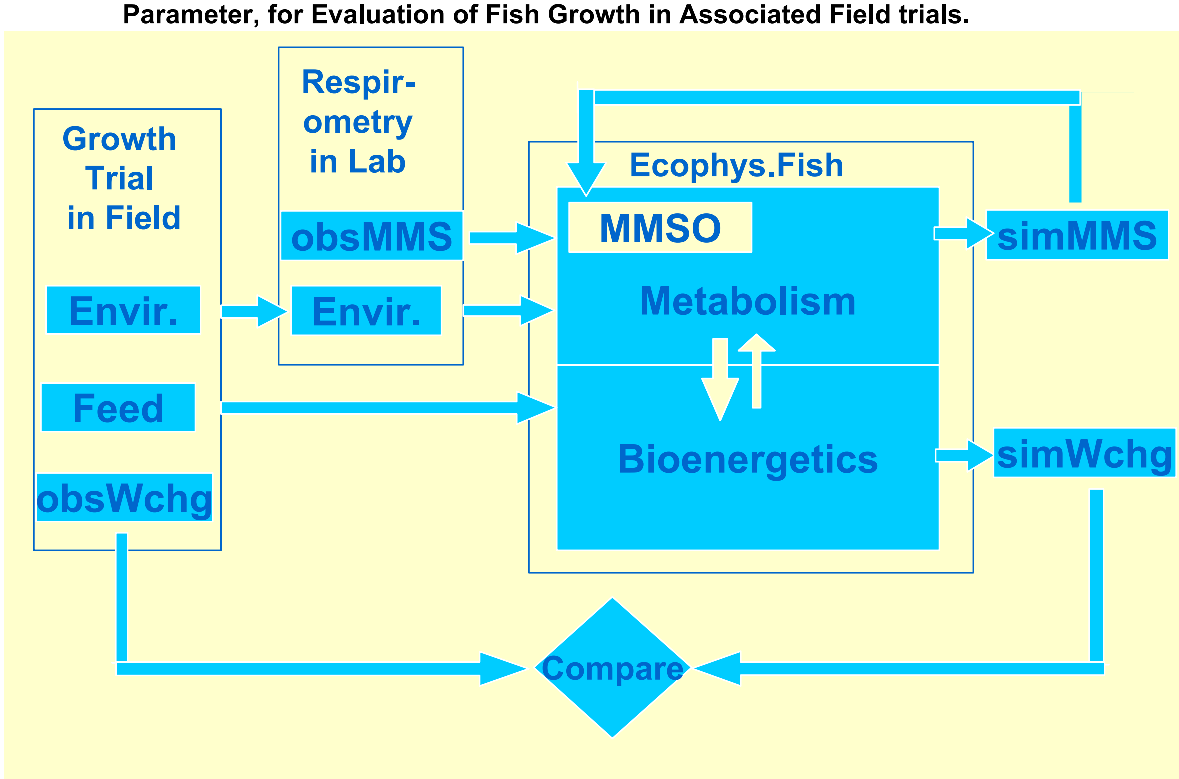


typhoid fever, hepatitis A, and other adverse health effects. The methods used to measure fecal coliform bacteria are prone to interferences, but continue to be used because they are the most cost-effective and practical methods currently available (APHA, 1997).

The detection of specific pathogenic microorganisms (bacteria, viruses, protozoa) is an extremely time consuming and expensive approach. For certain pathogens such as the caliciviruses (eg Norwalk virus) tissue culture systems are yet unavailable, which precludes an accurate estimation of pathogen loads in environmental samples. To overcome some of the limitations associated with the detection of specific microbial pathogens, the use of indicators to detect fecal contamination have been proposed. The rationale behind this approach is that if a particular water sample is fecally contaminated, then the probability that the sample contains pathogens is greater. A number of different microbial and chemical indicators of fecal contamination have been proposed. These include total coliforms, *E. coli*, male specific coliphages, *Bacteroides fragilis* phages, fecal streptococci, sulfate-reducing clostridia, H_2S producers, and fecal sterols.

Total Coliforms. Total coliforms are defined as a heterogeneous group of Gram negative, non-spore-forming bacteria that are rod shaped, and ferment lactose with the production of acid and gas within 24 to 48 hours at 35°C. The coliform group has been used as the standard for assessing fecal contamination of recreational and drinking waters for most of the past century (Gerba, 2000). There are a number of

Figure 8. Using Respirometry Data to Estimate Ecophys.Fish's MMSO Parameter, for Evaluation of Fish Growth in Associated Field trials.



limitations associated with the use of total coliforms as indicators. Studies have shown that there is significant regrowth in aquatic environments in both source and treated water (Gleeson & Cray, 1997). Additionally, the die-off rate of this organism is strongly influenced by the presence of organic matter and temperature. Other studies have shown that these organisms can originate from non-enteric sources such as wastes from the wood industry (Toranzos & McFeters, 1997) and epilithic algal mat communities in pristine streams. A primary limitation of total coliforms as an index of fecal contamination is that they are capable of regrowth even within distribution systems in the presence of free chlorine (LeChevallier et al., 1988). In water bodies rainfall events would cause a proliferation of these cells.

Fecal Coliforms. The fecal coliform group of organisms can be considered to be a subset of the larger "total coliform" group. Fecal coliforms are defined as Gram-negative, non-spore-forming, rod shaped bacteria that are able to ferment lactose with the production of acid and gas within 24 hours at an elevated temperature (44.5± 0.2°C). Studies have shown that for the most part, fecal coliforms show strong correlation with fecal contamination from warm-blooded animals (Pourcher et al., 1991). The rationale for the elevated temperature criterion is the elevated temperature found in mammalian enteric tracts. Though the incubation conditions are meant to select for those organisms that are indicative of fecal contamination, it does not distinguish human and animal contamination. There have been reports that fecal coliforms including *E. coli* have been detected in pristine tropical waters (Hazen & Toranzos, 1990; Toranzos, 1991; Solo-Gabriele et al., 2000) and that they can possibly proliferate outside the intestines of warm-blooded animals. Recent studies have also shown that fecal coliforms do not correlate positively with the presence of infectious enteric viruses. There have been studies that have documented viral infections from water samples that have been negative for fecal coliforms and *E. coli*.

E. coli. *Escherichia coli*, a member of the fecal coliform group of organisms, has been shown to be a useful microbiological indicator of water quality. A major drawback for their use, however, is the isolation of *E. coli* from tropical regions and the recent findings suggesting natural proliferation of the cells in subtropical coastal waterways (Hazen & Toranzos, 1990; Solo-Gabriele et al., 2000).

The Total Maximum Daily Load (TMDL) Process

A TMDL determination is an analysis used to calculate the maximum pollutant load a water body can receive (loading capacity) without violating water quality standards (Hession et al., 1995; Hession et al., 1996b). TMDLs establish waste load allocations (WLAs) for point sources, load allocations (LAs) for non-point sources, background loadings from natural sources, and a margin of safety to ensure achievement of the water quality goal (USEPA, 1991). The TMDL process has five distinct steps (USEPA, 1991):

1. Identify pollutants of concern
2. Estimate the waterbody's assimilative capacity for those pollutants
3. Estimate the pollution loading from all sources to the waterbody
4. Determine the total allowable pollutant load to the waterbody
5. Allocate pollutant loading limits to each source, including a margin of safety.

The difficulty with applying this approach to fecal coliform bacteria is that the units of measure, colony forming units (CFU), per 100 ml do not consider time or growth and reproduction based on source. The assumption is that fecal coliform bacteria, or those indicated on the American Society for Testing and Materials (ASTM) test method as fecal coliform bacteria do not grow *ex situ*. However, bacteria pathogenic to humans are known to survive in soil for defined periods of time. These bacteria may in fact persist and reproduce in warm, southern streams with high sediment and organic carbon loading (Ginnivan et al., 1980; Kudva et al., 1998; Kunte et al., 1998). Recently Solo-Gabriele et al. (2000) have shown that *E. coli* could proliferate in subtropical waterways in Florida.

Factors Affecting Fecal Coliform Kinetics

There are a number of factors, which affect the fate of bacteria in general and fecal coliform in particular. These factors can be divided into physical, physicochemical and biochemical and biological factors. Some of the important factors and the way they affect the fate of fecal coliform are given in Table 10. The current study was aimed at analyzing the HSPF model to find out its applicability as a prediction tool for fecal coliform bacterial concentration, to evaluate the sensitivity of the parameters associated with the bacterial simulation and to determine the fraction of the variance in peak in-stream fecal coliform concentrations due to the uncertainty of these most sensitive parameters.

Table 10. Factors Affecting the Fecal Coliform Die-off Rate

Category	Factors	Effect
Physical	Photo-oxidation	Light increases the mortality of fecal coliform
	Adsorption	Affects fecal coliform mortality, but inconclusive data available
	Flocculation	Affects fecal coliform mortality, but inconclusive data available
	Coagulation	Affects fecal coliform mortality, but inconclusive data available
	Sedimentation	May decrease the mortality rate by depositing the fecal coliforms to the bottom of the stream bed
	Temperature	This is the most important factor affecting the fate of bacteria. Other than directly affecting the mortality rate, temperature affects other factors which affect the mortality rate of bacteria
	Osmotic effects	Salinity is an important factor in the case of E. Coli.
Physicochemical	pH	Generally E. Coli survives longer in lower pH
	Chemical toxicity	In general the presence of heavy metals reduces the bacterial concentration
	Redox potential	The higher the redox potential the higher is the mortality rate of bacteria
	Nutrient level	Increase in nutrient level may increase amount of in-stream fecal coliform
Biochemical-biological	Presence of organic substance	May decrease mortality rate
	Predators	May increase the mortality rate
	Algae	In general, detrimental to bacteria because of production of toxic substance along with algal boom.
	Presence of fecal matter	Increase the concentration of fecal coliform.

Methodology

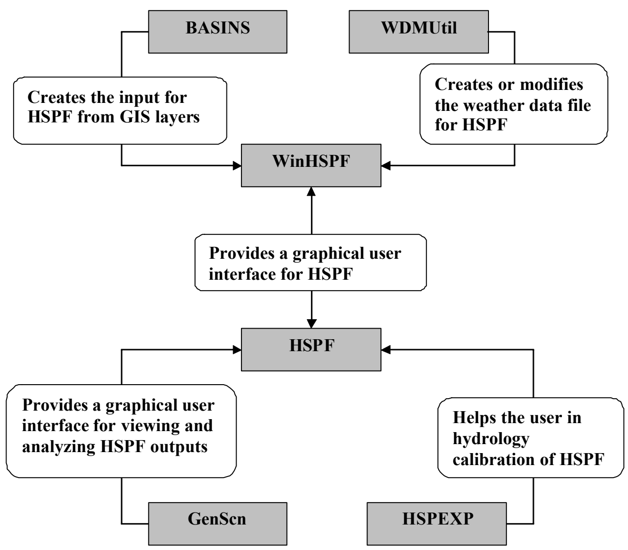
The study was conducted using the tools available in BASINS system framework. The main steps involved are:

1. Calibration of the watershed hydrology component of HSPF.
2. Validation of the fecal coliform component of HSPF.
3. Sensitivity analysis of HSPF to identify parameters that may have the most effect on variability in peak in-stream fecal coliform concentration predictions.
4. Uncertainty analysis using the First Order Analysis technique to identify which parameters contribute the most to output variability.

Model Description

BASINS was developed by the US EPA's Office of Water to support environmental and ecological studies in a watershed context (USEPA, 2001). BASINS works within a GIS framework and comprises various components including (1) national databases (2) assessment tools (3) a watershed delineation tool (4) classification utilities (5) characterization reports (6) an in-stream water model, *QUAL2E* (7) watershed loading and transport models, *HSPF* and Soil and Water Assessment Tool, (*SWAT*); and (9) a simplified GIS based model, *PLOAD* that estimates annual average nonpoint source pollutant loads. Different programs employed in the current study and their interactions are shown in Figure 9.

Figure 9. Computer Programs Employed in the Project



HSPF is a continuous hydrological modeling software that can be used to simulate a comprehensive range of hydrologic and water quality processes. HSPF has a modular structure and the watershed is divided into three groups: pervious land, impervious land and channels or reaches. The modules PERLND, IMPLND and RCHRES in HSPF simulate the processes that occur in a pervious land segment, an impervious land segment and a single reach respectively. The spatial distribution of different land segments within a particular watershed is not considered in HSPF.

Though HSPF is a versatile model, it is WinHSPF (Windows interface to HSPF), which provides the user an interactive graphical user interface (GUI). WinHSPF was created for the BASINS 3.0 system replacing the earlier program called Non Point Source Model (NPSM) used in combination with BASINS 1.0 and BASINS 2.0. WinHSPF helps the user to create a new HSPF input sequence or to modify an existing HSPF input sequence. WinHSPF also makes it easier to modify the input sequence for HSPEXP, an expert system developed by United States Geological Survey (USGS) for assisting users in calibrating HSPF. Figure 10 shows an HSPF project opened with the WinHSPF interface.

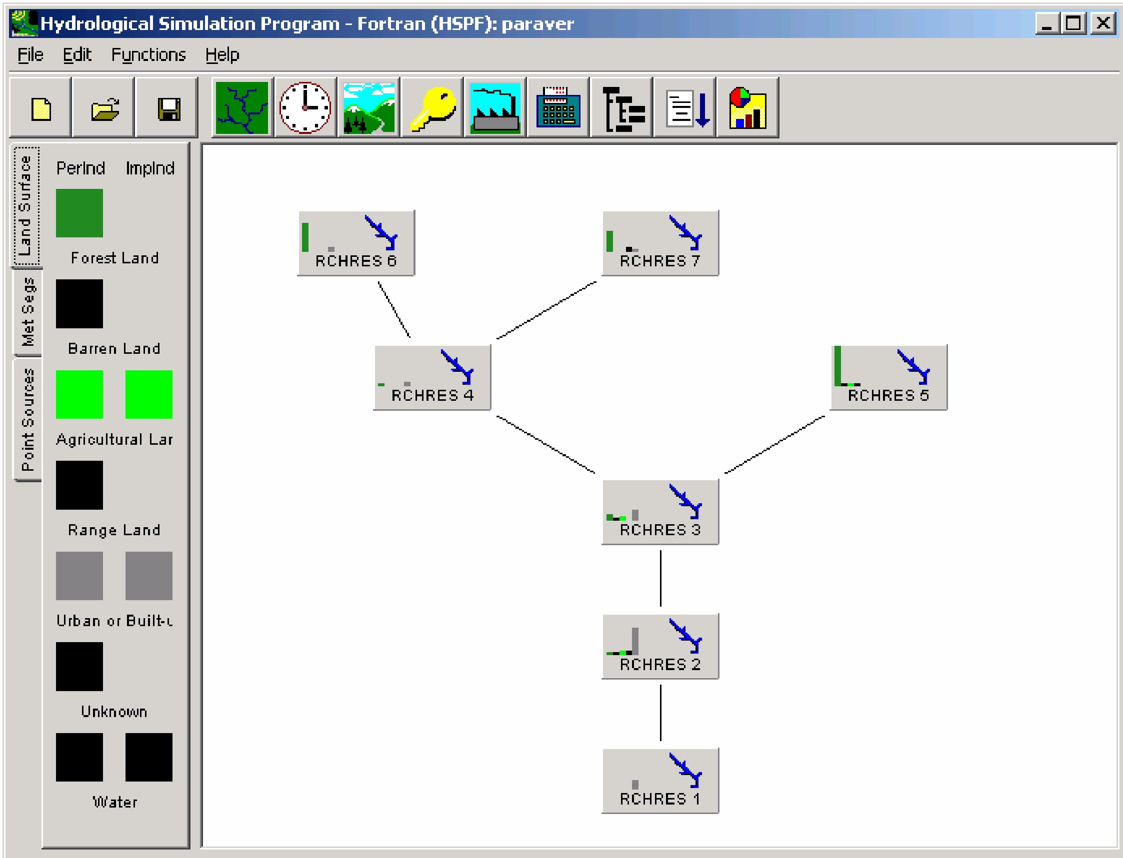


Figure 10. Graphical User Interface of WinHSPF opened with an HSPF project

The required meteorological data for the HSPF program is stored in a Watershed Data Management (WDM) file format. The WDMUtil program allows users to import available meteorological data into WDM files. The new WDMUtil 2.0 allows importing of data sets of various time steps and formats.

GENeration and analysis of model simulation SceNarios (GenScn) is a GUI based program for creating simulation scenarios, analyzing the results and comparing scenarios. The program can be used to run the HSPF model after changing the input sequence interactively and to view the results graphically for further analysis. Some of the plots that can be obtained from GenScn are a standard time-series plot, a residual time-series plot versus another time-series, a cumulative differences time-series plot and a scatter plot of two time-series including an optional 45-degree line and regression line. Three statistical analyses available in GenScn are comparing two time-series over a range of class intervals, constituent duration analysis, and frequency analysis.

HSPEXP, an Expert system for calibration of HSPF interactively allows the user to edit the input sequences of HSPF, simulates with HSPF, plots the output from the HSPF against different observed values and computes error statistics. Using a set of over 35 rules and 80 conditions, the system advises the users of necessary changes on different parameters to improve the calibration.

Study Area

The location of the Salado Creek study area is between latitudes 29.735 N and 29.147 N and longitudes 98.649W and 98.221 W. Salado Creek is one of the major tributaries of the San Antonio River with a total length of approximately 32 miles (Figure 11). The creek originates in the north central region of the San Antonio River basin and flows in the eastern region of the San Antonio metropolitan area and joins the San Antonio River at the south end. The total area of the delineated watershed is 123,155 acres.

Climatic Data Used

Climatic data stored in WDM file corresponding to Texas (tx.wdm) was used. The weather station used was located at the San Antonio International Airport.

GIS Data Layers for Salado Creek

The Salado Creek Watershed is located within the Hydrologic Unit Code (HUC) 12100301. BASINS core data corresponding to HUC 12100301 was obtained from EPA's BASINS data web site. This includes GIS data layers required to create the BASINS project. These data sets include different data layers such as Land Use/Land Cover, STATSGO Soils data, and Reach Network Version 1, all of which are at 1:250,000 scale.

The Digital Elevation Model (DEM) at 1:24,000 scale with a spatial resolution of 30m was obtained from Texas Natural Resources Information System (TNRIS). These are grids covering a full 7.5-minute USGS quadrangle. The DEMs for the entire Bexar County were obtained from TNRIS and mosaiced together for the current study.

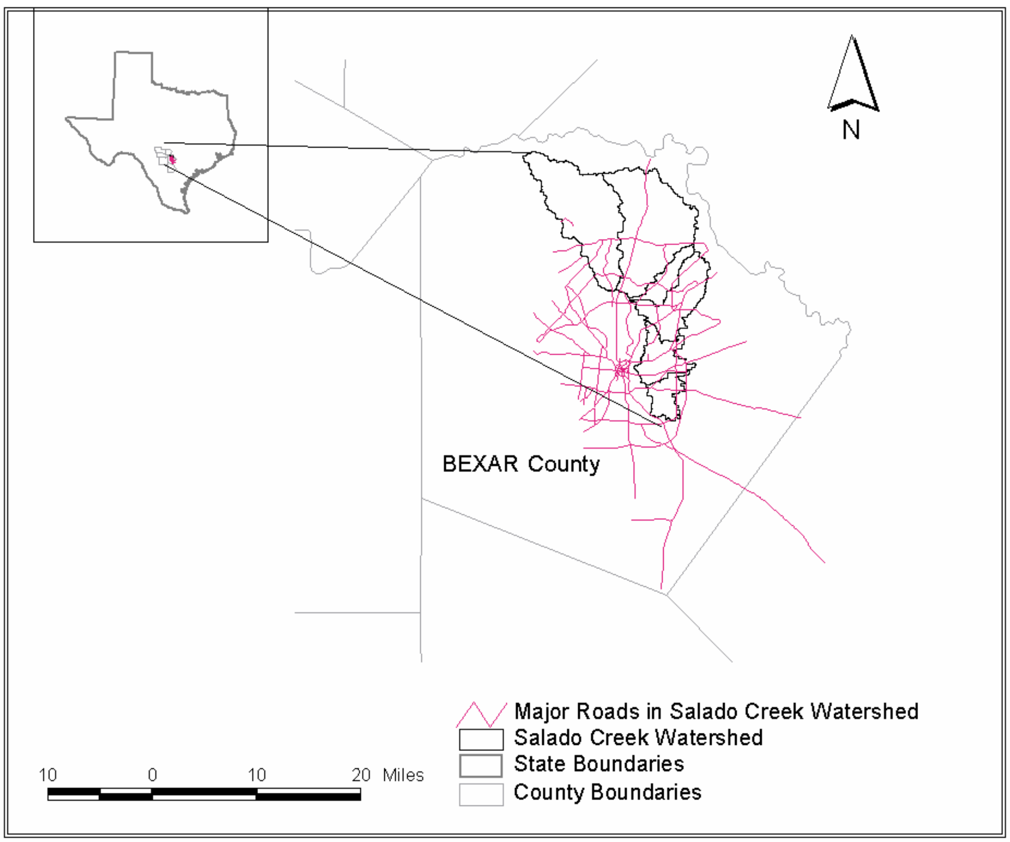


Figure 11. Salado Creek Watershed, Bexar County, Texas

The detailed stream network for the HUC was obtained from National Hydrography Dataset (NHD) using the NHD download tool available with BASINS 3.0. NHD is based upon the content of USGS Digital Line Graph (DLG) hydrography data integrated with reach-related information from the EPA Reach File Version 3 (RF3). It is a comprehensive set of digital spatial data that contains information about surface water features such as lakes, ponds, streams, rivers, springs and wells. NHD is an improvement of DLG and RF3 without replacing them. It is based on 1:100,000-scale data.

The Salado Creek main watershed was sub-divided into hydrologically connected sub-watersheds using the DEMs and the Automatic Watershed Delineation tool available with BASINS 3.0. The NHD stream network for HUC 12100301 was used to obtain the correct path of the streams (Figure 12).

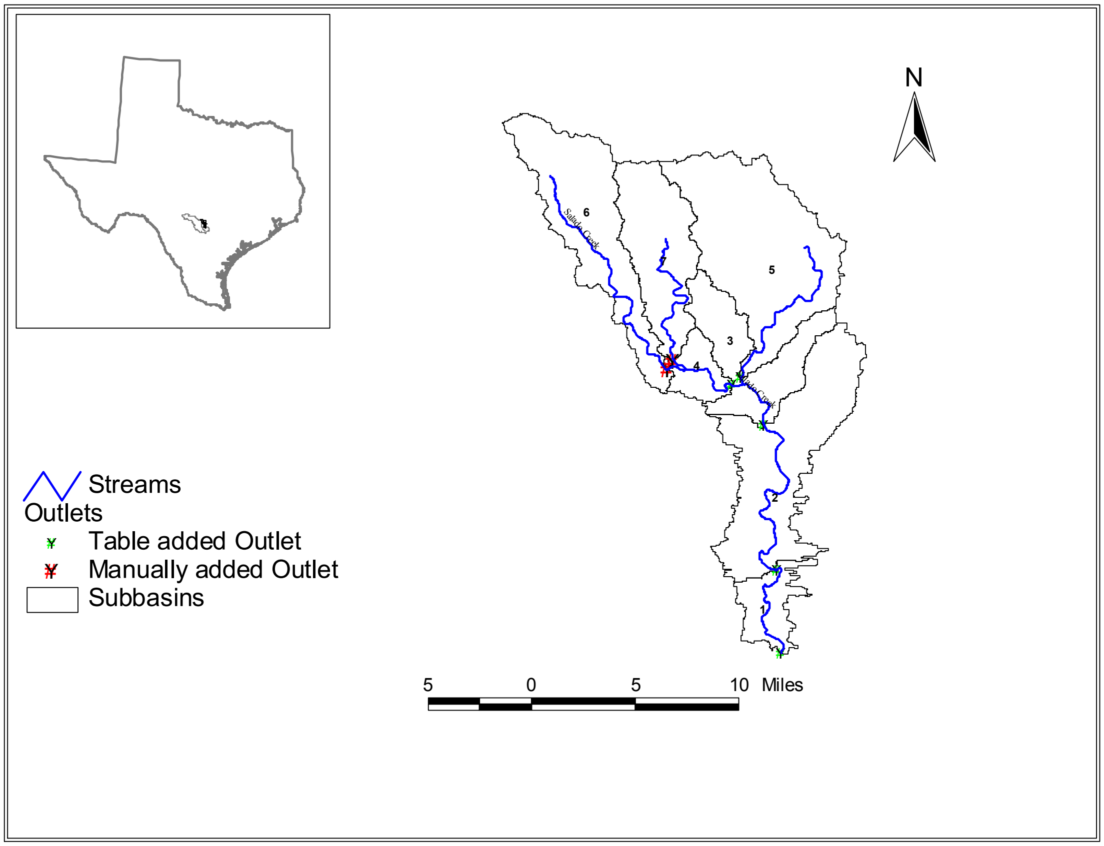


Figure 12. Sub-Watersheds of Salado Creek Watershed, Bexar County, Texas

The Geographic Information Retrieval and Analysis System (GIRAS) Land Use/Land Cover theme was obtained from EPA's web site. GIRAS data uses the Anderson Level II classification scheme. Some Anderson Level II classes were consolidated to obtain HSPF land use classes. Figure 13 shows the land use distribution for the study area with the reclassified groups.

Data for Calibration

Historical daily mean stream flow data for USGS gauging stations 08178800 and 08178700 were obtained from the USGS web site for the simulation period. Historical water quality data for fecal coliform at different water quality stations were obtained from both the San Antonio River Authority and also from EPA's STORET (short for STOrage and RETrieval) system. The location of the USGS gauging stations and the water quality stations are shown in Figure 14.

Sensitivity Analysis

Sensitivity analysis determines the sensitivity of model outputs to changes in model parameters, or, how much variation in the output is attributed to a particular input parameter. Two types of sensitivity coefficients, absolute sensitivity and relative sensitivity can be calculated (Haan, 1995a). The absolute sensitivity, S is calculated as:

$$S = \frac{\partial O}{\partial P} \qquad (2)$$

where, O is the output value corresponding to the parameter value P. The absolute sensitivity is the absolute change in the output for a unit change in the input value.

The relative sensitivity, S_r is calculated as:

$$S_r = \frac{\partial O}{\partial P} * \frac{P}{O} \quad (3)$$

The relative sensitivity is the percent change in the output for a unit percent change in the input. In the current project the relative sensitivity term is used to compare across parameters to select those parameters that when changed cause the greatest change in model outputs. The sensitivity of simulated peak in-stream fecal coliform concentrations at the main outlet of the Salado Creek to changes in model parameters was evaluated. Since the model was calibrated for the hydrology, only the parameters that affect the water quality modules in HSPF were selected for the sensitivity analysis.

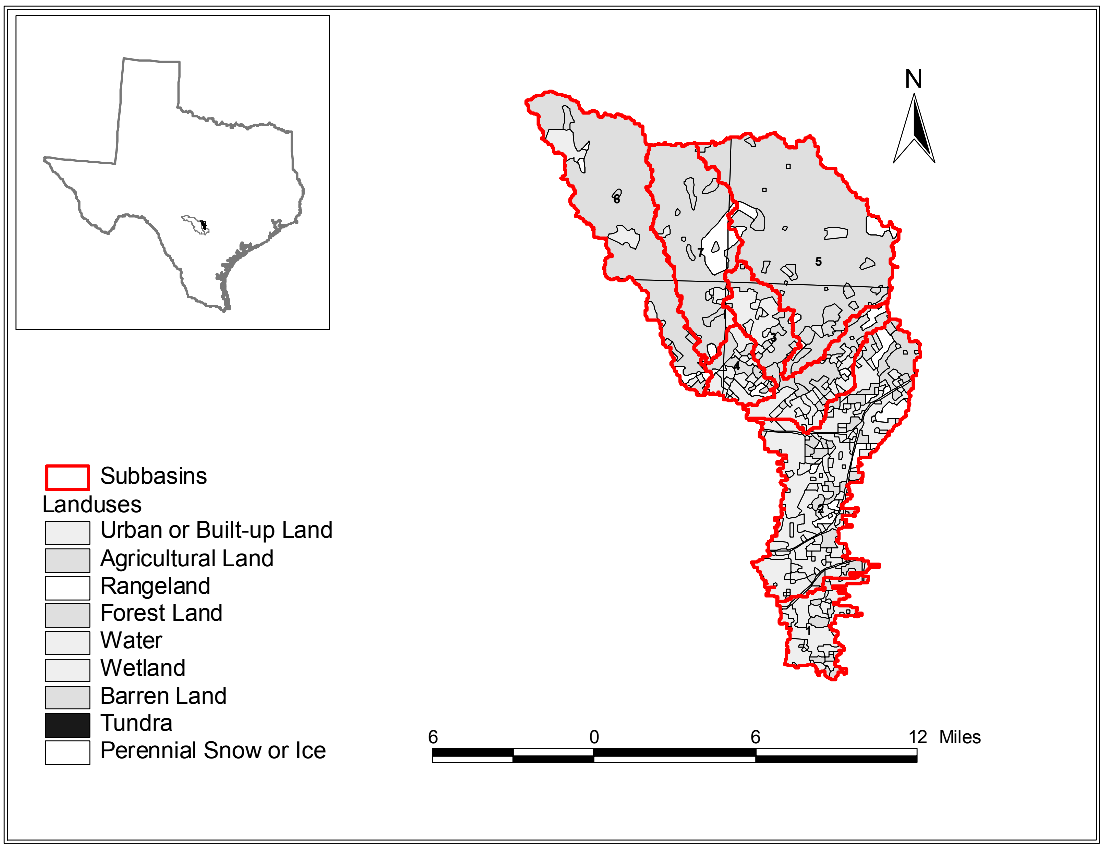


Figure 13. Land Uses of the Salado Creek Watershed, Bexar County, Texas

Uncertainty Analysis

There are various techniques available for propagating the uncertainty of a model. Some of the popular techniques are Monte Carlo Simulation, Latin Hypercube Sampling and First Order Approximation. These techniques vary in their conceptual approach and the effort required for computation. First Order Approximation was used for propagation of uncertainty in this study.

First Order Analysis (FOA)

According to FOA the estimate of the variance of the output is given by (Haan, 1995a):

$$Var(O) = \sum_{i=1}^{p} S_i^2 Var(P_i) \tag{4}$$

where S_i is the absolute sensitivity of the model output O with respect to parameter P_i.

Also the fraction of the total variance attributed to a particular parameter is given by:

$$F_i = \frac{S_i^2 Var(P_i)}{\sum_{i=1}^{p} S_i^2 Var(P_i)} \tag{5}$$

where P is the number of sensitive parameters.

This study was aimed at obtaining the fraction of the variance of the simulated peak in-stream fecal coliform concentration due to the uncertainty in the most sensitive parameters determined by the sensitivity analysis.

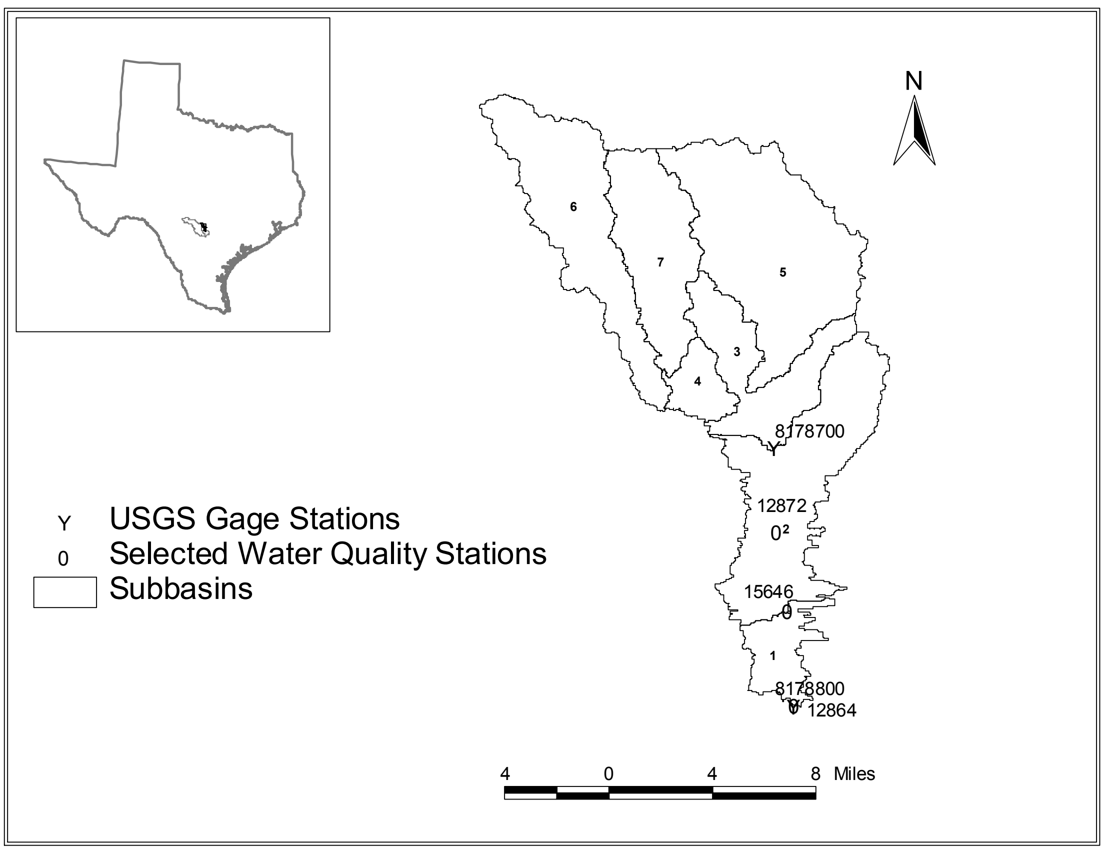


Figure 14. Locations of USGS Gage Stations and Water Quality Observation

Water Quantity Issues—Water Re-Use

Land use by humans in the form of agriculture, transportation, manufacturing, mining and construction invariably has an impact on the surrounding ecosystem. This includes increased nutrient loads from point and non-point sources (NPS), toxic loading from point sources, increase or decrease in the stream flows - all of which might lead to a decline in the overall water quality of the streams and make then unfit for recreation or human and wildlife consumption.

A 1996 Texas Natural Resources Conservation Commission (TNRCC) study indicated that water quality in Leon and Salado Creeks in the San Antonio River Basin is impaired due to elevated concentration of nutrients, fecal coliform bacteria and violation of DO. Subsequently, these water bodies have been included in the Federal Clean Water Act Section 303 (d) listing of impaired water bodies for Texas. The United States Environmental Protection Agency (USEPA) under its Clean Water Action Plan of 1998 is emphasizing the need for state, local and tribal authorities to carry out a watershed level study and management approach in order to address the issues of nonpoint source runoff and pollution and restore the health of impaired waters. The San Antonio River Basin traverses at least three eco-regions in Texas: the Central Texas Plateau, the Texas Blackland Prairies, and the Western Gulf Coastal Plain This basin is dominated by urban and industrial development from the city of San Antonio but agriculture is also a major economic source in the region. Since there is diverse economic activity in this region, the sources of pollution are also diverse. To add to this is the increasing ethnic diversity and income levels. The social and economic diversity in this watershed makes it a classic case study for urban areas. It is critical to analyze the ecological risks that various factors such as land use change and nonpoint source pollution pose to these ecosystems in order to restore them.

The major streams in this basin include Salado Creek, the Upper San Antonio River and Leon Creek. shows the location of Leon and Salado Creeks. Both creeks originate in the north central region of the basin. Leon Creek flows in the western region of the San Antonio metropolitan area whereas Salado Creek flows in the eastern region of the San Antonio metropolitan area. Both eventually join the San Antonio River south of the city.

The Leon Creek Watershed is fed by runoff, springs and small, undesignated streams. The drainage area crosses the Edwards Aquifer Recharge Zone, spanning from the Hill Country northwest of San Antonio through the western edge of the City of San Antonio to its convergence with the San Antonio River southeast of the city. Although the northern half of the segment is normally dry, this water body is a major source of aquifer recharge during heavy storm events, potentially including urban runoff and leaks from sewage collection (Harris, 2000). Water quality in these river segments must be restored by 2003 under the TNRCC Statewide Basin Management Schedule (TNRCC, 1997). Under the Federal Clean Water Act of 1972, restoration of these water bodies requires development of a Total Maximum daily Load (TMDL) for each component not in compliance.

Problem Statement

In keeping with the above-defined management objectives, it becomes imperative to understand the various physical, chemical and biological processes (anthropogenic as well as non-anthropogenic) that occur at the watershed level and their effects on various indicators of ecosystem health. Nonpoint source (NPS) pollution is of vital importance to water resources and land management activities. This is because the sources of nonpoint pollution are distributed over a landscape, both spatially as well as temporally. It is very difficult to pinpoint all sources of nonpoint pollution because it arises from varied land usage, which are spatially distributed over a geographic region. Also, there is no defined pattern for the release of this type of pollution. All this makes it very important to study and understand the spatial as well as temporal patterns of nonpoint source pollution in order to determine and reduce its impact on the ecosystem.

The concentration of DO in natural waters is a primary indicator of overall water quality and the viability of the aquatic habitat). A number of abiotic and biotic factors, such as reaeration, stream respiration, nutrient and organic loading affect DO concentration in streams. Hence DO is considered a non-conservative constituent (Greb & Graczyk, 1995). Low base-flow in streams can also aggravate DO problems. Various point and nonpoint sources of pollution such as wastewater treatment plant (WWTP) discharges and agricultural runoffs can affect DO concentration in streams. At the watershed level, DO is a key indicator of water quality in the receiving waters that can characterize risks sufficiently. Hence, understanding the impact of various anthropogenic as well as non-anthropogenic processes on the DO concentration in water bodies becomes imperative in any watershed management/rehabilitation plan.

Objectives

The objective of this study was to use a complex GIS-based hydrological/water quality model in order to simulate various physical, chemical and biological processes occurring in the Leon Creek Watershed, to simulate hourly DO concentrations in Leon Creek and to study the effect of water reuse on DO concentration in the creek and its tributaries.

Specifically, the model was used to evaluate stream flow augmentation as an alternative management practice for improving the water quality in Leon Creek and associated tributaries. The hypothesis tested in this study was that increasing the base flow during low-flow periods increases the mean daily DO concentration in the streams.

Materials and Methods

This project used historical and current water quality and quantity data from the San Antonio River Authority (SARA) and the United States Geological Survey (USGS) monitoring stations on Leon Creek. Watershed land use was determined using the USGS Geographic Information Retrieval Analysis System (GIRAS) land use classification database.

The US EPA's Better Assessment Science Integrating Point and Nonpoint Sources (BASINS, version 3-Beta) environmental modeling software was used for used for modeling purposes. BASINS brings key data and analytical components together in one framework. This is consistent with the new holistic approach, which makes watershed and water quality studies much easier. BASINS uses Arc View-Geographic Information System (GIS) as the integrating framework to provide the user with a fully comprehensive, state of the art watershed management tool for developing TMDLs that require the integration of both point and nonpoint sources (Battin et al*.,* 1999*).*

BASINS addresses three objectives: 1) to facilitate examination of environmental information, 2) to provide an integrated watershed and modeling framework, and 3) to support analysis of point and nonpoint source management alternatives (Battin et al*.,* 1999). Originally released in September 1996 (BASINS version 1.0), heart of BASINS version 2.0 is its suite of interrelated components essential for performing watershed and water quality analysis. These components are grouped into five categories:

- National databases with local data import tools;
- Assessment tools (TARGET, ASSESS and Data Mining) that address needs ranging from large-scale to small-scale and Watershed Characterization Reports;
- Utilities including Data import, Land use Reclassification, Digital Elevation (DEM) Reclassification, Watershed Delineation and Water Quality Observations Data Management Utilities;
- Watershed and water quality models including NPSM (HSPF), TOXIROUTE and QUAL2E; and
- Post-processing output tools.

BASINS (version 3-Beta) includes additions like an automated watershed delineation tool and an additional model SWAT. Among the various models available within the BASINS suite, the Non Point Source Model (NPSM) Model was used for the present study. NPSM can integrate watershed-based point and nonpoint loading and transport. NPSM is basically an abbreviated version of the HSPF (Hydrological Simulation Program-FORTRAN) model, with the added convenience of a graphical user interface (GUI). HSPF is a combined watershed based point and non-point source and receiving-water model that has been under development since the early 1980s with the help of USEPA grants. This project will use NPSM (HSPF) for its watershed based water quality modeling needs.

Building the Project for NPSM

Building the project for NPSM model run involves creating the input files for the selected watershed by using the USEPA's RF1 (reach file version 1) as a stream network, extracting information on point sources from its database and calculating what percent of the land surface belongs to each category. This is done in the ArcView-GIS environment within BASINS and the steps involved are discussed below.

Watershed Delineation

The first step in setting up the project is to delineate the sub-watersheds to be modeled. This first involves creation of a study area (i.e. main watershed) that will be delineated into sub-watersheds. USEPA's River Reach Files Version 1 (RF1) provided the stream network information for Leon Creek and its major tributaries – Helotes Creek and Culebra Creek. This data was developed for stream routing for modeling at 1:500,000 scale (Lahlou et al., 1998). The study area for this project was determined by visualizing the RF1 stream network and making sure that all major streams that contribute flows to Leon Creek were included. The next logical step is to divide the study area into sub-watersheds that contribute flows to each of the stream segments. The quick and easy way to do this is to use the automated watershed delineation tool available with BASINS. This tool allows for rapid definition of sub-watersheds based on user supplied coverages or point and click selection. Streams, DEM or even aerial photographs can be used to assist in delineation.

As can be seen, each sub-watershed is associated with a single stream segment and each stream segment is associated with a single "pour point" at its most down-stream location. NPSM calculates various quantities such as pollutant loading, surface-runoff, and sediment transport within each sub-watershed and "dumps" these at the most upstream point of the associated stream segment. The quantities are then routed through the length of the stream segment and the output is a concentration/flow measured at the most downstream point (pour point) in the modeled sub-watershed. NPSM is therefore a lumped-catchment model in the sense that it does not allow for a longitudinal resolution of flow or pollutant loading and output measurement. Hence, one must make the assumption that the output concentration or any other physical quantity is representative of the entire reach (stream segment). This assumption is more valid when considering only nonpoint sources as opposed to both point and nonpoint pollutant sources. This is because the point sources may have more localized effects on water quality.

BASINS allows the users to add/remove pour points before it delineates sub-watersheds. Taking advantage of this option a pour point was added to coincide with the location of USGS gauge station on Leon Creek at Interstate Highway 35. This will ensure that when the model is calibrated for flows, the observed and simulated values are for the same output point.

Invoking NPSM

With the sub-watersheds delineated, the next step is to select the sub-watersheds and invoke NPSM within BASINS. When invoked, NPSM first creates the base project file by taking the input files created in earlier steps and incorporating data such as parameter values for various watershed level and in-stream processes, stream cross-sections and land use contributions among others. The user then calls the project file within NPSM and uses the GUI to set up the input file for simulation.

Watershed Modeling: Application of the NPSM/HSPF Model

The BASINS software was downloaded from the USEPA website and also obtained via CDROM. All the necessary data for Leon Creek was extracted within BASINS and projected using the following parameters:

Projection: UTM Zone 14

Spheroid: GRS 80

Central Meridian: -99

Reference Latitude: 0
Northing:0
Easting: 500000
Scale Factor: 0.9996

This projection provided land use data and maps for the Leon Creek area and an USEPA RF3 reach file for the stream channel that is more detailed than the RF1 reach files. However, to keep things simple, the RF1 file was used for delineation and modeling purposes. An image theme of USGS topographic map called Digital Elevation Model (DEM) having 30m resolution was added to the GIS environment to aid in the delineation of watershed. This image theme was obtained from the website of Pacific Environmental Services, Inc. (http://home.pes.com/demprog.html).

Watershed Characteristics

The total length of the creek from its headwaters in the north central part of the San Antonio Basin to its confluence with the Medina River, south of the city of San Antonio, is about 36 miles. The Leon Creek watershed, as delineated by the BASINS software has mixed land use. Figure 12 shows Leon Creek and its associated sub-watersheds. The total area of the delineated watershed is 131906 acres. Figure 13 shows the land use characteristics of the watershed based on the Anderson Level II classification available in BASINS. The upper half of the watershed area is mostly evergreen forest. Crop land and pasture land covers the central and the lower region of the watershed. Most of the urban, built-up and commercial land is spread in the lower half of the watershed. Roughly, the upper third of the watershed lies in the Edwards Aquifer recharge zone. This land use inventory is from the 1996 data available within BASINS. NPSM requires specification of the percent impervious cover for various land use category. So all of the impervious area is lumped into the urban land use assignment. For the present study, the impervious cover was defined as 70 percent of the urban land use category. Similarly, impervious cover for agricultural land was defined as 5 percent of the total agricultural land. Table 11 gives the impervious cover defined for each type of land use based on literature values.

Table 11. Percent Impervious Cover for Each Land Use Type

Land-use type	Percent impervious
Residential	25
Open Land	5
Forest	0
Commercial	70
Agricultural	5
Barren	0

Source: Brun & Band, 2000

Reach Physiography

NPSM requires reach physiography to be defined in order to do the in-stream mass balance and routing calculations. The date required is depth of flow, surface area, volume and outflow for each reach. These data are contained in an “F-table” within the model. BASINS 3-Beta has the capability of automatically generating F-tables for each stream reach for use within NPSM. NPSM does not actually consider the cross-section of reaches for hydrodynamic calculations. Instead, it uses the depth, volume and surface area of each reach for flow and mass routing.

Water Quality Data Collection

Historical water quality data for dissolved oxygen was found to be limited. The San Antonio River Authority (SARA) maintains water quality data from 1990 onwards at several locations on Leon Creek.

However, the SARA water quality data consists of "grab samples" collected at a specific time of the day during low flow periods, normal flows or storm events. For studying DO in a stream, continuous 15-minute or hourly data is required in order to capture the diurnal variation of DO concentrations in a stream. As such, the water quality data from SARA was limited in scope and usability.

Supplementary water quality data was collected by deploying electronic Yellow Spring Instruments (YSI) Datasondes at two locations along the creek. Figure 14 shows the location of the two monitoring sites on Leon Creek. Water quality data was collected at 15-minute intervals for a period of two weeks in August 1999, January 2000 and July 2000 on both sites. Table 12 shows the various water quality constituents monitored during the two-week periods.

Table 12. Water Quality Constituents Monitored by YSI Datasondes

Constituent	Units of measurement
Temperature	^{o} C
Specific Conductivity	mS/cm
DO	mg/L
DO % saturation	-
Depth	feet
PH	-

This water quality data was later used in the modeling exercise to observe daily trends of DO and for the purpose of rough calibration.

Calibration, Validation and Sensitivity Analysis

The first step in using a hydrological/water quality model is to calibrate it for hydrology. Hydrology drives all other processes in a watershed. Part of the precipitation that impinges on the land surfaces infiltrates into the soil, a part of it evaporates and the remaining flows as surface runoff. Out of the fraction that infiltrates, a part of it may be lost to deep percolation and the remaining recharges an underground spring and/or aquifer. This shows up later as base flow when the aquifer/spring recharges a stream. The surface runoff flows on the land and finally finds its way to a receiving stream, river or a lake.

Calibration of hydrology in NPSM (HSPF) involves the adjustment of parameters that govern watershed response to precipitation and comparing simulated flows versus measured flows in streams. Hydrologic calibration is performed for long-term simulation (base flow) and for specific storm events depending upon the needs of the modeler. For the present project, long term simulation of the model for base flows was important since most of the DO problems occur during low base flows in the streams. Hence it was decided that the model be calibrated for long-term simulation only. The long-term simulation involves establishing an annual water balance and estimating initial storage conditions. If the estimated runoff over a time period is within reasonable limits of the observed runoff for that time period, the model is said to be well calibrated. A well calibrated model also takes into account seasonal variability in flow conditions.

Data

For the present project, historical daily mean stream flow data was obtained for USGS gauging station number 08181480 located on Leon Creek at Interstate Highway 35. Figure 14 shows the location of the gauging station on Leon Creek. While creating the NPSM project, care was taken to include this geographic point as the last "pour point" for the entire watershed. NPSM simulates flows at each user-defined pour points within a watershed, which are located on the stream being modeled. Thus, by ensuring that the most downstream pour point of the entire watershed coincides with a USGS gauging station, the calibration exercise becomes accurate. Incidentally, the drainage area delineated by BASINS for that pour point is within

6 percent of the drainage area reported by USGS for their gauging station. Table 13 gives the details of the USGS gauging station used for hydrological calibration purposes.

Table 13. Details of USGS Gauge Station Number 08181480 on Leon Creek

Station name	Leon Creek At I.H. 35 At San Antonio, TX
Station number	08181480
Latitude (ddmmss)	291947
Longitude (dddmmss)	0983502
State code	48
County	Bexar
Hydrologic unit code	12100302
Basin name	Medina
Drainage area (square miles)	219
Gage datum (feet above NGVD*)	573.49

NPSM Hydrological Concepts

The user's manual for HSPF (Bicknell *et al.,* 1996) describes the hydrological concepts in NPSM. The water balance is denoted by a simple equation:

$$P^* - ET - PERC - \Delta SM = RO \qquad (9)$$

where:

P^* = precipitation (inches)
ET =evapotranspiration (inches)
$PERC$ =deep percolation (inches)
SM =soil moisture storage (inches)
RO =runoff (inches)

This simple relationship has been characterized in NPSM (HSPF) via numerous variables and storage compartments. Precipitation that falls on an impervious surface can either runoff, be stored in a storage compartment and evaporate, or be temporarily stored in a storage compartment and runoff later. On a pervious surface, the response to precipitation is a bit more complex. On impingement, it can either runoff, be stored in a temporary compartment and evaporate, be retained in a storage compartment temporarily and flow later as interflow or enter the subsurface via infiltration. NPSM divides the sub-surface into three zones: the upper zone, the lower zone and the deep groundwater zone. Upon entering the upper zone via infiltration, the water can remain in storage, be available for evapotranspiration or enter the lower zone via infiltration. In the lower zone, water can remain as storage, be available for evapotranspiration or enter the deep groundwater zone via infiltration. Upon entering the deep groundwater zone, the water can remain as groundwater storage, exit as groundwater outflow, be available for evapotranspiration or be lost from the system due to deep percolation. All the above processes have an initial value or rate. These values and/or rates govern the flow of water on pervious or impervious surfaces. It is observed that runoff from pervious surfaces is more complicated than runoff from impervious surfaces. Runoff from impervious surfaces directly affects the peak flows and volumes where as runoff from pervious surfaces affect base flow in stream. From the standpoint of long-term calibration a sensitivity analysis was performed on three parameter variables that directly affect base flow and the overall annual water balance for the simulation period.

Hydrology Calibration

Calibration was performed for the year 1994. 1994 represents a moderate year with above average annual precipitation for the region. 1992 represents an extremely wet period due to a very high annual precipitation and hence was not used for calibration purposes. Similarly, 1995 was a record drought year for the region and hence was not used for calibration purposes. Initial parameter estimates for hydrology were made based on the BASINS Technical Note 6 that is available from USEPA's BASINS web page (http://www.epa.gov/ost/basins/bsnsdocs.html). These initial parameter estimates were then revised iteratively to arrive at the final values. The model was set to simulate flows from the period 1987-95. In doing so, a sufficient lead-time was given to reach dynamic equilibrium for various storage variables. For the initial runs, the estimated initial storage parameters at the start of the simulation period were checked against equilibrated values during a similar period a few years later. The initial estimates were then revised for the start of the simulation period and further simulations done. Four steps were followed for the final calibration exercise:

1. Development of an overall mass balance for the watershed by adjusting overall gains and losses from the watershed due to precipitation, evapotranspiration and loss to deep groundwater. This water balance should be compared to the observed flow data.
2. Adjusting the low-flow high flow distribution as compared to observed data by adjusting the rate at which the water infiltrates the soil, enters groundwater and recharges the streams.
3. Roughly match peak flows and adjust recession rates so that the peaks recede to normal levels as observed.
4. Fit the seasonal distribution of flow, taking into account seasonal distribution of evapotranspiration, soil moisture and changes in groundwater recharge to streams.

The first step was to achieve an overall annual water balance by reasonable distribution of gains and losses in the watershed. It has been reported that the eastern region of Edwards Aquifer receives up to 50 percent more rainfall at some places as compared to the western region of the aquifer. The weather station used for modeling purposes is located to the east of the aquifer region and the Leon Creek Watershed is located to the west of the aquifer region. Moreover, the weather station is about 16 miles away from the USGS flow gauging station under consideration. The farthest reaches of the watershed are about 21 miles away from the weather station. Owing to the above facts, it was concluded that the precipitation observed at the weather station is not representative of the precipitation over the entire watershed and hence a multiplication factor of 0.5 was incorporated in the input precipitation data. Deep percolation losses were increased by adjusting a parameter DEEPFR to 0.4. These deep percolation losses denote permanent losses from the watershed. Such losses might include lateral outflow of groundwater from one aquifer to another low-lying aquifer outside the watershed area. Sometimes, groundwater flows beneath the gauging station and shows up as base flow at a point downstream. These flows are not accounted for at the gauging station reading and hence constitutes a loss from the system. There is reason to believe that such losses are taking place in the Leon Creek Watershed, since the geology of Edwards Aquifer Recharge Zone consists of extremely porous limestone formations. All these losses are incorporated into one parameter called DEEPFR in NPSM. Actual evapotranspiration was observed to be close to potential evapotranspiration and hence no factor was incorporated for evaporation data. After doing the above adjustments, the overall annual runoff volume was observed to be over-predicted by about 39 percent. This water balance can be improved by incorporating better precipitation data and by studying the evapotranspiration parameters in detail.

The second step in the calibration process was to compare high-flow low-flow distribution with the observed data. This was achieved by adjusting model parameters representing infiltration (INFILT), interflow (INTFW) and groundwater recession (AGWRC).

The third step was to roughly compare the simulated peaks and observed peaks (not extreme storm events) for shape and recession. The model parameters adjusted in this step were interflow recession constant (IRC), and surface flow parameters (LSUR, NSUR and SLSUR).

The final step in hydrology calibration was to match the monthly flow distribution. It was observed that the model overestimated winter month base flow and underestimated summer month base flow. Adjusting monthly parameters for evapotranspiration (MON-INTERCEPT, MON-LZETPARM), upper zone storage (UZSN), evapotranspiration from base flow (BASETP) and groundwater recession (KVARY) reduced this difference. The table shows default values, calibrated values and the minimum and maximum possible values as defined in the BASINS user's manual (USEPA, 1996). A complete description of each parameter is provided in BASINS Technical Note 6, which is available from the BASINS web page (http://www.epa.gov/ost/basins/bsnsdocs.html). Figure 15 shows a graph of the observed vs. predicted flows at USGS gauging station on Leon Creek near I.H. 35 for the year 1994.

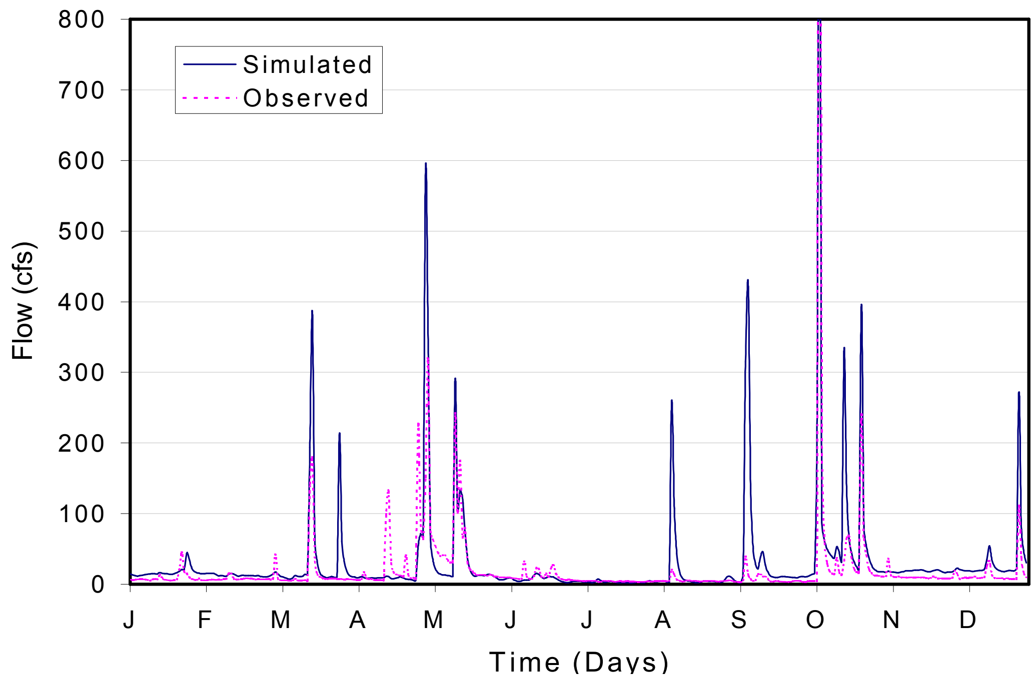


Figure 15. 1994 Simulated and Observed Flow

The following observations can be made from the graph. Base flow is over- predicted for some winter and spring months. Extreme storm events are not well predicted by the model. These deficiencies can be attributed to the following:

1. Data from only one gauging station was available for calibration of the entire watershed. This considerably limits the extent to which calibration can be done, especially of a watershed covering an area of about 130,000 acres. Technical experts suggest using at least 5 gauging stations for good calibration.

2. Weather data that is representative of the watershed area under study was lacking. Data from a weather station that is 20 miles away from the gauging station was used. Moreover, the precipitation was assumed to be uniform over the entire watershed, which in reality might not be the case for a big watershed. This introduces an uncertainty in the modeling process as some localized events get

wrongly introduced in the simulation. A distributed data covering the entire watershed area is needed for good calibration.

3. Calibration was attempted using the "lumped parameter" approach. In other words, all types of land segments were assigned same parameter values. Assigning parameter values based on type of land-use and/or location could have improved calibration. However, this involves a detailed study of the land-use characteristics and thorough knowledge of local conditions in the watershed, which was beyond the scope of this project.

Figure 16 shows a scatter plot of observed vs. predicted flows at USGS gauging station 08181480 on Leon Creek. R-squared value of 0.657 indicates a fairly reasonable prediction for hydrology. The slope of the line is about 0.9.

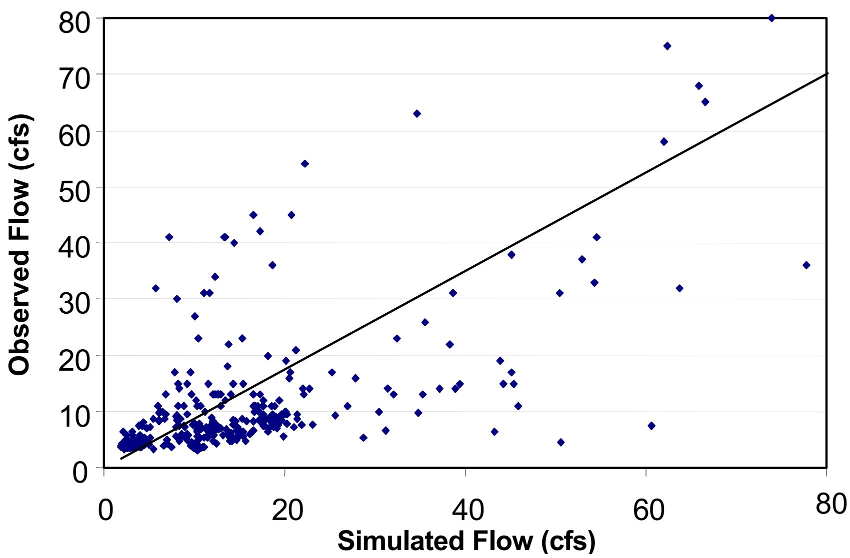


Figure 16. 1994 Scatter Plot of Observed vs. Simulated Flow (R^2 = 0.647; m = 0.875)

Validation

The model was validated for the year 1990, which represents a similar year to 1994 in terms of precipitation. Figure 17 shows a graph of the observed vs. predicted flows at USGS gauging station on Leon Creek near I.H. 35 for the year 1990. Figure 18 shows a scatter plot of observed vs. predicted flows at USGS gauging station 08181480 on Leon Creek for the year 1990. The R-squared value is about 0.59, which indicates a fairly reasonable validation. The slope of the line is about 1.1.

Hydrology Sensitivity Analysis

Three model parameters that affect base flow and overall water balance respectively were considered for sensitivity analysis. These parameters were soil infiltration capacity (INFILT) and deep percolation losses (DEEPFR) in the Pervious Land Module (PERLND) and retention storage capacity (RETSC) in the Impervious Land Module (IMPLND). A simple sensitivity analysis was performed by varying the parameters by +/- 50 percent of the calibrated values.

In NPSM/HSPF, INFILT is the parameter that controls the division of precipitation into surface and sub-surface flow and storage compartments (USEPA, 2000b). Higher INFILT values means higher base flow in streams as more water goes to the sub-surface. Lower INFILT values mean that more water flows as surface runoff and less water percolates, leading to lower base flow. Infiltration also affects the overall runoff

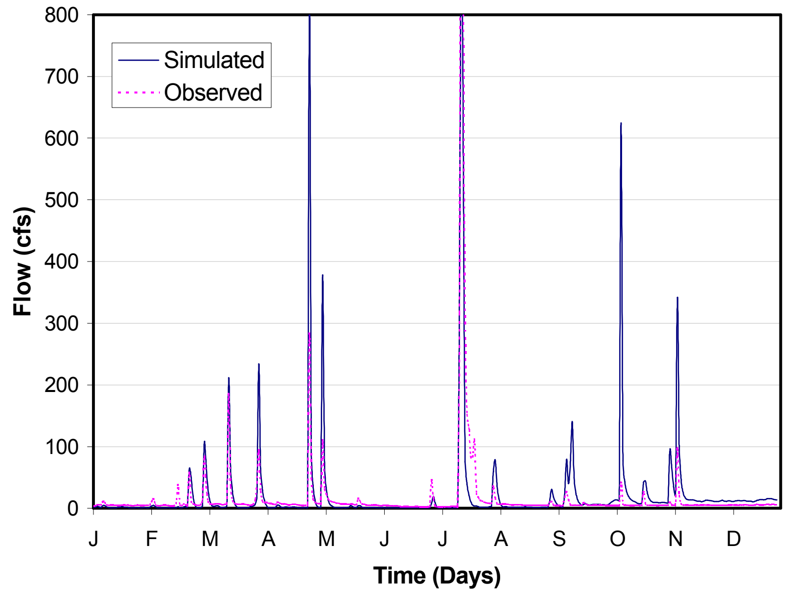


Figure 17. 1990 Simulated and Observed Flow

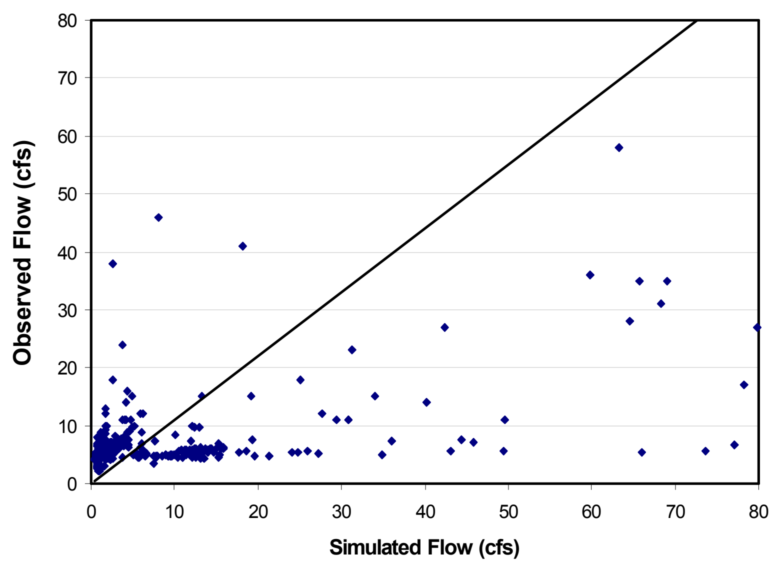


Figure 18. 1990 Scatter Plot of Observed vs. Simulated Flow (R^2 = 0.59; m = 1.112)

volume. The more the infiltration, the higher the chance for evapotranspiration losses from soil layers. Similarly, in conjunction with DEEPFR (deep percolation), higher INFILT values lead to larger permanent losses from the watershed. It was found that increasing the INFILT value by 50 percent lead to an increase of 23 percent in the overall runoff volume and an increase of 87 percent in average base flow for the year 1994. Decreasing the INFILT value by 50 percent decreased the overall runoff volume by 22 percent and the average base flow by 74 percent for the year 1994.

DEEPFR in NPSM/HSPF is the fraction of infiltrating water that is lost to deep percolation. 1-DEEPFR then is the fraction available as active groundwater storage and hence contributes to base flow in the streams. Portions of a watershed at higher elevations are more prone to groundwater losses since there might be lateral outflows to low-lying aquifers outside the watershed. An example of this inter-watershed transfer is the significant flow from Edward's Aquifer to Comal Springs. During 1980, nearly 48 percent of the spring discharge from Edwards Aquifer was from Comal Springs in Comal County (Ryder, 1996). DEEPFR is also used to denote losses that may not be measured at the flow gage used for calibration, such as flow around or under the gage site. On account of the above reasons, DEEFR was set to 0.4, which is a rather high value. However, it gave a reasonable water balance. A detailed study of groundwater conditions is needed to verify this value. A 50 percent increase in DEEPFR led to a 29.6 percent decrease in overall runoff volume for the year 1994. Decreasing DEEPFR by 50 percent increased the overall runoff volume by 41 percent for the year 1993.

RETSC is the depth of water that collects on the impervious surface before any runoff occurs. This directly affects the amount of storm water runoff in streams and hence the overall runoff volume. A 50 percent increase in RETSC led to a 10 percent decrease in overall runoff volume for the year 1993. Decreasing RETSC by 50 percent increased overall runoff volume by 14.3 percent for the year 1993. It should be noted that RETSC does not affect base flow in the streams, but only the surface runoff from impervious land.

Temperature

Temperature plays a significant role in the solubility of oxygen in water. Oxygen, being a non-polar molecular compound is not highly soluble in water. Solubility of oxygen depends on water temperature and salinity among others and can range from 4ppm[8] to 15ppm. Higher water temperatures demonstrate lower solubility of oxygen and vice-versa. However, water temperature and DO are not related linearly. To account for temperature therefore, it is necessary to consider the percentage saturation. DO is then denoted as a percentage of the saturation value for a particular temperature. For example, a DO concentration of 7.5 mg/l (or ppm) might represent a 90 percent saturation, which means a good turnover for DO at that particular temperature. However the same concentration of 7.5 mg/l might represent 60 percent saturation, thus indicating a poor turnover for that particular temperature.

Temperature of water is a linear regression function of air temperature in NPSM. It was observed that simulated water temperature followed the air temperature curve and was lower than the air temperature, which is usually the case for small streams. As the volume of water is not large compared to a reservoir or lake, the water is expected to gain and lose heat quickly and hence follow the air temperature curve in shape. For this project, calibration for water temperature was found to be difficult for the following reasons:

1. No hourly-observed data was available for the simulation period. Hourly data was available for a period of two weeks in August 2000, January 2000 and July 2000 from the water quality Datasondes deployed in the creek. The data availability constraints within BASINS and NPSM restrict the simulation of the model up to the year 1995. Hence, it was not possible to simulate the model for the year 2000.

2. Some “grab sample” data was available from SARA for the years 1994-97. However, the quantity of data was very less. For example, only 16 readings were available for the year 1994 at the SARA monitoring site near Leon Creek Wastewater Treatment Plant. These readings represent a “snap-shot” of the water conditions at a particular time of the day and by no means represent the hourly values or even daily means. Thus, this data set is of practically no use for calibration purposes. Moreover, the location of data collection was different from the location of the calibration point, although it was on the same stream segment as the calibration point. Depending on the conditions surrounding the sampling points, the data can vary considerably within the same reach segment. For example, the water temperatures at a location surrounded by shade can be lower than a down stream or upstream location where there is no shade.

Considering the above factors, it is evident that calibration for temperature is a not possible for the present study. The same holds true for DO. Moreover, the aim of the project was not to use the model for deterministic analysis, but rather to analyze trends. Using a complex and highly parameterized model such as NPSM (HSPF) itself involves a big learning curve, much less using it to simulate such a complex constituent as DO. The objective of the project was to be able to use NPSM to successfully simulate the various physical, chemical and biological processes taking place in the watershed and to compare simulated trends with the trends actually observed. In other words, the objective was to see how effective the model simulates complex processes, rather than how accurately it does it. Extensive calibration therefore was neither a priority nor a possibility within the given time frame and data constraints. In absence of calibration, a trend analysis is the best option to determine whether the model is doing what it is supposed to do.

Dissolved Oxygen

As pointed in the earlier section, calibration for DO is not possible for the present study. From the standpoint of the NPSM model, what is of importance however is whether that model can simulate diurnal DO variations within a stream segment. For the purpose of trend analysis, the data collected using Datasondes at the sampling site on Lackland Airforce Base was used. January and July 2000 data was chosen to represent cold and hot weather conditions. As explained in the previous section, the model could not be setup for year 2000 simulation period. Hence, the 1994 water year was chosen for simulation of DO with the model. 1994 happens to be the most recent year that has similar precipitation and flow trends as the year 2000. The following criteria were used:

1. Annual precipitation comparison
2. Monthly precipitation comparison
3. Mean daily stream flow comparison during the period for which data was collected using Datasondes.

Various module sections within NPSM were chosen to simulate DO. These represent to a large extent, almost all of the complex processes taking place in the watershed and in the streams. These processes include fate and transport of BOD and nutrient loading on land surfaces, benthal oxygen demand, phytoplankton influences, alga respiration and sediment oxygen demand among others. The table also gives the default values and the maximum and minimum values as defined for the model. A few parameters were changed from their default values. Quantities such as BOD and nutrient application rates were supplied by the user

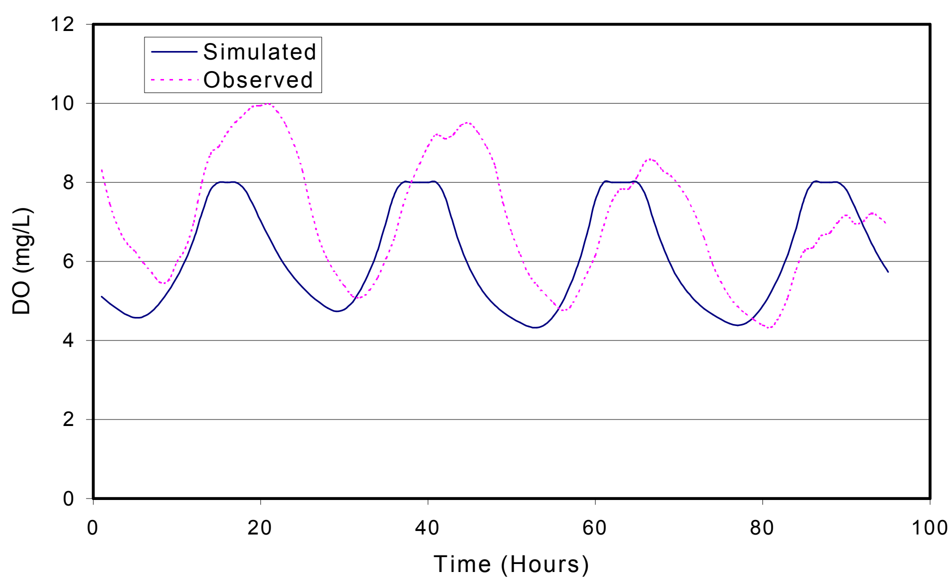


Figure 19. 1994 Simulated and 2000 Observed Hourly DO During January 26 to January 31

since there were no default values. Nutrient application rates were estimated from a report of the United States Department of Agriculture (Lander & Moffit, 1996). These rates were chosen just to demonstrate that nutrient and BOD loading processes could be simulated by NPSM. The model was then run for the simulation period 1994. Figure 19 shows simulated hourly DO for January 1994 as compared to observed hourly DO for January 2000 during the period starting 26th January and ending 31st January. Similarly, Figure 20 shows simulated hourly DO for July 1994 as compared to observed hourly DO for July 2000 during the period starting 26th July and ending 29th July.

The following points need to be considered before analyzing trends:

1. The simulation site and the sampling site are located on the same stream reach as designated by NPSM and are roughly 8 miles apart. The simulation site is the most downstream point of stream no. 6 as designated by NPSM.
2. There could be differences in water temperatures, flows, shading, riparian vegetation and algae concentration at the simulation site and the sampling site.

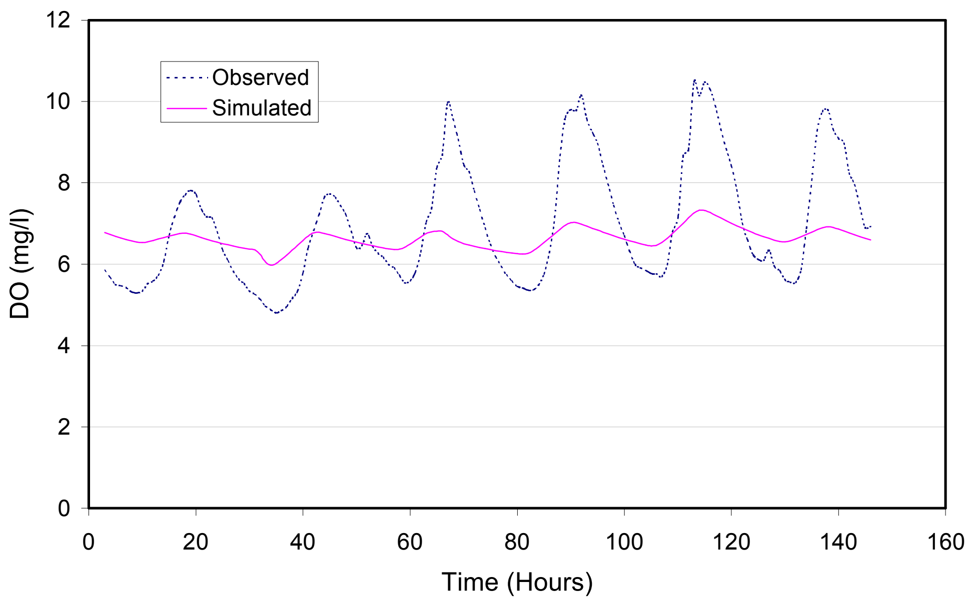


Figure 20. 1994 Simulated and 2000 Observed Hourly DO During July 26 to July 29

Two means of trend analysis were used for comparison of data. The first was a visual analysis of the curves and the second was a comparison of the mean daily DO values. The following observations were made:

1. The simulated DO curve follows the observed DO curve in shape. Its peaks roughly at the same time that the observed curve peaks and sags roughly at the same time that the observed curve sags.
2. The daily variation of simulated DO is within the same range as the daily variation of observed DO.
3. The daily mean simulated DO is within 11 percent of the daily mean observed DO for January period and within 21 percent of the daily mean observed DO .for July period.

Based on the above observations, the following conclusions can be drawn:

1. The model is capable of simulating the diurnal variation of DO for a stream segment.

2. For the scope of the present study, the DO simulation done by the model is satisfactory.
3. The model presently cannot accurately reproduce the diurnal swings in DO in amplitude. This might be due to the fact that simulated organic loading, nutrient loading and algal concentrations in the streams is not representative of real conditions. With extensive calibration and continuous sampling data, the simulation curve could be improved to represent actual conditions.

Sensitivity Analysis for DO

Simulation of the entire DO source and sink cycle in a stream is highly complex and requires an in-depth understanding of in-stream processes. These include BOD, BOD settling, sediment oxygen demand, benthic influences, algal respiration and photosynthesis among others. Besides, these processes can vary significantly along the length of a stream depending on local conditions such as flow, land-use, shading and nutrient loading. Surface processes such as the BOD loading and washoff, nutrient application and washoff, plant uptake of nutrients and atmospheric deposition of nitrogen compounds also indirectly affect the in-stream processes. NPSM has the capability of simulating all these processes involving a large number of parameters. Understanding enough about these parameters in order to get calibrated values involves extensive research that is often not possible due to time constraints or lack of data. As such, there is an inherent uncertainty built in the model because of over-parameterization.

Due to the above reasons, it was thought best to keep default values for most in-stream parameters that affect DO. A few parameters were changed from their default values. Performing a detailed sensitivity analysis was beyond the scope of this project. It was however found that in-stream DO concentration was extremely sensitive to groundwater concentration of DO. This is reasonable since groundwater recharges Leon Creek via springs. A simple sensitive analysis was performed by varying the groundwater DO concentration (GRNDDOX) by +/-50 percent from the calibrated monthly values. An increase of 50 percent in GRNDDOX led to an average increase of 35 percent in the predicted average daily DO concentration during the period January 26, 1994 to January 31 1994. Decreasing GRNNDOX by 50 percent led to a 44.5 percent decrease in average daily DO concentration over the same period. Similarly, an increase of 50 percent in GRNDDOX led to an average increase of 12.7 percent in the predicted average daily DO concentration during the period July 26, 1994 to July 29, 1994. Decreasing GRNNDOX by 50 percent led to a 20.8 percent decrease in average daily DO concentration over the same period.

Low-flow Augmentation to Rectify DO Problems: A Rehabilitation Approach

The ultimate objective of this project was to evaluate the effects of low flow augmentation on the DO concentration of Leon Creek. As stated earlier, most of the DO problems in streams occur during low flow conditions when the water is stagnant, the concentration of nutrients, algae and BOD is high and the reaeration rate is low. The July 2000 observed DO graph (Figure 20) is an example of such conditions. It shows that the DO criterion was violated at many instances with concentrations as low as 4.3 mg/l. Such low flow conditions usually occur during dry periods when there is no rainfall and the temperatures are hot. It should be noted that Texas experienced one of the hottest summers that year with record temperatures and record number of consecutive days without significant precipitation.

The city of San Antonio boasts of a history of water recycling dating back to the 1960s when the city public service (CPS), the electric utility started using treated wastewater from the Calaveras and Braunig lakes in its cooling towers (Texas Water Savers, 1998). In the recent years, in an effort to maintain the sustainability of Edward's Aquifer, SAWS started to develop the San Antonio Water Recycling Project. As part of this project, treated wastewater from the Leon Creek Waste Water Treatment Plant will be used for irrigation, instream flows and industrial purposes. Pipelines painted in purple have already been installed. In a pilot project in 1996, water treated at the Leon Creek plant and stored at Lake Mitchell was delivered to Mission del Lago Golf Course for irrigation purposes. The Water Recycling Project is expected to save about 11 billion gallons of Edwards Aquifer water annually for potable use (Texas Water Savers, 1998).

Approach

For the present study, a hypothetical quantity of water having quality representative of the Leon Creek Wastewater Treatment Plant discharge was assumed to be conveyed from the plant to an upstream site via the recycling pipe. This treated wastewater was discharged at an arbitrarily chosen upstream point to augment in-stream flow. This scenario can be represented in NPSM by adding a "point source" to the stream of interest. The exact location of the point source can be specified by entering a "mile point" number i.e. the number of miles from the most downstream point on the stream of interest. For the present study, stream segment number 5 (Culebra Creek) in NPSM was chosen as the receiving stream for flow augmentation. The mile point for the point source was fixed at 3.75 miles, which is the headwater for the creek. Four parameters were chosen to represent the quality of water entering as point source in NPSM. These were nitrate-nitrogen, phosphate (dissolved ortho-phosphorus), DO and BOD. Representative values for nitrate-nitrogen and phosphate were obtained from the data obtained from SARA at its monitoring site located near the treatment plant. No data for BOD concentration were available from SARA. BOD data was acquired from effluent data collected by the San Antonio Water System (SAWS) monitoring site near the treatment plant. Annual average BOD concentration for the 6-year period 1992-98 was used as a representative value. Table 14 shows the representative values for water quality for the Leon Creek Wastewater Treatment Plant discharge.

Table 14. Representative Values for Water Quality Constituents at Leon Creek WWTP

BOD	2.25 mg/l
Nitrate Nitrogen	5.5 mg/l
Phosphorous (Phosphate)	1.3 mg/l
Temperature	27 ^{o}C
PH	7.0
Dissolved Oxygen	7.0 mg/l

Scenario Analysis

A 3-month period with no significant rainfall was chosen between 1st July and 30th September of years 1993, 1994 and 1995. This constituted a hot weather low-flow period as compared to other months. The periods were chosen based on the following four criteria:

1. Palmer Drought Severity Index
2. Monthly mean precipitation as observed at San Antonio International Airport.
3. Monthly mean temperatures as observed at San Antonio International Airport.
4. Monthly mean observed flows as observed at USGS gauging station number 08181480, which was used for hydrology calibration.

Six scenarios were evaluated. They are as follows:

1. Control (CTRL) – Simulated flow and DO without any external flow source. This is the "as is" scenario.
2. Scenario 1 (S1) – Flow augmented by adding a point source with 0.25 times base flow.
3. Scenario 2 (S2) – Flow augmented by adding point source with 0.5 times base flow.
4. Scenario 3 (S3) – Flow augmented by adding point source with 1 times base flow.
5. Scenario 4 (S4) – Flow augmented by adding point source with 2 times base flow.
6. Scenario 5 (S5) – Flow augmented by adding point source with 4 times base flow.

Daily mean DO concentrations were then calculated by averaging over the 24-hour period for each day, for 92 days. Plots of daily mean DO for each scenario during the 1993, 1994 and 1995 study periods are shown in figures 21, 22 and 23 respectively.

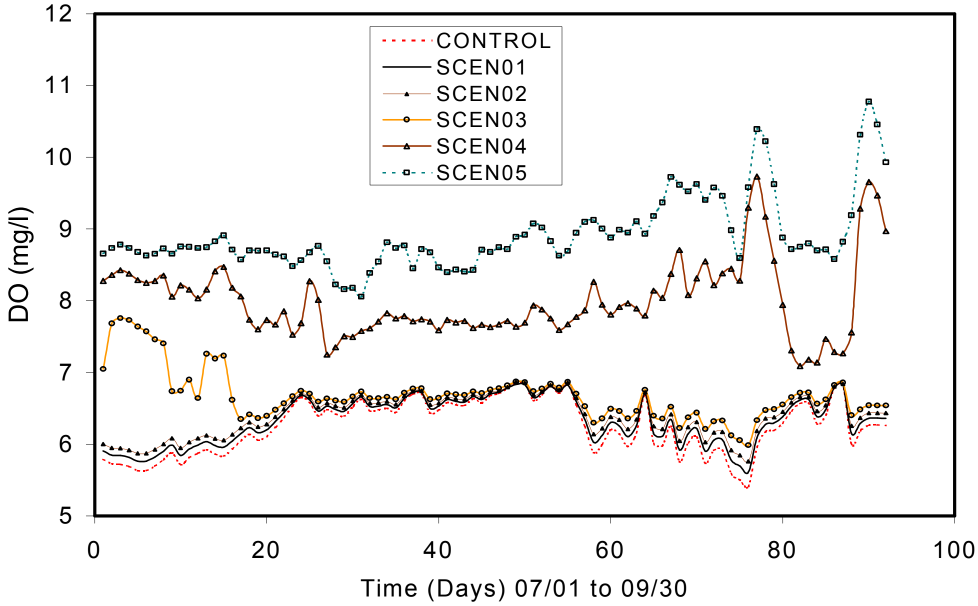


Figure 21. Daily Mean DO During 1993 Scenario Analysis

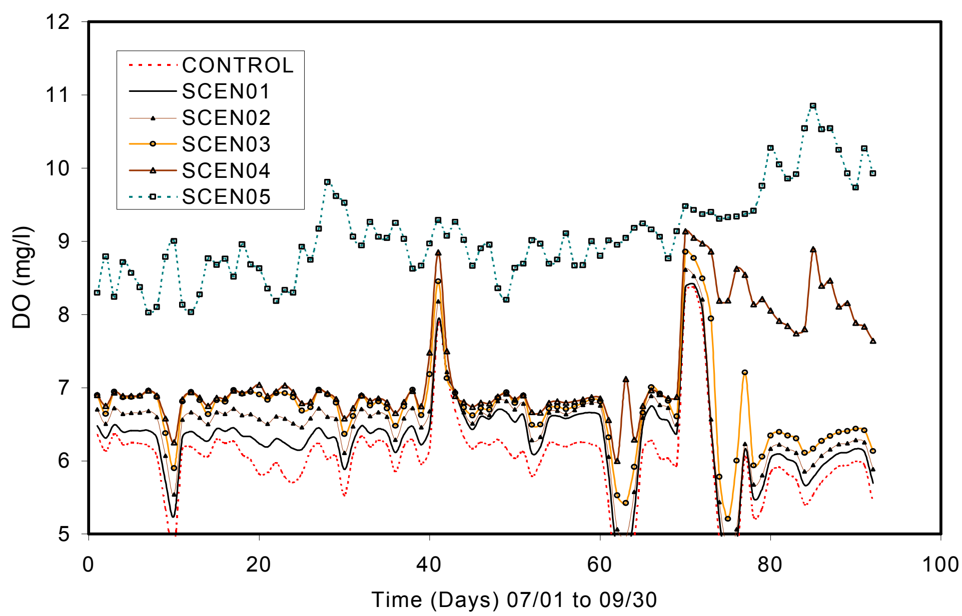


Figure 22. Daily Mean DO During 1994 Scenario Analysis

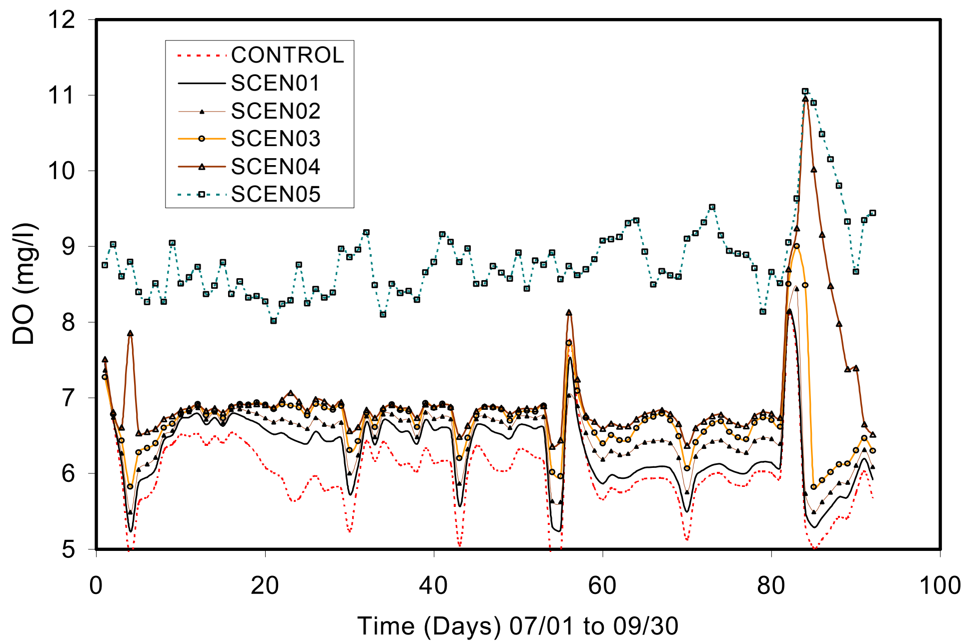


Figure 23. Daily Mean DO During 1995 Scenario Analysis

To evaluate the scenarios, statistical analysis was performed using the Statistical Analysis System (SAS) software. The sample population consisted of 92 data points for each scenario. Each data point represented the daily mean DO concentration in mg/l over the chosen three-month low-flow period for 1993, 1994 and 1995. Following methods were used for comparison of the datasets and means:

1. Kruskal-Wallis Chi-Square Approximation Test – Pre-test for differences.
2. Student-Neuman-Keuls (SNK) Test
3. Tukey's Studentized Range (HSD) Test

Leon Creek Riparian Zone Modeling

The urban development of San Antonio, Texas through 1987-1999 may have impacted the water quality of Leon Creek, one of the main tributaries in the San Antonio River Basin of Bexar County. By 1990, the population of the San Antonio River Basin had reached 1.3 million residents, and it is estimated that the population will increase to 3.3 million by 2050 (TNRCC, 1996). In 1998, an assessment by the Environmental Protection Agency (USEPA) found that there were three stream segments in the San Antonio River Basin that were not supporting water quality standards according to the Clean Water Act (CWA). One of these stream segments included Leon Creek (TNRCC, 1996).

The objective of the Clean Water Act, as stated in Section 101, is to "restore and maintain the chemical, physical, and biological integrity of the Nation's waters" (USEPA, 1996). The technical component of the Watershed Protection Approach is the Total Maximum Daily Load (TMDL), required under CWA Section 303(d), (1), and determines the maximum load of a pollutant that a water resource can absorb without violating a water quality standard. The TMDL process involves characterizing watershed conditions to determine the nature of the impairment and determine the viability of using restoration to meet water-body goals. (USEPA, 1996).

The San Antonio River Basin has two segments that require a TMDL from the EPA: Leon Creek and Salado Creek. A major initiative of the Clean Water Act is the Watershed Protection Approach and the Total Maximum Daily Load (TMDL) process. The Watershed Approach prioritizes stressors within a watershed using water quality data, biological monitoring, habitat suitability data, and information on land use and location of critical resources (USEPA, 1996). Leon Creek in Bexar County, Texas is an ecological corridor through an urban and agricultural environment with changing patterns of soil and vegetation.

The riparian zone of Leon Creek is of ecological importance in that it is the buffer zone between terrestrial and aquatic systems. Through the use of satellite imagery and thematic data, an assessment of riparian vegetation for the river margin of Leon Creek was mapped out in a Geographic Information System (GIS). The temporal and spatial patterns of riparian vegetation removal may provide understanding into the water quality problems of Leon Creek. The United States Agricultural Department (USDA) has found that riparian vegetative buffer zones may filter out up to 50% of nutrients and pesticides, 60% of pathogens, and 75% of sediments (USDA, 1999). Hopefully, such information can be used to provide data for watershed analysis and modeling, environmental assessment and restoration strategies.

The riparian vegetation surrounding Leon Creek needs to be evaluated both in urban and rural environments for possible effects upon water quality. Urban impacts upon riparian areas are difficult to assess, and the scale of urban impacts on riparian vegetation is unknown as well.

We examined the riparian areas surrounding Leon Creek to ascertain if this is a factor in decreased water quality. The evaluation of a riparian area involves the development of characterizing changes that occur over time within the landscape. The steps in any landscape examination should involve a systematic approach in defining the composition and structure but also determining the rate of change (Figure 24)(Jensen & Bourgeron, 1994). The following are key objectives in the use of GIS for assessing land cover changes and how these changes affect water quality:

1. Determine whether the intensity of vegetation change is greater in rural or urban areas between 1987 and 1999. An intense decrease in vegetation between 1987 and 1999 may result in increased water quality problems.
2. Determine percent area of dense vegetation in both urban and rural areas between 1987 and 1999. A decreased percent area means less vegetative cover, which may result in decreased water quality.
3. Determine patterns of vegetation patches in both rural and urban areas to ascertain if fragmentation is occurring more in an urbanized area or agricultural area between 1987 and 1999. Increased fragmentation results in less connectivity, which may result in decreased water quality.

Methodology

Leon Creek's riparian vegetation was evaluated through Landsat TM data, where different methods were compared and integrated (Figure 24). Through the use of an Image Difference Calculation, areas of 20% or more change (increase or decrease) in pixel value from 1987 to 1999 were determined. These areas show the most intense loss or gain of dense vegetation and whether these intense changes are occurring in rural or urban zones. An unsupervised and supervised classification on 1987 Landsat 5 and 1999 Landsat 7 images were performed of major land use/land cover types to determine areas of dense vegetation in the riparian areas. The class of dense vegetation was analyzed for their percent area in defined riparian zones surrounding Leon Creek between 1987 and 1999 in both rural and urban zones. Spatial landscape indices were used to compare 1987-99 changes in the riparian vegetation area. A land cover theme was created from the supervised classification images where only the dense vegetation theme was analyzed for fragmentation patterns from landscape metrics.

Study Site

The Leon Creek Watershed is approximately 233 square miles located on the southwest edge of San Antonio (Figure 25). Leon Creek itself is approximately 36 miles in length (Figures 26 and 27). There are two dimensions to the Leon Riparian Site. There is the urban/rural division as well as buffer zones designated at specific distances from Leon Creek. The Leon Watershed will be studied in its totality as well as divided into urban and rural areas. This urban area theme was designated by high population and housing density. The "densely settled surrounding territory" adjacent to place consists of the following: territory made up of one or more contiguous blocks having a population density of at least 2,500 people per square mile (U.S. Census Bureau, 1990).

The water quality varies along Leon Creek with Segment 1906 of Leon Creek as the major concern for the TMDL. Segment 1906 of Leon Creek (Figure 28), bacteria levels exceed established standards to assure safe contact recreation (San Antonio River Authority, 1998). The lower section of Leon Creek of Segment 1906 has dissolved cadmium concentrations, which exceed standards established by the EPA to protect aquatic life (San Antonio River Authority, 1998). Excess nutrient loads from nearby rural land are also considered a non-point pollution problem in Leon Creek (Table 15).

Precipitation rates in Bexar County can vary drastically between severe droughts to floods. The average precipitation rates for the area is 29 inches per year (USGS, 2000). In 1998, there were severe flooding conditions in the San Antonio River Basin, including Leon Creek. There are only 20 days a year of freezing temperatures, with warm to mild temperatures from March to Late November (USGS, 2000). Both Landsat images were taken in September, a typically warm month for Bexar County. There were no cold fronts during this time period to affect vegetative growth.

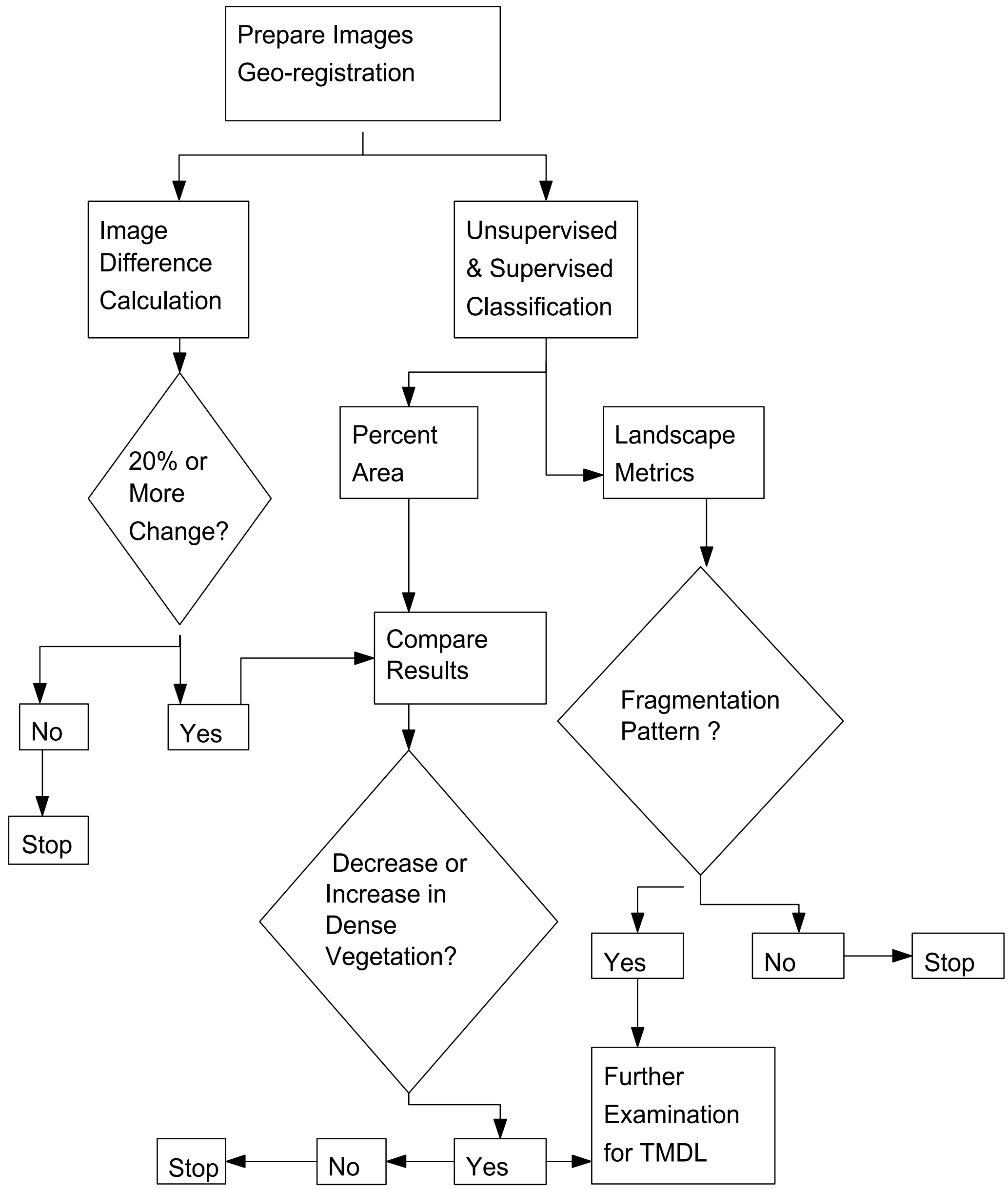


Figure 24. Flow Chart of GIS Assessment of Riparian Vegetation. Includes assessment on intensity of change, percent area, and landscape indices to evaluate whether or not riparian vegetation plays a role in decreased water quality in Leon Creek.

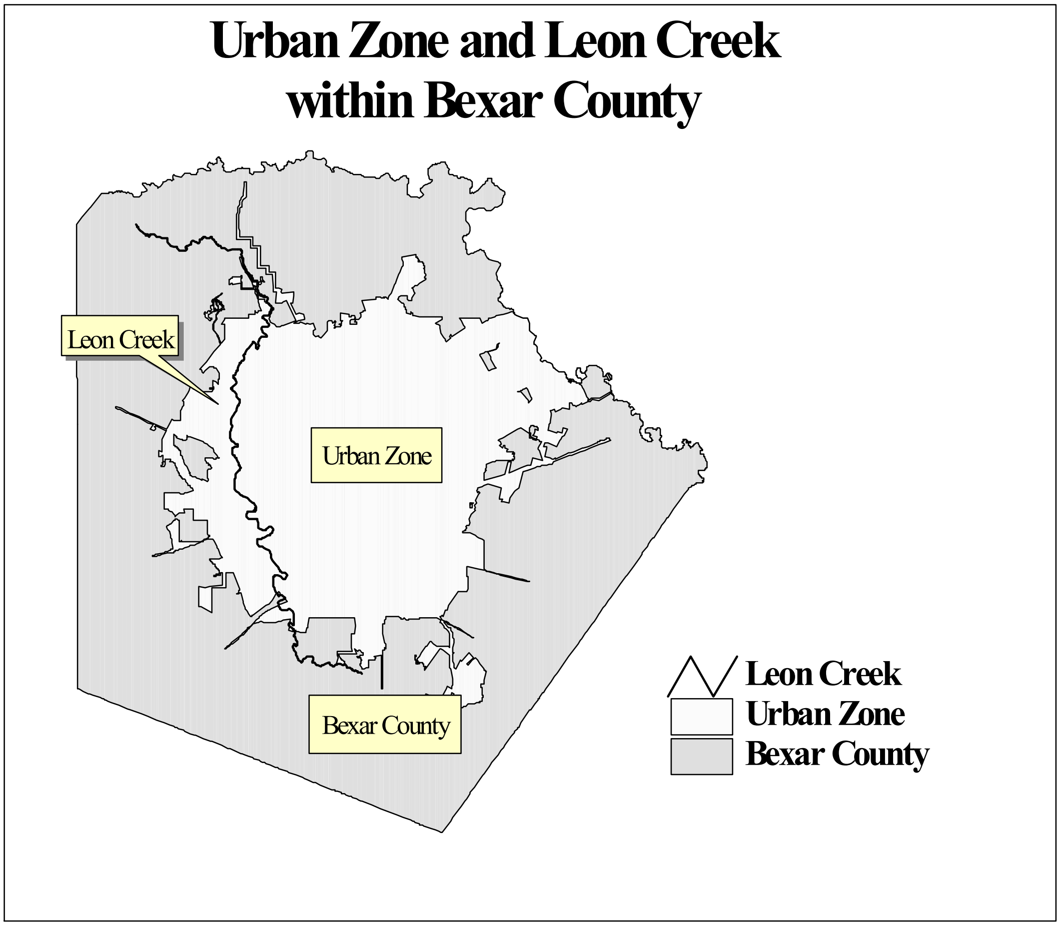


Figure 25. Map of Urban Zone in Leon Watershed, Bexar County, Texas. Depicts position of Leon Creek and Leon Watershed in relation to the urban area of San Antonio.

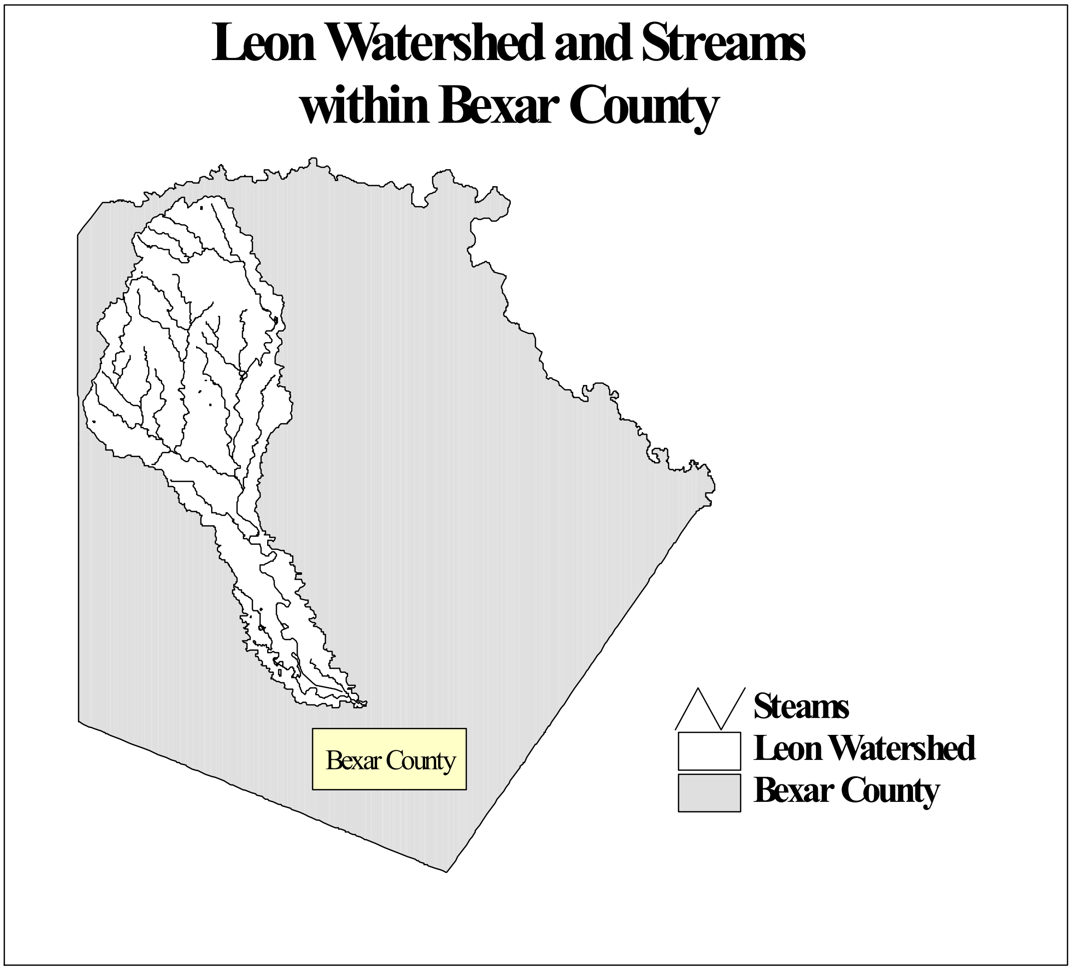


Figure 26. Map of Streams in Leon Watershed, Bexar County, Texas. Depicts position of Leon Watershed and waterways in relation to Bexar County.

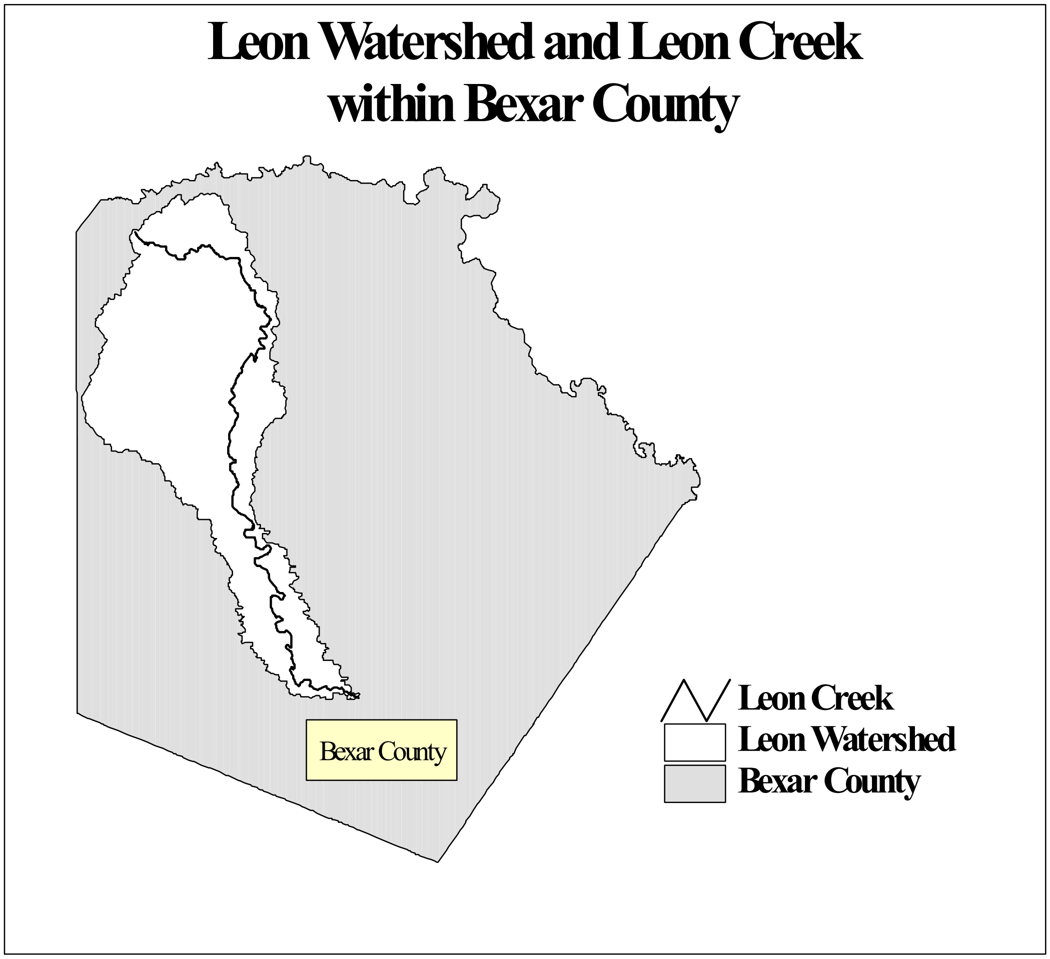


Figure 27. Map of Leon Creek in Leon Watershed, Bexar County, Texas. Depicts position of Leon Creek and Watershed in relation to Bexar County.

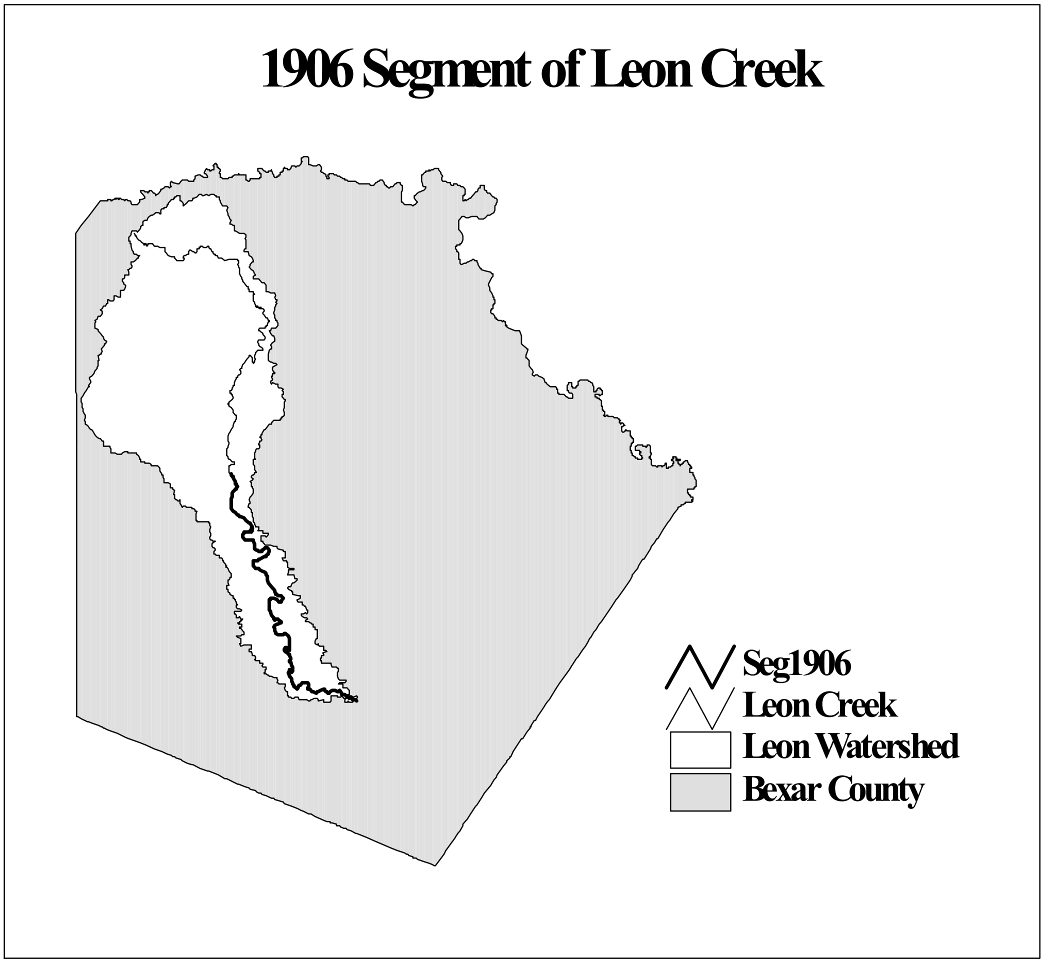


Figure 28. Map of Segment 1906 of Leon Creek. Depicts position of Leon Creek and Watershed in relation to Bexar County.

Table 15. Texas Water 2000 Clean Water Section 303(d) List, April 28, 2000. Segment Number: Classified segment number assigned to a water body or portion of a water body in the Texas Surface Water Quality Standards (TNRCC, 2000)

Segment Number	Water Body Name	Priority*	PS*	NPS*	Parameters of Concern
1906	Lower Leon Creek	M	Y	Y	Pathogens, Low dissolved oxygen

*Priority: Priority rank of the water body for TMDL development. With "H=high", "M=medium", and "L=low" for impaired water bodies.

*PS: "Y" indicates pollution mainly comes from a point source and "N" indicates it does not come from a point source.

*NPS: "Y" indicates pollution mainly comes from a non-point source and "N" indicates it does not come from a non-point source.

Buffer zones around Leon Creek at 0.5 mi., 0.25 mi., 0.125 mi., and 100 ft. were examined for their riparian vegetation patterns as well (Table 16, Figures 29 and 30). The 0.125 mi. and 100 ft. zones are the most critical areas of riparian vegetation, while the 0.25 mi. to 0.5 mi. may help to detect trends beyond the riparian vegetation zone (USDA, 1999). "The appropriate size of a riparian reserve for a particular site depends on the physical and biological functions of the riparian zone at the site." (USDA, 1994).

Table 16. Area in Square Hectares for all Buffer Zones

Buffer Zones	Watershed Area (ha)	Urban Zone (ha)	Rural Zone (ha)
0.5 mi.	11837.5	6862.3	4975.2
0.25 mi.	6487.7	3662.6	2825.1
0.125 mi.	3338.5	1690.6	1647.8
100 ft.	516.7	260.1	256.6

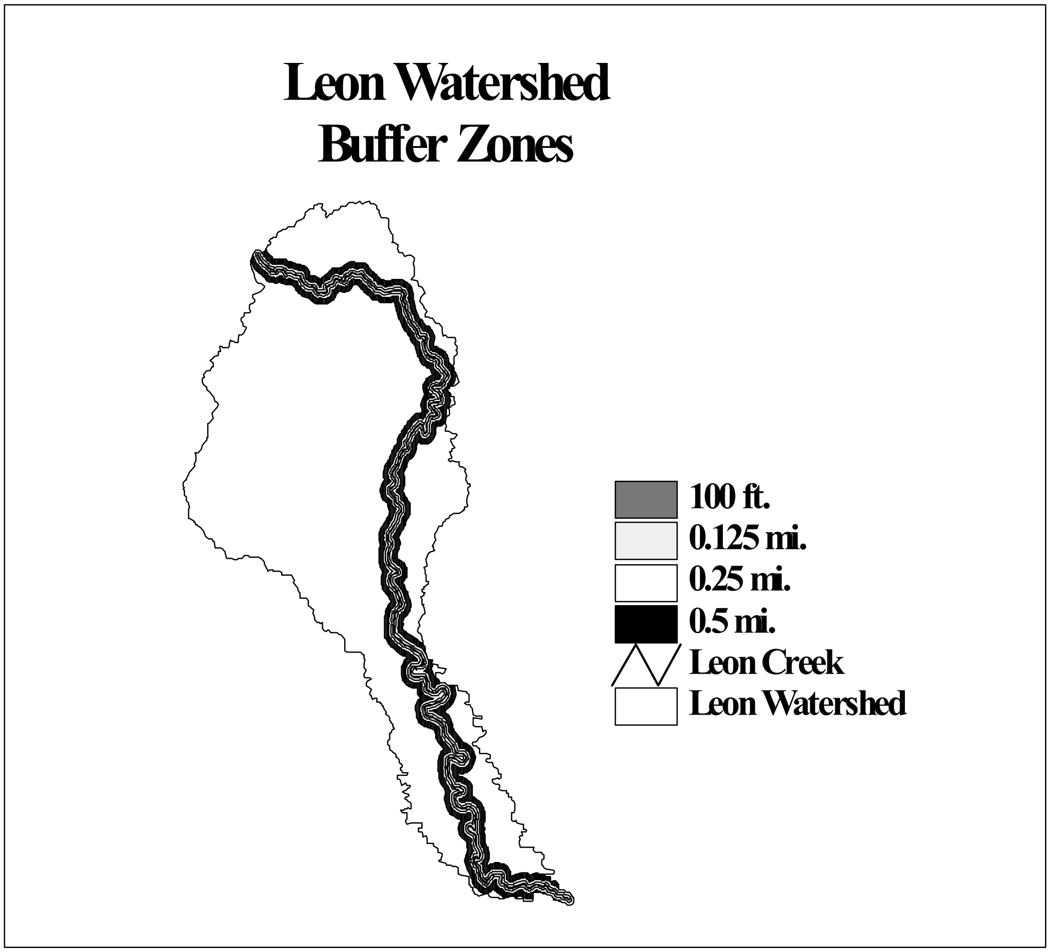


Figure 29. Leon Creek Study Site Buffer Zones. Depicts total watershed area with the buffer zones at 0.5 mi., 0.25 mi., 0.125 mi., and 100 ft.

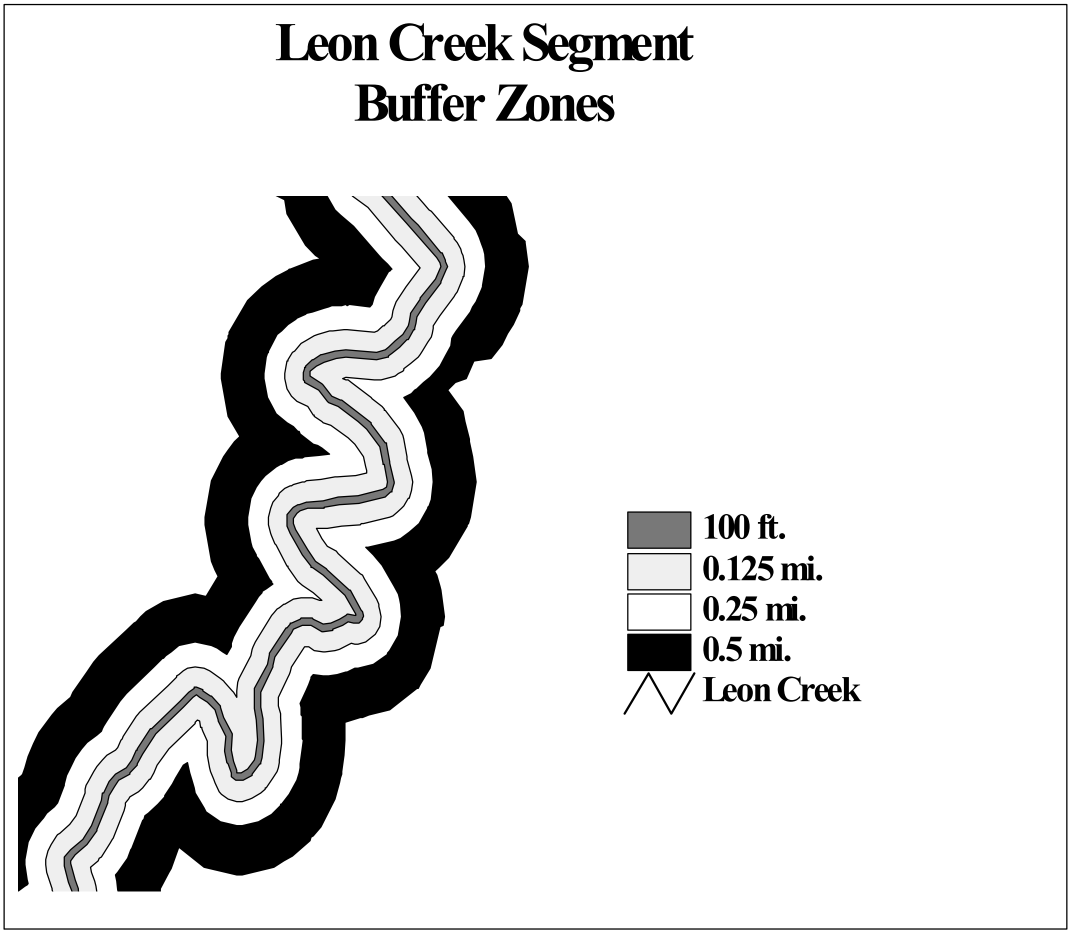


Figure 30. Leon Creek Segment of Study Site Buffer Zones. Depicts stream segment with all the buffer zones to be analyzed at 0.5 mi., 0.25 mi., 0.125 mi., and 100 ft.

The dominant soils surrounding Leon Creek are Lewisville Silty Clay, Tarrant Association, and Patrick Soils (Figure 31). A large amount of the area around Leon Creek is also composed of pits and quarries. Lewisville soils are found mainly in Texas near streams in both the Blackland Prairies and the Grand Prairie. The dominant Lewisville Series soils are described as well-drained, semi-permeable soils where the runoff is slow to medium. The taxonomic class is a Thermic Udic Calciustolls with a texture that is a silty-clay and a coloring, which is gray-brown (USDA, 1995).

The dominant vegetation in Bexar County includes perennial grasses and forbs, as well as woody vegetation. The dominant woody plants in Bexar County include vine mesquite (*Panicum obtusum*), post oak (*Quercus stellata*), and live oak (*Quercus virginiana*). Dominat forbs include trailing wildbean (*Strophostyles helvola*), while dominant grasses include the following: big bluestem (*Andropogon gerardii*), sideoats grama (*Bouteloua curtipendula*), buffalograss (*Buchloe dactyloides*), Canada wildrye(*Elymus Canadensis*), Little Bluestem (*Schizachyrium scoparium*), Indiangrass (*Sorghastrum nutans*), and tall dropseed (*Sporobolus asper*) (USDA, 1995). The type of vegetation typically found on Lewisville series soils includes various grasses, sugar hackberry (*Celtis laevigata*), honey mesquite (*Prosopis glandulosa var. glandulosa*), and winged elm (*Ulmus alata*) (USDA, 1995).

Bexar County Soils Data from the United States Department of Agriculture (USDA) has information on the suitability of the soil to produce a specific wildlife habitat. This suitability index may give insight into the type of vegetative habitats common in Bexar County, as well as the resiliency of such habitats to disturbances. The suitability of a habitat is within a range of "Good", "Fair", "Poor", or "Very Poor". For Bexar County, the suitability of the soil for herbaceous plants, shrubs, and rangeland habitats is described as "Fair", while the suitability for wetlands, wetland plants, and shallow water is described as being "Poor" or "Very Poor" (Table 17) (USDA, 1995). Resiliency for herbaceous plants, shrubs, and rangeland may be only moderate since soil suitability habitats are described as only "Fair", however, riparian habitats may be in an advantageous position because of increased moisture levels.

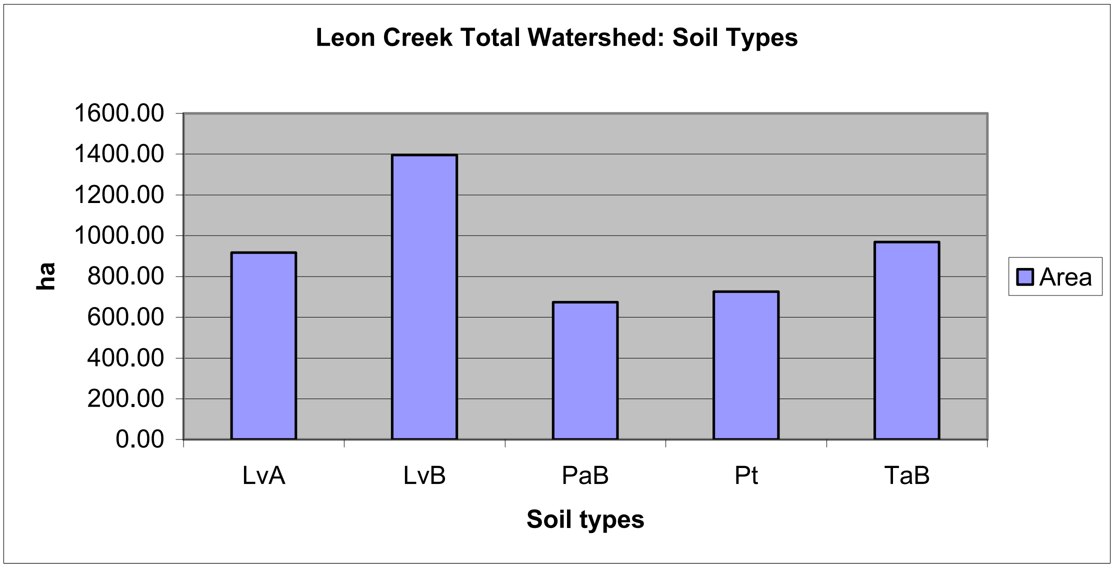


Figure 31. Soil Types within Leon Riparian Zone. Area (in ha) in Leon Riparian Zone. Soil Type Names: LvA (Lewisville Silty Clay, 0 to 1 percent slopes), LvB (Lewisville Silty Clay, 0 to 3 percent slopes), PaB (Patrick Soils, 1 to 3 percent slopes), Pt (Pits and Quarries), TaB (Tarrant Association, Gently Undulating).

Data Compilation

Satellite images from Landsat 5 and 7 are the primary data sources to be used in examining the integrity of the riparian vegetation along Leon Creek.

The following satellite images were obtained from the USGS-EROS Data Center, which was used in the analysis of land use and land cover patterns:

1. Landsat 5 image of Bexar County (path 27, row 40) taken on September 25th, 1987.
2. Landsat 7 image of Bexar County (path 27, row 40) taken on September 18th, 1999.

Landsat 5 scenes were obtained from the USGS-EROS Data Center. The quality for path 27, row 39 and row 40 was described as "very good" to "good" (USGS, 1987). The cloud-cover percent for row 39 and 40 was zero. The spectral range for each of the Landsat 5 bands is described in Table 18. The Landsat 7 satellite images were obtained from the USGS-EROS Data Center. The scenes were ordered as an "L1R" product, which means the scenes have been radiometrically corrected but not geometrically corrected. The Landsat 7 image consists of 8 bands with an Enhanced Thematic Mapper (ETM) Plus sensor (Table 19). This ETM+ sensor is able to filter and collect radiation from the Earth in a 183-km swath (USGS, 1999).

Table 17. Soil Suitability Index for Wildlife Elements. Shows habitat types including wild grain, grassland, herbaceous plants, shrub land, wetland plants, shallow water, open land, wetland, and rangeland. Suitability Index ranges from "Good" to "Very Poor".

SSURGO National Cooperative Soil Survey USDA	
*suitability of soil to produce wildlife element	
Habitat Type	**Suitability Index**
Wild grain	POOR
Grassland	POOR
Herbaceous plants	FAIR
Shrub land	FAIR
Wetland plants	POOR
Shallow water	VERY POOR
Open land	POOR
Wetland	VERY POOR
Rangeland	FAIR

Table 18. Spectral and Spatial Information on Landsat 5 TM. The spatial resolution of each band signifies the land area that each pixel covers in that particular band. A 30m spatial resolution means each pixel represents a 30X30m area.

Band	Color	Wavelength (um)	Spatial Resolution (m)
1	Blue	0.45-0.52	30m
2	Green	0.52-0.60	30m
3	Red	0.63-0.69	30m
4	Near infrared	0.76-0.90	30m
5	Mid-infrared	1.55-1.75	30m
6	Thermal infrared	10.40-12.50	120m
7	Mid-infrared	2.08-2.35	30m

Table 19. Spectral and Spatial Information on Landsat 7 ETM. The spatial resolution of each band signifies the land area that each pixel covers in that particular band. A 30m spatial resolution means each pixel represents a 30X30m area.

Band	Color	Wavelength (um)	Spatial Resolution (m)
1	Blue	0.45-0.52	30m
2	Green	0.52-0.60	30m
3	Red	0.63-0.69	30m
4	Near infrared	0.76-0.90	30m
5	Mid-infrared	1.55-1.75	30m
6	Thermal infrared	10.40-12.50	60m
7	Mid-infrared	2.08-2.35	30m
8	Panchromatic	0.520-0.900	15m

Supplementary spatial data was also be used to examine past land use/land cover in Bexar County as well as basic hydrologic and soil data in the study area. This information was useful in examining urban patterns as well as the remaining ecological resources. This supplementary spatial data has been obtained from a number of different agencies, which include the Census Bureau, the Texas Natural Resource Information System, and the United States Department of Agriculture.

1995 TIGER Files from the Census Bureau were obtained from the ESRI ArcData Online service, or otherwise known as the ADOL Tiger data sets. These TIGER Files are in Geographic Projection with a datum of NAD 27 at a 1:100,000 scale, with map units in decimal degrees, which were then transformed to Universal Transverse Mercator Projection with a datum of NAD 83, with map units in meters. The following TIGER files were used as informational data for Bexar County, Texas to assist with the analysis of the area:

- County (1995 Boundaries)
- Land and Water Outlines
- Landmark Areas
- Urban Areas
- Water Bodies
- Lines-streams
- Lines-streets
- Points-landmarks
- 1990 Census Tract Demographics

Data layers from the Texas Natural Resource Information System (TNRIS) data site were used both for the hydrology information and as a reference for the supervised classification process. 1976 Anderson Land Use and Land Cover themes were provided by the USGS for Bexar County in Universal Transverse Mercator Projection with a datum of NAD 83, with map units in meters. The Anderson Classification System showed Level II land use/cover themes including the following: agriculture, urban, forestland, barrenland, and reservoirs. The following data sets of Bexar County were included:

- Land Use/Land Cover Data (based on the Anderson land use codes)

- Streams
- Watersheds

The Soil Survey Geographic (SSURGO) Database for Bexar County, Texas in 1995 was obtained from the U.S. Department of Agriculture. It is the most detailed level of geographic soil information from the National Cooperative Soil Survey. The map unit delineations are polygons, which are dominated by a single soil type, non-soil components, or a mixture of soil types (USDA, 1995). SSURGO soil data is in Universal Transverse Mercator Projection with a datum of NAD 83, with map units in meters. The map data was in 7.5-minute quadrangles that were merged together for all of Bexar County and then clipped to analyze the data for the Leon Watershed. The following attribute tables were used:

- County Soil Survey
- Soil Suitability for Habitat Type
- Plant Species Survey for Soil Survey

Data Preparation

The images were subset to Bexar County, which comprised the northeast corner of the image. The map coordinates of San Antonio were selected, and ERDAS' Imagine was able to select out a smaller image from the original. The coordinates selected for the Leon Watershed were as follows:

- Northwest Coordinate: -98.80 longitude, 29.73 latitude
- Northeast Coordinate: -98.49 longitude, 29.73 latitude
- Southwest Coordinate: -98.80 longitude, 29.26 latitude
- Southeast Coordinate: -98.49 longitude, 29.26 latitude

The 1987 Landsat image was then put through the process of rectification, since the image had not been geometrically corrected. The image was altered from having no projection to Universal Transverse Mercator (UTM), NAD 83 projection. The use of the TIGER street data layer was used for this process, since the street theme was an easily identifiable feature on the image. ArcView's Image Analyst extension was used to align the image to the UTM projection. Five control point pairs were selected in both the image and feature themes. A first order transformation was used to calculate the transformation of the image to the map coordinate system (Environmental Systems Research Institute, 1998). The projected UTM image had a total Root Mean Square error of 0.90. The Root Mean Square is the distance between the image input point and the feature point. A resampling takes place, so the lower the RMS error, the more accurate the resampling. A RMS error that is below 1.0 is considered good and an RMS error close to 5.0 is considered poor (Environmental Systems Research Institute, 1998).

Both images were then set in the same projection so they could then be geo-referenced through the use of Ground Control Points (GCPs). Both the 1987 and 1999 images were viewed together and saved in the same projection, which was the Universal Transverse Mercator, NAD 83. The 1987 image was selected as the input image, and the 1999 image was considered the reference image. In the input image viewer, the Geometric Correction menu was selected and the Source Viewer Dialog Box was used to set the type of geometric model to be used, which was a polynomial geometric model with 2 orders. The projection type was again set to the Universal Transverse Mercator, with a datum of NAD 83. GCPs were then placed in the Input Image (1987 image) and the Reference Image (1999 image). The output image for 1987 was then a geo-referenced image in a Universal Transverse Mercator. The process was repeated for the 1999 image.

Through the use of ERDAS Imagine these images were co-registered through the operation GCPs. This registration process changes the spatial structure of the input image and resampling alters the spectral content. By registering one image to another referenced image (using a reference system such as Lat/Long or UTM), change analysis over time can better be performed as well as altering the resolution of an image.

Image Difference Calculation

To help detect the intensity of change, an Image Difference Calculation was used for the Landsat 1987 and 1999 images. The two satellite images for 1987 and 1999 were input for an Image Difference Calculation in ArcView's Image Analyst to detect changes in pixel values. This type of Image Difference Calculation can only be performed upon continuous data, not thematic data.

The objectives of the Image Difference Calculation are outlined below:

- Perform the Image Difference Calculation in ArcView's Image Analyst.
- Determine percent value to be used in detecting intensity of change. Create a theme change image to determine the percent value that best indicates land use/land cover change.
- From the Image Difference function, find the percent area going through a 20% or more change (increase or decrease) in the total watershed, urban and rural zones.
- Compare results from both the Image Calculation and the Percent Area from the Supervised Classification.

Bands 4,3, and 2 were selected from both images to be included in the Image Difference Calculation to specifically examine vegetation changes between 1987 through 1999. Band 4 (0.76-0.90 wavelength) is the near-infrared band, which displays vegetative biomass. Band 3 (0.63-0.69 wavelength) is in the visible red, which displays chlorophyll absorption or vegetative distinction. Band 2 (0.52-0.60 wavelength) is within the visible green wavelength, which emphasizes plant vigor or reflectance peak for plants (Schowengerdt, 1997).

Differences in pixel value can also be highlighted with the resultant image difference according to the intensity of change, as a percent or value. Highlighted percent values were examined from 10% to 90% to determine which percent value would be the most useful in determining the intensity of vegetation change in rural and urban areas. The optimal percent value would be the one that was sensitive enough to increases or decreases in pixel value to the point of land use/land cover change by 1999, yet was not too sensitive as to highlight inconsequential changes. For example, at the lowest percent value of 10% slight pixel changes were included, which would not be helpful as a true representation of vegetation change. At the other extreme a 50% value was not sensitive enough to include increases in vegetation.

A theme change image was used to find areas that had either been changed from dense vegetation in 1987 or changed into dense vegetation by 1999 from another land use/land cover class. Using the supervised classification images, a theme change image was created in ArcView's Image Analyst. This resultant image showed the class that was identified in 1987 and what it changed into in 1999. Image Analyst performed a change calculation for thematic data, which showed the before and after themes as attributes in the resultant image. This assisted in determining which percent value to use in the Image Difference Calculation. The percent value image that matched best with the theme change image was the percent value considered most useful. A highlighted percent value of 20% was found to be the most sensitive to increases and decreases in pixel value, but not so sensitive that it included minimal change.

Unsupervised Classification

An unsupervised classification was performed on the 1987 and 1999 Landsat images to assist in the supervised classification of both images. "An unsupervised classification is performed to assure separation clusters for known categories and check for mistakes and errors" (Richards, 1986). ERDAS Imagine was used to perform an Iterative Self-Organizing Data Analysis (ISODATA) Technique unsupervised land use and land cover classification. By using ISODATA, a classification occurs repeatedly, and organizes the clusters of similar data. This method uses the minimum spectral distance to establish clusters, and repeatedly re-establishes cluster groups determined by the spectral signature of the data (Schowengerdt, 1997). A total of fifty classes were created for both 1987 and 1999 images.

Anderson Land Use/Land Cover data layer themes provided by the USGS in 1976 were used to provide assistance with the unsupervised classification process of the 1987 and 1999 Landsat images. The northern region of the Leon Watershed is depicted as a forested area with evergreen trees. It is primarily hill country terrain that is sparsely populated and used for ranching. The central urban area of San Antonio, Texas is densely populated with residential, commercial, and industrial activities. There is intense suburban/urban development occurring on the western edge of San Antonio. Leon Creek also passes through Kelly Air Force Base and Lackland AFB and Training Annex. The urban zone of Leon Creek is described as having little riparian vegetation, composed mainly of deciduous forest patches. The southern end of the watershed is rural with dense riparian vegetation surrounded by farms and ranches (San Antonio River Authority, 1998).

The images were viewed and the grayscale maps were then transformed into a color-coded map. Each grayscale level, which identified a class, was converted into a classification color. Each classification color represents a land use type or land cover type. These color classifications may result in classification misrepresentation, since dark colored classes may appear similar. For example, water may be in the same dark class as a densely vegetated area. Therefore, the true color and near infrared composites were referred to in the final analysis process. Six main classes were narrowed downed and assigned a color-code. These six classes were as follows:

- Heavy urban-industrial, commercial, airports, and quarry areas
- Light urban-roads, streets, residential areas
- Dense vegetation-dense areas of trees and shrubs
- Light vegetation-areas sparsely populated with trees, shrubs, or herbaceous material
- Water-lakes, ponds, reservoirs, tanks, streams
- Bare soil

Supervised Classification

A supervised classification will be performed upon both the Landsat 5 and 7 images to analyze differences in riparian vegetation patterns along Leon Creek between 1987 and 1999. In a supervised classification, the analyst controls the land use/land cover types in a satellite image, whereas unsupervised classification is computer-automated. With supervised classification, clustered pixels are selected that represent land cover areas defined by other GIS sources such as aerial photos, ground truth data, or land use/land cover maps (ERDAS Imagine, 1994). By identifying pixel clusters as a specific land use/land cover type, the algorithms are created find other pixel groups with the same spectral characteristics as those identified by the analyst. Anderson Land Use/Land Cover data layer themes provided by the USGS in 1976 and the results from the unsupervised classification were used to provide assistance with the supervised classification process of the 1987 and 1999 Landsat images.

There are parametric signatures, which are statistical, and non-parametric, which is based on discrete objects in a feature space image. In this project, non-parametric supervised training was used to define specific land use/land cover classes. A parallelepiped classifier, a type of decision rule for non-parametric signatures, determines a range of pixel values for each identified class. In a parallelepiped decision rule, the values of the selected pixels are compared to upper and lower limits, which can be based on any of the following methods (ERDAS Imagine, 1994):

- Minimum and maximum data file values of each band in the signature
- The mean of each band, plus and minus the standard deviations
- Any specified limits based on personal knowledge of the data

For this project, the mean of each band, plus or minus the standard deviations was practiced. Based on similarities of high or low pixel numbers for each class, a pixel is placed in a designated category (Schowengerdt, 1997). Digital Number (DN) values for each pixel are quantitative and continuous, which means they relate to one another. These relationships are represented by their contrast or color value. A pixel that is black in appearance has a high value in black, when the DN value is assigned as such.

With the use of a seed tool in ERDAS Imagine's program, areas were identified that have similar spectral characteristics in the images. Bands 4, 3, and 2 were used to create a pixel value range for specific land use and land cover classes. Training samples were defined for each land use and land cover type by selecting a set of pixels as a distinct pattern or class. The seed tool was set for a radius of five pixels to define the extent of a pixel value range for my defined classes. When more than one pixel is chosen, the mean of the sample is calculated from the selected pixel set. In this method, the training sample was determined from a set of pixels with similar spectral characteristics. The classification scheme was defined in the following manner:

- Heavy urban-industrial, commercial, airports, and quarry areas
- Light urban-roads, streets, residential areas
- Dense vegetation-high density of trees and shrubs
- Light vegetation-low density of trees, shrubs, or herbaceous material
- Water-lakes, ponds, reservoirs, tanks, streams
- Fertilized vegetation-active cropland, golf courses, lawns
- Bare soil

The spectral reflectance value of a surface was used to identify urban land cover (Colwell & Sadowski, 1993). In color infrared images, urban surfaces, such as concrete and asphalt, are characteristically blue to white in color. Vegetation appears light to dark red, and soil appears yellow to dark brown. Individual buildings can be identified if they are large and contrast with the background reflectance color. The generalized urban boundary may contain undeveloped land that is completely surrounded by developed areas. Parks, golf courses and other areas whose natural vegetation has significantly altered is included in the urban category (Bell et al., 1995).

Landscape Indices

The land use patterns in Bexar County will be interrelated with the riparian vegetation patterns and fluvial processes along Leon Creek. To study the pattern and process of the landscape in these river margins, a series of area, density, and size metrics will be used to provide a statistical analysis. "The characteristics of the reflected spectral radiances and their geo-statistical properties may be potential indicators of differences in vegetation conditions," (Benkelman et al., 1992). ArcView Patch Analyst 2.0 will be the spatial analysis program used to develop the necessary metrics.

The objectives of this analysis are outlined below:

- Analyze the riparian vegetation change along Leon Creek 1987-1999 using landscape indices.
- Examine the spatial location of vegetation change in defined riparian zones at ½ mi, ¼ mi., 1/8 mi, and 100 ft.
- Analyze whether the vegetation change or fragmentation is more intense in an urban or rural zone.

Buffer zones were designated around Leon Creek at 0.5 mi., 0.25 mi., 0.125 mi., and 100 ft. A land cover theme was created from the supervised classification images where only the dense vegetation theme was selected inside each of the buffer zones. To analyze whether there was more fragmentation in an urban area as opposed to the fragmentation in the total watershed area, the U.S. Census Bureau Urban Zone theme was used for more in-depth analysis.

To examine fragmentation on the rural, urban and watershed level, a number of landscape indices were used to measure patch size, shape, and core areas of the vegetation patches. A combination of these indices can be more valuable than examining them separately. When examined separately, the indices can be misleading whereas if they are viewed together the indices can reinforce or question probable landscape patterns. Also, landscape metrics can only be fully understood "when the limitations of each measure are fully understood and the range of attainable values is known," (Hargis et al, 1997).

To analyze the size of patches Number of Patches (N), Mean Patch Size (MPS), and Patch Size Coefficient of Variation (PSCOV) were the landscape indices used together to ascertain how the landscape pattern change between 1987 and 1999. For the shape indices, Mean Shape Index (MSI), Area Weighted Mean Shape Index (AWMSI), were used as indicators of patch shapes. Total Core Area (TCA) was used in conjunction with the shape indices to provide further insight into shape complexity.

Justification for Deviation From Plan

Social Science

There were two changes made to the original research plan for the social science project activities. First, Phase III was not conducted, thus the council groups ended their work in December 2000 with the production of the action plans. The primary reason for this change was that the stakeholder groups decided that further meetings beyond the action plan were not desired. There were also budgetary constraints that prevented the research team from continuing to provide financial support beyond the December 2000 meeting. The research team did offer to continue to meet with the two council groups to help facilitate the meetings. Inspection of the action plans described in the Research Findings section reveals, however, that some of the recommended actions were quite specific, suggesting that participants had discussed implementation of the goals listed in their action plans. Second, due to the change in plan for Phase 3, we conducted one post-survey of the council groups in December 2000, instead of two waves of post-surveys as proposed in the original plan. We did, however, administer a third survey to council members on issues of concern in the watershed in February 2000, and then used this information to help facilitate the work of the council groups in subsequent meetings.

Ecological Science

Fish were obtained from a commercial supplier rather than from the subject streams, because of uncertainty about the sufficiency of stream bluegill populations to support the required levels of sampling without replacement. Fish were acclimated in cages, instead of mesocosms, because written permissions from land-owners/managers from all four sites could not be completed in time for the Summer 1999 field work. Because fish were held in cages instead of in streamside mesocosms, it was impractical to apply artificial loading treatments involving nutrients or metals. In fact, while metals were a suspected contaminant during the preparation of this proposal, subsequent investigation has suggested that bacteria rather than metals are the concern in these streams. Tangential funding was acquired to support investigation of fecal coliform contamination of these streams.

ENDNOTES

[1] The dependent variable we use to test this hypothesis is respondent's self-report of willingness to participate in a collaborative stakeholder process. As can be seen from the question wording, included above, respondents were not told in detail what exactly would be involved in the collaborative stakeholder process. Instead, they were asked whether they would be willing to participate in discussions "…on the topic of clean water…". There is no doubt that had survey interviewers taken the time to explain what exactly is entailed in a collaborative learning process, some respondents might have changed their answer to this survey question. This was, however, beyond the scope of our phone survey.

[2] Not all researchers agree with this direction. Some believe that more trusting citizens withdraw from the arena, feeling they are already represented, while it is the more mistrusting ones who feel they must participate to protect their interests.

[3] Entries in bold text in the second column indicate that the estimated effect of the variable in the first column of that row was significant at conventionally accepted levels for statistical hypothesis testing in the social sciences (a two-tailed p-value less than or equal to .05). Entries in *italics* in the second column indicate that the estimated effect of the first column variable of that row was close, but not quite significant at conventionally accepted levels (a two-tailed p-value between .05 and .10). An asterisk symbol ("*") entry in the second column indicates that the estimated effect for the first column variable was not significant at conventionally accepted levels (a two-tailed p-value of .10 or greater).

[4] This instrument has been used extensively in past research on environmental concern and is presumed to assess three dimensions: the desire to protect the balance of nature, the belief that there should be limits to growth, and that humans should be regarded as part of rather than ruling over nature (Scott & Willits, 1994). We used the original four-response version of this scale (see Dunlap & Van Liere, 1978) with respondents being asked if they strongly agree, agree, disagree, or strongly disagree with each of the 12 statements (see Table 4 for a listing of the 12 items in the NEP scale).

[5] Note the disparities in sample size in the stakeholder groups relative to the phone survey subsamples. All individuals in each group who attended at least one Council meeting were asked to complete a survey. The combined response rate for the stakeholder groups was *90 %*. However, because the marked differences in sample size preclude reliable tests of statistical significance, we restrict our descriptive analysis to the largest differences (in percentages) between the stakeholder council groups and the corresponding watershed subsamples drawn from Bexar County.)

[6] These educational differences between the watershed subsamples and the stakeholder councils may provide an explanation for the large differences between these groups in perceived water quality evident in Table 5. Thus, college and/or graduate education may be positively correlated with perceived severity of water quality problems in the watershed, which in turn, may lead these individuals to participate in the councils.

[7] We report descriptive data for responses to all 12 questions in Table 4 because the NEP scale has been shown to be multidimensional (Scott & Willits, 1994) and thus, collapsing the items into a single scale score may be misleading. We believe this more detailed presentation is preferable at this stage of our understanding of representation issues.

[8] parts per million

EVALUATION OF STAKEHOLDER COLLABORATIVE LEARNING (CL) PROGRAM

We employed both quantitative and qualitative approaches to evaluate the effects of the CL program on members of the stakeholder councils. The quantitative approach used standard survey methodology to assess any changes that occurred between pretest (Wave 1) and posttest (Wave 2) administration. Unstructured interviews were conducted following the completion of the final CL meeting in December 2000 to provide a qualitative evaluation of the impact of the CL program on participants in the council groups. In addition, we discuss in this section the issues of representation and participation, based on the survey results presented in the Research Activities section. Conclusions are presented regarding the effectiveness of the CL intervention in terms of the 4 social science hypotheses and the success of the recruitment methods employed to obtain representative samples of citizens to participate in the two council groups.

QUANTITATIVE EVALUATION OF COLLABORATIVE LEARNING PROGRAM EFFECTIVENESS

Analysis Strategy

To test the hypotheses regarding the effects of the Collaborative Learning program, we included a series of 24 questions in the pretest survey (Wave 1) and the posttest survey (Wave 2). This set of questions had virtually identical wording in both surveys. Differences in question content across the two surveys were minor, usually involving changing item wording from future (Wave 1) to present tense (Wave 2). The questions were grouped into five categories: (1) knowledge/learning outcomes, (2) communication competence, (3) relationships outcomes, (4) water quality beliefs, and (5) personal efficacy beliefs. In addition, there was one question that asked for an overall evaluation of the council member's experience with the CL format used in the restoration council meetings. Appendix B presents these 24 questions on both Wave 1 and 2 surveys (wording changes indicated in brackets). Response scales for each item are also included.

The statistical analysis involved testing for changes between the pretest and posttest responses of stakeholders. In the Salado Creek Restoration Council (SCRC), we obtained 38 matched pairs of respondents who completed both Wave 1 and Wave 2 surveys. Because the response scales used for these survey questions can be considered to be either interval or ordinal scales, we conducted both dependent sample t-tests (interval scale) as well as Wilcoxon matched pairs signed-ranks tests (ordinal scale) on stakeholders' survey responses. In most cases, the statistical results led to identical conclusions. However, the nonparametric Wilcoxon procedure makes fewer assumptions of the underlying data distribution and is a more conservative test for statistical significance. Due to the small number of matched pairs in the Leon Creek Restoration Council (LCRC) (N= 21), we do not report parallel analyses for this stakeholder group.[1] Thus, the tests of the social science hypotheses (1-4) reported here are based on the survey data for the Salado Creek Restoration Council. The significance (alpha) level was set at $p = .05$ for all statistical tests.

Knowledge/Learning Outcomes

Hypothesis 1

- *Stakeholders' general understanding of non-point source (NPS) pollution issues in this watershed will increase as a result of the Collaborative Learning intervention.*

Six questions were included dealing with Hypothesis 1. Respondents were presented with a series of statements and asked to indicate their response on a Likert-type scale (1=Strongly Agree; 2=Agree; 3=Neither Agree or Disagree; 4=Disagree; 5=Strongly Disagree). We presented these items as "opinion" statements to avoid the appearance that this was a test of objective knowledge. Members of the ecological science team (Matlock, Kenimer, Neill) reviewed these questions and provided "expert" answers so that we would have a basis for comparing stakeholder knowledge to an external standard defined by scientific expertise. Thus, each of the 6 questions in this category (W1Q22, W1Q23, W1Q24, W1Q25, W1Q26, W1Q27) had a "correct" answer, as defined by our ecological science team.

Table 20 presents the statistical results from analysis of change scores on all 24 questions, listed by category. Using p= .05 as the decision criterion, responses to two of the 6 questions demonstrated significant change from Wave 1 to Wave 2 (W1Q25, W1Q26). Question 25 (Wave 1) read: "It is easy to predict how water quality will affect living organisms in and near a stream." Positive values for the t and S statistics indicate that respondents changed in the direction of greater agreement with this statement from pretest to posttest.[2] Question 26 (Wave 1) read: "Water quality problems in Salado Creek change depending on factors such as weather and time of year." The significant positive t and S values for this question show that stakeholders agreed more with this statement at posttest than at pretest.

The direction of change (toward agreement) for Question 26 agreed with the expert scientific judgment of our ecological science team. Weather patterns and time of year have a significant impact on non-point source pollution and the amount of water flowing in streams. For Question 25, however, the ecological science team indicated the "best" answer would be either disagreement (4=Disagree) or neutrality (3=Neither Agree or Disagree). Predictions about how changes in water quality affect living organisms in or near streams are possible, but they are not easy; such predictions tend to time consuming, expensive, and the resulting data is often difficult to interpret. The observed change on this question was in the direction of agreement. It appears that there was a shift of respondents from the Disagree category (4) at pretest to agreement (1 or 2) with this statement at posttest. Thus, the support for Hypothesis 1 was mixed: participants changed toward the "correct" answer on one item (Q26), but changed too much toward the "incorrect" answer on the second question (Q25).

The fact that the other 4 items in this category showed no significant change could be interpreted in one of three ways: (1) the CL intervention had no effect on their understanding of these issues, (2) respondents' initial answers to these questions at Wave 1 were already close to the "correct" answer and thus the amount of change possible at Wave 2 administration was restricted by the limited range; or (3) some respondents shifted toward the "correct" answer while others shifted in the opposite direction, thereby canceling out the overall pattern of change observed.

Hypothesis 2

- *Stakeholders' understanding of and ability to use scientific information about Total Maximum Daily Load (TMDL) and rehabilitation options will increase as a result of the Collaborative Learning intervention.*

Three questions were included to test Hypothesis 2. Table 20 reveals that there was significant change in the predicted direction on stakeholders' responses to all 3 questions. These questions used a self-report format to assess learning outcomes. Question 28 (Wave 1) read: "I am familiar with the scientific information used to make decisions about water quality management in the Salado Creek watershed." (1=Strongly Agree; 5=Strongly Disagree). Question 39 (Wave 1) read: "My ability to understand and use scientific information regarding water quality problems and related environmental issues is:" (1=Poor; 2=Fair; 3=Good; 4=Very Good; 5=Excellent). Question 29 (Wave 1) read: "I am familiar with the Total Maximum Daily Load (TMDL) program for ensuring water quality in Texas rivers and streams." (1=Strongly Agree; 5=Strongly Disagree). The positive t and S statistics on Questions 28 and 29 indicate that respondents agreed more with these statements at posttest than at pretest. The negative t and S statistics for Question 39 are due to the reverse scoring on the scale; higher scores indicated greater ability to understand and use scientific information regarding water quality problems. Moreover, the negative t and S values are consistent with Hypothesis 2: stakeholders should change toward greater understanding and higher ability to use scientific information as a result of the CL intervention.

Hypothesis 3

■ *Stakeholders' understanding and ability to use systems thinking in ecosystem management will increase as a result of the Collaborative Learning intervention.*

One survey item was included to test Hypothesis 3. Question 36 (Wave 1) read: "My understanding of systems thinking principles is:" (1=Poor; 2=Fair; 3=Good; 4=Very Good; 5=Excellent). Table 20 shows that respondents felt their understanding of systems thinking principles improved from pretest to posttest. Given the emphasis on systems thinking in the CL program, this result provides strong support for Hypothesis 3.

Communication Competence

Hypothesis 4

■ *Communication competence of stakeholders will increase as a result of the Collaborative Learning intervention.*

Three survey questions were designed to test Hypothesis 4. Question 30 (Wave 1) focused specifically on the quality of communication among stakeholders in the Salado Creek watershed. This item read: "Constructive communication regarding watershed management issues is occurring among affected parties in the Salado Creek watershed." (1=Strongly Agree; 5=Strongly Disagree). Table 20 reveals a significant change in stakeholders' responses between Wave 1 and Wave 2, but the negative sign of the t and S statistics indicates that the direction of this change was contrary to expectations. Respondents reported greater disagreement with this statement at posttest than at pretest. Thus, for this item dealing with communication at the level of the stakeholder group, there was no support for Hypothesis 4.

Two additional questions dealt with communication competence at the individual level. Question 37 (Wave 1) read: "In general, my ability to listen to others is:" (1=Poor; 5=Excellent). Question 38 (Wave 1) read: "In general, my ability to ask good questions is:" (1=Poor; 5=Excellent). Table 20 shows a significant change for Question 37, but no change in responses to Question 38 between pretest and posttest. However, the direction of change on Question 37 is in the opposite direction to Hypothesis 4. Respondents reported less ability to listen to others at Wave 2 administration compared to Wave 1.

Relationships Outcomes

It is commonly believed that collaborative methods will produce valuable "social capital" in the form of new and more positive relationships among participants. We included three questions to evaluate this assumption. The first two items addressed the quality of interpersonal relationships among members of the stakeholder group: Question 31 (Wave 1) read: "Members of the Salado Creek Restoration Council get along well with each other" (1=Strongly Agree; 5=Strongly Disagree). Question 33 (Wave 1) read: "I have high personal regard for the other members of the Salado Creek Restoration Council." (1=Strongly Agree; 5=Strongly Disagree). Contrary to popular belief, statistical analysis showed that stakeholder members reported that relationship quality decreased from pretest to posttest (see Table 20). Question 32 (Wave 1) focused on the issue of trust in other stakeholder group members: "I trust other members of the Salado Creek Restoration Council to do what is best for the long-term health and integrity of Salado Creek." (1=Strongly Agree; 5= Strongly Disagree). Table 20 reveals that there was no significant change in trust from Wave 1 to Wave 2 survey administration. In summary, the survey results suggest that the CL intervention did <u>not</u> improve the quality of relationships between stakeholder members, nor did it enhance feelings of trust in other members to act in ways that would preserve the long-term ecological health of the creek. In fact, the data suggest that interpersonal relationship quality decreased from the beginning of the CL meetings to the conclusion.

Negotiation Effectiveness

We included one question to measure stakeholders' perceived effectiveness in negotiating with other members of the Salado Creek Restoration Council. Question 34 (Wave 1) read: "I will be effective in negotiating with other members of the Salado Creek Restoration Council during future watershed policy discussions." (1=Strongly Agree; 5=Strongly Disagree). Table 20 reveals that there was a significant negative change in perceived negotiation effectiveness: stakeholder group members reported that they felt less effective at the conclusion of the CL meetings than at pretest. While we had no formal hypothesis about this issue, the philosophy behind CL would suggest that negotiation effectiveness should increase with participation in CL activities. We found no support for this assumption.

Overall Evaluation of Collaborative Learning Method

As a summary index, we asked respondents to evaluate their overall experience with the CL program. Question 35 (Wave 1) was worded as follows: "Overall, my past experience with collaborative planning/decision making methods has been: (1=Very Positive; 2=Positive; 3=Neutral; 4=Negative; 5=Very Negative). The Wave 2 version (Question 60) read: "Overall, my experience with the Collaborative Learning format used in the Salado Creek Restoration Council meetings has been: (1=Very Positive; 5=Very Negative). Thus, the comparison in this analysis was between stakeholder members' previous experiences with collaborative methods for planning/decision making in general vs. their specific experiences with Collaborative Learning in this project. The results displayed in Table 20 show that a significant change in respondents' experience with CL occurred over the course of the project. Stakeholder members reported more positive reactions to CL compared to their previous experiences with other collaborative methods.

Water Quality Beliefs

Four questions were designed to assess perceptions of water quality in Salado Creek. We had no expectations that the CL intervention would change these water quality evaluations. However, for exploratory purposes, we performed statistical analyses to test for change from Wave 1 to Wave 2 survey administration. Surprisingly, there was a significant improvement in beliefs about water quality across all four items (W1Q18, W1Q19, W1Q20, W1Q21). These four questions read as follows: Question 18 ("In your opinion, how safe is water from Salado Creek for drinking?" 1=Very Safe; 2=Safe; 3=Not Sure; 4=Unsafe; 5=Very Unsafe); Question 19 ("How safe do you think Salado Creek is for swimming?" 1=Very Safe; 5=Very Unsafe); Question 20 ("How safe do you think it is to eat fish that come from Salado Creek?" 1=Very Safe; 5=Very Unsafe); Question 21 ("In your opinion, how safe is Salado Creek water for livestock to drink? 1=Very Safe; 5=Very Unsafe). Table 20 demonstrates that, across all four questions, respondents believed that water in Salado Creek was safer at posttest than at pretest. This pattern of results was unanticipated based on existing theory about CL, but is intriguing nonetheless. It is possible that the increased knowledge about ecological principles and dynamics gained from the CL program may have stimulated stakeholders to revise their earlier, more negative perceptions of water quality in Salado Creek.

Personal Efficacy Beliefs

Two additional survey items assessed respondents' beliefs about personal efficacy. Question 2 (Wave 1) read: "People like me do not have any say about what the **national** government does." (1=Strongly Agree; 2=Somewhat Agree; 3=Neither Agree or Disagree; 4=Disagree Somewhat; 5=Disagree Strongly). Question 3 (Wave 1) read: "People like me do not have any say about what the **city** government does." (1=Strongly Agree; 5=Disagree Strongly). It might be expected that involvement in the meetings of the Salado Creek Restoration Council could increase perceptions of personal efficacy among participants, particularly at the local community level. Statistical analysis of these two items (see Table 20) showed no significant change on either survey item. Thus, there is no evidence that the CL intervention affected stakeholders' beliefs about personal efficacy at either the national or city government level.

Table 20. Statistical Analysis of Pretest - Posttest Survey Change Scores

QUESTION	S	N *	p	t	N	p
W1Q22-W2Q27	34.0	19	.152	1.67	35	.103
W1Q23-W2Q28	16.0	14	.351	1.14	36	.263
W1Q24-W2Q29	9.0	18	.720	0.55	38	.584
W1Q25-W2Q30	64.5	22	.028	2.22	37	.033
W1Q26-W2Q31	57.5	21	.037	2.21	38	.033
W1Q27-W2Q32	30.5	16	.124	1.76	38	.086
W1Q28-W2Q33	74.5	20	.003	3.34	36	.002
W1Q29-W2Q34	85.0	23	.004	2.97	36	.005
W1Q36-W2Q56	-126.5	22	.0001	-5.93	37	.0001
W1Q39-W2Q59	-193.0	31	.0001	-5.01	37	.0001
W1Q30-W2Q43	-121.0	28	.003	-2.84	37	.007
W1Q37-W2Q57	61.5	18	.005	3.11	36	.004
W1Q38-W2Q58	-11.5	25	.749	-0.31	36	.757
W1Q31-W2Q51	-211.0	30	.0001	-6.95	36	.0001
W1Q32-W2Q45	2.5	18	.633	0.57	36	.571
W1Q33-W2Q47	-258.5	32	.0001	-8.81	37	.0001
W1Q34-W2Q49	-277.5	33	.0001	-11.25	36	.0001
W1Q35-W2Q60	129.0	24	.0001	5.24	37	.0001
W1Q18-W2Q38	77.5	21	.002	3.07	38	.004
W1Q19-W2Q39	68.0	24	.035	2.18	38	.036
W1Q20-W2Q40	112.5	22	.0001	4.46	37	.0001
W1Q21-W2Q41	195.0	28	.0001	6.49	37	.0001
W1Q2-W2Q1	19.0	26	.632	0.45	38	.653
W1Q3-W1Q2	-7.5	19	.776	-0.33	38	.745

**Note*: The reduction in the size of N in this column is explained by the fact that all matched pairs with difference scores equal to zero are omitted from the statistical analysis in the Wilcoxon matched pairs signed-ranks test.

Summary and Conclusions

Our discussion of the results focuses on the four social science hypotheses tested in this project. First, with respect to learning/knowledge outcomes, we found support for Hypothesis 1 on one of the 6 objective knowledge survey questions relating to non-point source pollution. Hypothesis 2 received strong support on the self-report learning/knowledge items: stakeholder members reported that their knowledge of and ability to use scientific information in water quality management improved over the course of the CL program. These results are also supported by a qualitative analysis of post-interviews conducted with participants following the completion of the project (see Qualitative Evaluation in next section). With respect to Hypothesis 3, we found solid support: stakeholders reported greater familiarity and ability to use system thinking principles in ecosystem management at the conclusion of the CL meetings compared to the beginning of the project.

Hypothesis 4, however, was not supported by the statistical analysis of the survey data. In terms of communication competence, stakeholders reported that the quality of communication at the group level (Salado Creek Restoration Council) decreased from pretest to posttest. At the individual level, we found either no change or change in the opposite direction from Hypothesis 4. For example, respondents believed that their ability to listen to others had decreased from the beginning of CL meetings to the end of program. We return to this surprising pattern of results later in this discussion.

With regard to interpersonal relationships, the results were clear: stakeholder members believed that relationship quality declined from Wave 1 to Wave 2 survey administration. There was also no change in trust in others over the course of the 14-month series of CL meetings.

Overall, we found that stakeholder members evaluated the CL format used in the meetings more positively compared to their previous experiences with collaborative methods. Similar to the results for Hypotheses 4 and for relationship outcomes, participants felt their effectiveness in negotiating with other members of the Salado Creek Restoration Council declined from pretest to posttest. Thus, while stakeholder members viewed the CL program favorably overall, there was no evidence of improvement in members' negotiation effectiveness over the course of the project.

Ancillary analyses of water quality beliefs revealed that, overall, participants changed their perceptions of water quality in Salado Creek in a positive direction: water was believed to be safer for drinking, swimming, fishing and livestock at posttest compared to the pretest. Analysis of personal efficacy beliefs showed no changes as a function of participation in the CL program.

Before we accept these results concerning the effects of the CL intervention, we must address one potential threat to the internal validity of these findings. That is, can we attribute the positive changes observed for learning/knowledge outcomes to the CL intervention or are there alternative explanations? One such alternative explanation is that participants who reacted negatively to the CL program dropped out and stopped attending the meetings over the course of the project. This threat is referred to as attrition (or mortality) in the program evaluation literature (Cook & Campbell, 1979). If this occurred between pretest and posttest, it could be argued that the significant change results are due to the fact that only the participants who felt positively toward the CL program remained to complete the Wave 2 posttest surveys. Thus, the final sample of matched pairs of respondents may not represent accurately the original group of stakeholders who participated in the CL meetings at the outset of the project. If this were true, we would expect that any changes on the survey questions from Wave 1 to Wave 2 would be in the positive direction. That is, there would be a "halo" effect such that stakeholder members would report positive changes in their reactions to the CL program across all the dimensions of the CL assessment. However, this pattern was not observed. For example, for the communication and relationship quality questions, the changes reported were in the negative direction. Anecdotal observations of the pattern of attrition, based on attendance records, also do not support this alternative explanation: there were a significant number of participants who had been regular attendees at the CL meetings throughout the 14-month period who did not complete the Wave 2 surveys. We conclude that, while attrition did occur between pretest and posttest, this alternative explanation does not account fully for the mixed pattern of results across the set of 24 survey items.

In summary, what can be concluded from this quantitative evaluation of Collaborative Learning in this project? First, the survey results suggest that significant learning about non-point source pollution, TMDLs, and rehabilitation options did occur over the course of the CL meetings. There is also clear evidence that participants improved their knowledge about systems thinking in the context of ecosystem management. Thus, in terms of learning/knowledge outcomes, we conclude that the CL intervention was successful in achieving its objectives. Second, the survey data provide little support for the prediction that communication competence would improve with the use of CL. There is also no evidence in the survey results to support the assumptions that interpersonal relationships among stakeholders and negotiation effectiveness improved due to the CL program. Third, it appears that the CL intervention did change beliefs about water quality in the Salado Creek watershed toward greater confidence in the safety of the water for human and animal uses.

An obvious question is why the CL intervention did not improve communication competence and negotiation effectiveness? One explanation is that these skills were not the primary focus of the majority of the activities in the CL program that we designed. For example, listening and questioning skills were covered in only one short session at the beginning of the CL meeting series. Negotiation skills were not addressed directly in any of the CL meetings. A strong emphasis of the CL meetings was providing scientific information from the ecological science team and other experts on ecosystem management in the Salado and Leon Creek watersheds. This exchange of information with stakeholder members was accomplished through a combination of lecture, small group discussion, and active learning exercises. Thus, the lack of positive changes in communication and negotiation skills may have been due to the particular implementation of CL that we used in this project. It is also possible that the specific questions that we used to measure communication competence and negotiation effectiveness were not sufficiently sensitive to detect changes over the course of the CL meetings. The qualitative analysis reported in the next section provides an alternative, complementary method for assessing the development of communication competence. The conclusions from this qualitative analysis suggest a more positive effect with respect to the communication competence question.

Another puzzle is why interpersonal relationships did not improve with the use of CL in this project. The survey results suggest that the assumption that relationships will always be enhanced by collaborative methods may be incorrect, at least with respect to the version of CL that we implemented in this project. Daniels and Walker (2001) are careful to note that the use of CL may improve relationships among parties, but it should not be viewed as a certain outcome. Our quantitative results strongly reinforce their warning about this issue. Significant learning did appear to occur during the CL meetings, but that does not mean that the participants also came to like each other more. In fact, the survey data suggest that relationship quality appeared to decline over the course of the meeting sequence. This is an important caveat that should be recognized by future practitioners and researchers interested in using CL for local watershed management groups.

QUALITATIVE EVALUATION OF COLLABORATIVE LEARNING PROGRAM EFFECTIVENESS

Post Interviews

We interviewed 27 participants, either within the final month of the Collaborative Learning workshops or after the workshops were concluded. We conducted 23 post interviews, most with individuals and some with small groups.

During the post-interviews we invited informants to share their impressions of the workshops, and tell us what had been of most and least value to them. Although we were interested in learning whether they thought their communication competency had improved, we explicitly avoided asking them to discuss competency issues. After working closely with these community members for over a year, we assumed the need to guard against enabling informants to make preferred statements (statements they thought would please the researchers). With no prompting toward this topic, when asked to recall their experience in the workshops, informants in 20 of the 23 interviews talked about how the workshops had helped them, or other council members to become more effective citizens. Some informants made direct statements to this effect, while others were less focused.

Generally speaking, informants identified the opportunities to make connections with other interested citizens and the knowledge they gained during the workshops as most important. Agency representatives also noted that participation had improved their working relationships with other agencies. Citizen comments ranged from cautiously optimistic to extremely optimistic. All claimed that participation had increased the likelihood that they would participate in additional opportunities for self-governance. Most based this statement on the increased confidence and knowledge (of watershed and political processes) and communication skills they gained through participating in the council workshops.

Below are examples drawn from the post-interview transcripts:

Participant Feedback

From Agency Participants

- "[Regarding interaction with citizens] It was just the opposite of our fears."
- "We've always had a good relationship, but I think that those meetings helped the two groups [two agencies] kind of really start working together on what the goals of the streams."
- "I hadn't worked with citizens that closely before. And quite honestly, I wasn't all that thrilled about it initially. But, as we got into the process, I really enjoyed it a lot, dealing with the different people."
- "They [citizens] learned a little bit about our watershed and learned how to talk to each other and come to consensus and all that."
- "Most of the people have learned to communicate with each other and come to a consensus–those that have stayed with it. And so that's been real good. And I think that's–I guess it gave them the boost they need to reorganize what I guess used to be the old Save Our Salado group. And so it got a lot of new people interested and organized, and they organized it better this way–this time around to where they've got active people that go from up in the watershed through the bottom of the watershed and stuff like that so. And they're all well versed and they know what's important when it comes to a watershed–monitoring and restoration and development, that type of thing. They've got a better feel. They understand the jargon better and they're a lot more enlightened as a result of y'all's work. . . ."
- "And they've got several groups of citizens working up and down the San Antonio River. And that's not working out as well, from my observation, as the Salado and the Leon groups did. Mostly because these citizens . . . didn't go through the workshops y'all put on . . ."
- "They [a new watershed council] see those people [past participants of the Salado Council] working with the city council, working with the elected officials and the neighborhood groups and all that. So they're trying to model the San Antonio group after the–after the–what the Salado Creek and the Leon Creek groups have done…"
- [Our agency is] starting to use the–what transpired with Salado and Leon as a model of how to work with these citizens groups now."
- "Y'know everything [from the Leon Creek Watershed Council] has just kind of disbanded and people scattered, but not everybody just went home. There's some people that were involved in that process that are now involved with groups like Leon Creek Green Belt Coalition. And that–the opportunities that they had here have enabled them to really participate and even lead in this other group."
- Even if they [citizen participants] never get involved in an organization, they have all this background that–the training that they get from this process that y'all have developed. And they, I'm sure–although I haven't really talked to anybody since then–but I'm sure they're sharing that with other people."
- "I think the more people that are educated in these type of environmental issues, the better the decision making is gonna be for our community–or any other community."

From Citizens with a History of Activism

- "The ones [activities/presentations] that we remember the most are the ones where we had to get up out of our seat and go do something, y'know whether it was make a chart or–I think probably one of the–really one of the first things I remember is y'know when we had the–the . . . the map of the–the whole creek bed and you were supposed to find whereabouts you lived or your business was or whatever. And that was actually a wonderful opening exercise because for one thing you had to talk to the other people in finding your own location. So it was a great way to get to know the other people that participated."
- "I can visualize all of us going to your classes and learning these things and then taking it y'know to our constituents and y'know patterning–y'know duplicating this."
- "That kind of leadership and infrastructure isn't–isn't present in Leon. And work like yours encourages its development."
- "What was novel was the comprehensive nature of what y'all were doing. It wasn't just talking about one element of the creeks, but it seemed kind of getting the sociology of things. That was different than probably anything I've experienced along those lines."
- "There's just all these spinoffs from the groups."
- "The great thing about this program was how it was bringing everybody up to a certain level of education on the topic. I think that's just really great."
- "I think they [the workshops] gave people something that they could use when they needed to talk to agency personnel."
- "When you talked to us about listening, I thought that was really useful. And the way that you handled the meetings worked–was also excellent. It was important that they didn't just become gripe sessions. So many of these become gripe sessions and that's all they are and nobody ever gets anywhere."
- "I thought the people who came learned to respect each other and each other's ideas and each other's issues, and I think that was an important part of those meetings."
- "I liked the process. It really facilitated learning."
- "I think probably the most effective thing of it was the fact that it got people from a whole diverse arena of backgrounds together that did not know each other, knew that they had a little stake in the green belt–not in the green belt but in the; in the creek itself. And now they have learned, if nothing else, that there's a whole lot of other people that are interested in it. And some of them are still kind of staying active and will continue to stay active. So I think from the standpoint of it was very useful for that purpose alone, if nothing else."
- "And it may be difficult to quantify, but where it will make a difference is people have gotten to know each other. Y'know they've made some connections. And assuming those–that same group stays on the scene–the San Antonio scene–say, for the next five years, you find y'know that they can make a difference and maybe not creek issues but other issues–community issues–down–down the road--y'know people who know how to get things done."
- "It's kind of hard to–it's hard to know even now. I mean you're–clearly Salado Creek Foundation, that's one thing that–it may have happened on its own, but I think y'know ______ got–made some connections through the meetings that y'all helped make happen. I think she got some networking done and she also got excited about it and got other people excited about it."
- "But y'know when they find a–when something comes their way–an opportunity to help out, those are the sorts of things that I'm pretty sure y'know will come out of this. The connections."

From Citizens for whom this was the First Participation Experience in Governance

- "What my work on the council did for me was to give me a little more confidence in moving into other arenas, and I currently sit on the board of ________. . . . So I would never have thought to actually do that unless I had participated with y'all in the process of Leon Creek, feeling that I didn't know enough about government and the way things were done. I met a lot of really nice people, and I find that that's–that's good for me and useful for me in my position."
- "This was the first time I had been through the kind of experience that you all organized, which I thought was just excellent. It was just simply excellent the kind of stuff–if you want somebody to be able to be good stewards of the earth or in this case the water or our watershed, they gotta have some real information about the science of it all and the history of it all, and then you can expect them to make the decisions. And usually that doesn't happen."
- "And so that was the wonderful thing about what you all did. And then I–maybe by happenstance or maybe by design, the group that stayed interested in it was willing to ride the river, so to speak."
- "But I didn't really concern myself with the broader picture. But since then I have. I do sit on the ______ panel. And I don't think I–I may not have done that before."
- "I come better prepared to be a citizen because of your program–in terms of understanding the whole system."
- "You've educated us. And then after you educated us, you empowered us because you gave us that knowledge. And you gave us the exposure–not only to the knowledge and the hazards or the benefits of Leon Creek and all that–freshwater–but we got to where we connected with other people."
- "I'm more involved, not only for creeks and fresh water and all that–but once you empowered me with education, someone nominated me for a school bond committee."
- "Now that–those meetings down there, y'know I figure they did some good. Well–well, yeah. Cleared my mind up on a few things. . . . And–but it gives people an idea of what–what you run into when you start talking about flood plains and flow lines and all that y'know."
- "People are getting a lot of information…. Before [participating in these workshops] I couldn't have told people about --I'm in a better position…I can tell people, 'I'm working with SAWS, and the River Authority, and Texas A&M'…so now I can say this is what happened and why it happened. . . Now I can explain it."
- "If good wasn't being done, people wouldn't be there. If it's a Friday or Saturday, people could be doing a lot of other things."
- "People ask me, 'What are you doing these days?' Then I can talk to them about the council."
- "I stayed with this council because I saw people I knew from other neighborhoods. And these are busy people. You take __. She's got plenty of money, she doesn't have to be there. She's there because she cares. And ___, he's a farmer. You don't think he has about a thousand things he could be doing? Mrs. __. She's not so much for herself but for the community. And ___. He's fine. He's a businessman, ok. He wants to develop something. There's nothing wrong with that. We can't hold that against him."
- "You brought us together, you educated us–even members of the city staff. We helped pass one proposal out of many, but that was to buy land over the recharge zone and for linear parks along Salado and Leon Creek. . . . We went back to our neighborhoods, our meetings, our associations and what–and said we need to, if nothing else, vote for this. The city–remember ______wrote a letter."

CL Meeting Participants Demonstrate New Skills

There were also some surprises that occurred during the course of the CL meetings. One particular meeting provides a vivid example of the changes that took place among members of the Salado Creek Council. Their response to a challenge demonstrates the new skills that had been developed through participation in the series of CL meetings. Below is a summary account of the events at the 10th meeting of the SCRC.

Ironically, the SCRC's hard work was nearly derailed after council members assisted in obtaining sufficient electoral support for a sales tax that would provide additional funds for watershed restoration. Both councils had set aside most of their tenth meeting to make final decisions regarding the recommendations they would present to the Mayor, and the future of the groups. Extensive publicity of the new funding drew the attention of many new stakeholders along Salado Creek, and attendance more than doubled. Participants who had developed a systemic understanding of the watershed through a year of intense learning activities glanced uncomfortably at each other as new attendees vented, making statements such as:

> "I know what I own and ain't no one gonna mess with it. I turned 88 last month, and I'm gonna be out there another 50 years cause Genesis 6 and 3 gives me 120 years. On top of that, you don't want to deal with my relatives. I got 50 kids, I had five, and they went out and brought home five. And that gives me ten. They gave me twenty grandkids and they gave me twenty great-grandkids. And I got a man sitting here that is the grandfather of two of my great-grandkids. So you don't want to deal with them. You better deal with my wife of 57 years and me on buying that water rights and buying that land, because other people won that land up and down that Salado and you ain't just gonna walk in and say we're gonna take it. Because a lot of them are meaner than me, and I'm as mean as a junkyard dog."

Because the research team had essentially turned management of the group over to local participants, we did not intervene. We sat quietly, waiting to learn whether council veterans would have developed the skills necessary to redirect the energies of the new participants. After listening to similar inflammatory comments for approximately 30 minutes, Cathy, one of the seasoned participants attempted to redirect the conversation. Others, who recognized that the behaviors of the new members were threatening the group's consensus regarding how to improve the watershed, as well as how to behave as a council member, soon joined her. They had developed these informal norms through months of difficult interaction, and were not prepared to give them up lightly. Cathy started by explaining the CL process they were using. She added that the facilitators "are here to help us make our own decisions about the creek, not to tell us what to do." She then introduced a group of people designated as "the nominating committee" (made up of participants who, at the request of the entire group, had spent the past two months developing a slate of nominees who were willing to serve as officers in a continuing council). At Cathy's request, Hank, John, Janet, and Tina each delivered extemporaneous remarks about the work the Council had done, and its future direction. Janet summarized the concerns of council members when she explained, "we are concerned that we need a place at the table to spend this money. We are concerned that people will come out from under rocks to spend the money on things we don't want. We need to protect the trees. We need to help people and we need safety and protection. We need to institutionalize ourselves." Janet's presentation reminded everyone in the room that most council members had reconceptualized the watershed as a system, within which multiple entities held legitimate interests, and that they had worked hard to develop this understanding. It was not something to give up lightly.

Following the nominating committee's remarks, Cathy took charge of the meeting. Each time a new member attempted to return the conversation to a single interest at the expense of the more systemic perspective, she turned to an experienced member to help move the discussion back to the goal of watershed improvement. At the next meeting, when participants piloted their simulation model, the attendance pattern changed again. The seasoned participants returned along with less than one quarter of the new attendees. The new attendees, for whom this was only a second meeting, began accepting behavioral norms established over the past year. Each of them worked in a small group with two or three seasoned veterans, who had conceptualized the simulation model they were evaluating. When the group broke for lunch, they joined in planning an upcoming field trip designed to further enhance understanding of the entire watershed.

Lessons Learned

The research team left the groups in December 2000 and the councils have since taken on lives of their own. One has developed into a private foundation that has won city-wide awards. Electronic messages received from members indicate continuing participation in city governance issues relevant to the watershed. The stakeholders were able to connect to each other, and add their contribution to ongoing management debates. Members of these citizen councils have maintained themselves as self-organizing bodies, recognized and respected by existing institutions.

In summary, the qualitative evaluation approach offers a somewhat different picture than the survey results presented earlier. Many of the comments in the post-interviews reinforced our conclusion that significant learning had taken place about non-point source pollution, ecological principles and dynamics, and rehabilitation options. The interview data, however, also suggest that some participants did improve their communication skills and develop new relationships with others over the course of the program. Agency representatives were also positively impressed with the quality of the CL meetings and there is evidence from the interview data that the CL approach is now being used as a "model" for other watershed collaborative groups in San Antonio. Many interview comments focused on how the information gained during the project meetings helped them to become more active in other community groups and self-governance activities.

We conclude that the qualitative results of this project provide additional, and in some cases, stronger support for the original social science hypotheses proposed.

One of the main obstacles in moving toward sustainable, multiple use of resources is our inability to synthesize knowledge and perspectives from many distinct disciplines within a single problem-solving philosophy. Collaborative learning can provide a basis for sustainable, multiple resource use by facilitating multidisciplinary planning and by creating an effective communication interface between scientists, citizens, and policy makers, thus promoting identification and communication of policy options to decision makers.

Participation and Representation in Watershed Restoration Councils

Who Participates in Watershed Collaborative Groups and Why

In the Research Activities section, we began by posing three fundamental questions about stakeholder participation and representation in collaborative processes: (1) Who participates in these collaborative processes?, (2) Why do they participate?, and (3) What basic values, beliefs, and attitudes about democracy, society, and the environment do they represent? In this Research Findings section, we begin by discussing the empirical results with respect to these questions. Next, we consider whether the stakeholder recruitment techniques used to structure the watershed councils satisfied the normative criteria—autonomy, welfare, justice—proposed by Trachtenberg and Focht (2003) for evaluating procedural legitimacy. Finally, we explore implications of empirical results for practitioners and researchers working on collaborative processes in watershed management.

The survey and qualitative results suggest some initial answers to our three questions. First, with respect to the first question—who participates—three attitudinal variables in the large-sample phone survey positively predicted willingness to participate: optimism about personal finances, negative evaluations of water quality, and pro-environmental attitudes. Survey responses of actual participants in the CL stakeholder groups also confirmed the importance of negative water quality evaluations and pro-environmental beliefs. They also suggested the important role of personal efficacy, specifically relating to city government. In terms of demographic characteristics, the larger survey of San Antonio residents indicated that Latino citizens would be more willing to participate in discussions on clean water in San Antonio. However, data on actual participation in collaborative stakeholder groups supported the opposite conclusion: Latino citizens were significantly under-represented in both stakeholder forums.[3] In addition, women in both survey samples were found to be less willing to participate than men.

With respect to the second question concerning why individuals participate in collaborative processes, we collected additional data at the fourth meeting (February 2000) of the series of CL stakeholder workshops. Facilitators asked participants at this meeting to complete the following statements: (1) "My participation in the Council will have been worthwhile if: ______"; and (2) "My participation in the Council will **NOT** have been worthwhile if: ______." Providing context for this request was the meeting agenda, which focused on having the groups discuss their mission as a Council, with the unstated process goal of developing their collective identity as citizen groups. Forty-eight participants provided responses across the two Councils (N=28 in Salado Creek; N=20 in Leon Creek). This sample of meeting attendees represented 47% of the total number of survey respondents in the Salado Council (N=60), and 57% of the total number of survey respondents in the Leon Council (N=35).

Fifty-nine responses to the first question were content analyzed and divided into 8 distinct categories.[4] Table 21 presents these 8 response categories along with two representative examples of statements coded in each category. The categories are arranged in Table 21 from most to least frequent. The final category ("Idiosyncratic Motives") includes a number of diverse statements that did not fit clearly into the other seven categories.

Table 21. Citizen Goals/Reasons for Participation in Watershed Councils

Protection/Restoration of Watershed (N=11)	"Creek is brought back to lush, green, living, flowing creek that I grew up with." "Nature is preserved and enhanced. Wildlife is conserved."
Information/ Education (N=9)	"Other participants have a better understanding of watershed and water quality management issues." "Useful information to help preserve the creek in its natural state will be gained from Council meetings."
Political Action/ Implementation (N=8)	"City Council and public approval of Council plan." "Council recommendations are implemented."
Community Good (N=7)	"Something progressive and meaningful comes out of it. We accomplish good for the city as a whole." "Quality of life for Bexar County residents in watershed has improved 10 years from now."
Water Quality/ Quantity Improvement (N=7)	"We get a reliable/sustainable quantity of quality water." "We can clean up the creek."
Parks/Greenbelt/ Recreation (N=7)	"Three miles of hike/bike trails opened by June 2001 on the creek." "Process furthers city/county commitment to watershed linear parks and dedicated buffer zones along San Antonio's watersheds."
Flood Control (N=4)	"Necessary flood prevention of lower creek is accomplished." "Flooding and erosion is prevented. Master plan for drainage adequate to development is created."
Idiosyncratic Motives (N=6)	"Develop working relationships with stakeholders from different areas of watershed to protect the resource." "Incorporate procedure for contacting Native American descendents when sites/remains are uncovered."

The responses to the second statement (N=43) about what would make participation **NOT** worthwhile were more tightly clustered: 77% of responses were captured by 4 categories. The modal response (42%, N=18) was expressed as follows: "Nothing is done/No implementation/Council recommendations-plan gathers dust on shelves." The next most common response (19%, N=8) was: "Council disbands/Can't reach agreement on plan." Less frequent were the following two categories: "No useful information is obtained for community education/Willingness to inform and be informed stops" (9%, N=4); and "Channelization of creek with concrete occurs" (7%, N=3). The remaining 10 responses fell into a fifth category labeled "Idiosyncratic". The majority of these latter responses were simply the opposite of the reasons stated in response to the first question.

Two general observations can be made on the results from these content analyses. First, the stated goals/reasons for worthwhile participation are quite varied, being distributed fairly evenly across the eight categories. Note that over 40% of the responses (3 categories) focus on rather abstract reasons not tied to the specific content of the Council's recommendations—Information/Education, Political Action/Implementation, and Community Good. The remaining majority of responses, however, do reflect objectives that depend on specific issues (i.e., protection/restoration of creek environment, water quality, parks/greenbelts, flood control) being addressed directly by the Council's final action plan. Second, the outcomes of the Council's work making citizen participation **NOT** worthwhile are not necessarily mirrored by those cited that make participation worthwhile. Citizens seem most concerned about two possible outcomes: (1) No implementation of Council recommendations by relevant political entities, and (2) Inability of Council to reach consensus on a set of policy recommendations. It is instructive that neither of these concerns depend on the substantive issues addressed by the Council's action plan. One possible implication, at least for a majority of citizen participants (61%), is that the minimum criteria for worthwhile participation in the CL process are that: (1) the Council agree on some set of specific recommendations, and (2) that "on the ground" implementation of at least some Council recommendations occurs by the relevant political authorities.

Answers to the second question—why do people participate?—are suggested by our content analyses of CL council members' stated goals/reasons for participation. People participate in collaborative engagement processes for two types of reasons.[5] Not surprisingly, the majority stated that they participate because they believe specific policy issues can be addressed directly by the collaborative process, such as environmental protection/restoration, flood control, or parks/greenbelt development. Participation is worthwhile for this subgroup if the final recommendations that emerge from the collaborative process include their personal issues and deal with them satisfactorily. The second category of citizens participated because they believed that the collaborative process had the potential to enhance the public good by improving the quality of life for all residents in the watershed. This group was less concerned with specific issues and more concerned that the collaborative process produced any tangible, positive results "on the ground" for their community. Personal efficacy appeared to be the underlying theme for this subgroup. Participation was valued by these citizens as long as something worthwhile gets done in their watershed. This response pattern reveals that many citizens in the stakeholder groups were often frustrated by numerous planning processes soliciting citizen input and then not culminating in successful implementation by elected political authorities and government agencies.

Finally, this research addressed the third question of what basic values, beliefs, and attitudes were represented by those who actually participated in collaborative engagement processes. Survey results from both watershed restoration councils present a fairly consistent picture. In terms of political beliefs/ideology, while participants represented both ideological positions on the role of government in society, there was greater participation among those who believe that "government can do more" compared to citizens who subscribe to the "less government, the better" ideology. We also found that personal efficacy (both city and national government) was generally higher in both watershed restoration councils than in the watershed subpopulations from which these stakeholders were recruited. With regard to environmental attitudes, clear evidence existed in both watershed restoration councils of greater support for the New Environmental Paradigm (Dunlap & Van Liere, 1978), relative to those residents surveyed in the respective watershed populations. Finally, citizens who rated water quality in the watershed as "unsafe" were over-represented in the watershed restoration councils compared to citizens in the respective watershed subsamples.

One interpretation of this pattern of participation is that our councils may have over-represented "problem perceivers". Prospect theory (Kahneman & Tversky, 1979) from the behavioral decision literature may help explain this outcome. According to this theory, people value losses more than gains; thus, people may be more willing to accept the costs of participating in a council if they perceive problems to be solved (i.e., impending losses), rather than opportunities for gains. In these council groups, water quality perception and education level were positively related: citizens with higher levels of education also believe that water quality is poor in the watersheds. From a representation perspective, this is not necessary a problem. In fact, one could argue that "problem perceivers" should be over-represented because these individuals have stronger beliefs and attitudes toward the policy issue in question. An implication for practitioners is that they should expect these "problem perceivers" to represent a significant segment of their stakeholder audience when using recruitment methods similar to those used here. Educational programs may then need to be tailored to reach this stakeholder subgroup if change in beliefs and attitudes is a desired objective.

Theoretical Implications: Evaluating Normative Criteria for Legitimacy

Trachtenberg and Focht (2003) have proposed a normative framework for evaluating procedural and substantive legitimacy of watershed management policy. Our research design permitted a quantitative evaluation of whether the recruitment techniques succeeded in meeting the normative criterion of autonomy. Specifically, do the empirical results support the conclusion that the demographic composition of the watershed council stakeholder groups matched the corresponding statistical profiles of the watershed subpopulations from which these participants were selected? Overall, we conclude that the actual demographic profile of the stakeholder councils differed in important respects from the statistical characteristics of the Leon and Salado Creek watershed subpopulations. As noted earlier, women and Hispanic residents were clearly under-represented in the watershed restoration councils.

Note that representatives from the San Antonio business/development community were visibly absent from the majority of the watershed restoration meetings. State/local agency and city government officials were active participants, as were citizens from a wide variety of interest groups and neighborhood associations. While there were a relatively small number of small business owners who participated actively in the watershed restoration councils, despite dedicated efforts to recruit additional business/development representatives, large corporations in the watersheds declined to participate in these collaborative processes.

Why didn't stakeholders from the business/development community participate? There are probably numerous reasons, but we speculate that at least one important factor was the belief that the watershed councils had enough regulatory authority to recommend actions that might harm their interests. While the councils were created with the knowledge and support of the relevant federal, state, and local water regulatory agencies (e.g., U.S. Environmental Protection Agency, Texas Natural Resources Conservation Commission, San Antonio Water System, San Antonio River Authority), neither the Salado Creek or the Leon Creek Restoration Council possessed formal regulatory authority. The end goal of these forums was to present a set of recommendations to local political authorities on how to improve the current situations in Salado and Leon Creeks. Thus, another plausible reason for non-participation among the business/development community was that these stakeholders did not view the councils as powerful enough to change the environmental regulations in ways that would adversely affect them. The validity of this explanation, of course, rests on the assumption that business/development stakeholders participate in these councils primarily for defensive reasons.

How serious are the problems of under-representation of women, Hispanics, and business/community representatives? That is, to what extent does the failure to achieve adequate representation for these subgroups compromise our primary objective of satisfying the autonomy criterion? In our view, the under-representation of Hispanics is the most serious threat to the goal of ensuring autonomy because they represent the largest non-Anglo group in San Antonio with substantial numbers living in proximity to the watersheds in question. Any policy with regard to the management of these watersheds will affect this population and vice versa. While the proportion of women in our watershed restoration groups was lower than the statistical incidence in the respective watershed populations, we still had roughly 40-45% women in these stakeholder groups. It is harder to assess whether the under-representation of large business organizations undercut our efforts to meet the autonomy criterion because we have no reliable, objective standard for establishing what proportion of our watershed restoration councils should have been composed of business/development representatives. We did, however, ensure that the recruitment process for the watershed council stakeholder groups was open to the business/development community by making numerous efforts to publicize the meetings among relevant business interest groups (e.g., Chamber of Commerce) within both watersheds. The openness of the process for recruiting watershed restoration council participants helps mitigate the potential for negative consequences of having under-represented business/development interests.[6]

Implications for Practitioners and Researchers

What do these results and field experiences imply for practitioners working in agency/government organizations or researchers who study collaborative processes in watershed management? In general, this project's experience underscores how difficult it can be to fully satisfy the legitimacy criterion (see Trachtenberg & Focht, 2003). Researchers and practitioners must think carefully about what will constitute adequate representation in any given geographical area and about how they will recruit from socioeconomic groups with which they have little familiarity. Even though we expended considerable time and effort to achieve adequate representation of all important stakeholders in these collaborative processes, there were gaps in representation. Despite devoting resources to the recruitment of council members representative of their larger watershed populations, we failed to adequately represent one critical ethnic group (i.e. Hispanics) in the San Antonio community.

The decision about which groups to include as stakeholders requires, at a minimum, familiarity with the social categories of people who make up the population potentially affected by policy and management decisions. While it may be relatively easy to acquire information about the population characteristics of the general geographical area in question, it may well take more effort to acquire information on the population nearest the management area. This, however, is the population most likely to be directly affected by decisions and most likely to have strong feelings about the feature or area in question, both of which have obvious implications for the sustainability of management policies and practices.

Once target groups are identified, research must confront issues of appropriate recruitment strategies. Researchers and practitioners may be at considerable disadvantage in the recruitment process. For example, social situations defined by unfamiliar expectations place the researcher or practitioner in an intercultural context (see Lustig & Koester, 1999). It has long been recognized that communicative effectiveness may be diminished simply by differences in speech patterns (Philipsen, 1975). Consistent with this literature, it is common practice to match for gender and ethnicity in many kinds of survey and interview efforts. However, we found that short of having "recruiters" from the relevant groups, it was very helpful when a "sponsor" was available to introduce the recruiter to persons in these under-represented groups (Lindlof, 1995). This strategy worked with people from the least advantaged and the most advantaged economic groups.

Almost a quarter of the respondents (24.1%) from the initial telephone survey indicated interest in participating in a citizen forum on watershed issues. This was also true of the Hispanic citizens who were contacted. While only 5 of the survey respondents (2 percent) who expressed interest in participation actually became members of one of the watershed councils, other recruitment methods were relatively more successful, except for Hispanic residents of the watershed.

Our experience suggests that issues of community consultation should be of special concern to practitioners and researchers who seek to involve underrepresented populations, including Hispanics. This consultation should begin during the planning stage of the project and should continue through the project's completion. Community consultation integrates several approaches, including public forums, immersion and interaction with key informants, and the use of consultants. Public forums should be integrated into ongoing functions of community associations such as social clubs, fraternal groups, and churches. Key informants and consultants can provide feedback to practitioners and researchers on how their plans relate to ongoing community interests and activities. Field researchers should become well informed about the local community, going beyond census data and obtaining information that will help the researchers better understand the way in which a community functions. In research projects where personal contact is involved, same-ethnicity data collectors enhance rapport, willingness to disclose, and the validity and reliability of the data provided. Other similarities in background between researchers and participants are equally valuable in motivating participants to complete the research process and also to provide accurate information.

A practical implication of the elaborate process needed to acquire representation is the time and staff needed to accomplish it. Research teams can budget adequately for the effort, but agencies rarely have the staff and other resources required. These observations are consistent with the findings of the EPA Science Advisory Board (USEPA Science Advisory Board, 2001).

The final lesson in this project relates to the potential survival of watershed coalitions after external facilitators depart. In this watershed, the Leon Creek Restoration Council did not persist after the completion of the collaborative learning activities. The Salado Creek Restoration Council, on the other hand, resurrected an inactive nonprofit entity, the Salado Creek Foundation (www.saladocreek.org). The Salado Creek Foundation is active in community policy advising, education, and cultural preservation of the riparian zone of the Salado Creek system. The variables that give longevity to collaborative organizations are certainly too complex to define in this project; however, we speculate that several fundamental differences in the two watersheds resulted in the different ends. The Salado Creek system is heavily urbanized, relatively well developed, and has a history of conflict over management issues. The Leon Creek watershed, on the other hand, is only recently urbanizing, resulting in rapid land use changes and degradation of water quality. The citizens in the Leon Creek watershed are a broad mix of urban fringe poor with little formal education, and well-educated ranchette owners, resulting in very little common experience among the stakeholders (Table 2). A few motivated citizens who committed themselves to the task organized the Salado Creek Foundation. No such leadership emerged from the Leon Creek Watershed Restoration Council. The obvious complexity of this process suggests that designing watershed collaboratives for persistence may be very difficult indeed.

Program Outcomes

One important goal of both council groups was to develop a set of recommendations by the end of the CL meeting series. Both the SCRC and the LCRC achieved this objective by the end of the October 2000 meetings. Representatives from each council made oral presentations of these action plans to the Mayor of San Antonio, who attended the November and December 2000 meetings. A brief summary of the key elements of these recommendations is provided below. Appendix C contains the full text of the action plan recommendations for each council.

Action Plans

The SCRC recommendations included the following two goals: (I) Minimize damage from flooding and maintain sufficient flow to restore water quality; and (II) Restore and maintain water quality to a level sufficient to support a recreational fishery and wildlife habitat. The specific means for achieving these goals was detailed in 9 recommendations: (1) Increase citizen input in implementation of Proposition 3; (2) Strengthen city and county ordinances (including enforcement procedures) to prevent inappropriate development, floodplain fills, and dumping; (3) Assist grandfathered floodplain developments in working toward compliance with current regulations; (4) Establish and maintain city-wide program requiring responsible clearing and cleaning of waterways; (5) Eliminate hazards such as low water crossings (put in bridges as needed, e.g., Roland Ave.); (6) Develop retention facilities; (7) Establish meaningful base flow; (8) Improve solid waste disposal process; (9) Investigate impact of quarries on aquifer recharge, water quality, and flow.

The LCRC recommendations included the following overall goals: (I) Include guidance for enhanced citizen involvement in watershed management; and (II) Maintain and restore (as necessary) Leon Creek so that it is an asset to the community. Specific suggestions for achieving these goals were grouped into two categories: Watershed Council Led by Citizens, and Watershed Restoration. Under the Watershed Council concept, 7 specific recommendations were made: (1) Begin with existing stakeholder groups (LCRC); (2) Expand to include youth, specific interest groups, additional neighborhoods; (3) Encourage strengthening of neighborhood coalitions; (4) Include a public education and outreach component; (5) Use agency expertise in its deliberations; (6) Facilitate citizen oversight of agency performance; (7) Hold quarterly meetings with relevant agency personnel. Under the Watershed Restoration category, 5 specific recommendations were made: (1) Management should enhance the ecological sustainability of the watershed; (2) Use a watershed approach; (3) Maintain and restore green space (prefer using brushy natural vegetation where possible) alongside waterway to provide wildlife habitat, recreation opportunities, improved water quality, reduced flood danger, and reduction of non-point source pollution; (4) Impose a moratorium on building in the 100-year flood plain; (5) Consider an ordinance creating a 100-ft buffer to the 100-year flood plain.

Meeting Attendance

There were a total of 109 individuals who attended one or more meetings of the Salado Creek Restoration Council (SCRC). This total included guest presenters who typically only attended a single meeting to make a presentation or serve on an expert panel. The mean number of participants attending each meeting was 27. Attendance varied substantially across meetings from a high value of 40 to a low of 13. At the individual level, the mean number of meetings attended was 3.24, with a range from 1 to 12 meetings. The number of SCRC participants attending 6 or more of the meetings was 21.

In the Leon Creek Restoration Council (LCRC), there were a total of 65 individuals who attended one or more meetings. This total also included the guest speakers. The mean number of participants attending each meeting was 17. There was also wide variability across meetings, from a high of 24 to a low of 9. At the individual level, the mean number of meetings attended was 3.40, with a similar range to the SCRC of 1 to 12 meetings. Thirteen (13) LCRC members attended 6 or more of the meetings.

We made significant efforts to assure maximum attendance at each meeting. All participants were sent reminder messages either by phone or email several days prior to each meeting. Given that we had no direct experience in conducting such CL workshops in the San Antonio area, it was difficult to evaluate the attendance levels observed relative to a reference standard. It was clear that the LCRC was always a smaller group than the SCRC. We do not have a compelling explanation for why this attendance pattern occurred.

Program Costs

Conducting the series of 26 CL workshops in San Antonio was an expensive proposition. Over a 19-month period (October 1999-April 2001), the total amount spent was $14,291, an average of $752 per month. Providing food for the council groups was a major expense ($3,872, or 27% of total), in addition to supplies needed to facilitate the meetings ($3,937, or 27.5% of total). Part of the reason for the high cost was the distance of the research team from San Antonio, approximately 3 hours by car. Transportation and hotel costs totaled to $5,848, nearly 41% of the total amount spent on the meetings. A local research/facilitation team could conduct these meetings without incurring these travel expenses. Nevertheless, the costs of supporting the CL meetings were substantial. Prospective researchers and facilitators of collaborative community groups need to be fully aware of the high costs of running a CL program that extends over a full calendar year.

ECOLOGICAL SCIENCE INVESTIGATION

WATERSHED LAND USE CHANGE

Estimated Land Use Changes

It is important emphasize that, due to a lack of reference data, a limitation of retrospective land-use classification is uncertainty about the accuracy of the estimated size of land-use categories (Congalton and Green, 1999). Therefore, observation of changes in the size of land-use categories must be treated with caution. However, if the magnitude of the estimated changes in land use is substantial it may still be possible to draw general inferences about the effect of perceived changes in land use patterns on ecosystem services.

In 1976, the rangeland category dominated all three watersheds within the study areas and totaled an estimated 80,497 ha, but by 1991 the total area of this category was estimated to have decreased by about two-thirds to 27,896 ha. The effective annual rate of decrease appears to have accelerated from about 3% per year during the period from 1976-1985 to 12% per year during the period from 1985-1991. The woodland category, by contrast, increased substantially in all three watersheds and overall more than quadrupled from 8,886 ha in 1976 to 44,654 ha in 1991, with an estimated annual growth rate of 10-12% per year. The sizeable decrease in open rangelands and the concomitant increase in the size of the woodland category are consistent with the widespread and rapid encroachment of woody plants in the region (Smeins and Merrill, 1988; Smeins et al., 1997). Intensive livestock grazing and reduced fire frequency has resulted in a change in the grass-woody plant interaction on many rangelands in the USA (Scifres, 1980; Archer and Stokes, 2000), which has led to widespread transformation of grasslands to shrublands and savannas and woodlands (Archer, 1989; 1994; Schlesinger et al., 1990). As a result of these changes and the associated decline in water yields in some areas (Thurow et al., 2000), brush control has become a dominant issue in rangeland management throughout the Edwards Plateau and the Edwards Aquifer Recharge Zone.

The bare soil category, which was assumed to predominantly represent post-harvest cropping areas, was estimated to have increased in all three watersheds from 6,353 ha in 1976 to 13,057 ha in 1991. However, while the size of this land cover category increased by an estimated 113% between 1976 and 1985, it appeared to have decreased by about 3% during the subsequent six years. The initial increase may have been associated with clearing of rangelands for crop production as well as suburban development but it may also have been partly due to misclassification because of seasonal variations in vegetative ground cover at the time that the three satellite data sets were captured. For example, more rapid spring growth due to above average spring rains in 1976 could have resulted in an underestimate of cropland due to misallocation of cropland (bare soil) to rangeland in the 1976 image (NCDC, 1994).

Although the estimated size of the residential land cover category increased in all three watersheds from 11,449 ha in 1976 to 16,655 ha in 1991, the rate of growth averaged only 3% per year during this 15-year period, and most of the growth appears to have occurred during the 1985-1991 period (9% per year). The estimated size of this category actually decreased slightly in the Salado Creek and Leon Creek watershed between 1976 and 1985. The apparently small change in the residential land cover categories may be an underestimate of the actual growth for two reasons. First, because asphalt shingles are widely used for roofing and residential area road networks are dense, some residential areas could have been misallocated to the urban and transportation categories. Secondly, the potentially increasing masking effect of turf and tree canopy cover in residential areas could have resulted in some misallocation of maturing residential areas to the rangeland or woodland land cover categories in 1985 and 1991.

According to our analysis there was a 151% increase in the area of the urban land use category from 1976 to 1991. By contrast, the transportation category appeared to decrease marginally during the 15-year period of our study, mainly from 1976 to 1985, but reduction in transportation is inconsistent with expanding urban sprawl. This anomaly may be explained by the fact that there are inevitable inaccuracies in delineations of urban and transportation land uses when using remotely sensed spectral reflectance data because asphalt in parking lots, roofs, and roads produces identical reflectance, and concrete has the same spectral signature regardless of its location. Thus it is better to consider changes in the urban and transportation categories together. In combination, these two land use categories increased 23% from an estimated 31,864 ha in 1976 to 39,219 ha in 1991, and the average annual rate of growth appeared to have accelerated from 0.6% per year during the 1976-1985 period to 3.5% per year during the 1985-1991 period.

Estimation of Changes in Ecosystem Services

Using the estimated change in the size of each land cover category together with the ecosystem service value coefficients reported by Costanza et al. (1997), we found that land-use changes in the 141,671 ha of our study area resulted in a $0.78 million per year (i.e., 4% or $5.58 ha^{-1} yr^{-1}) net decline in ecosystem services from 1976 to 1991. Assuming a linear decline in ecosystem services from 1976 to 1991, this decline represents a cumulative loss of $6.24 million in ecosystem services over the 15-year period of the study. However, this cumulative estimate must be regarded with caution because ecosystem services may fluctuate somewhat; our analysis detected a slightly higher ecosystem service value in 1985 than in 1976. In order to accurately estimate the cumulative loss of ecosystem services, it would be necessary to measure ecosystems services at regular intervals, preferably annually.

The estimated change in the annual value of ecosystem services was small in all three watersheds between 1976 and 1991. However, while the change was negative in both the Leon Creek and San Antonio River watersheds ($0.41 and $0.96 million yr^{-1}, respectively) it was positive in the Salado Creek watershed ($0.59 million yr^{-1}). The primary reason for these small changes is that the effect of the estimated decline in rangelands, valued at $232 ha^{-1} yr^{-1}, was largely offset by the increase in woodlands that were valued at $302 ha^{-1} yr^{-1}, 30% higher than rangelands. Thus, while the value of ecosystem services provided by the rangeland category was estimated to have declined by $12.0 million (65%) between 1976 and 1991, the value of the woodland land cover category was estimated to have increased by $10.8 million (403%) during the same period. The estimates also indicated a small increase ($0.62 million yr^{-1}) in the contribution of bare soil to the total ecosystem service value during this same time period.

Our estimated 15-year conversion of 9,511 ha of rangeland, woodland and bare soils to urban and suburban land-uses (residential, commercial and transportation) and the associated loss in ecosystem services appeared to have a small net effect on the annual value of ecosystem services provided within the study area. This is a surprising result given that the residential, commercial and transportation land cover categories were assigned no ecosystem service value while the ecosystem services provided by rangelands, woodlands, and bare soils were assigned values of $232, $302, and $92 ha^{-1} yr^{-1}, respectively. One explanation for the apparent small net effect of land use conversion on the value of ecosystem services in the study area is that the loss of ecosystem services on land being developed was offset by the apparent conversion of other ecologically "less" valuable bare soil (cropland) and rangelands to ecologically "more" valuable woodlands. This ecological conversion is consistent with the increase in distribution and density of woody plants throughout much of central Texas (Smeins et al. 1997).

Ecosystem Services Sensitivity Analysis

The coefficients of sensitivity (CS) of these analyses were less than unity in all cases. The estimated value of the ecosystem service value for the study area increased from a low of 0.03% to 0.06% for a one percent increase in the value of the bare soil coefficient, to a high of 0.31% to 0.85% for a one percent increase in the value of the rangeland coefficient. This indicates that the total ecosystem values estimated for the study area are relatively inelastic with respect to the ecosystem service coefficients. While this implies that our estimates were robust, highly under or over valued coefficients can substantially affect the veracity of estimated changes in ecosystem service values over time even when the coefficients of sensitivity are less than unity. For this reason we also report on the effect of large variations in coefficient values on the estimated value of land-use related changes in ecosystems services.

When the rangelands coefficient was increased by 50% the ecosystem services in the study area decreased in value by 22.0% (-$6.89 million yr^{-1}) between 1976 and 1991, which was a substantially greater decrease than the initial 4% decline. A 50% decrease in the rangeland coefficient resulted in a 43.6% ($5.62 million yr^{-1}) gain in ecosystem services during the same time period. Increasing or decreasing the rangeland coefficient by 50% affected the estimated 1976 ecosystem service value more (±42.6%) than the 1991 value (±15.3%). Conversely, when the coefficient for woodlands was increase by 50% the value of ecosystem services grew by 19.8% (\$4.62 million yr^{-1}) between 1976 and 1991, but a 50% decrease in the woodland coefficient led to a 28.10% (-\$5.88 million yr^{-1}) decrease in value. In contrast to rangelands, increasing the woodland coefficient by 50% affected the estimated 1976 ecosystem service value less (±6.1%) than the 1991 value (±31.9%). These differences between land over types can be accounted for by the decrease in size of the ecologically "less valuable" rangelands between 1976 and 1991 and a concomitant increase in size of the "more valuable" woodlands.

Because of the uncertainty about the relative ecological value of rangelands and woodlands, perhaps most relevant was the result of the sensitivity analysis in which the woodland coefficient was equated to that of rangelands (\$232 ha^{-1} yr^{-1}), the underlying assumption being that the transformation of rangelands to woodlands results in no overall change in ecosystem services. In this scenario, the annual value of ecosystem services between declined by 15.4% (\$3.29 million yr^{-1} in the 141,671 ha of our study area, i.e., \$23.22 ha^{-1} yr^{-1}) between 1976 and 1991. This is a substantially greater loss than the originally estimated 4% (\$5.58 ha^{-1} yr^{-1}) decline in annual ecosystem service values. Assuming a straight-line decline this would equate to a cumulative loss of \$26.32 million over the 15-year period of our study compared to the estimated cumulative loss of \$6.24 million using the original coefficient values.

Increasing the value coefficient of bare soils by 50% resulted in a reduced loss of annual ecosystem services from 4% to 2.1% between 1976 and 1991, while decreasing the coefficient by 50% resulted in an increased loss of 5%. These comparatively small changes were mainly because of the small size of this land cover category.

Discussion

Remote sensing from satellites may be the only economically feasible way to regularly gather information with high spatial, spectral, and temporal resolution over large areas (Verstraete, et al., 1996). This advantage will increase as the cost of obtaining such data declines and computational power to cope with larger data sets from higher resolution sensors increases. However, one limitation for conducting time series analyses of land-use changes using remotely sensed data is that satellite data from high-resolution detectors have a relatively short history. Even LANDSAT data cannot be used for analyzing land-use changes prior to 1972. Because of the lack of recorded historical land cover data, one further limitation of using remotely sensed data to determine changes in land-use patterns is the difficulty of measuring uncertainty about land-use classification. Congalton and Green (1999) state "To date, no standard accuracy assessment technique for change detection has been developed." They propose the use of an error matrix, but this assumes that some reference data exist. Without such reference data, the investigator is left with two choices: do no analysis, or conduct the analysis with the full knowledge that there is uncertainty associated with historical land use classification. Even with uncertainty, simple classification schemes such as those used in this study are reasonable because the distinction between gross land covers is relatively high (Congalton and Green, 1999). Thus, despite the uncertainty of classification, the relatively course spectral resolution of early sensor technology, such as LANDSAT MSS, can be useful for estimating broad scale land-use changes within watersheds.

One challenge for identifying land-use changes was that different land uses might produce similar spectral signatures. For example, it was impossible to determine whether changes in the extent of bare ground were due to changes in the area of cropland, denuded rangeland resulting from drought or overgrazing, or new construction. In Bexar County, most crops are harvested by August, the month in which two of the three images used in the study were taken. This resulted in a lack of distinction between post-harvest cropland and other bare soil surfaces. In addition, areas designated as rangelands in 1976 appear to have converted to woodland in the later images. Such increases in woody plant distributions have been associated with long-term overgrazing in many areas in Texas since the 1880s as well as fire suppression, which is exacerbated in areas with rapid development (Smeins and Merrill, 1988). Nevertheless, there was some uncertainty about the true extent of the rangeland to woodland conversion. As satellite data with greater spectral and spatial resolution becomes more readily available such land-use ambiguities are likely to decrease.

While Costanza et al.'s ecosystem service values that we used in our analysis have been challenged on theoretical and empirical grounds, they represent the only set of valuation coefficients for a wide array of biomes each of which encompass several related ecosystems. Also important is the realization that absolutely accurate coefficients are often less critical for time series than cross-sectional analyses because coefficients tend to affect estimates of directional change less than estimates of the magnitude of ecosystem values at specific points in time. We were primarily interested in changes in ecosystem services over time, and the scope of our project precluded us from deriving area specific ecosystem value coefficients.

Based on the estimated size of six land cover categories and Costanza et al.'s ecosystem services values for related biomes, we determined that the total annual ecosystem service values in Bexar County declined from $21.94 to $21.16 million per year from 1976 to 1991. Thus, while there appeared to be a 65% decrease in the size of rangelands and a 29% increase in the area of the urban land-use categories, we estimated there to have been only a 4% ($5.58 ha^{-1} yr^{-1}) loss in the annual value of ecosystem services. Assuming a straight-line decrease, this represents a $6.24 million 15-year cumulative loss. This relatively small decline is largely attributable to the fact that the estimated $12.2 million per year loss in ecosystems services delivered by rangelands was largely offset by the estimated $10.8 million per year increase in ecosystem services from woodlands. When we assumed that the shift of rangelands to woodlands produced no net change in the value of ecosystem services per hectare, the loss in the estimate annual ecosystem service value between 1976 and 1991 grew to 15.4% ($23.22 ha^{-1} yr^{-1}), representing a $26.32 million 15-year cumulative loss if the decline is linear.

While our study showed that urban sprawl in Bexar county resulted in a decline in the value of ecosystem services delivered by the affected land, it also showed that changes in the value of ecosystem services over time depend on the interaction of changes in various land cover types. Our study suggests that urban spread may not necessarily lead to a large net decline in ecosystem services if there is a concomitant increase in size of other land cover types that provide a greater level of ecosystem services. This is not to say that urban sprawl is beneficial for the delivery of ecological services, but that the negative impacts of the spread of urban and suburban land uses can be potentially offset by other mitigating changes in land cover. This is important because it is unlikely that with increasing human population pressure, the conversion of land to urban and suburban land use will cease or even dissipate in the near future.

Our study showed that LANDSAT data can be used to obtain coarse estimates of changes in ecosystem values at the watershed level. However, in order for this type of analysis to become valuable for policy formulation affecting land-use, it is imperative to obtain a wider array of value coefficients for ecosystem services that more accurately reflect local conditions. This is no easy or costless exercise. One approach could be to identify benchmark ecosystem service values for dominant ecosystem types within a region and then to evaluate the ecosystem services provided at specific locations relative to the representative benchmark. Since ecosystem services are not traded and, therefore, have no market-based price, indirect valuation techniques will be required to obtain these estimates. Such an approach would be analogous to the evaluation procedure developed by Dyksterhuis (1949) for determining the ecological condition of rangelands, and it would allow systematic estimation of changes in the ecological services delivered at specific locations over time. Such information could be extrapolated to larger watershed and regional scales though the use of remotely sensed data and GIS tools to classify land into representative ecosystems for which benchmark values have been established.

Tools for this sort of analysis are becoming increasingly available in common GIS systems. While we limited the use of GIS tools to quantifying land-use change, such tools can also be used to conduct a variety of additional analysis because the data are spatially explicit. Ecological economists could incorporate variables such as patch size, edge effect, contiguity, wildlife corridors, biodiversity potential, to name a few, in their analyses to better evaluate changes in land-use from an ecosystem service perspective. For example, in future watershed level studies, quantifying the location and sequence of different land cover types relative to watercourses could be used to facilitate ecosystem value estimates. Thus, increasingly sophisticated GIS tools could be used in conjunction with ecosystem benchmarking to efficiently and accurately estimate to impact of land use conversion on the services provided by the affected ecosystems.

From a policy stand it is also critical that the factors facilitating urban sprawl be clearly understood and that rural landowners be provided with incentives that minimize the negative impacts from such changing land use. For example, weak local government, developer pressure, high estate taxes for rural landowners, and greater urban affluence are major causes of rapid rural land fragmentation and urban sprawl west of San Antonio and Austin. Unless landowner's have positive incentives to sustain rural land-uses, urban sprawl will increasingly impact ecosystem services in these areas even when we are able to accurately quantify the loss of ecosystem services associated with such changes in land-use.

Periphyton as Bio-Indicators

Eutrophication is an increasingly recognized threat to the health of streams and rivers in the United States. The U.S. Environmental Protection Agency’s 1998 National Water Quality Inventory Report to Congress reports that 35% of assessed streams and rivers do not meet water quality standards set for them, and excess nutrients are a leading pollutant in these impaired waters (USEPA, 2000a). Elevated concentration of nutrients—particularly nitrogen and phosphorous—is widely recognized as a cause of increased algal growth (USEPA, 2000b). The negative effects of increased algal growth include increased drinking water treatment costs, interference with water–based recreation, and fish kills caused by excessive swings in diurnal dissolved oxygen.

Reducing nutrient loads to streams and rivers is an especially formidable task given the diverse and diffuse sources in any given watershed. There are considerable technical, legal, social, and financial uncertainties associated with identifying and quantifying nutrient sources and imposing restrictions on them. Given this challenge, it may be preferable in some cases to address problems of eutrophication by targeting factors other than nutrients. Specifically, there may be cases where reducing periphyton growth can be effectively and inexpensively controlled by factors such as light and temperature, in lieu of or in addition to nutrient control.

Light and those factors affecting light availability on substrate surfaces regulate periphyton growth (Weitzel, 1979). Numerous researchers have investigated the relationship between light and the structure and function of periphyton in streams (DeNicola et al., 1992; DeNicola and Hoagland, 1996; Hetrick et al.,1998; Mosisch et al., 1999). Heppinstall and Fuller (1994) used cloth shading to manipulate light levels and found that periphyton growth was primarily limited by light and secondarily, during seasons when light intensity was greatest, by phosphorous. DeNicola et al. (1992) provide a more detailed analysis, reporting the growth of periphyton by taxon at four sites with different light availability. They reported that the distributions of several species were related to the degree of canopy cover at the sites.

Other researchers have investigated photoinhibition in periphyton, defined by Dodds et al. (1999) as “a statistically significant decrease in photosynthetic rate when irradiance increases past the point of maximum photosynthesis.” Examining fifteen whole communities, Dodds et al. (1999) found that none exhibited photoinhibition at the highest values of irradiance to which they were naturally exposed, a result consistent with that reported by Boston and Hill (1991). Hill (1996) suggested that whole periphyton communities should only demonstrate photoinhibition when they are acclimated to shaded conditions and then moved to full sun.

There have been efforts to link structural measures of periphyton (e.g., biomass, chlorophyll-a, and taxon) to functional characteristics (e.g., carbon fixation). Dodds et al. (1999) found a positive correlation between periphyton expressed as aerial chlorophyll-a and maximum photosynthesis. The correlations were negative when periphyton was expressed as biomass (instead of chlorophyll-a), suggesting that algal community mass and chlorophyll-a content may not be strongly correlated. Boston and Hill (1991) also observed a negative, but insignificant, correlation between photosynthesis and both light and biomass, possibly due to shading of cells deep within the periphyton matrix. Hill et al. (1995) reported that shaded periphyton grown on clay tiles was twice as efficient at low irradiance than was periphyton from open sites.

Integrated primary productivity at these sites was still four times lower than open sites because the increased efficiency only partially compensated for the low irradiance in the shaded sites.

Missing from the literature is a quantified relationship between periphyton growth and light availability that is of direct use to water resource managers. There is ample evidence that light and chlorophyll-a are positively correlated, but efforts to quantify that relationship are hampered because varying nutrient levels and other factors may be responsible for the observed response. The purpose of this study is to determine if light availability is the primary determinant of periphyton growth under both ambient and nutrient-enriched conditions. Specifically, this study tests the following null hypothesis: light availability does not explain the majority of observed differences in periphyton growth between locations.

Methods

Matlock Periphytometers (Matlock et al., 1998) were used to measure the response of periphyton under controlled and nutrient enriched conditions at four locations near San Antonio, Texas during July and August 2000. Relative light availability at each location was measured with inexpensive, battery-powered light meters. Linear regression was used to analyze the relationship between light and periphyton growth.

Site Description

The four locations used in this study include the natural stream channels and adjacent artificial channels on Leon and Salado Creeks, San Antonio, Texas (Figure 1). Leon and Salado Creeks originate in from springs in the limestone formations north of San Antonio, and flow along the western and eastern sides, respectively, of the metropolitan area. At the study sites, the streams are fourth-order, perennial waterways. The artificial channels were constructed on the banks of the creeks approximately four meters from the water's edge. The sites were selected for their accessibility, perennial flow, and range of available light conditions. Table 22 describes the shade characteristics of each site.

Table 22. Shade Characteristics at the Four Study Sites

Location	Shade Characteristics
1. Leon Creek Stream	No overhead canopy. Tall vegetation and steep bank on one side. Open for 100 degrees between horizons.
2. Leon Creek Artificial Channels	No overhead canopy. Open for 150 degrees between horizons.
3. Salado Creek Stream	Approximately 40% canopy cover.
4. Salado Creek Artificial Channels	Approximately 80% canopy cover.

Measuring Periphyton Growth

Modified Matlock Periphytometers (Matlock, 1998) were used to deliver nutrients via diffusion to an artificial growth surface. In this method, nutrients dissolved in water diffuse out of a submerged plastic bottle through a nylon membrane and glass fiber filter (Figure 4). Algae colonize the filter of each bottle. The Matlock Periphytometer consists of forty bottles (10 control replicates and 10 replicates of three different nutrient treatments) are attached to a rigid galvanized steel frame. PVC pontoons on two ends allow the rack to float with the bottles submerged several inches below the water surface (Figure 5).

For this study, the 1.0 liter bottles of the original Matlock Periphytometer were replaced with 0.25 liter bottles. The bottles used in the natural stream channels were attached to a floating rack as shown in Figure 5. The bottles used in the artificial channel were attached to a 0.61 by 2.4 meter rack that fit inside the artificial channel. Twenty bottles were attached to each side, and the rack was then secured 5 cm below the water surface in two of the four channels (Figure 6). Two artificial channels were used as insurance against equipment failure. The remaining two artificial channels at each site were not used in this experiment.

The forty bottles used in each natural stream and in each artificial channel were divided equally as follows:

1. Control (C): water treated by reverse osmosis (RO).
2. Nitrate (N): 26.1 mg/L NO_3 -N in the form of $NaNO_3$ in RO water.
3. Phosphate (P): 86.7 mg/L PO_4-P in the form of $Na_2HPO_4*7H_2O$ in RO water.
4. Combined Nitrate and Phosphate (NP): 26.1 mg/L NO_3 -N and 86.7 mg/L PO_4-P in the forms listed above in RO water.

The Matlock Periphytometers were retrieved after fourteen days. Each glass fiber filter growth surfaces was removed and placed in 7 ml of 90% acetone solution saturated with magnesium carbonate, wrapped in aluminum foil, and placed on ice for transport back to the laboratory. Chlorophyll-a analysis was performed in the laboratory using the tri-chromatic method (APHA, 1989). The results were expressed as mass chlorophyll-a per unit surface area of the exposed filter.

Measuring Light Availability

Light Meters

The light intensity at each site was measured and logged with HOBO LI meters (Onset Corporation, 1996). These inexpensive, battery powered, palm sized meters detect intensities from about 0.01 to 10,000 lumens/ft2 and store the data at user-specified intervals for later downloading to a computer. The meters are sensitive to wavelengths from approximately 350 nm and 1100 nm, and are most sensitive to wavelengths of near 900 nm. The meters are calibrated at the factory to provide accurate readings of incandescent light.

The HOBO LI meters are designed to provide reliable comparisons of the *relative* light available in different locations. Except for incandescent light sources, they are not designed to provide accurate *absolute* measures of light intensity. The peak sensitivity of the meters is considerably higher than the peak intensity of sunlight (approximately 500 nm), which results in readings of sunlight that are low by a factor of approximately 6 (Onset Corporation, 1995).

We programmed the meters to take readings at 15-minute intervals and then sealed them in clear plastic bags to seal out moisture. The meters were secured to the Matlock Periphytometers approximately 5 cm above the water surface and deployed at the four sites (Figure 32).

Figure 32. HOBO LI Meter Attached to Matlock Periphytometer

Correcting for Light Meter Bias

After retrieving the light meters and downloading the data, the meters were again set to record at 15 minute intervals, sealed in plastic, and then set side-by-side outside for 72 hours. We calculated the average measurement of the data from all four meters over the 72 hours, and compared this overall average to the average reading for each for each individual meter. The ratio of the overall average to the individual average was used as a correction factor. For example, the overall average of the meters over 72 hours was 76.6L/ft^2, the individual average of one meter was 77.7 L/ft^2, which gives a correction factor of 0.985. So, the data recorded by this particular meter during the fourteen-day deployment were multiplied by 0.985 to arrive at a corrected value.

A more complex correction approach that adjusted each individual measurement over the 14 days (as opposed to correcting a single, average value) was also explored. This complex correction yielded very similar results to the simple approach, so the added complexity was deemed unwarranted.

Results and Discussion

Periphyton Growth as a Function of Light

The relative average light intensity and the mean water temperature recorded at the four locations are shown in Table 23. The temperature data are means of the 15-minute samples recorded continuously during the 14-day study. The Leon Creek Artificial Channels had the greatest relative light intensity (1.00), and the Salado Creek Artificial Channels had the lowest (0.24). The measured chlorophyll-a values are presented in Table 24.

Table 23. Relative Light Intensity and Mean Temperature at the Four Study Sites

Location	Relative Light Intensity (0.0-1.0)	Mean Water Temperature (˚C)
1. Leon Creek Stream	0.74	27.33
2. Leon Creek Artificial Channels	1.00	28.36
3. Salado Creek Stream	0.31	27.28
4. Salado Creek Artificial Channels	0.24	27.23

Table 24. Measured Chlorophyll-a Under Control and Nutrient Enriched Conditions

Location	Control			Nutrient Enriched		
	Mean Chl-a (mg/m²)	Stnd Dev (mg/m²)	Sample Size	Mean Chl-a (mg/m²)	Stnd Dev (mg/m²)	Sample Size
1. Leon Creek Stream	9.16	2.07	10	58.00	8.19	10
2. Leon Creek Artificial Channels	11.63	2.58	19	85.91	27.85	20
3. Salado Creek Stream	8.06	1.48	10	8.93	1.48	9
4. Salado Creek Artificial Channels	6.98	1.35	20	6.90	1.62	20

The chlorophyll-a data for both control and nutrient enriched conditions were strongly correlated to the light data as demonstrated by coefficients of determination of 0.919 and 0.997, respectively. These data indicated that the availability of light explains 91.9% of the variation in periphyton levels under control conditions (no nutrient addition), and 99.7% under the combined nitrate and phosphate treatment. Relative light (dimensionless, between 0 and 1) predicts chlorophyll-a (mg/m^2) according to the following equations:

Control Conditions: Chl-a = 5.30 * Light + 5.92 (1)
Nutrient Enriched Conditions: Chl-a = 107.36 * Light - 21.59 (2).

The relationship between relative light and chlorophyll-a are shown in Figures 33 and 34. The data points on each plot were compared using the Student-Newman-Keul's (SNK) Test (α=0.05) in SAS/STAT© (SAS Institute Inc., 1990), and data points on each plot with the same letter are not significantly different. Under control conditions, all mean four chlorophyll-a values are significantly different. Under enriched conditions, the two lowest mean chlorophyll-a values (found at the more shaded Salado Creek Stream and Artificial Channels sites) are not significantly different from each other, but are different from the values measure at the Leon Creek sites.

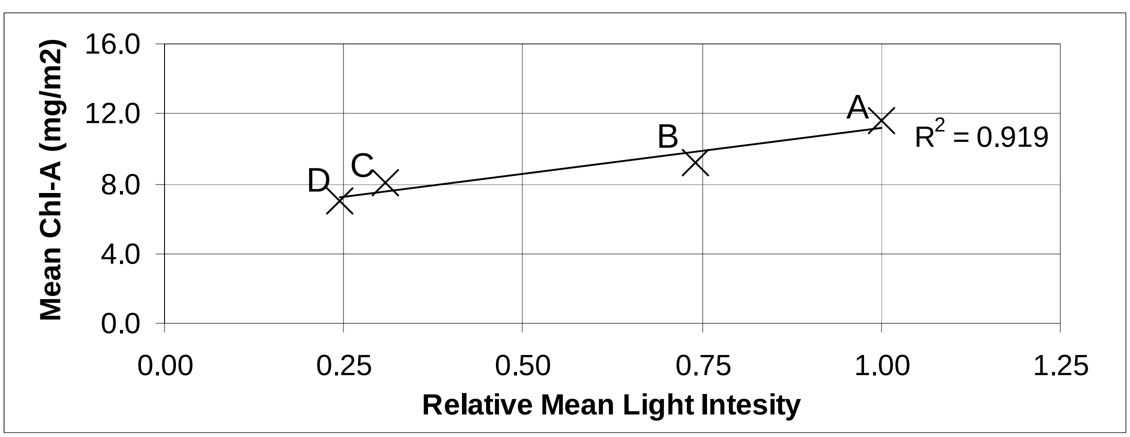


Figure 33. Chlorophyll-a as a Function of Relative Light Intensity Under Control Conditions

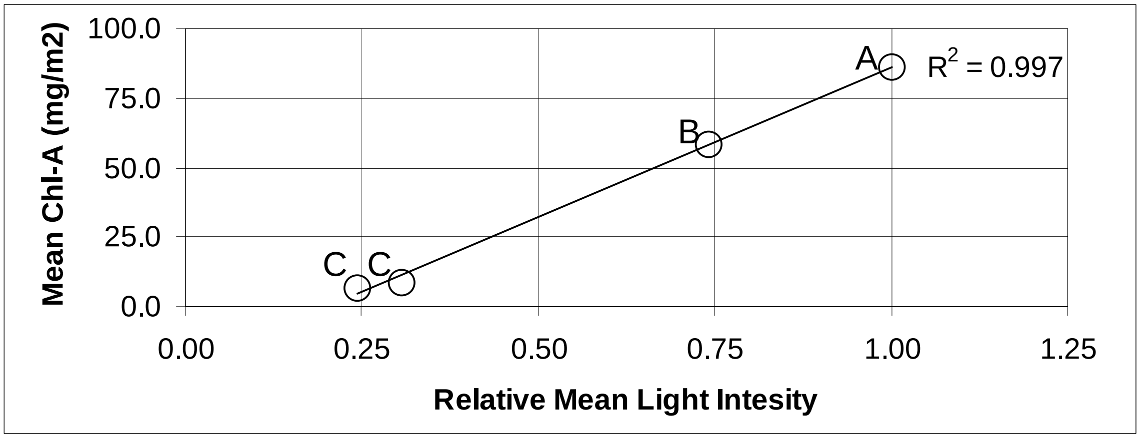


Figure 34. Chlorophyll-a as a Function of Relative Light Intensity Under Nutrient-Enriched Conditions

Establishing Causality Between Light Availability and Algal Growth

Correlation does not guarantee causality, but there is a strong case to be made for it here. The generally accepted factors controlling algal growth and accumulation are water movement (particularly as it relates to scouring), macro and micronutrient availability, temperature, and light (Weitzel, 1979). Under the NP treatment, macronutrients are not limiting. It is reasonable to assume that micronutrients are also not limiting the growth of algae given the Karst origins of these waters and the thriving communities of rooted macrophytes that are present in many shaded reaches. It is unlikely that scour reduced the periphyton measurements made here since the algal growth surfaces of the Matlock Periphytometers are recessed slightly below the level of the drilled-out bottle caps, and are further protected by the aluminum screening wrapped over the top of each bottle. With nutrient availability and scour removed as controlling factors, the periphyton growth observed here becomes a function of only light and temperature.

Analysis of the temperature and relative light data (Table 23) shows that they are relatively strongly correlated ($R^2 = 0.67$), as would be expected in these relatively slow moving streams in which the impact of direct and indirect solar radiation likely controls water temperature much more than evaporative, convective, and conductive fluxes (Walling and Webb, 1992). So, the methods used in this study make periphyton growth a function of light and temperature. And since temperature can be explained largely by light, it is reasonable to present a causative model of periphyton growth as a function solely of light availability.

Applicability of the Model

While the light data are reported here only as relative measures between sites, the four sites represent a wide range of light conditions. Based on our knowledge of the area, these conditions capture all but a small fraction of the riparian conditions found in the watersheds, which suggests that additional data points will confirm the observed relationship between light and chlorophyll-a. The models presented here (Equations 1 and 2) are therefore relatively robust.

Conclusion

Light availability explains 91.9% of the variability in chlorophyll-a measured under ambient (control) conditions in these streams, and 99.7% of the variability when nutrient limitation is removed (NP Treatment). These findings have important implications for the management of streams. They suggest that a focus on nutrient control in these waters may be misguided since algal growth is so strongly linked to light availability.

Additional data collected from other streams, during different seasons, and under different light regimes would enhance the applicability of the model. Calibrating the HOBO LI meters against more accurate light meters so the actual, rather than relative, light intensity could be reported would further enhance this work and allow comparisons with previous light-algae studies.

Ultimately, this work would be most valuable if accompanied by a correlation between light and density of riparian vegetation. This added piece of information would allow for a model of periphyton growth as a function of riparian cover. Such a model would allow resource managers and planners to make direct use of the results by prescribing critical tree densities to achieve a target chlorophyll-a level. For example, managers of a stream may wish to keep periphyton levels below 100 mg/m^2. Using models of periphyton as a function of light and light as a function of vegetation density, they could determine the necessary density of riparian plantings. A similar approach could be used to determine the necessary shading to prevent unacceptable swings in diurnal dissolved oxygen brought on by high periphyton levels.

Summary

Eutrophication of streams and rivers is recognized as a growing threat to the health of aquatic ecosystems throughout the United States. There is an immediate need for methods to quantify the extent of eutrophication, identify the causes, and predict future conditions given changes in watersheds. These methods must be accessible—in technical and financial terms—to those in charge of managing water resources. Nutrient-diffusing substrates have shown promise as a method to study the impact of nutrients on periphyton. The Matlock Periphytometer is particularly effective because constructing it requires only materials that are commonly available, and analyzing the data requires skills and equipment common in most laboratories. The first study showed that nutrient delivery via diffusion elicits a greater periphytic algae response than nutrient enrichment via whole-channel dosing. This finding is important because it illustrates a limitation of all diffusing substrate methods currently available. Specifically, the results of a diffusing substrate study are not necessarily applicable to the natural stream channel. This has been a reasonable criticism of artificial substrates, and it is even more legitimate in the case of diffusing artificial substrates where both the growth surface and the nutrient delivery method are artificial. While the findings presented here illustrate some limitations of diffusing substrate methods, there is nothing to suggest that they do not remain legitimate tools for comparing the relative influence of different nutrient treatments as might be done to determine limiting nutrients.

The modified Matlock Periphytometers (Matlock, 1998) used in this study replaced the 1.0 liter bottles of the old system with 0.25 liter bottles. This change reduced the weight of the system by 30 kg and reduced its dimensions by nearly half. The result is a system that is far easier to transport and deploy than the original, making it a more viable method for others to use.

The artificial channels used in the study were successful in that they required little maintenance, allowed individual adjustment of flows in each channel, allowed only minor temperature increases, and provided reasonably precise nutrient dosing rates. The major limitation of the design was the flow rate in the channels. Logistical and financial constraints limited us to using 12 Volt DC-powered pumps to deliver water from the stream to the artificial channels. These pumps did not have the capacity to simulate the flow in the natural channel. As a result, the velocity through the artificial channels, and thus the hydraulic residence time, was approximately an order of magnitude less than that of the stream.

The second study showed that light availability explains 91.9% of observed variations in periphyton growth under control conditions, and 99.7% under nutrient-enriched conditions. Equations to predict periphyton as a function of relative light were presented. These results are important because they suggest that a focus on nutrient management in these streams may be misguided since the periphyton growth is so strongly linked to light availability. Nutrient control efforts might still be justified because of nutrient-related problems further downstream. At the study sites, however, more successful anti-eutrophication efforts would focus on increasing riparian vegetation and stream shading.

The HOBO LI meters used to measure light availability have distinct strengths and weaknesses. Because they are relatively inexpensive and easy to operate, they make light measurement a viable option for people investigating eutrophication processes. The meters provide accurate measures of *relative* light intensity, but they do a poor job of measuring *absolute* intensity. As a result, measurements taken in this study were reported only as relative values. It may be possible to calibrate the meters using more sophisticated light meters in order to determine reasonable absolute values, which would allow these results to be compared to studies conducted elsewhere and reported in the literature.

FISH (BLUEGILL) AS ECO-INDICATORS

Median MMS accounted for 35% of the variation in median Wchg of our caged bluegill. When MMS and VO2r (routine metabolic rate) were transformed to their MS equivalent at DO = 3.0 mg/L, median MS accounted for 41% of the variation in median Wchg. Finally, when MMS and VO2r were adjusted for the expected effects of median in-stream temperature (Ta), the percentage of variation in median Wchg accounted for by Ta-adj. MS increased to 47%. The regression of Wchg on Ta-adj. MS had an intercept of -0.40 and a slope of 8.66. We interpret these regression coefficients to mean that the metabolic scope necessary for weight maintenance of our San Antonio bluegill was about 0.40/8.66 = 0.05 mgO2 $\cdot g^{-1} \cdot h^{-1}$, and that, above this maintenance value, about 1/8.66 = 0.12 mgO2 $\cdot g^{-1} \cdot h^{-1}$ of metabolic scope was required for each 1% $\cdot day^{-1}$ change in body weight.

It is of interest to ask, "What might have been the metabolic expenditures of the fish while in the cages relative to those in the respirometers?" If one assumes the caged fish used all their metabolic capacity (at DO = 3 mg/L), then they must have expended about 40% more in routine metabolism in-steam than in the respirometers–i.e., the median values of Ta-adj. VO2r used to compute Ta-adj. MS must be multiplied by 1.4, to shift the regression equation's intercept from -0.40 to zero.

The foregoing analysis of the bluegill ecoassay data, while instructive, was less than gratifying, both because more than half the variation in median Wchg remained unexplained, and because only a small fraction of the explained variation could be attributed to variation in the components of stream environment that we measured. But, up to this point, our attempt to explain the variation in weight-change rate observed for bluegill caged in San Antonio streams had not involved the ecophysiological model–because there was not one for bluegill.

This, we set about trying to remedy. First, the existing ecophysiological model was parameterized for bluegill, using data from the published literature and from the present study. Then the bluegill version of the model was engaged in the process of Wchg evaluation for the San Antonio ecoassay by fitting it to each of the 10 available sets of environmental data (2 of the 12 stream-site-year/season datasets had been lost, owing to environmental-data logging failure) and to the corresponding median value of observed MMS. Model fitting was performed by setting Wfish0 (initial fish weight) to the median initial weight of fish yielding the median MMS; setting post-cage environmental values to the medians of those measured during the respirometry sessions, including terminal DO at the median LOCr (limiting DO for routine metabolism); and, solving for the parameter MMSO (residual intercept of MMS) giving the observed values of median MMS and median VO2r = Mact (active, or maximum, metabolic rate). The resulting set of 10 MMSOs explained 74% of the variation in the corresponding values of median Wchg. Examination of the Wchg vs. MMSO plot suggested that additional variation in median Wchg was associated with some difference between fish responses in summer 1999 and those in summer and winter 2000; and, in fact, exclusion of this variation via multiple regression, indicated that MMSO would have explained 89% of the variation in median Wchg, had the temporal difference not been a factor.

What caused the difference in response between summer 1999, on the one hand, and summer and winter 2000, on the other? Two possibilities are respirometry-system and feeding-regime changes, both of which occurred between the summer-1999 and summer-2000 research sessions. In summer 1999, we used manually controlled respirometers with circulation provided by peristaltic pumps; in summer and winter 2000, we used computer-controlled respirometers with circulation provided by centrifugal pumps (Fontaine 2002). But, in our opinion, this difference in respirometry probably had no systematic effect on the magnitude of MMSO (and certainly could have had no effect on Wchg). Instead, we prefer an explanation linked to the change in feeding regime. In summer 1999, the caged fish were fed chopped, fresh earthworms to apparent satiation on alternate days; in summer and winter 2000, the fish were fed live mealworms to apparent satiation each day. Not only were the earthworm-fed fish fed only half as often as the mealworm-fed fish, but also earthworms have less than half the energy content (per unit live weight) of mealworms–about 770 cal/g for earthworms vs. about 2000 cal/g for mealworms. Earthworm-fed fish achieved about 0.5%$\cdot day^{-1}$ less growth than did mealworm-fed fish at the same level of MMSO. Although the polarity of the offset is logical, favoring the fish with the richer diet, its quantitative significance is obscured by the fact that all fish had potential access to the natural forage that may have entered their cages.

Assuming that the difference in the Wchg vs. MMSO relationship between years was largely the difference between feeding regimes, one could interpret the progressive increase in R^2–from 0.31 for Wchg vs. MMS (using only the 10-pair subset of data for which MMSO could be computed), to 0.74 for Wchg vs. MMSO, to 0.89 for year-adjusted Wchg vs. MMSO–as follows: Metabolic "health" of the fish at the end of the cage trial accounted for 31% of variation in Wchg during the cage trial; physicochemical environment, independent of its effects on metabolic health, accounted for an additional 43% (74 - 31%); quantity or quality of feed accounted for yet another 15% (89 - 74%); leaving only 11% (100 - 89%) unaccounted for. Obviously, much of the variation both in median Wchg and in median MMS and MMSO was seasonal, and presumably temperature-related. Values of Wchg were typical of bluegill in other systems. *No significant fraction of variation in median Wchg (Fontaine 2002), median MMS (Fontaine 2002), or median MMSO (this analysis) could be related to differences between the two streams or to differences between sites within streams.*

The foregoing analysis has this important implication: Much of the variation in fish performance from time to time and place to place can be attributed to variation in the ordinary components of proximal environment. Thus, even in urbanizing streams like those we studied in San Antonio, fish are stunted in their growth or killed, not by mysterious contaminants, but by sub-optimal values of normal environmental variables like temperature and DO and forage. This is not to argue that urbanization has no degrading impacts on aquatic systems. It is to argue that those impacts—arising mainly as nutrient excesses and reduced water inflow—are primarily on stream DO and on carbon-nitrogen dynamics, and only indirectly on the fish by way of their manifestation as DO and forage availability.

These are difficult relationships to grasp, even for trained scientists and engineers. Perhaps, the most important lesson our ecological research team learned in this project, was that our stakeholder groups could and would engage the important technical aspects our findings—if we worked to make the communication process meaningful and efficient. The modeling approach that we used to facilitate our research team's own understanding and interpretation of experimental results turned out to be a natural tool for really substantive communication within the larger team. And, by showing the models in action (please see http://wfscdisted.tamu.edu/wfscdisted/courses/wfsc616/EPAdemo.exe)—as opposed to only showing modeled outcomes—we developed the kind of reciprocal exchange of ideas within the larger team that we researchers found not only rewarding and heartening but, at moments, actually astonishing. We were on the verge, by project's end, of capturing the vision stated in our title: *Development of an Urban Watershed Rehabilitation Method Using Stakeholder Feedback to Direct Investigation and Restoration Planning*. It was a privilege to have been a part of the effort.

Salado Creek Fecal Coliform Modeling

Model Calibration for Hydrology

The hydrology calibration of HSPF was done in four steps:

1. Developing an overall water mass balance by adjusting precipitation, evapotranspiration, and loss to deep groundwater
2. Adjusting the high-flow/low-flow distribution by adjusting percolation rates, groundwater recharges, and re-emergence of water to streams
3. Matching peak storm volumes and adjusting the number of days required for flow to return to normal levels
4. Fitting the seasonal distribution of flows considering the seasonal variation in evapotranspiration, soil moisture, and changes in groundwater recharge to streams.

The hydrology calibration was carried out using the expert system HSPEXP. For running HSPEXP, eight output time series, simulated total runoff (SIMQ), simulated surface runoff (SURO), simulated interflow (IFWO), simulated baseflow (AGWO), potential evapotranspiration (ET) (PETX), simulated actual ET (SAET), upper zone storage (UZSX), and lower zone storage (LZSX) were required. The input file for HSPF called as User's Control Input (UCI) was modified to incorporate these output time series. Following advice from the HSPEXP expert system and also in accordance with the four calibration steps, the input parameters were modified. To get an overall water balance the parameter that controls the loss of water to deep groundwater, DEEPFR in the pervious land segment (PLS) was increased considerably. Increasing DEEPFR decreases the runoff from the PLS. Lower zone nominal storage (LZSN), which controls evapotranspiration, was adjusted next. The other parameters adjusted during the first round of calibration were, monthly lower zone ET parameter (MON-LZETPARM), monthly interception storage capacity (MON-INTERCEP), fraction of potential ET that can be satisfied from baseflow (BASETP), and fraction of potential ET that can be satisfied from active groundwater storage (AGWETP).

The second step in the hydrology calibration was to match the high-flow/low-flow distribution in the monitoring data. This was achieved by adjusting model parameters representing infiltration (INFILT), interflow (INTFW), and groundwater recession (AGWRC).

The third step in the hydrology calibration was to match storm flow. The parameters in the PLS such as the interflow recession parameter (IRC), upper zone nominal storage (UZSN), and surface flow parameters such as the length of the overland flow plane (LSUR), Manning's n for the overland flow plane (NSUR), and slope of the overland flow plane (SLSUR) were adjusted to match observed and simulated storm flow.

The fourth and final step in the hydrology calibration was to match the seasonal distribution of flows. This was done by adjusting the parameters that control the seasonal characteristics of the model like MON-INTERCEP, MON-LZETPARM, UZSN, BASETP, and a parameter that affects the behavior of groundwater recession flow (KVARY).

Additional calibration was done to parameters like retention storage capacity (RETSC), length of the assumed overland flow plane (LSUR), and slope of the overland flow plane (SLSUR) in the impervious land segment (ILS). After adjusting the above parameters the annual average observed flows and the simulated flows were found to have very good correlation with an R^2 above 0.90. However, the base flows were under-predicted throughout the simulation period. The seepage from the Edwards Aquifer beyond the extent of the Salado Creek Watershed caused an increase in observed base flows. In order to account for this additional flow, a point source of inflow was added at the middle of the watershed with a mean value of 5 cfs.

Hydrology calibration started with long term overall matching of observed and simulated flows. The model simulations were done for a time period of January 1, 1970 to December 31, 1993. Figure 35 shows observed flow at USGS gauging station 08178800 and the simulated flow at the main outlet of the Salado Creek corresponding to a long term calibration plotted against time. Also the comparison between the observed and simulated annual flows at the same stream location is given in Figure 36.

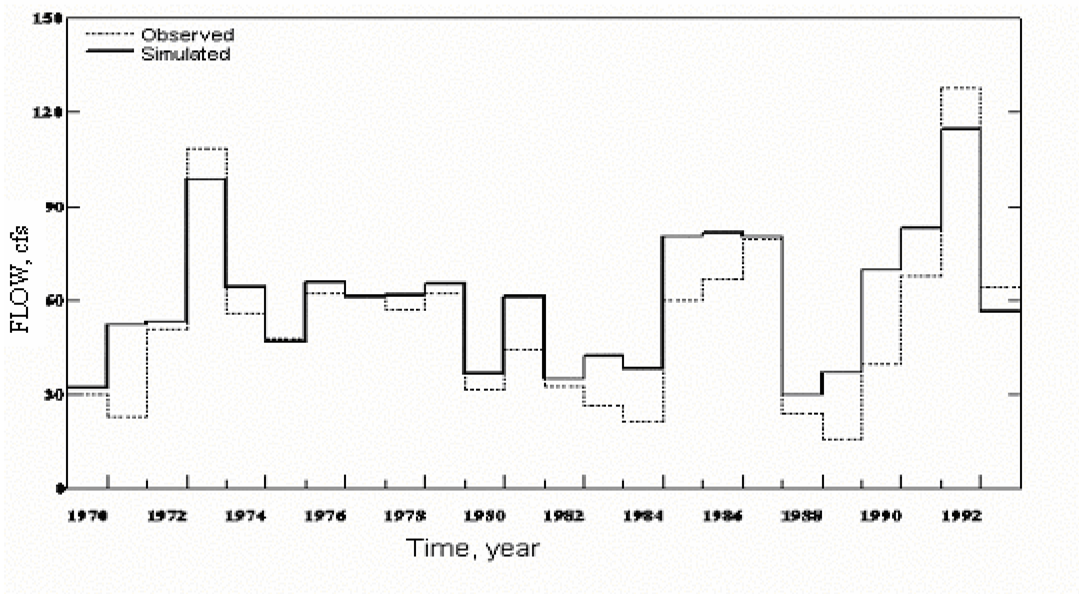


Figure 35. Analysis Plot for Yearly Mean Streamflow at the Outlet of Salado Creek

The model parameters were further adjusted to get a close correlation between the observed and simulated daily flows for three years, from 1991 through 1993. The comparison of the flows during this time period for the above gauging location is shown in Figure 37.

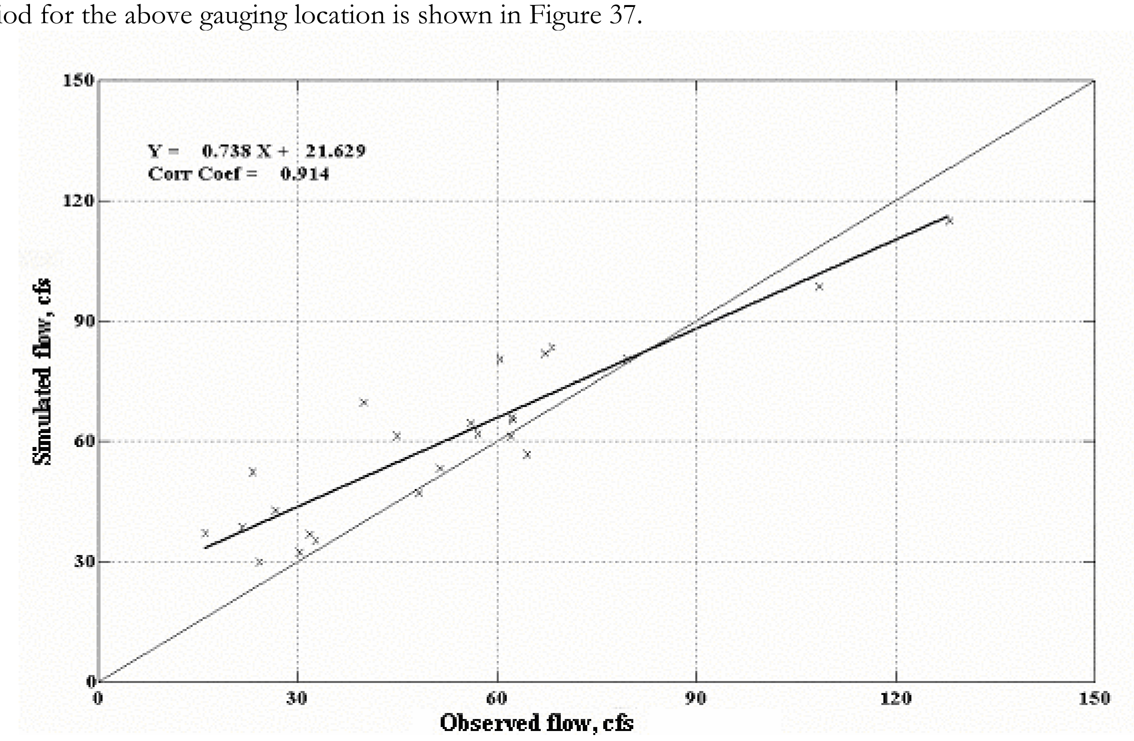


Figure 36. Scatter Plot for Yearly Mean Streamflow at the Outlet of Salado Creek

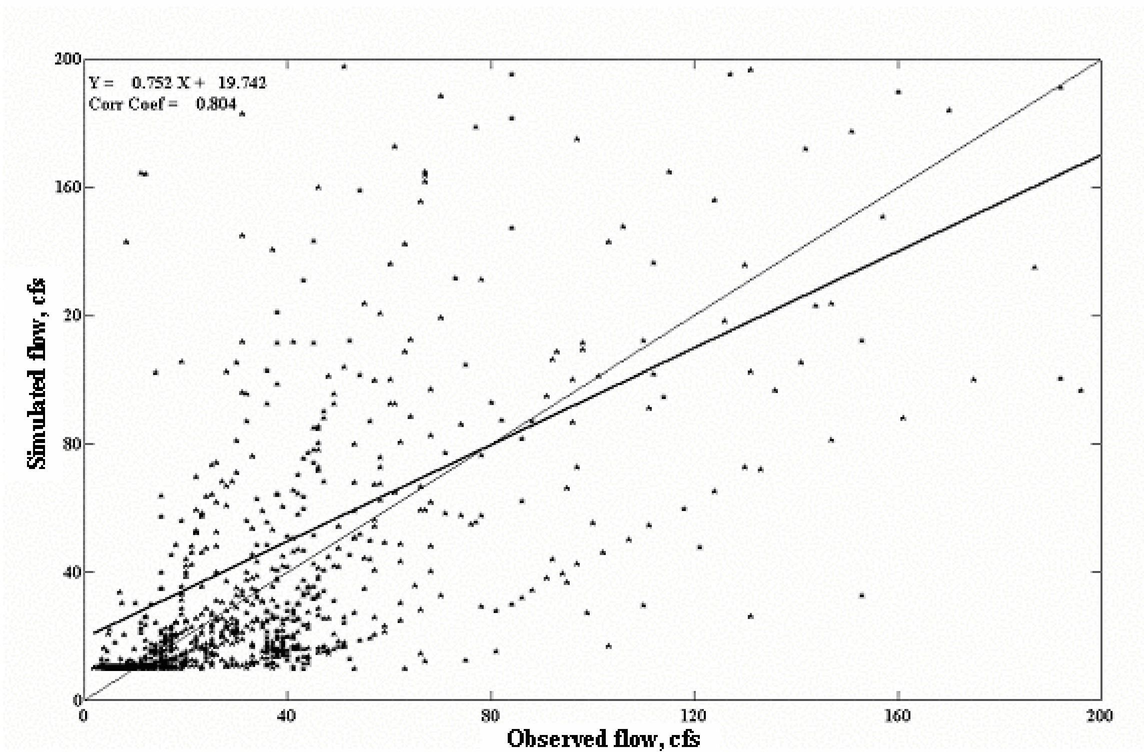


Figure 37. Scatter Plot for Daily Mean Streamflow at the Outlet of Salado Creek

Model Validation for Bacteria

Once the hydrology component of the model was properly calibrated for the study area, the next step was to simulate the in-stream fecal coliform concentrations and compare them with the observed data. Availability of observed water quality data was very limited, making it difficult to do a reasonable calibration. The current study aims only to quantify parameter uncertainty of the HSPF model, therefore, calibration of the bacterial model was not necessary. However, HSPF model was tested to observe how well the model can represent the dynamics of the fecal coliform in-stream. Figure 38 and 39 show simulated and observed in-stream concentrations of fecal coliform at the main outlet of the Salado Creek corresponding to years 1994 and 1996 respectively. The HSPF model produces a continuous output of in-stream fecal coliform concentration whereas the water quality observations correspond to grab samples obtained at discrete time periods.

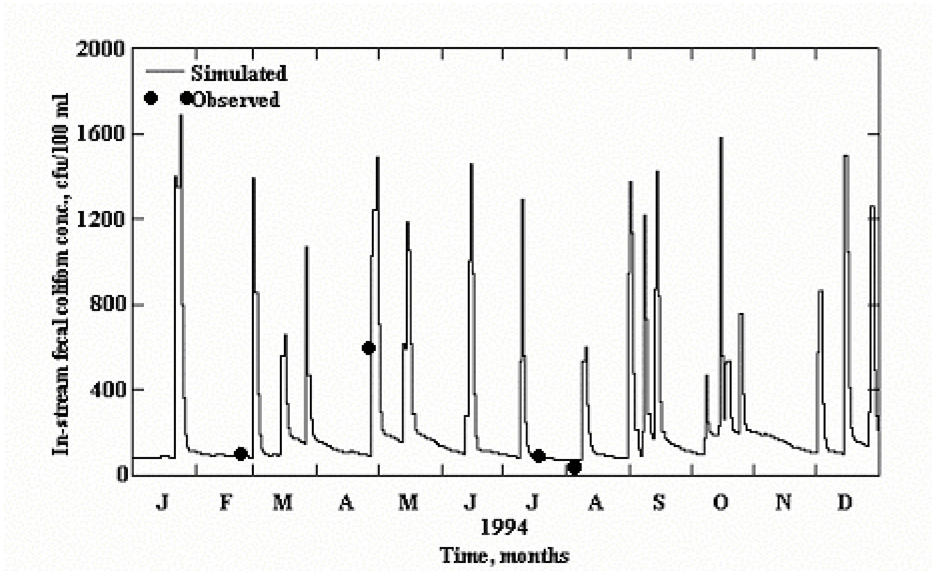


Figure 38. Fecal Coliform Concentration at the Outlet of Salado Creek for 1994

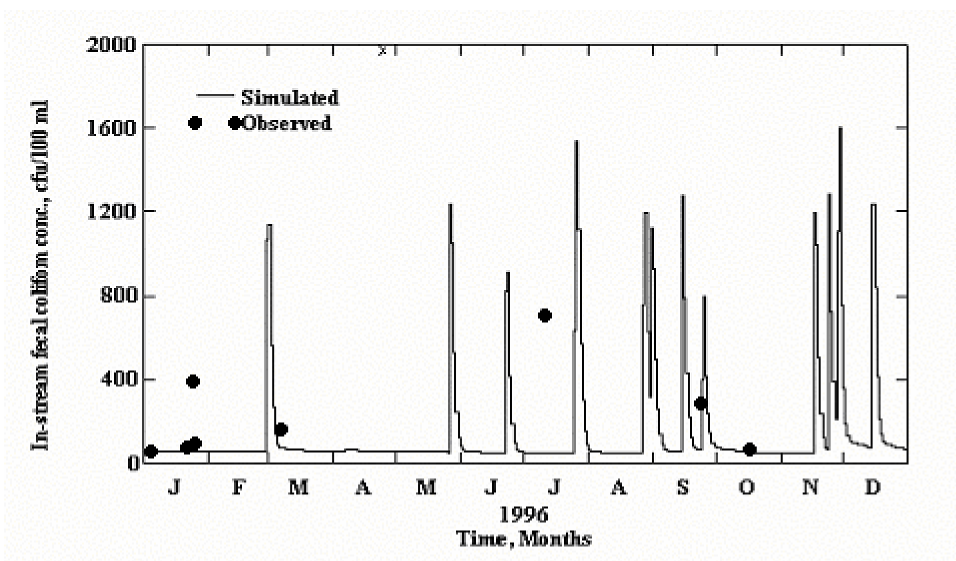


Figure 39. Fecal Coliform Concentration at the Outlet of Salado Creek for 1996

Sensitivity Analysis

A sensitivity analysis of the HSPF model was done to determine which parameters, when changed, caused the greatest change in the peak in-stream concentrations of fecal coliform in at the outlet of the Salado Creek. Input parameters were varied by ±10% of P to observe the effect on the output parameter O. Table 25 shows the results of the sensitivity analysis. The peak concentration of fecal coliform at the outlet of Salado Creek was most sensitive to the maximum storage of fecal coliform on the land surface (SQOLIM), the rate of surface runoff that will remove 90 percent of fecal coliform from the land surface (WSQOP) of PLS, the temperature correction coefficient for first-order decay of fecal coliform (THFST), the in-stream water temperature (TWAT), and the first-order decay rate for fecal coliform (FSTDEC) of RCHRES. Further uncertainty analysis was concentrated on these parameters.

Table 25. Absolute Sensitivity (S) and Relative Sensitivity (Sr) of Peak In-stream Fecal Coliform Concentrations (PFC, cfu/100 ml) to HSPF Input Parameters for the Salado Creek Watershed

Group of Activity in HSPF Model	Parameter	Parameter Mean P	PFC For P O	P – 10%P P_1	PFC For P_1 O_1	P + 10%P P_2	PFC For P_2 O_2	S	S_r
PERLND:QUAL-INPUT	**SQOLIM**	8.79E+12	428,000	7.91E+12	385,000	9.67E+12	470,000	4.84E-08	0.99
RCHRES:GQ-GENDECAY	**THFST**	1.05	428,000	1.00	409,000	1.15	458,000	3.23E+05	0.79
PERLND:QUAL-INPUT	**WSQOP**	1.64	428,000	1.48	473,000	1.80	390,000	-2.53E+05	-0.97
RCHRES:GQ-VALUES	**TWAT**	60.00	428,000	54.00	440,000	66.00	414,000	-2170.0	-0.30
RCHRES:GQ-GENDECAY	**FSTDEC**	0.40	428,000	0.36	436,000	0.44	419,000	-2.13E+05	-0.20
PERLND:QUAL-INPUT	**SQO**	8.79E+12	428,000	7.91E+12	428,000	9.67E+12	428,000	0.00	0.00
PERLND:QUAL-INPUT	**ACQOP**	5.84E+12	428,000	5.25E+12	428,000	6.42E+12	428,000	0.00	0.00
PERLND:QUAL-INPUT	**IOQC**	10,000	428,000	9,000	428,000	11,000	428,000	0.00	0.00
PERLND:QUAL-INPUT	**AOQC**	10,000	428,000	9,000	428,000	11,000	428,000	0.00	0.00
IMPLND:QUAL-INPUT	**SQO**	2.06E+07	428,000	1.86E+07	428,000	2.27E+07	428,000	0.00	0.00
IMPLND:QUAL-INPUT	**ACQOP**	1.29E+07	428,000	1.16E+07	428,000	1.42E+07	428,000	0.00	0.00
IMPLND:QUAL-INPUT	**SQOLIM**	2.06E+07	428,000	1.86E+07	428,000	2.27E+07	428,000	0.00	0.00
IMPLND:QUAL-INPUT	**WSQOP**	1.64	428,000	1.48	428,000	1.80	428,000	0.00	0.00

Data for Calculation of Parameter Statistics

The availability of historical values for storage of fecal coliform on the land surface (SQOLIM) was very limited. This parameter depends on the land use distribution and the numbers of various animals present for each land use. The parameter SQOLIM was calculated for a 26-year period from 1975 to 2000, using the land use distribution for Bexar County and yearly population of livestock and wildlife. The statistics corresponding to temperature correction coefficient for first-order decay of fecal coliform (THFST), rate of surface runoff that will remove 90 percent of fecal coliform from the land surface (WSQOP) of PLS, and first-order decay rate for fecal coliform (FSTDEC) were obtained from literature (USEPA, 1985; USEPA, 2000). The statistic for water temperature in the stream (TWAT) was calculated using historical values available from EPA's STORET database.

Livestock Data

Historical data for livestock counts for Bexar County were obtained from the National Agricultural Statistics Service (NASS) web site (NASS, 2002), (Table 26). The yearly data pertaining to Bexar County obtained from NASS include the number of beef cows, dairy cows, chickens (both commercial broilers and layers), goats, sheep and hogs. Most of the data were available for a period ranging from 1975 to 2000, except for the number of chickens. County data for chickens were available only for the period from 1975 to 1980. However, the number of chickens was available for the South Central District D81 (NASS, 2002) for a period of 1975-2000. These numbers were used to calculate the average ratio of chickens in Bexar County to the number of chickens in District D81, during the period of 1974 to 1980. The number of chickens in Bexar County for the period from 1981 to 2000 was estimated by multiplying the number of chickens for the corresponding year in District D81 with the calculated ratio. Animal counts were not available for the period from 1988 to 1992 and hence those years were not considered for the calculation of the parameters.

Table 26. Number of Livestock in Bexar County, Texas 1975-2000

Year	Numbers of Animals				
	BEEF	DAIRY	CHICKEN	HOG	SHEEP
1975	34,000	6,600	275,000	9,400	3,500
1976	33,000	6,200	272,000	10,500	3,000
1977	32,000	6,100	357,000	9,900	2,500
1978	31,000	5,200	343,000	11,100	2,700
1979	31,000	4,800	372,000	12,500	2,400
1980	24,000	5,000	297,000	11,300	2,000
1981	20,000	4,000	358,974	8,700	1,200
1982	35,000	3,600	310,464	7,500	2,500
1983	41,000	3,300	282,744	8,500	2,600
1984	34,000	3,000	287,109	5,400	1,800
1985	26,000	3,500	295,190	4,500	1,800
1986	21,000	4,000	303,984	4,900	2,000
1987	22,000	2,900	332,536	5,613	1,500
1993	28,000	700	313,832	3,330	1,900
1994	28,000	800	441,946	2,730	2,500
1995	25,000	700	464,545	1,798	2,500
1996	26,000	600	485,370	2,331	3,000
1997	26,000	0	484,739	2,331	1,100
1998	25,000	0	518,260	2,131	5,900
1999	22,000	0	530,061	1,864	6,100
2000	25,000	0	485,370	1,864	5,000

Wildlife Data

The information regarding the number of wildlife was downloaded from the Texas Parks and Wildlife (TPWD) website (TPWD, 2002). The information available was limited to the deer population. Wildlife data for Bexar County was available for four years, 1993, 1995, 1996 and 1997 and these are given in Table 27. For other periods, the bacterial loading from wildlife was calculated using the average number of wildlife for the above four-year period.

Table 27. Number of White-tailed Deer in Bexar County, Texas

Year	1993	1995	1996	1997
Number	34,000	55,000	37,000	36,000

Historical Land Use

Since the calculation of the parameters depends not only on the number of different animals present on the watershed, but also the land use distribution of the watershed it is important to consider the change in the land use patterns over the period of study. Harris (2000) gives the land use distribution corresponding to 1976, 1985, and 1991. Harris (2000) found that there was 57% decrease in cropland in Salado Creek watershed in Bexar County during the period from 1976 to 1991. For the same period there were considerable increase of area under forestland (394%) and commercial land use (100%). For the current study, land use distributions corresponding to 1976, 1985, and 1991 were assigned to periods 1975-1982, 1983- 1987 and 1988-2000 respectively. The land use distribution data used for the calculation are given in Table 28.

Calculation of Parameter Statistics

Storage of fecal coliform on the land surface (SQOLIM) values were calculated based on the number of livestock and wildlife over a period from 1975-2000. The values for SQOLIM were calculated for four different land uses, cropland, forestland, pastureland, and urban or built-up land. Mean and standard deviation of SQOLIM for the different land uses are given in Table 29. The overall mean and standard deviation of SQOLIM are found out to be 24,770,000 and 86,240,000 Mfc/ac (million fecal coliform per acre) respectively.

Table 28. The Land Use Areas (acres) Distribution in Bexar County During Different periods

Land uses	Periods		
	1975-1982	1983-1987	1988-2000
COMMERCIAL AND SERVICES	14,074	20,913	28,640
CROP	429,309	376,402	299,087
FOREST	128,958	187,217	241,440
INDUSTRIAL	1,319	1,878	2,510
MIXED URBAN OR BUILT-UP LAND	1,766	2,565	3,467
OTHER	25,309	31,753	31,753
OTHER URBAN OR BUILT-UP LAND	8,397	13,169	18,561
PASTURE	105,282	88,556	64,114
RESERVOIR	6,718	6,718	6,718
RESIDENTIAL	69,304	62,301	94,874
TRANSPORTATION, COMMUNICATIONS AND UTILITIES	12,011	10,974	11,282

Table 29. Mean and Standard Deviation of SQOLIM for the Different Land Uses

Land use	Mean (million fecal coliform per acre)	Standard deviation (million fecal coliform per acre)
Cropland	1,514	1,459
Pastureland	99,101,993	149,814,130
Urban or built-up land	21	2
Forestland	43	14

The first-order decay rate for fecal coliform (THFST), values were obtained from the literature (USEPA, 1985). Based on the values given in literature, THFST was assumed to have a uniform distribution ranging from 0.95 to 1.1. The mean and variance of THFST are 1.025 and 0.00188 respectively.

The temperature correction coefficient for first-order decay of water quality constituent (FSTDEC) was based on the literature (USEPA, 1985). Values obtained were assumed to have a uniform distribution ranging from 0.1 to 1.0. The mean and variance of FSTDEC are 0.55 and 0.0675 respectively.

The rate of surface runoff per hour that will remove 90 percent of the water quality constituent stored over the land surface (WSQOP) was assumed to have a uniform distribution ranging from 0.5 to 2.0 based on limited information. The mean and variance are 1.25 and 0.1875 respectively.

The historical water temperature data for the Salado Creek were obtained from the US EPA's STORET database. The data were analyzed to find out mean and standard deviation. The mean and variance of TWAT are 70.99 and 75.08 respectively.

Results of First Order Approximation

The availability of historical values for SQOLIM was very limited. This parameter depends on the land use distribution and the numbers of animals present for each land use. The parameter SQOLIM was calculated for a 26-year period from 1975 to 2000, using the land use distribution for Bexar County (Harris, 2000) and yearly population of livestock (NASS, 2000) and wildlife (TPWD, 2002). The values corresponding to THFST, WSQOP and FSTDEC were obtained from USEPA (1985, 2000). Based on the recommended values found in USEPA (1985, 2000a, 2001b) THFST, WSQOP and FSTDEC were assumed to have uniform distributions with intervals [0.95, 1.1], [0.5, 2.0], and [0.1, 1.0] respectively. The mean and the variance of TWAT were calculated using historical values available from EPA's STORET database. Table 30 shows the means and variances of the five parameters.

The five parameters that the maximum value of in-stream fecal coliform concentrations at the outlet of the Salado Creek showed the most sensitivity to were used in a First Order Approximation. The First Order Approximation was used to determine which of these sensitive parameters contributed the most to the variance of in-stream fecal coliform concentration. The results of the FOA are given in Table 8. The FOA showed that 99.9% of the variance in simulated peak concentration of fecal coliform concentration in-stream was contributed by the parameter that represents the maximum storage of fecal coliform over the pervious land segment. This is due to the very large variability in this parameter. Small portions of the variance were attributed to WSQOP (0.07%) and FSTDEC (0.02%). There were negligible contributions from THFST and TWAT to the output variance.

Table 30. First Order Analysis of HSPF for Fecal Coliform

Parameter	Mean	Variance	Sensitivity S	$S^2 \times$ Var	Fraction of Model Variance	% of Variance
SQOLIM	2.477E+07 [1]Mfc/ac	7.44 E+27	4.84E+08	1.74E+13	9.99E-01	99.91
THFST	1.025 (/day)	0.00188	3.23E+05	1.96E+08	1.13E-05	0.00
FSTDEC	0.55	0.0675	-2.53E+05	1.20E+10	6.89E-04	0.07
TWAT	70.99 (F)	75.08	-2.13E+05	3.05E+09	1.75E-04	0.02
WSQOP	1.25 (in/hr)	0.1875	-2.17E+03	3.52E+08	2.02E-05	0.00

[1]Million fecal coliform colony forming units/ac

Conclusion

The Salado Creek watershed in Bexar County, Texas was modeled using the HSPF model in BASINS. The model was calibrated for hydrology. A sensitivity analysis and first order approximation were performed to determine the parameters that most influence the in-stream fecal coliform concentration predictions. The specific findings from the study include:

1. The parameters that peak in-stream fecal coliform concentrations are most sensitive to are those that represent the maximum storage of fecal coliform bacteria over the pervious land segment (SQOLIM), and surface runoff that removes 90 percent of quality constituent from pervious land segment (WSQOP) of PERLND section (corresponding to pervious land segment) of HSPF model.
2. Other parameters in-stream fecal coliform concentration predictions are sensitive to are stream water temperature (TWAT), first-order decay rate of quality constituent (FSTDEC) and temperature correction coefficient for the first order decay rate (THFST) of RCHRES section.
3. Though the model is highly sensitive to the parameters listed above, results of the First Order Analysis showed that a major portion of the model output variance is caused by the variation in the parameter representing the maximum storage of fecal coliform bacteria over the pervious land segment.

These results point out the importance of parameterization in modeling with any complex, process-based watershed model. Small errors in assigning values to the maximum storage of fecal coliform over a given land use class may result in large errors in predicted coliform counts.

Further study is recommended using Monte Carlo Simulation techniques to evaluate the risk of exceeding some TMDL based on current land-use practices. This will provide us a better understanding of the system and thereby helps us in reducing the uncertainty associated with bacterial loading into the streams.

LEON CREEK RIPARIAN ZONE MODELING

Image Difference Calculation Results

The result is a grayscale image, displaying brighter areas indicating an decrease in pixel value and darker areas depicting an increase in pixel value. Areas where the pixel value is displaying an increase in brightness from 1987 to 1999 are actually showing areas of decreased vegetation. This may be a result of increased urbanization or because of natural causes such as flood damage. Likewise, an area where there is a decrease in brightness, there is an increase in vegetation. To be sure these areas are an increase in dense vegetation and not some other class that displays dark pixel values, such as water, the supervised classification theme layers for 1987 and 1999 were used to identify dense vegetation areas.

Areas undergoing either a 20% or more change in pixel value were highlighted and selected for the buffer zones (0.5 mi., 0.25 mi., 0.125 mi., and 100 ft.) (Figure 40). Twenty percent was a significant percent value for this project in showing changes from 1987 to 1999. Only the 1987 class of dense vegetation was to be examined with the 20% or more change in pixel value with the image difference calculation. Examining these two data layers assisted me in analyzing the total area of dense vegetation that had gone through a 20% or more change in pixel value in all of the buffer zones (0.5 mi., 0.25 mi., 0.125 mi., and 100 ft.) as well as in an urban zone versus a rural zone.

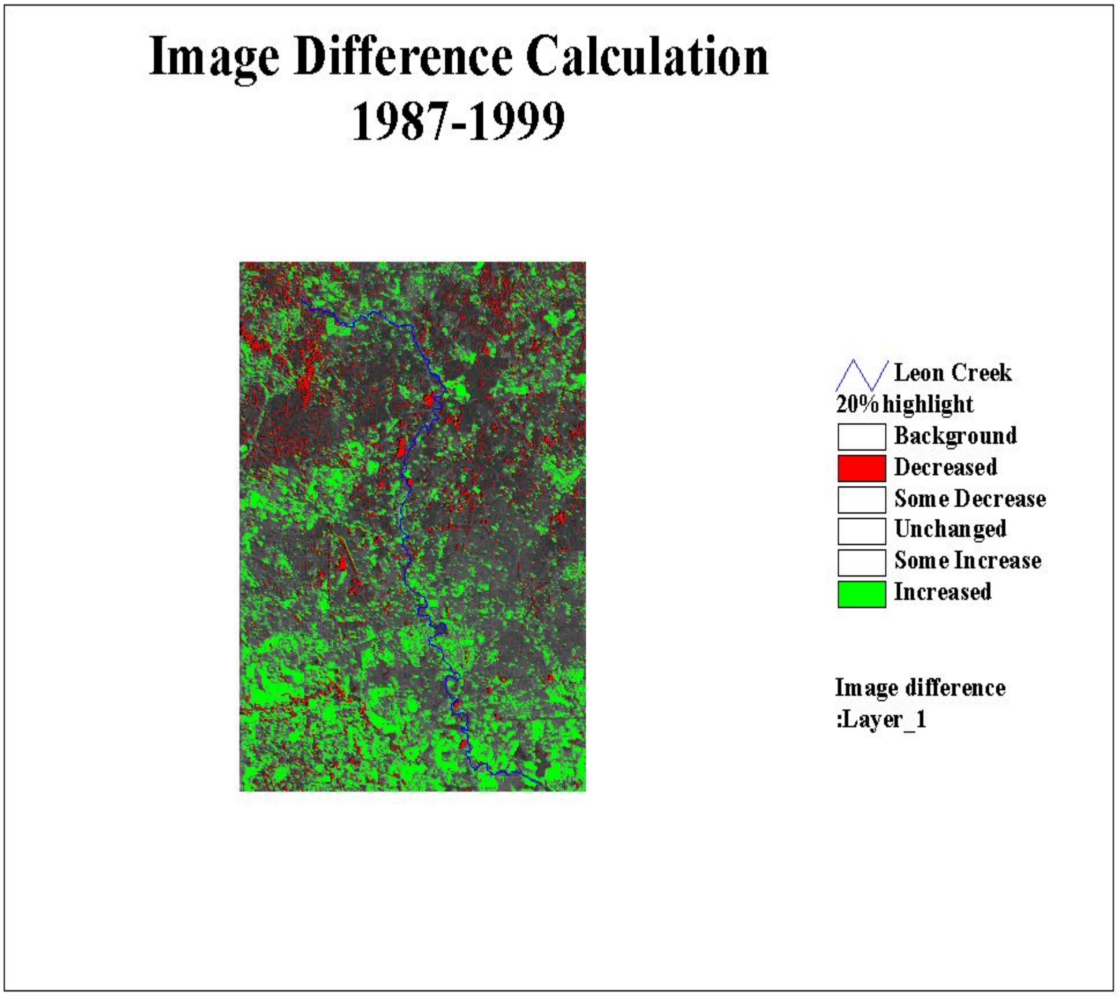


Figure 40. Image Difference Result with Highlighted 20% or More Change. Depicts resultant grayscale image, with the Highlighted Image. The highlighted image shows pixels that are going through a 20% or more change in pixel value. The red areas show pixel that are going through an increase in pixel value, where the pixel is darker. The green areas show decreases in the pixel value of 20% or more, which are brighter in color.

Within the total watershed area, there is a larger total percent of land area that is going through a brightness increase of 20% or more than a decrease (Figure 41). This means there is a larger land area that is going through urbanization or that is bare soil. In the 100 ft buffer zone, the most intense change is going on. In this buffer zone, there is both the greatest percent of area going through a 20% increase in brightness (3.5%) but also a 20% decrease in brightness (3.1%). This suggests that there are areas in the 100 ft buffer zone going through the greatest increases and decreases in vegetation. In all other buffer zones (0.5 mi. to 0.125 mi.), there is a small percent area that is increasing in dense vegetation, ranging from 0.2 to 0.6%. All buffer zones that are going through a decrease in dense vegetation are relatively higher (2.1 to 3.6%) than those going through an increase in vegetation (0.2 to 3.1%). This suggests that there is, in general, a greater percent area is going through decrease in vegetation than an increase in all buffer zones throughout the entire watershed.

In the urban zone, percent area of dense vegetation going through a 20% increase or decrease in brightness were also evaluated. The total area going through a 20% or more decrease in brightness was found to be fairly consistent through the buffer zones, ranging from 0.1 to 0.2% (Figure 42). There was a larger percent area going through a 20% or more increase in brightness ranging from 1.2 to 2.6%, with the largest amount of change occurring in the 100 ft buffer zone at 2.6% area. This means there is a 2.6% area going through a 20% or more decrease in vegetation, versus a 0.2% area going through an increase in vegetation in the 100 ft buffer zone. In the urban zone, there appears to be a larger percent area going through a 20% or more decrease in vegetation in all buffer zones than an increase in vegetation.

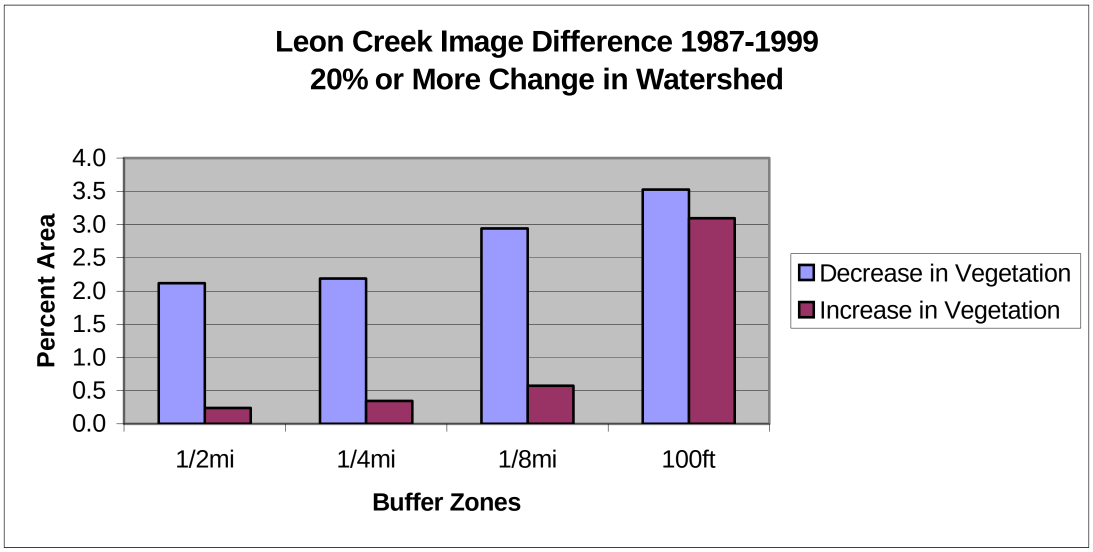


Figure 41. Total Watershed Area with 20% or More Change. Depicts the percent area of increase or decrease in vegetation in each buffer zone. All buffer zones that are going through a decrease in dense vegetation are relatively higher (2.1 to 3.6%) than those going through an increase in vegetation (0.2 to 3.1%).

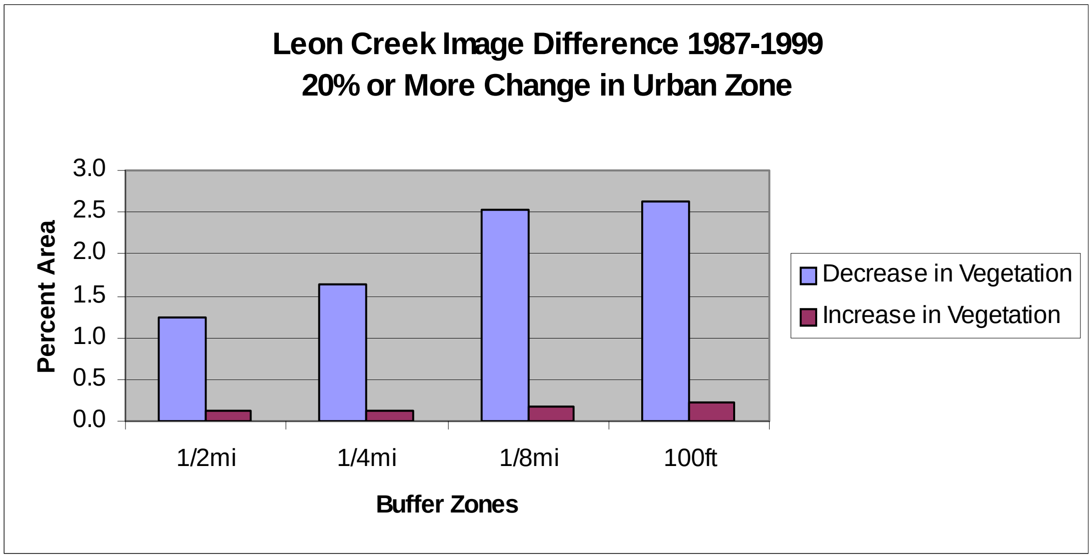


Figure 42. Urban Zone Area with 20% or More Change. Depicts the percent area of increase or decrease in vegetation in each buffer zone. In the urban zone, there appears to be a larger percent area going through a 20% or more decrease in vegetation in all buffer zones than an increase in vegetation.

The rural zone also shows higher percent areas going through a decrease in vegetation, except for the 100 ft. buffer zone (Figure 43). In the 100 ft. buffer zone, there is a 4.4% decrease in vegetation but there is a 6.0% area going through a 20% or more increase in vegetation. In the 0.5 mi., 0.25 mi., 0.125 mi. buffer zones there are respectively 3.3%, 2.9%, and 3.4% areas going through an increase in brightness from 1987 to 1999, which means these areas are going through a 20% or more decrease in vegetation. The 0.5 mi., 0.25 mi., 0.125 mi. buffer zones have small percent areas going through an increase of 20% or more vegetation at 0.4%, 0.6% and 1.0% area respectively.

It may be concluded, that the greatest amount of increase and decreases in vegetation are occurring in the 100 ft. buffer zone, within the entire watershed (Figure 44). On average, the rural zone is going through both the greatest decreases (3.5%) and increases (2.0%) in vegetation percent area. The total watershed, the rural zone, and urban zone exhibit greater decreases in vegetation than increases. The lowest averages with increases in vegetation are occurring in the rural zone at 0.2% average percent area. The greatest amount of vegetation change both in decrease and increase averages occurs in the rural zone, with the urban zone being the least productive in terms of increases in vegetation.

The Image Calculation results indicated greater degrees of change occurring in the rural zone, with a greater percent area going through a decrease by 1999 in dense vegetation in all buffer zones than in the urban zone. The greatest amount of both decreases and increases in the rural zone occurs within the 100 ft. buffer zone, closest to Leon Creek. In the urban zone, the Image Calculation results are very straightforward in that there are obvious decreases in vegetation in all buffer zones by 1999, especially in the 0.125 mi. and 100 ft. buffer zones.

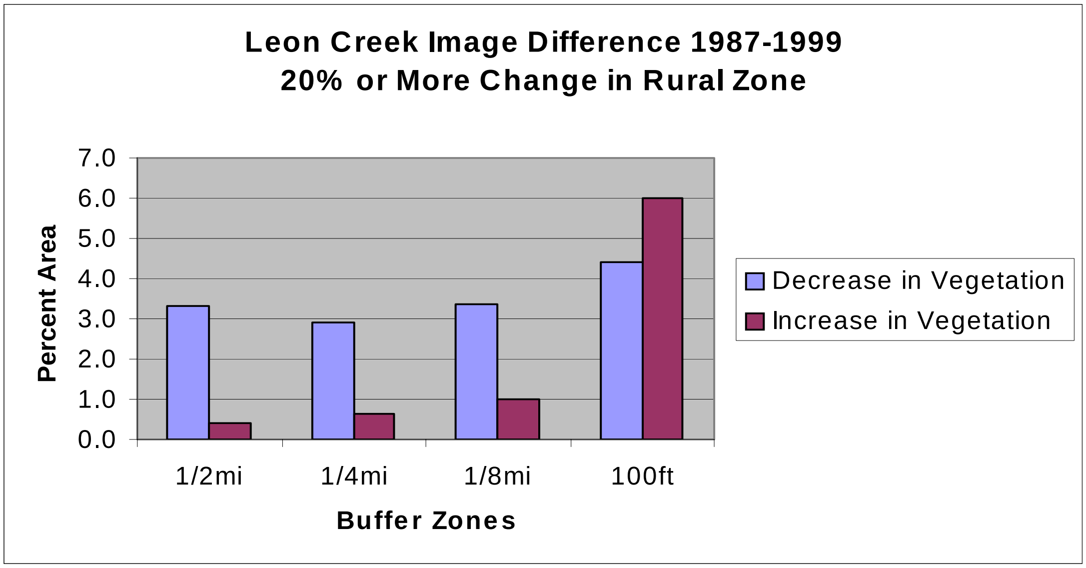


Figure 43. Rural Zone Area with 20% or More Change. Depicts the percent area of increase or decrease in vegetation in each buffer zone. In the 0.5 mi., 0.25 mi., 0.125 mi. buffer zones there are respectively 3.3%, 2.9%, and 3.4% areas going through an increase in pixel value from 1987 to 1999.

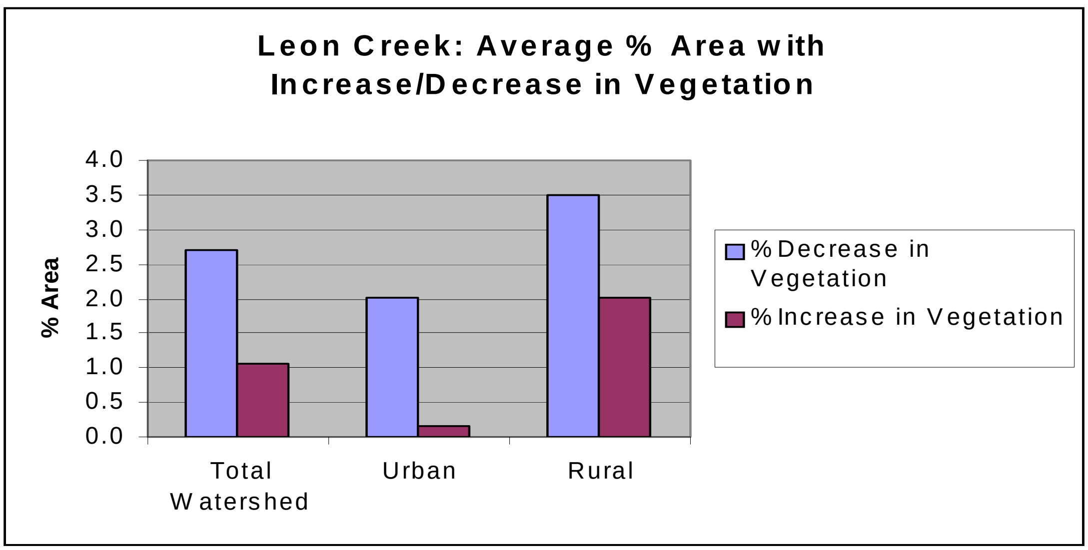


Figure 44. Average Area of 20% or More Change in Watershed, Urban, and Rural Zones. The greatest amount of vegetation change both in decrease and increase averages occurs in the rural zone, with the urban zone being the least productive in terms of increases in vegetation.

Supervised Classification Results

The supervised class images for 1987 and 1999 were smoothed with a focal majority, which uses a 3X3 matrix to find the value that occurs most frequently within the matrix (Schowengerdt, 1997). This will decreased speckling and visually smoothes the classes (Figures 45 and 46). The homogeneous pixels were then converted from raster pixels to polygons. The dense vegetation class was determined for the total watershed, the urban zone, and the rural zone within each of the buffers zones at 0.5 mi., 0.25 mi., 0.125 mi., and 100 ft. (Figures 47-70).

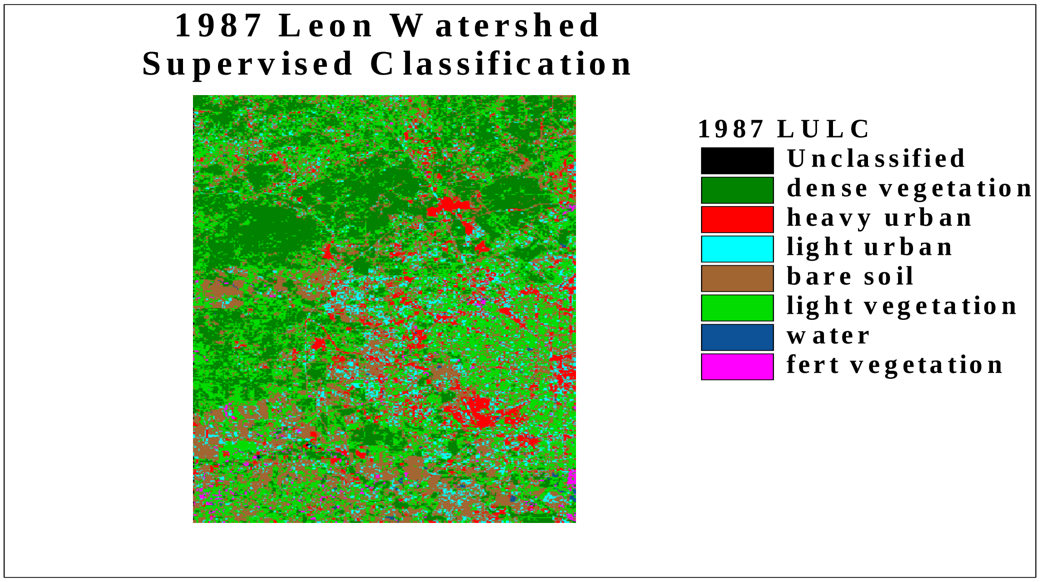


Figure 45. Results of Supervised Classification for 1987. Depicts all the defined classes Land Use and Land Cover Classes (LULC).

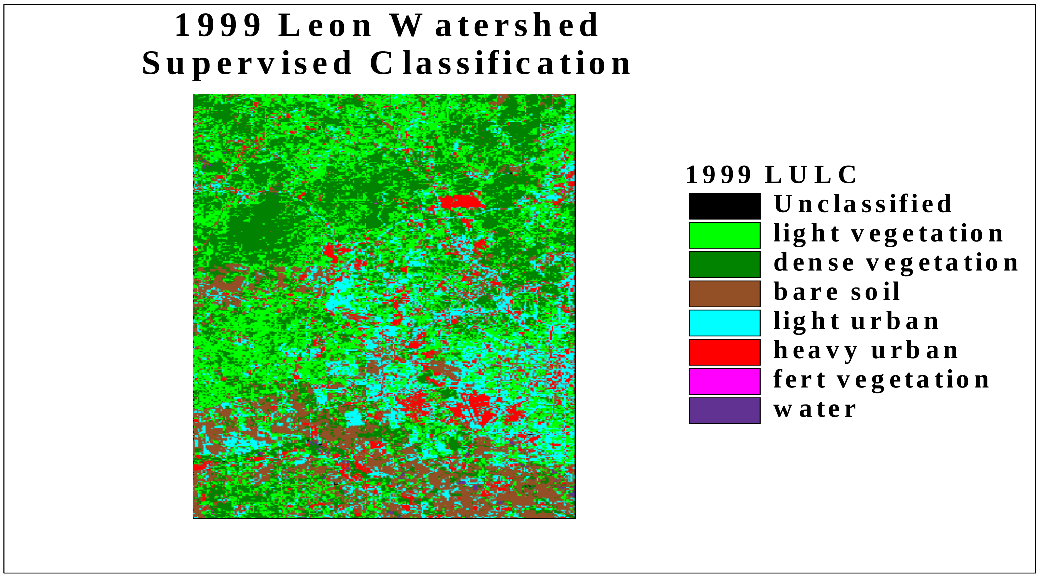


Figure 46. Results of Supervised Classification for 1999. Depicts all the defined classes Land Use and Land Cover Classes (LULC).

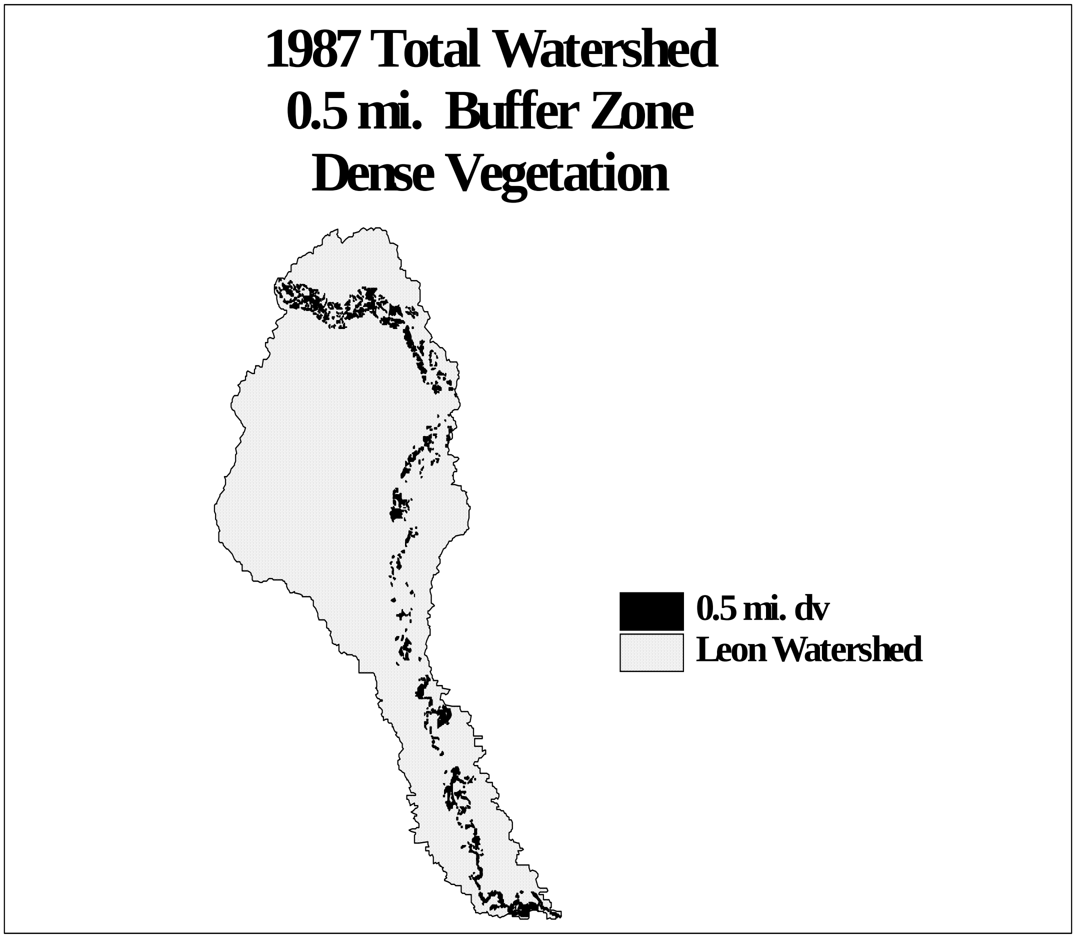


Figure 47. 1987 Dense Vegetation Class within 0.5 mi., Total Watershed. Depicts dense vegetation class within 0.5 mi. of Leon Creek in the Total Watershed. Polygons were created from the supervised classification and the dense vegetation class was examined for percent area.

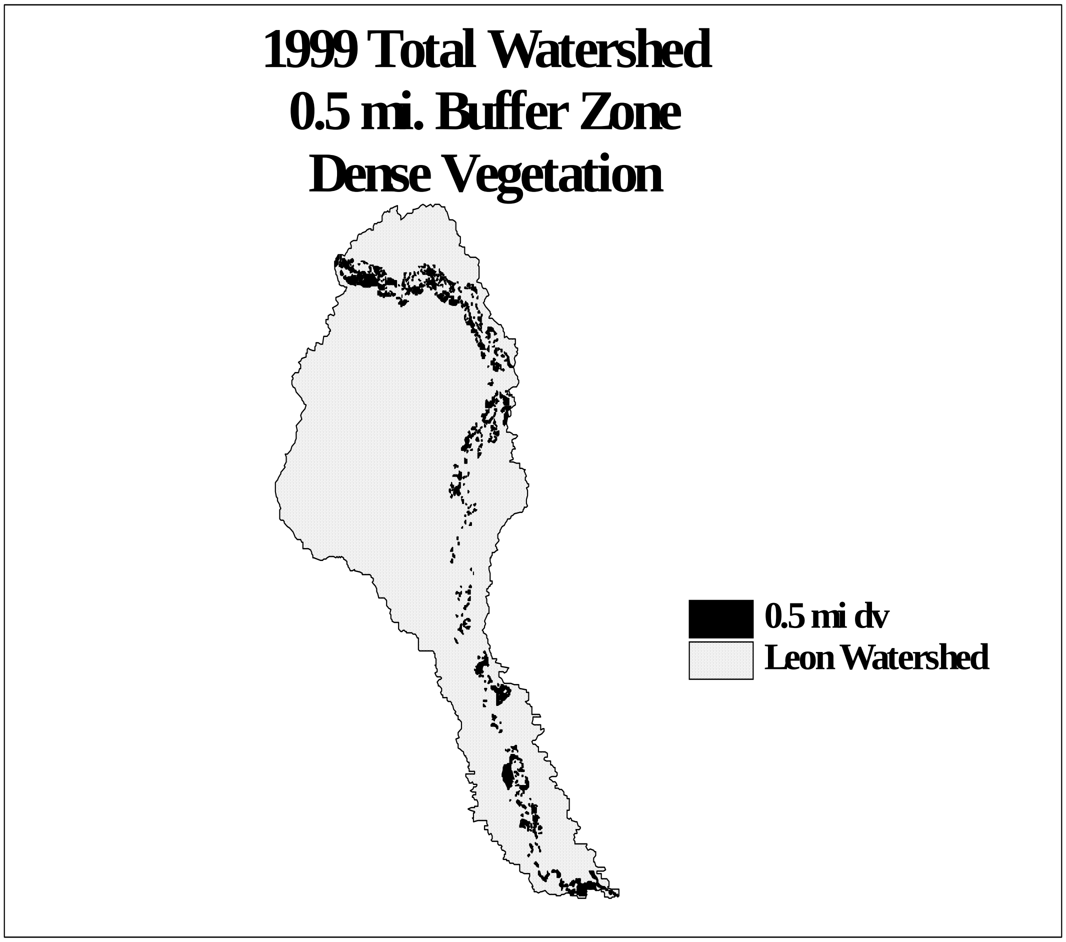


Figure 48. 1999 Dense Vegetation Class within 0.5 mi., Total Watershed. Depicts dense vegetation class within 0.5 mi. of Leon Creek in the Total Watershed. Polygons were created from the supervised classification and the dense vegetation class was examined for percent area.

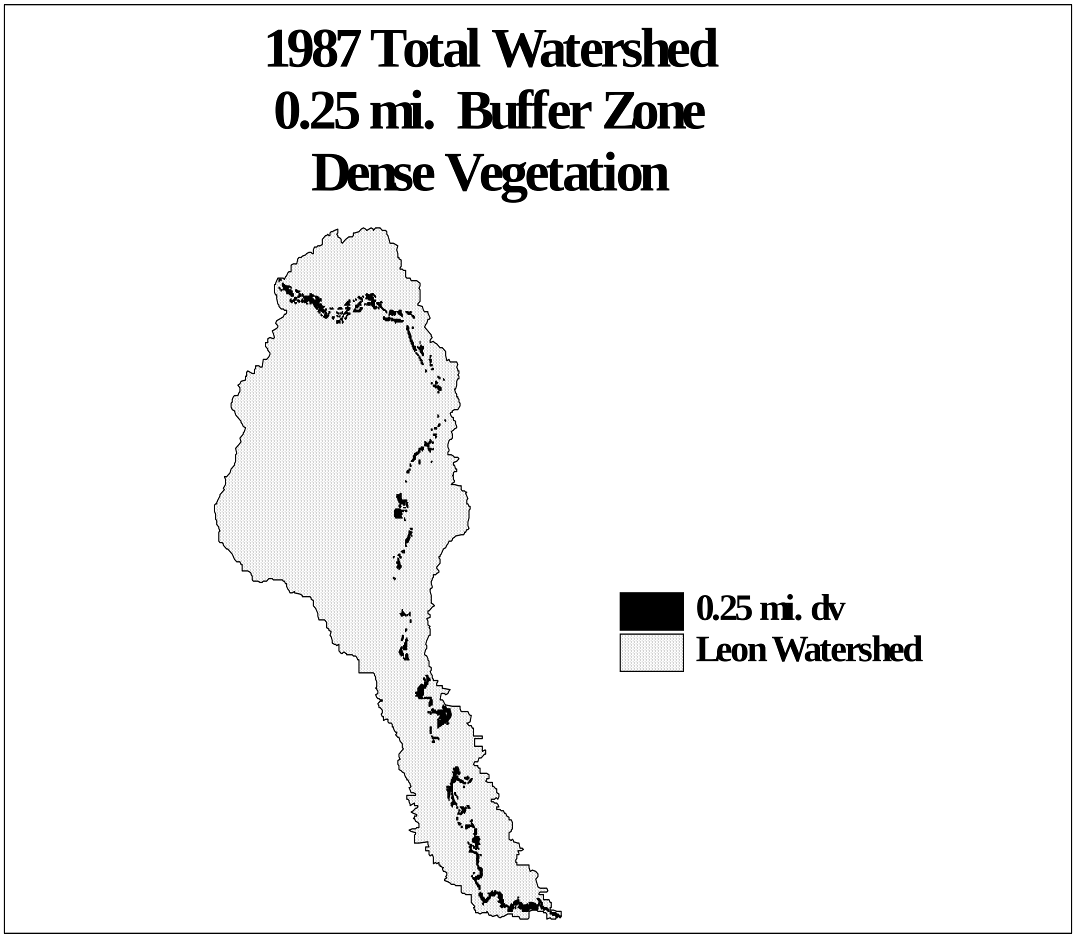


Figure 49. 1987 Dense Vegetation Class within 0.25 mi., Total Watershed. Depicts dense vegetation class within 0.25 mi. of Leon Creek in the Total Watershed. Polygons were created from the supervised classification and the dense vegetation class was examined for percent area.

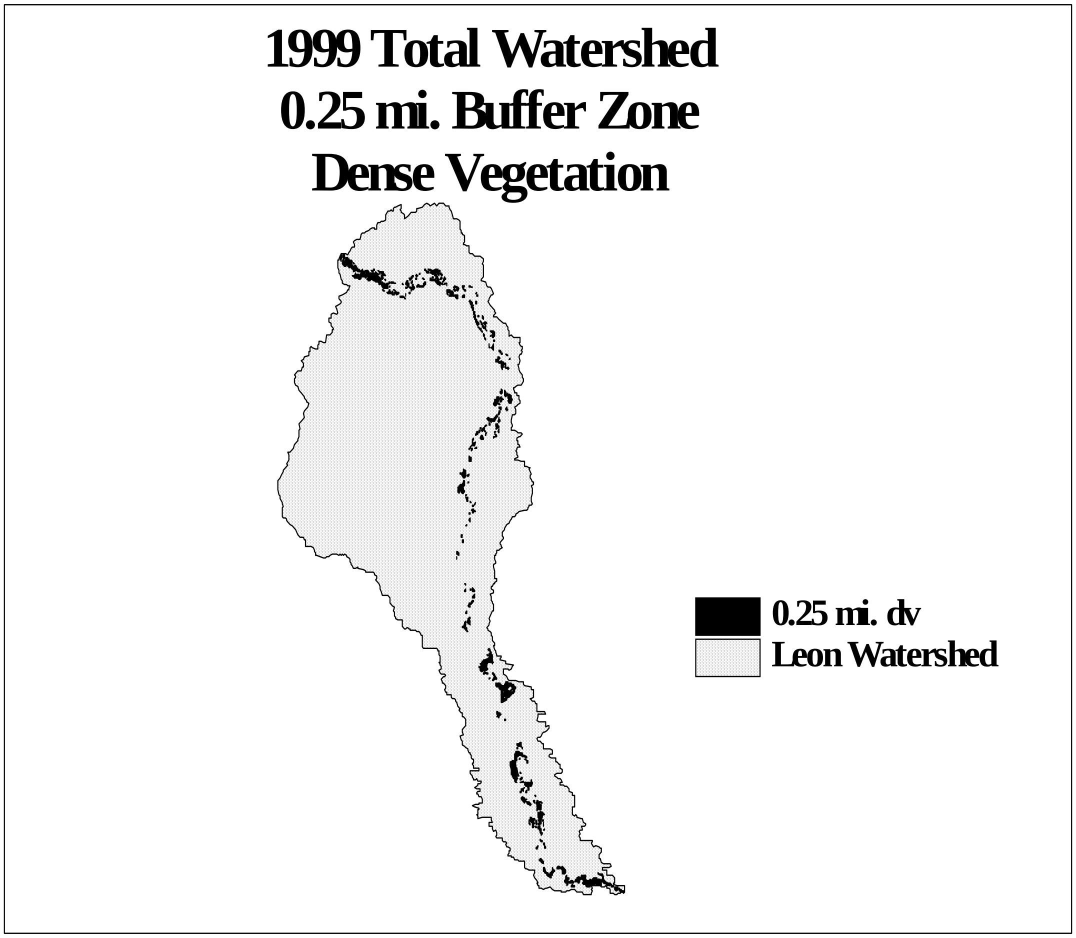


Figure 50. 1999 Dense Vegetation Class within 0.25 mi., Total Watershed. Depicts dense vegetation class within 0.25 mi. of Leon Creek in the Total Watershed. Polygons were created from the supervised classification and the dense vegetation class was examined for percent area.

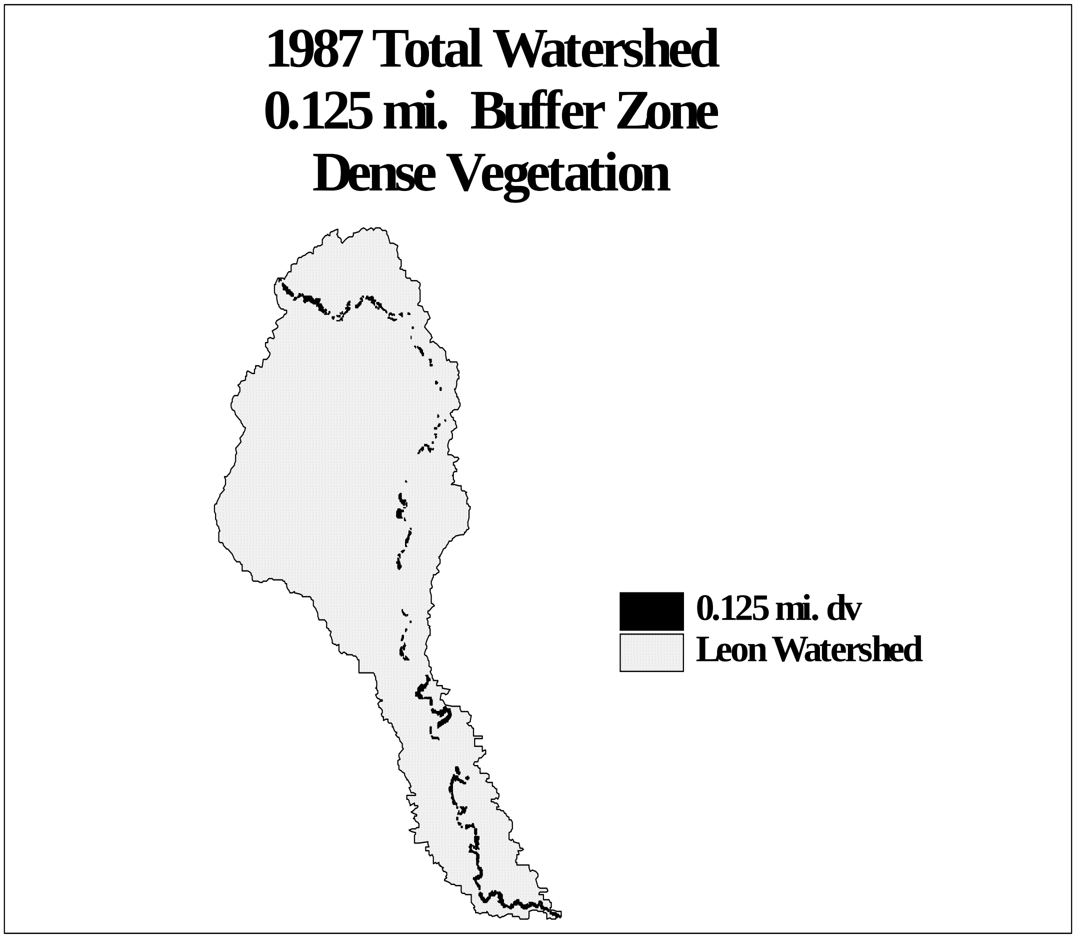


Figure 51. 1987 Dense Vegetation Class within 0.125 mi., Total Watershed. Depicts dense vegetation class within 0.125 mi. of Leon Creek in the Total Watershed. Polygons were created from the supervised classification and the dense vegetation class was examined for percent area.

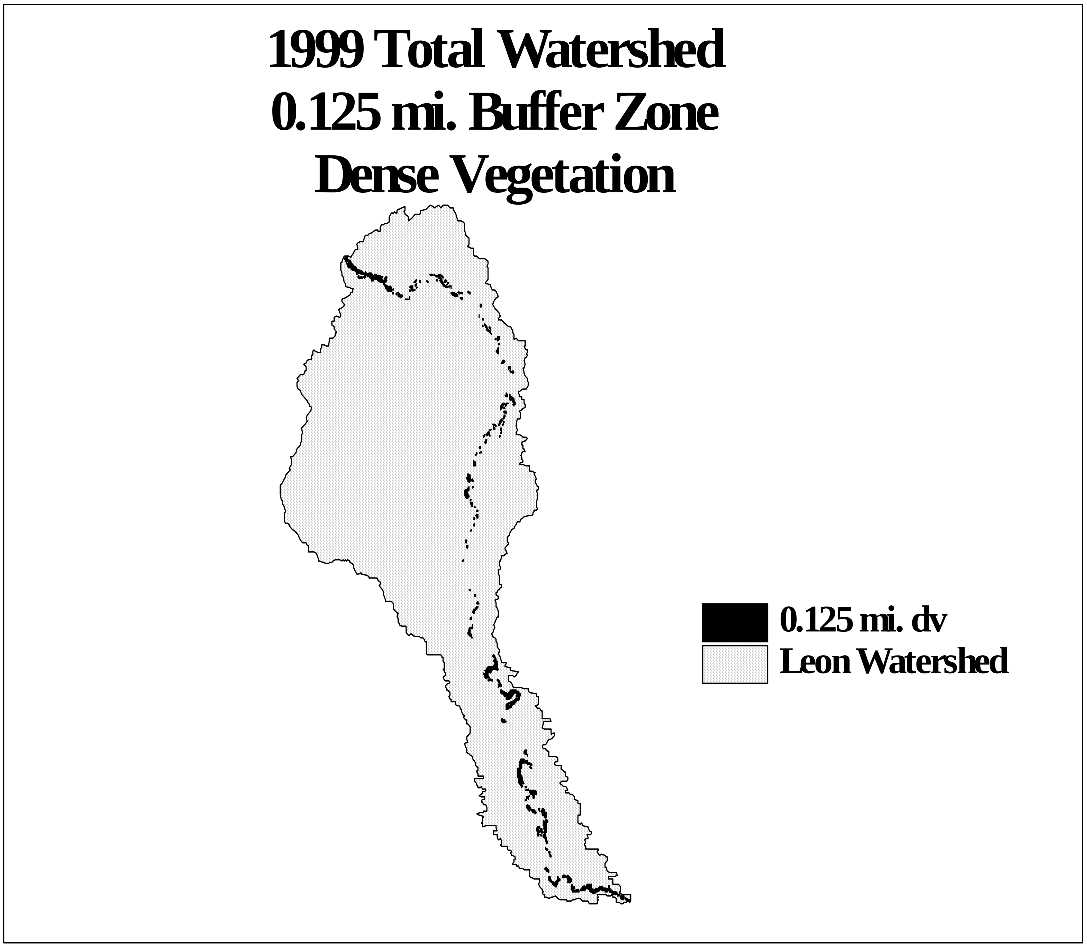


Figure 52. 1999 Dense Vegetation Class within 0.125 mi., Total Watershed. Depicts dense vegetation class within 0.125 mi. of Leon Creek in the Total Watershed. Polygons were created from the supervised classification and the dense vegetation class was examined for percent area.

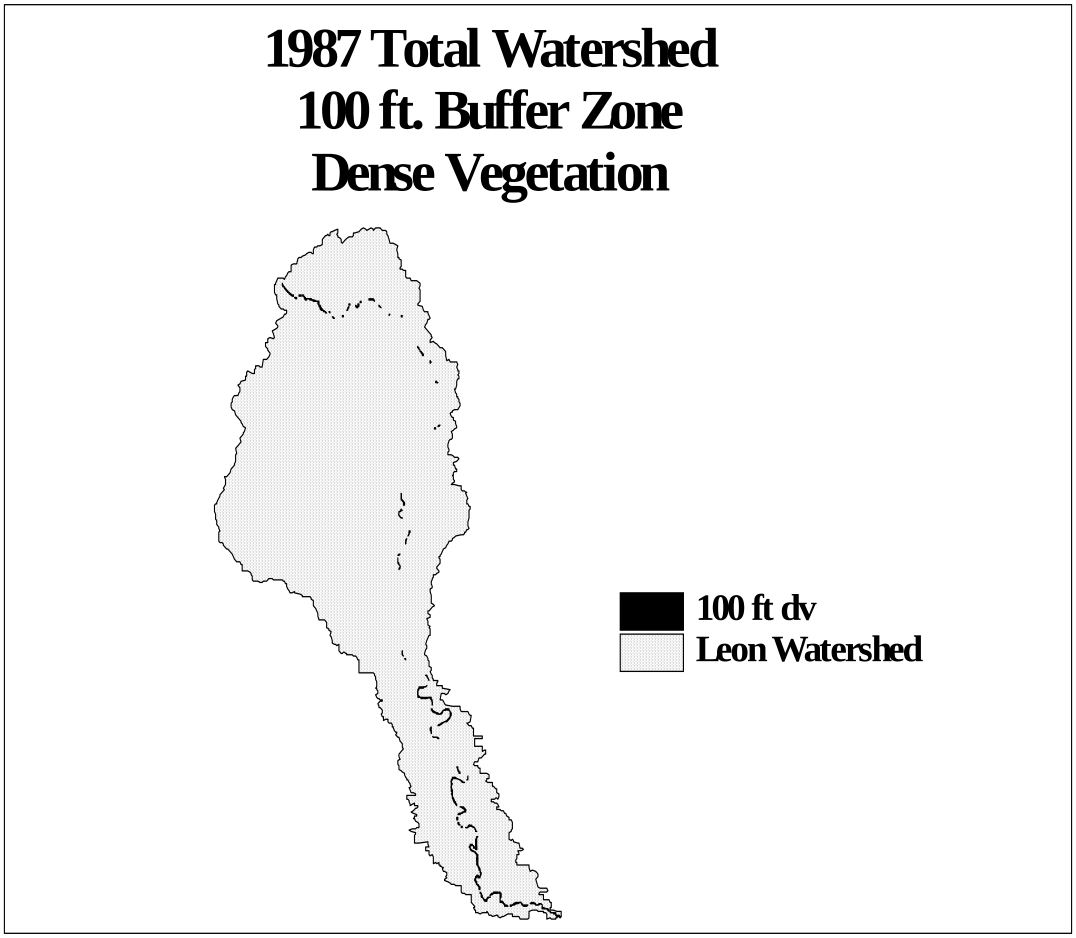


Figure 53. 1987 Dense Vegetation Class within 100 ft., Total Watershed. Depicts dense vegetation class within 100 ft. of Leon Creek in the Total Watershed. Polygons were created from the supervised classification and the dense vegetation class was examined for percent area.

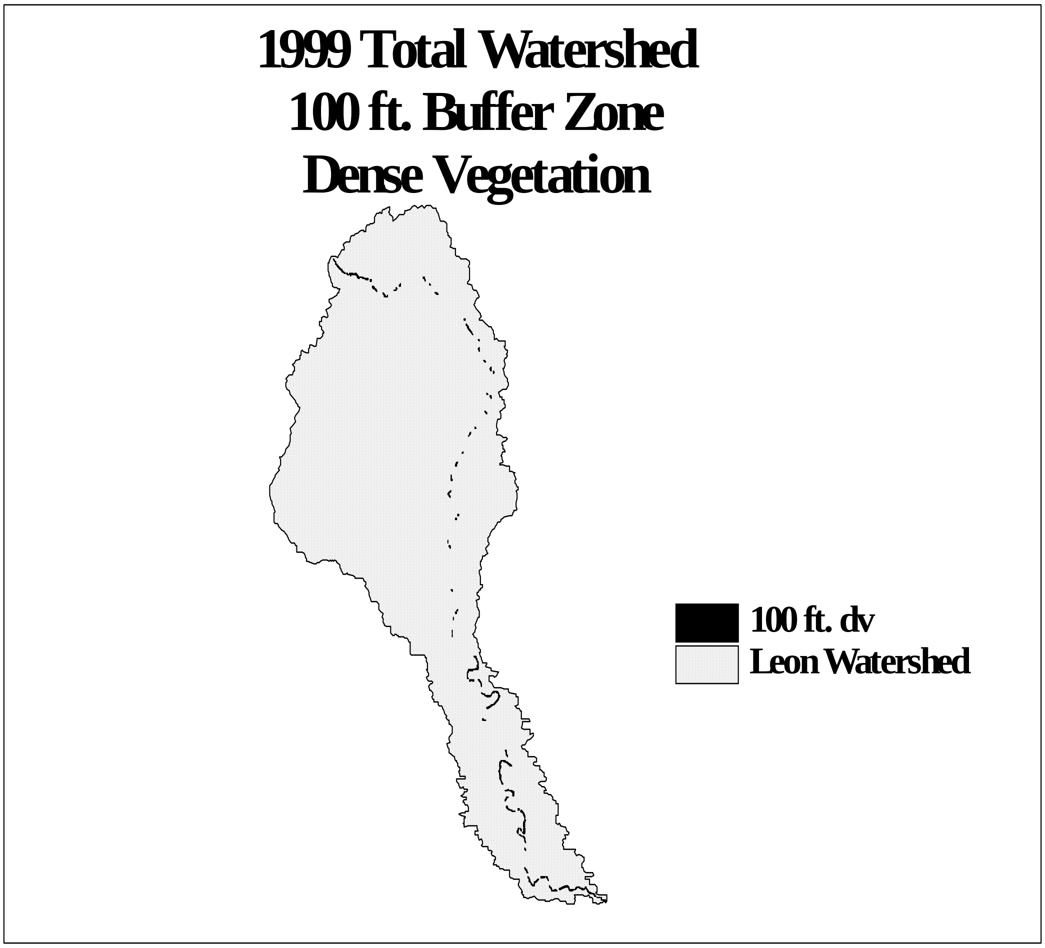


Figure 54. 1999 Dense Vegetation Class within 100 ft., Total Watershed. Depicts dense vegetation class within 100 ft. of Leon Creek in the Total Watershed. Polygons were created from the supervised classification and the dense vegetation class was examined for percent area.

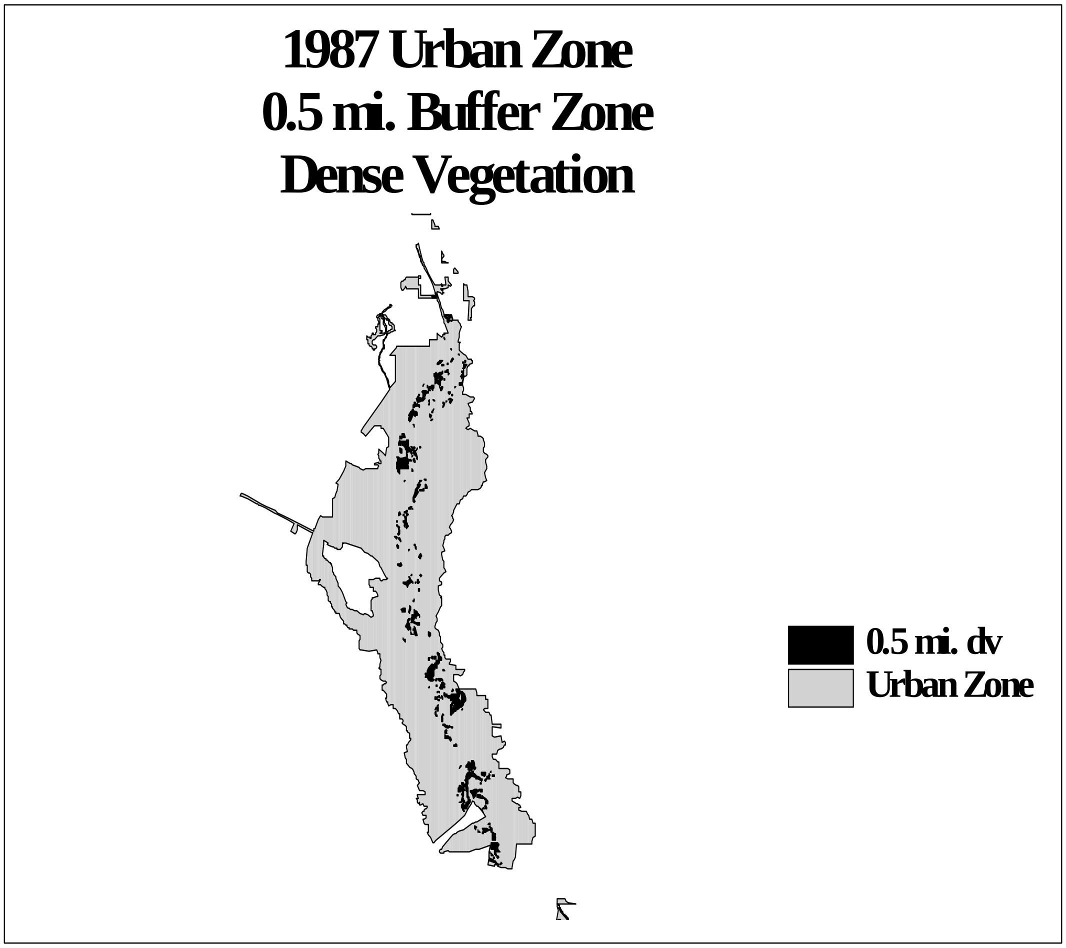


Figure 55. 1987 Dense Vegetation Class within 0.5 mi., Urban Area. Depicts dense vegetation class within 0.5 mi. of Leon Creek in the Urban Area. Polygons were created from the supervised classification and the dense vegetation class was examined for percent area.

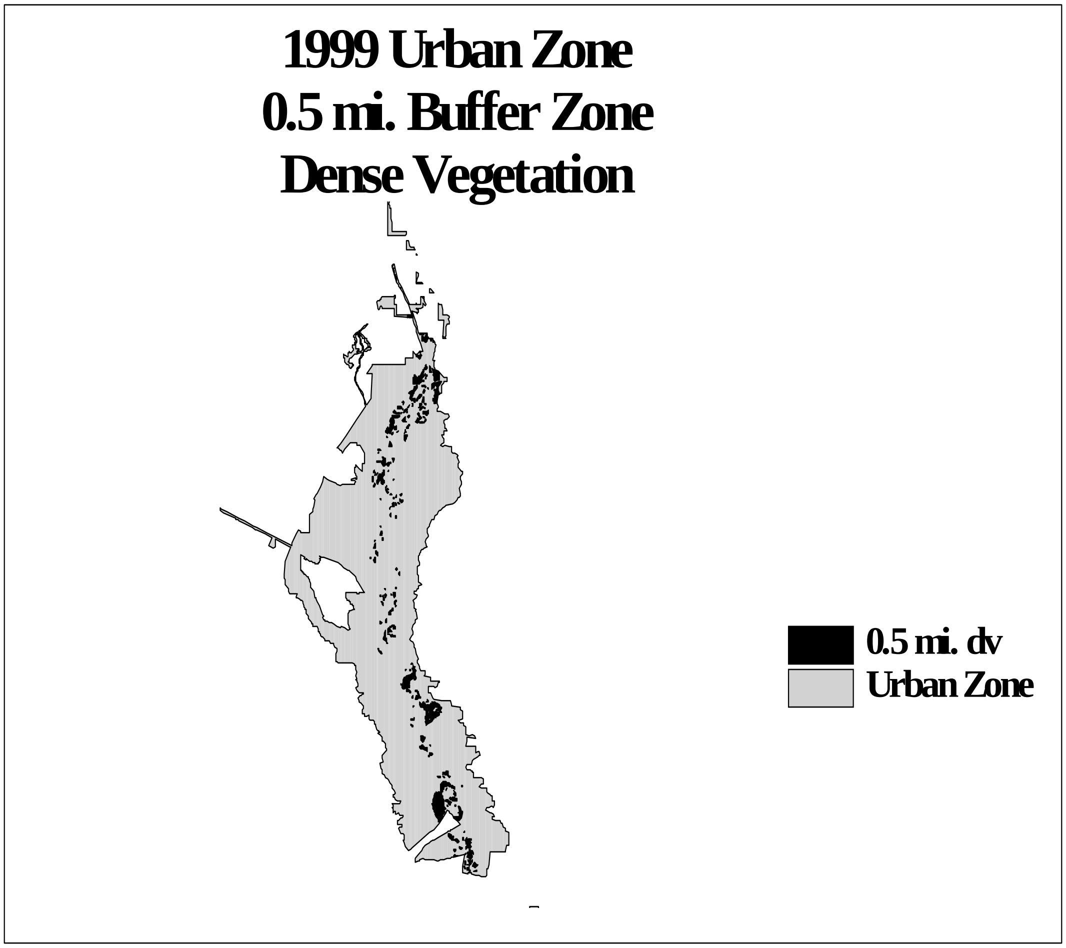


Figure 56. 1999 Dense Vegetation Class within 0.5 mi., Urban Area. Depicts dense vegetation class within 0.5 mi. of Leon Creek in the Urban Area. Polygons were created from the supervised classification and the dense vegetation class was examined for percent area.

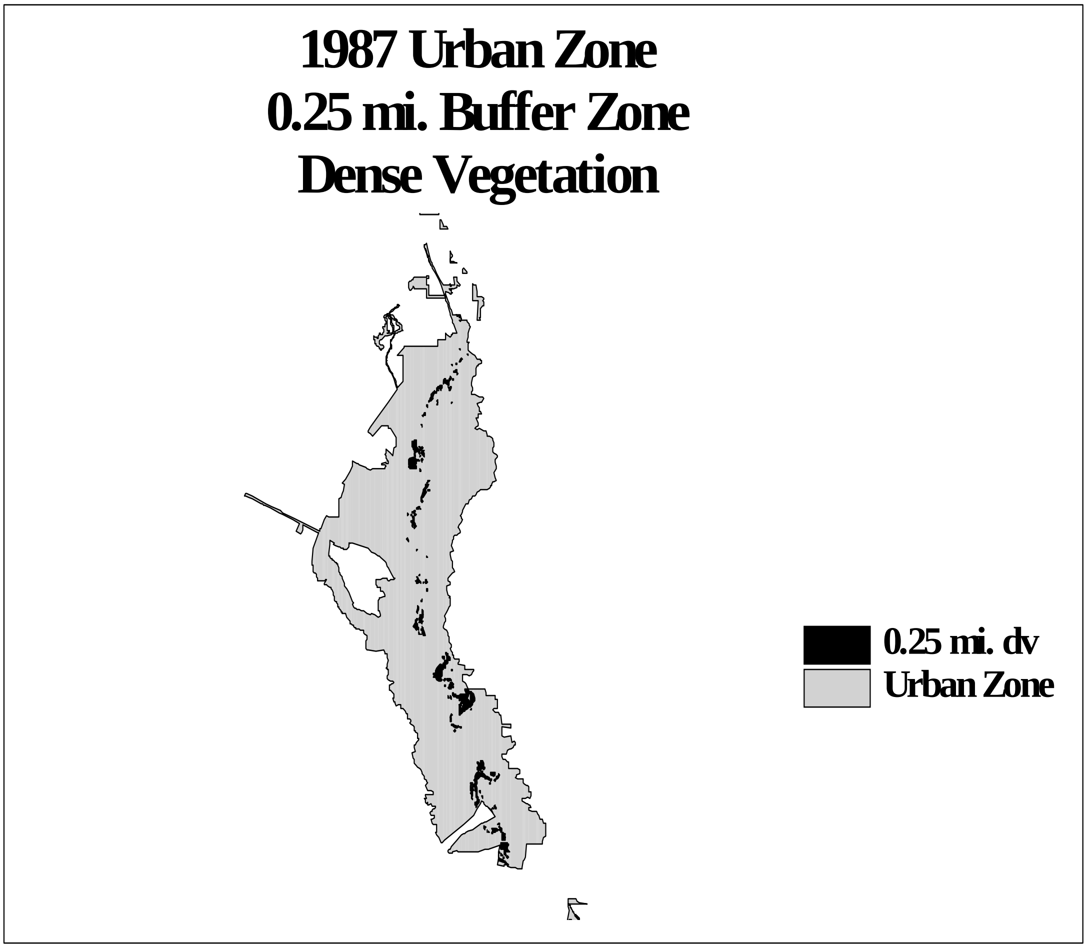


Figure 57. 1987 Dense Vegetation Class within 0.25 mi., Urban Area. Depicts dense vegetation class within 0.25 mi. of Leon Creek in the Urban Area. Polygons were created from the supervised classification and the dense vegetation class was examined for percent area.

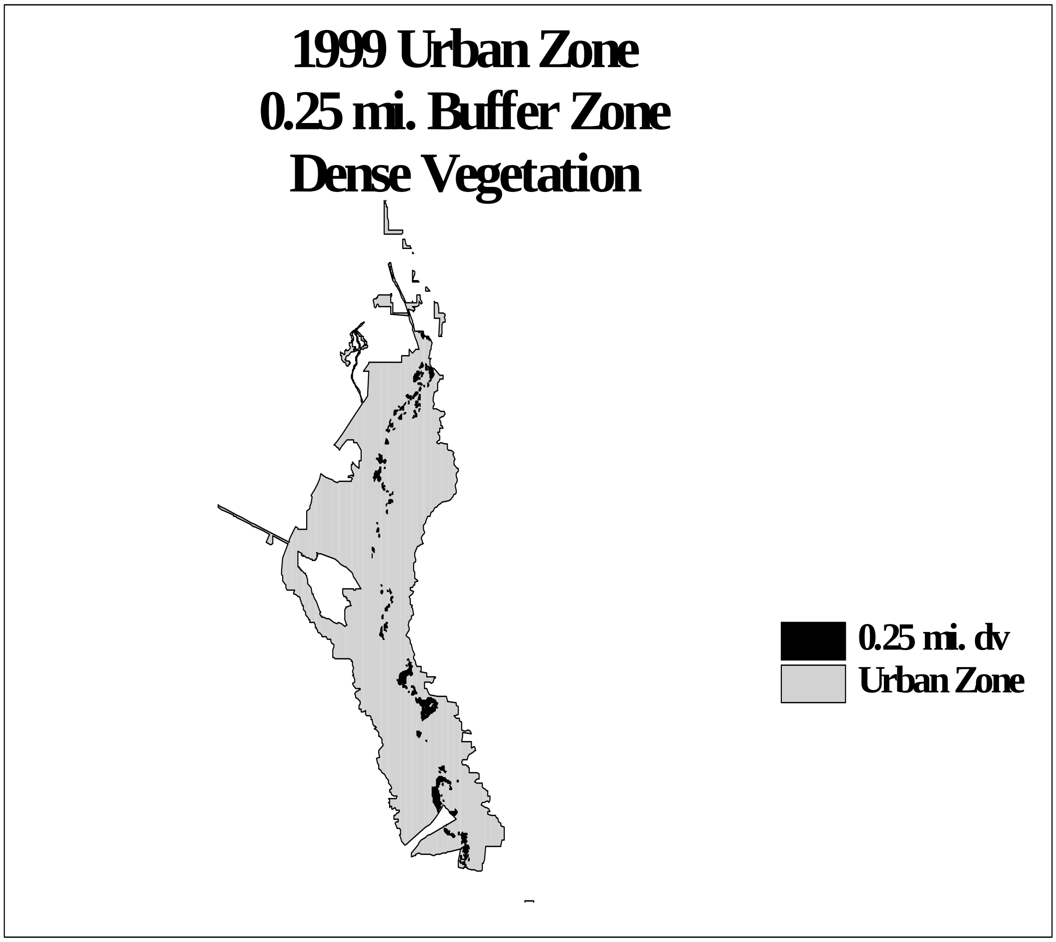


Figure 58. 1999 Dense Vegetation Class within 0.25 mi., Urban Area. Depicts dense vegetation class within 0.25 mi. of Leon Creek in the Urban Area. Polygons were created from the supervised classification and the dense vegetation class was examined for percent area.

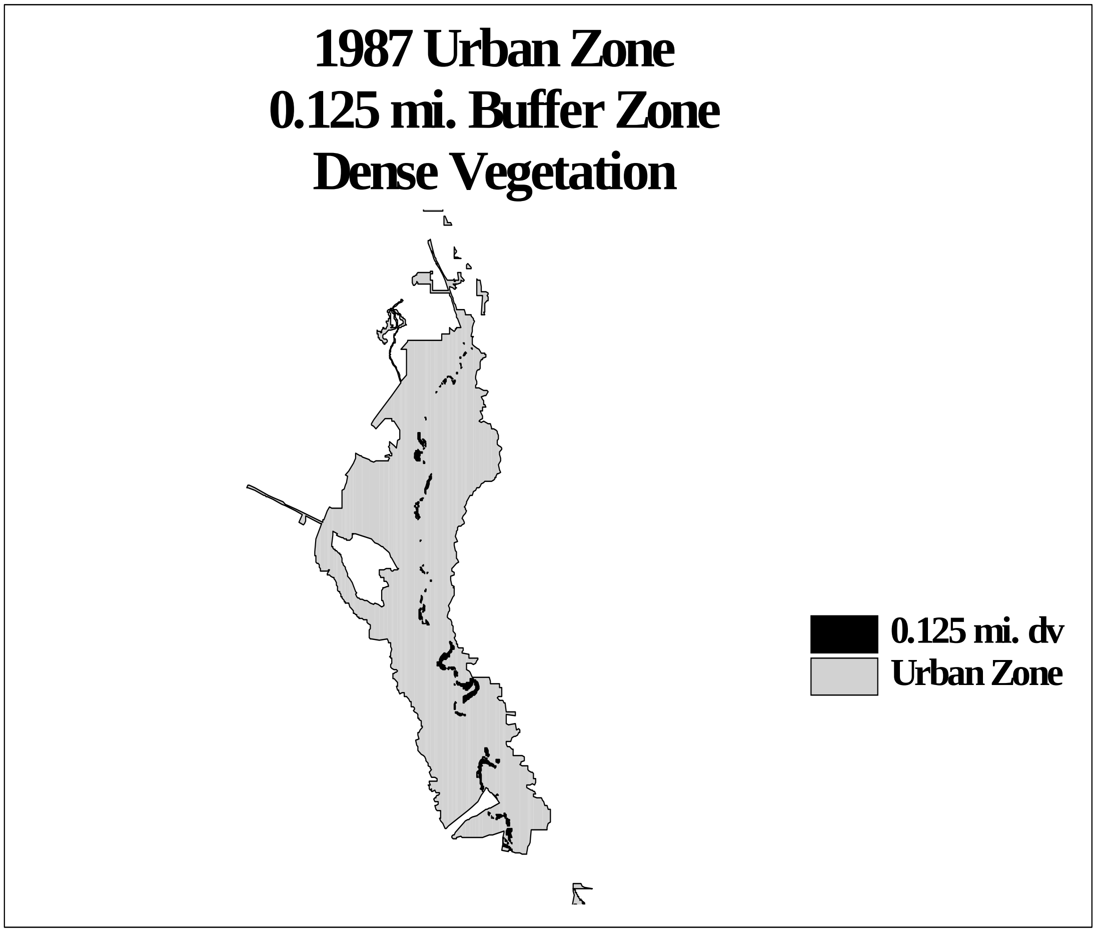


Figure 59. 1987 Dense Vegetation Class within 0.125 mi., Urban Area. Depicts dense vegetation class within 0.125 mi. of Leon Creek in the Urban Area. Polygons were created from the supervised classification and the dense vegetation class was examined for percent area.

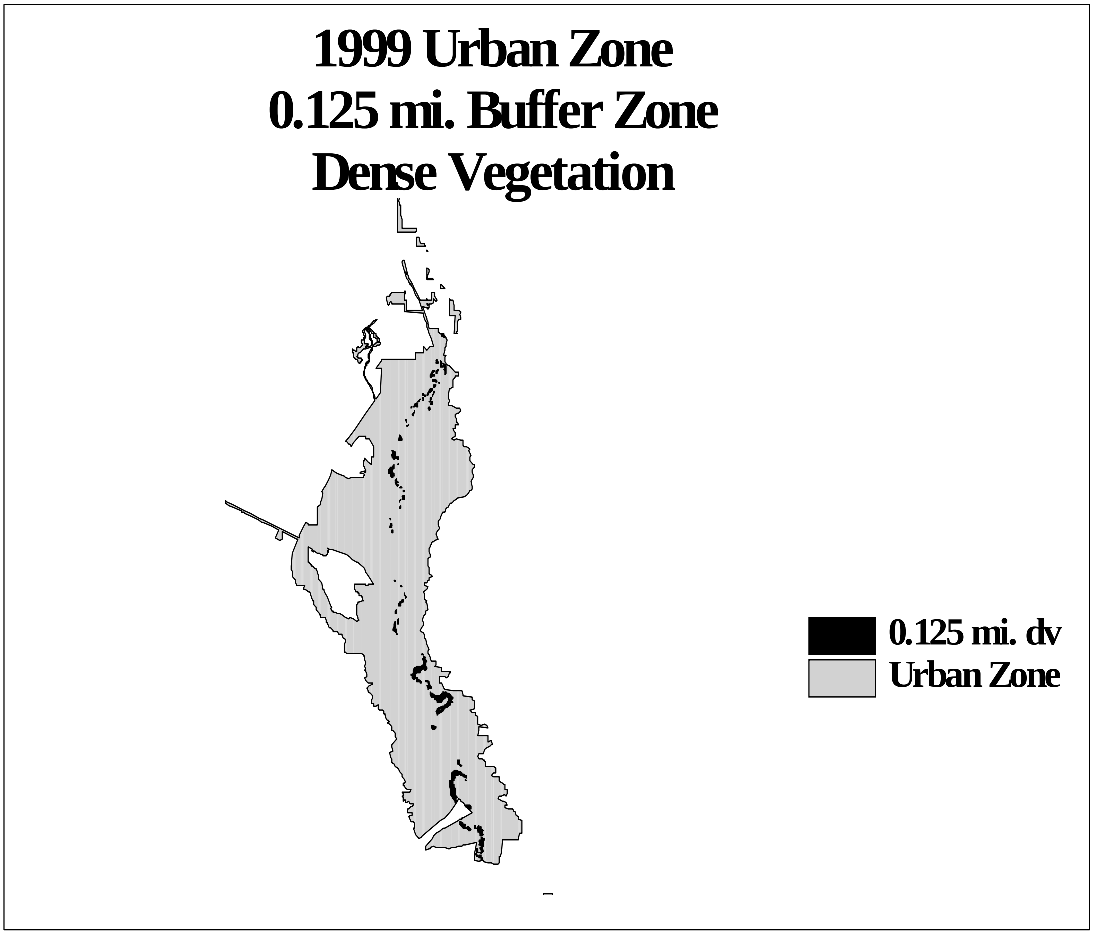


Figure 60. 1999 Dense Vegetation Class within 0.125 mi., Urban Area. Depicts dense vegetation class within 0.125 mi. of Leon Creek in the Urban Area. Polygons were created from the supervised classification and the dense vegetation class was examined for percent area.

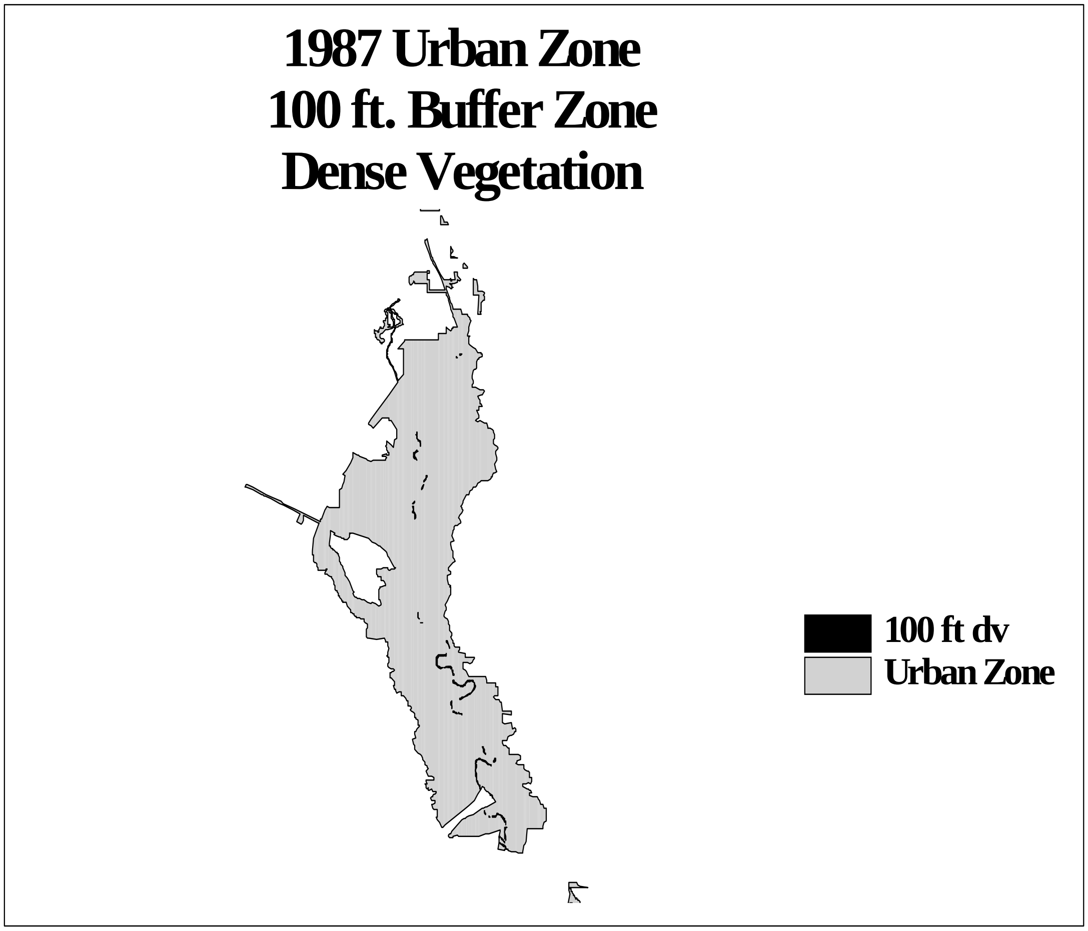


Figure 61. 1987 Dense Vegetation Class within 100 ft., Urban Area. Depicts dense vegetation class within 100 ft. of Leon Creek in the Urban Area. Polygons were created from the supervised classification and the dense vegetation class was examined for percent area.

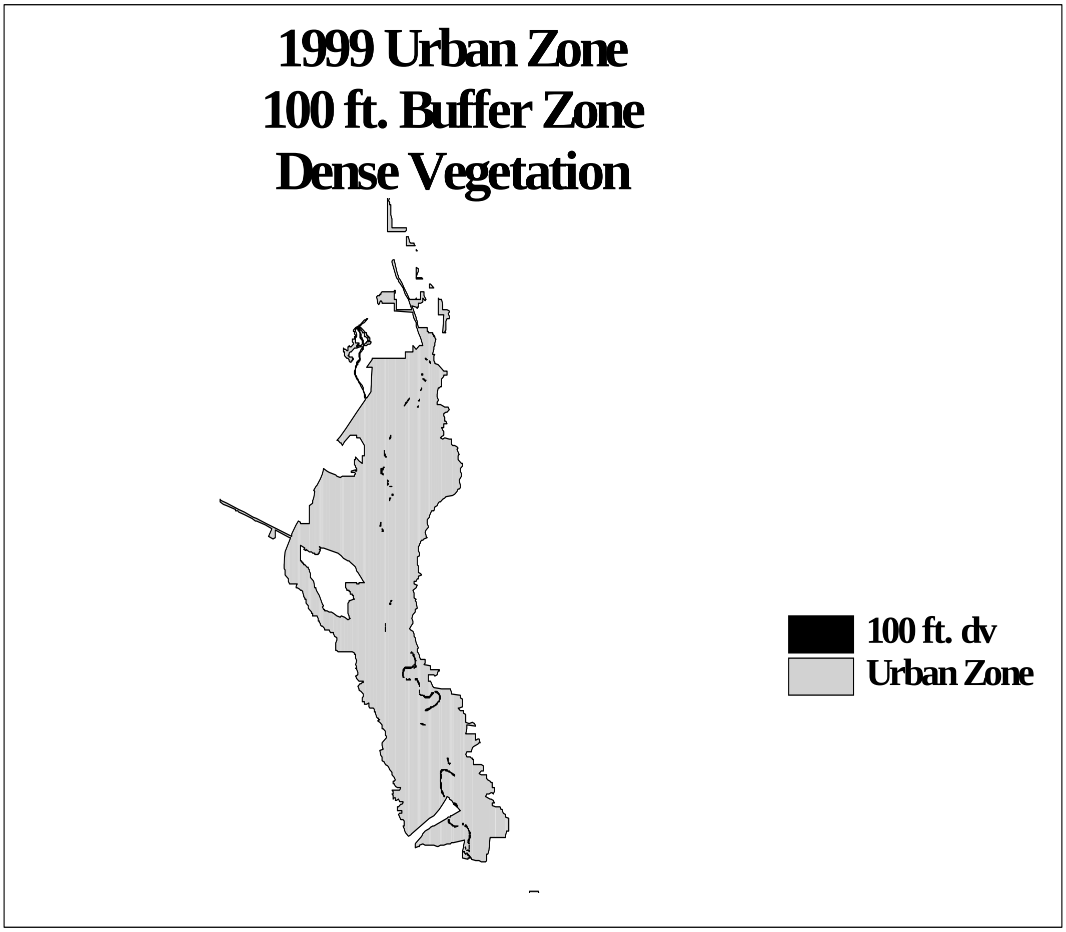


Figure 62. 1999 Dense Vegetation Class within 100 ft., Urban Area. Depicts dense vegetation class within 100 ft. of Leon Creek in the Urban Area. Polygons were created from the supervised classification and the dense vegetation class was examined for percent area.

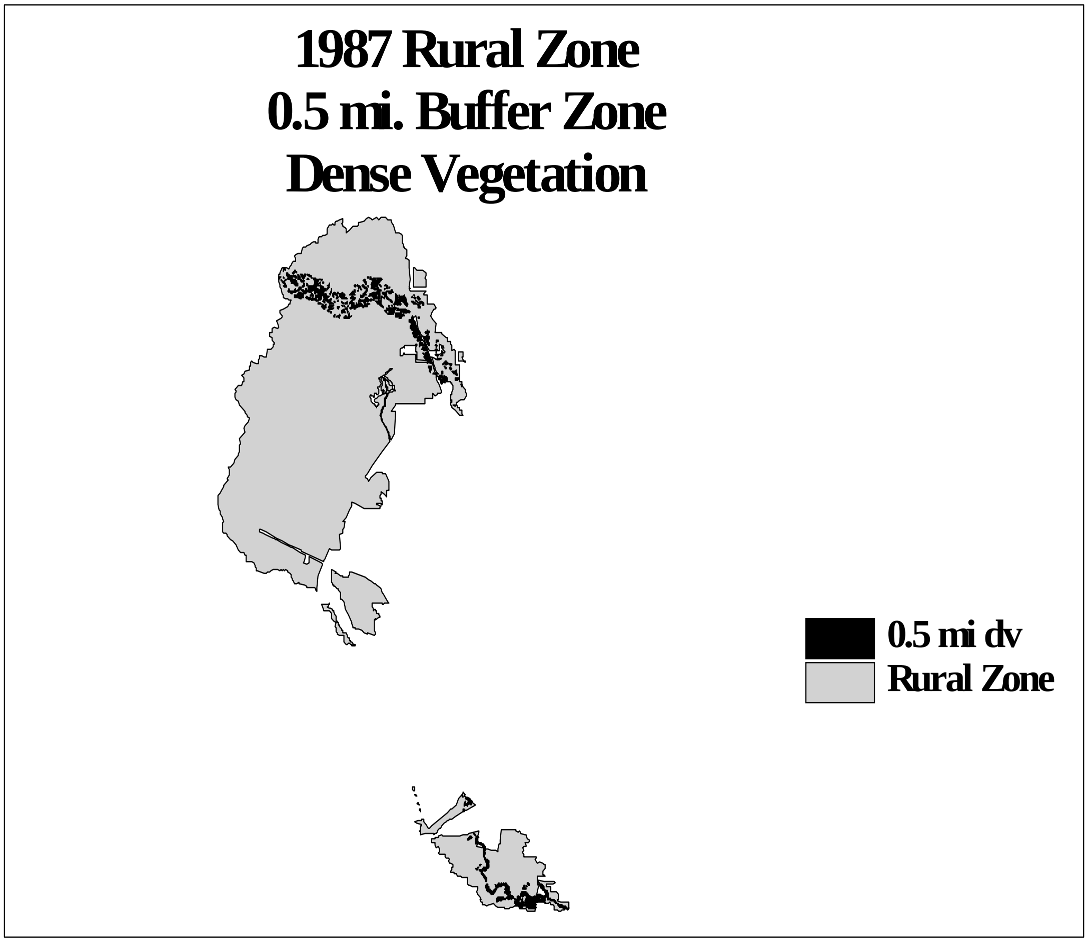


Figure 63. 1987 Dense Vegetation Class within 0.5 mi., Rural Area. Depicts dense vegetation class within 0.5 mi. of Leon Creek in the Rural Area. Polygons were created from the supervised classification and the dense vegetation class was examined for percent area.

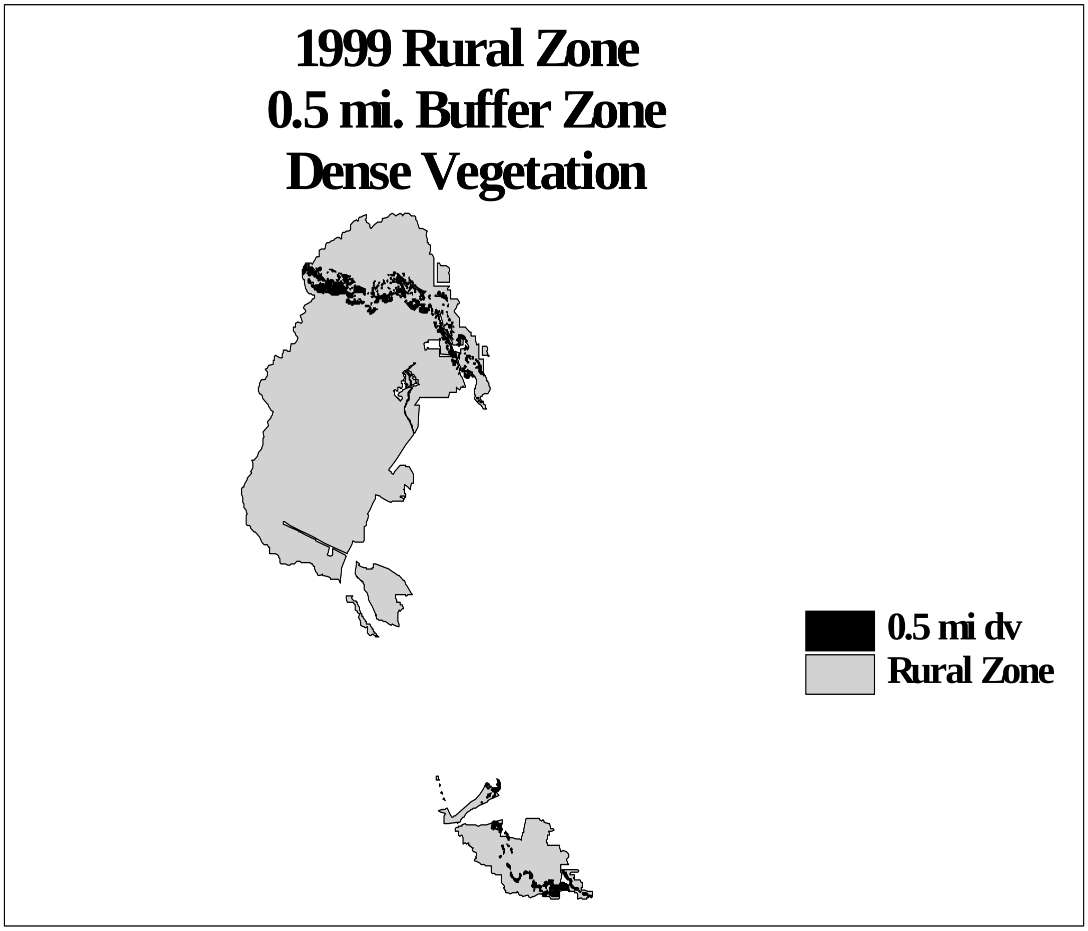


Figure 64. 1999 Dense Vegetation Class within 0.5 mi., Rural Area. Depicts dense vegetation class within 0.5 mi. of Leon Creek in the Rural Area. Polygons were created from the supervised classification and the dense vegetation class was examined for percent area.

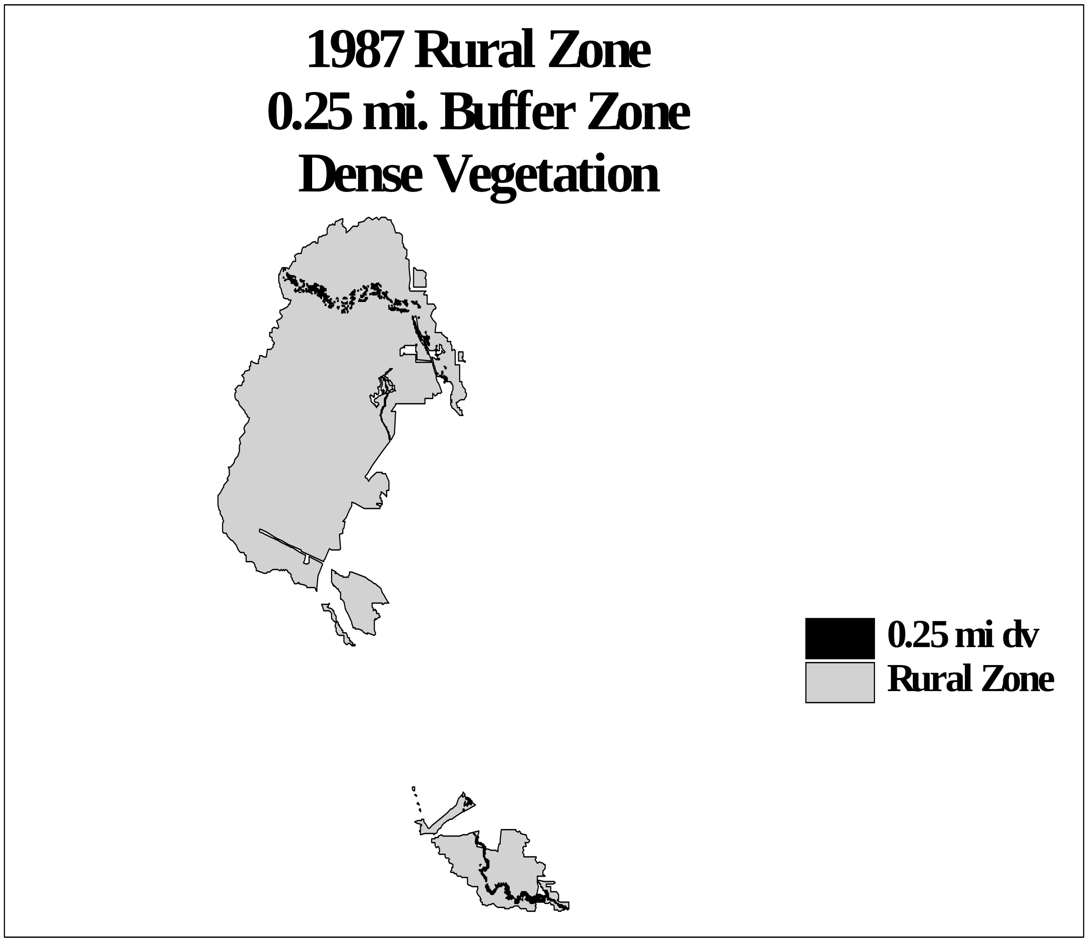


Figure 65. 1987 Dense Vegetation Class within 0.25 mi., Rural Area. Depicts dense vegetation class within 0.25 mi. of Leon Creek in the Rural Area. Polygons were created from the supervised classification and the dense vegetation class was examined for percent area.

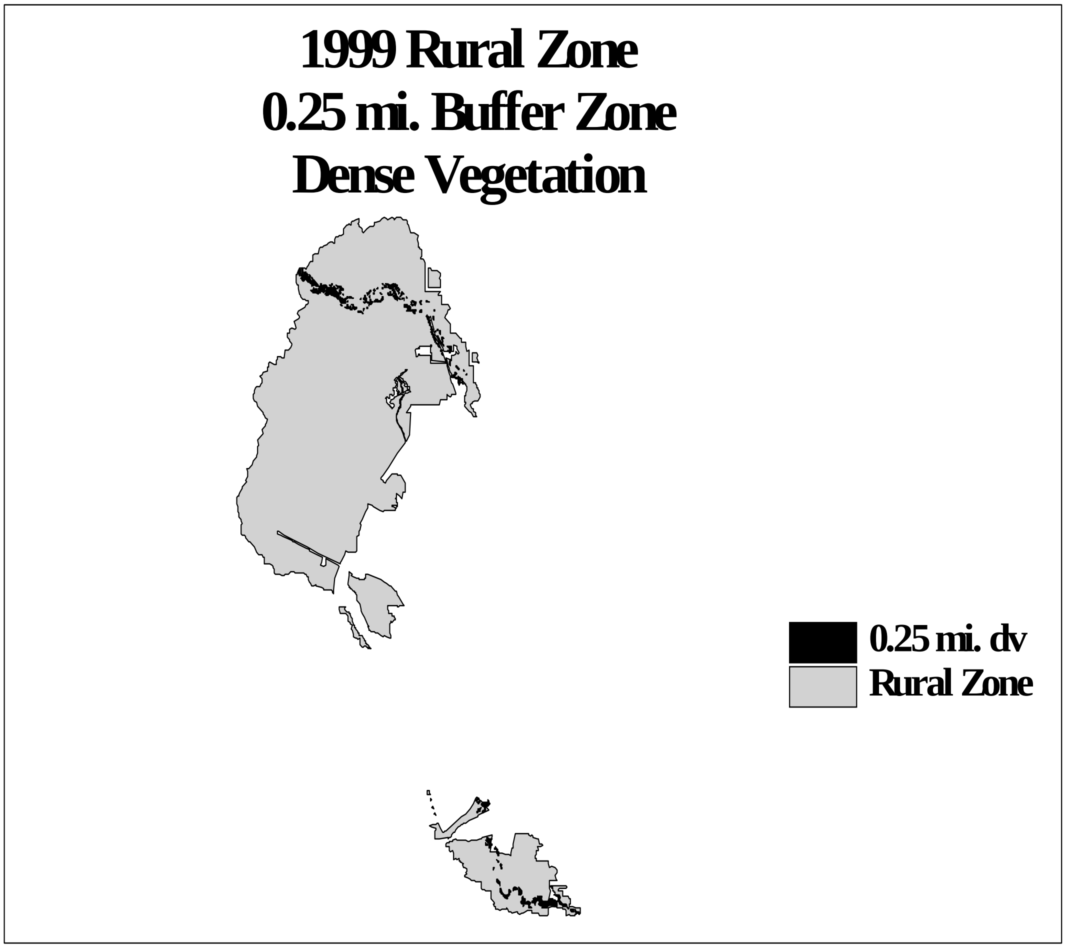


Figure 66. 1999 Dense Vegetation Class within 0.25 mi., Rural Area. Depicts dense vegetation class within 0.25 mi. of Leon Creek in the Rural Area. Polygons were created from the supervised classification and the dense vegetation class was examined for percent area.

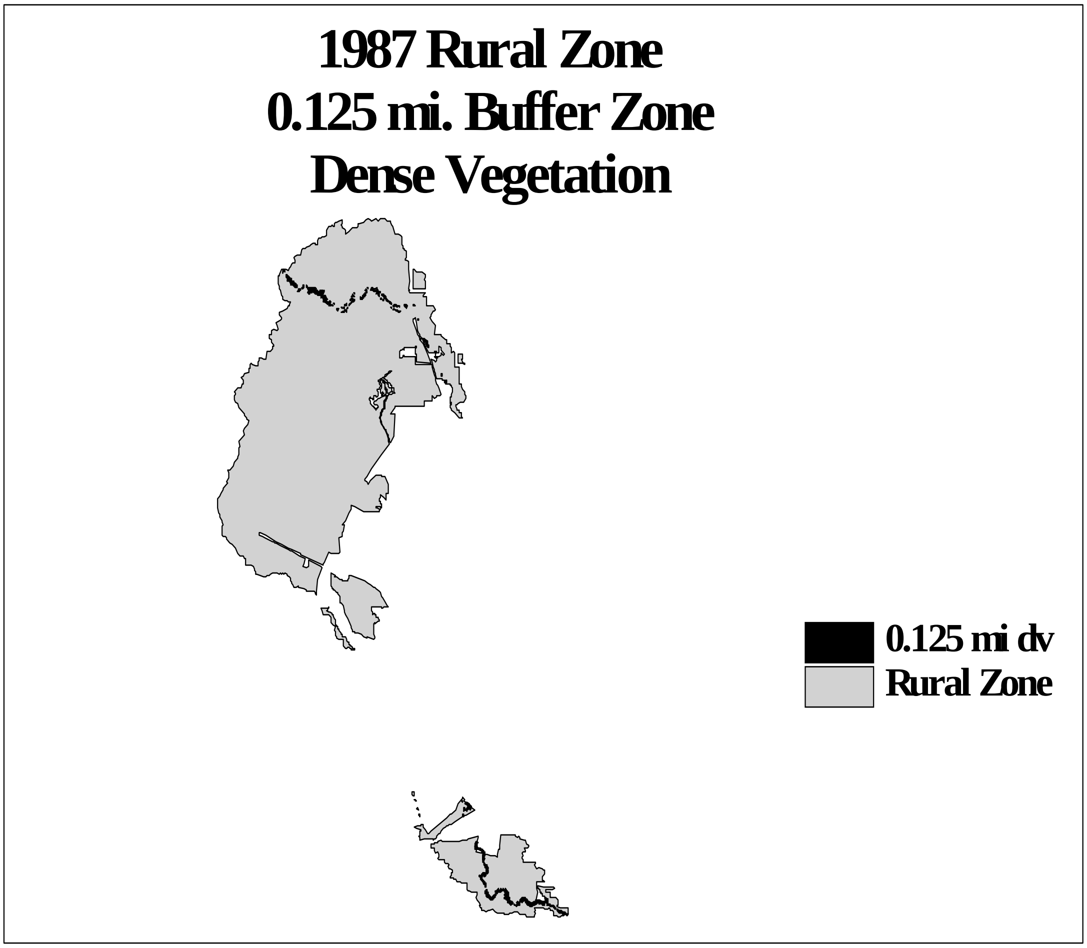


Figure 67. 1987 Dense Vegetation Class within 0.125 mi., Rural Area. Depicts dense vegetation class within 0.125 mi. of Leon Creek in the Rural Area. Polygons were created from the supervised classification and the dense vegetation class was examined for percent area.

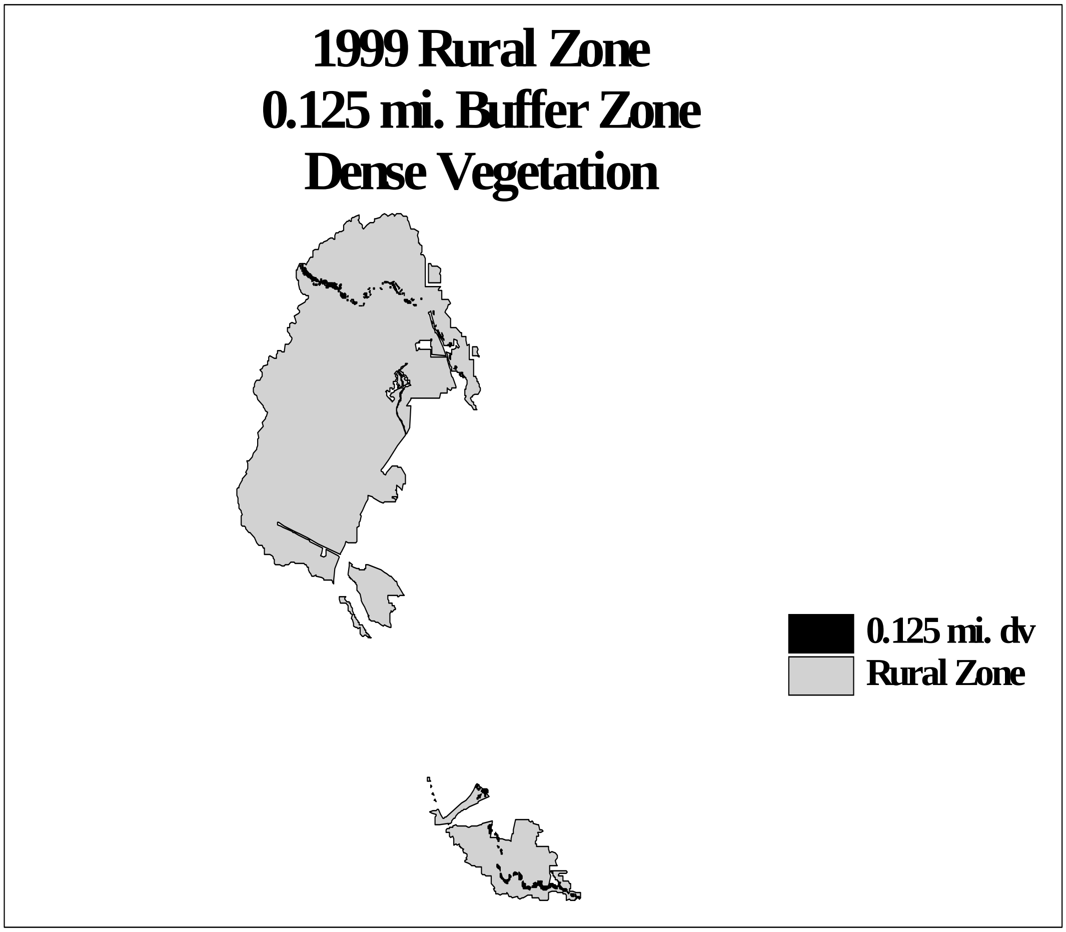


Figure 68. 1999 Dense Vegetation Class within 0.125mi., Rural Area. Depicts dense vegetation class within 0.125 mi. of Leon Creek in the Rural Area. Polygons were created from the supervised classification and the dense vegetation class was examined for percent area.

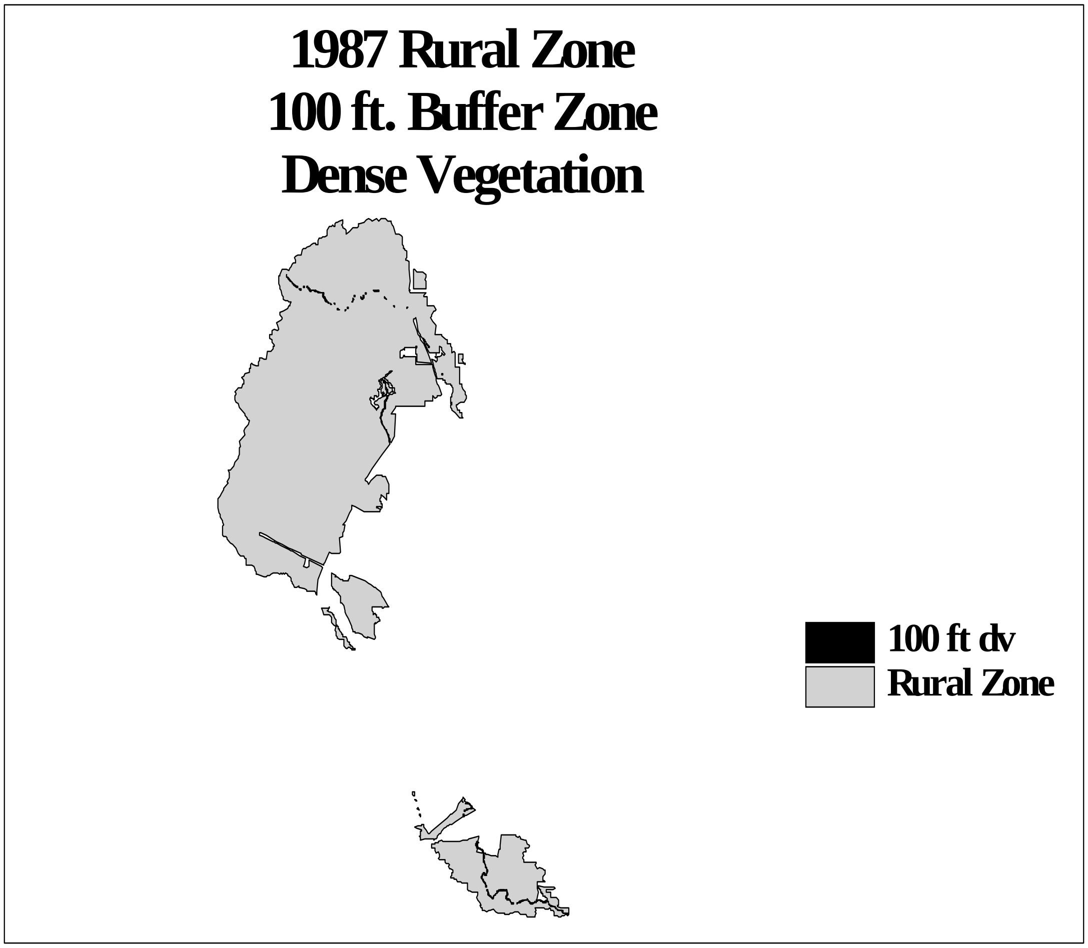


Figure 69. 1987 Dense Vegetation Class within 100ft., Rural Area. Depicts dense vegetation class within 100 ft. of Leon Creek in the Rural Area. Polygons were created from the supervised classification and the dense vegetation class was examined for percent area.

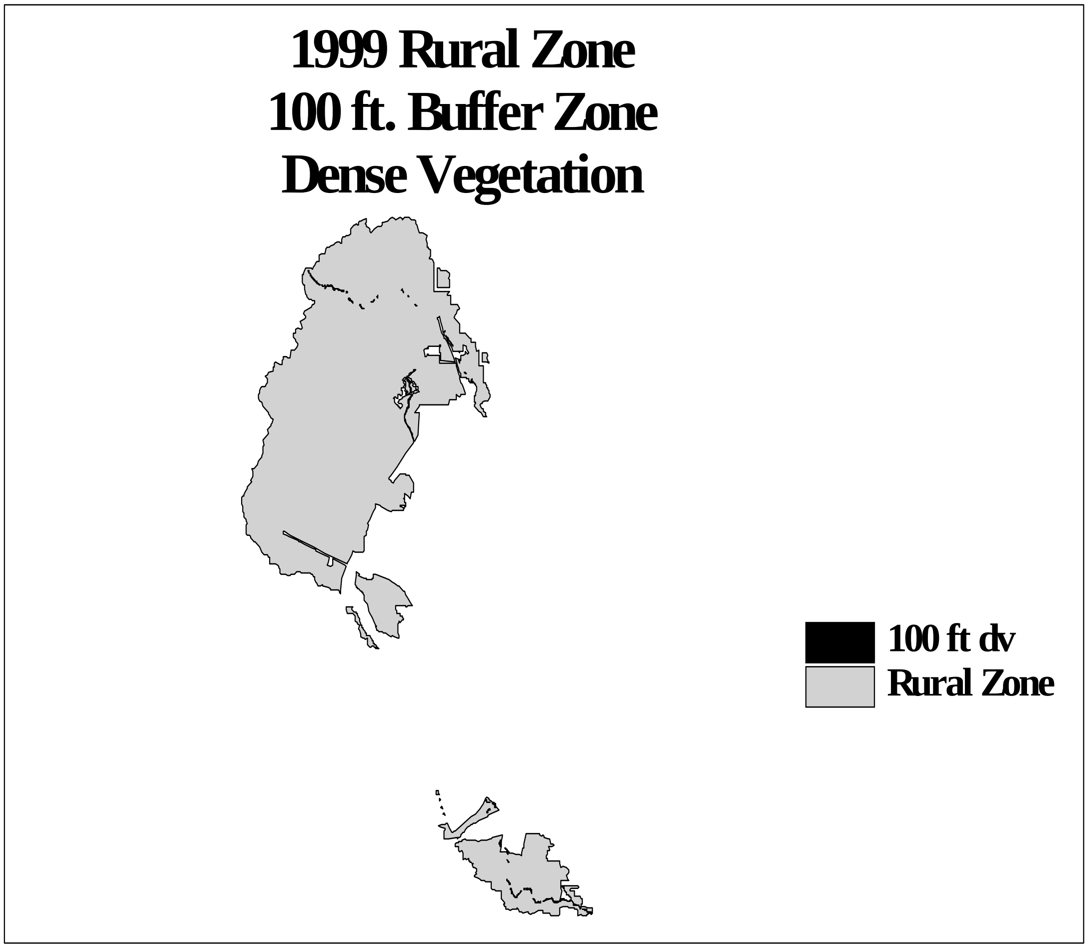


Figure 70. 1999 Dense Vegetation Class within 100ft., Rural Area. Depicts dense vegetation class within 100 ft. of Leon Creek in the Rural Area. Polygons were created from the supervised classification and the dense vegetation class was examined for percent area.

The percent area of dense vegetation in the 1987 and 1999 was then calculated from polygons created from the supervised classification to support or refute the image difference calculation findings. Within the total watershed, there was a much higher percent area in the 100 ft. buffer zone in both 1987 and 1999 (Figure 71). There was a slight decline in 1999 in the 100 ft. buffer zone from 38.7% in 1987 to 32.8% in 1999. There was a gradual decline in percent area of dense vegetation as the buffer zones increased to 0.5 mi, with little difference between 1987 and 1999. This suggests that most of the decrease in vegetation is occurring farther from the creek, where most development activities would occur.

The urban zone has lower percentage values in all buffer zones in 1999 (Figure 72), suggesting decreases in vegetation areas. The urban zone exhibits the same pattern as the total watershed with a decline in percent vegetation areas as the buffer zones extend out, with the 0.5 mi. zone having the least vegetation (approximately 25%) and the 100 ft. zone (approximately 50%) as having the most for both years. This is also true for rural zone where there is a higher percent area in 1987 than 1999 but only slightly (Figure 73). All the other buffer zones (0.5 mi, 0.25 mi, 0.125 mi) are at equal levels of percent area, all showing increases in 1999 (approximately 11%). This suggests that there are vegetation increases in the rural area and declines in the urban area between 1987 and 1999.

When comparing the results from the percent area dense vegetation with the findings of the Image Difference Calculation, there are similar findings as well as contradictions. While the percent area dense vegetation, supports the findings of both the total watershed and the urban zone with the Image Calculation, it is contradictory to the findings in the rural zone. The percent area of dense vegetation in the total watershed supports the image difference results in that it also showed there were higher decreases in vegetation in all the zones with both the highest increase and decrease in vegetation occurring in the 100 ft. buffer zone. Percent area of dense vegetation also supported the findings of the image difference in the urban, where there are higher decreases in all zones, with the greatest amount of decrease occurring in the 100 ft. zone. There are also low increases in all of the buffer zones.

Contradictions were found in the rural zone. From the Image Difference Calculation, there should be slight increases in vegetation in the 100 ft. buffer rather than a slight decrease in vegetation, which the percent area calculations show. The percent area calculations also suggest there would be decreases in vegetation in all other buffer zones, other than the 100 ft. zone, however these outer buffer zones show slight increases.

The findings between the percent area and image difference are opposite from what would be expected in the rural zone. This may be due simply to calculation errors or it could possibly have to do with the spatial configuration of what is defined as "rural". Both the total watershed area and the urban zone are cohesive land units, whereas the rural zone is broken up into two sectors (in the north and in the south). This may give confounding results.

The percent area results in the rural zone shows that overall in 1987 and 1999, there is less of a percent area of dense vegetation than in the urban zone. Also, in the rural zone there is an increase shown in dense vegetation in 1999 in all buffer zones, except at the 100 ft. buffer zone, which showed the highest percent area in both 1987 and 1999. This may indicate why the Image Difference Calculation shows the highest percent changes in the rural 100 ft. buffer zone. The percent area showed a greater percent area of dense vegetation in all urban buffer zones, with slight decreases in 1999 in all buffer zones.

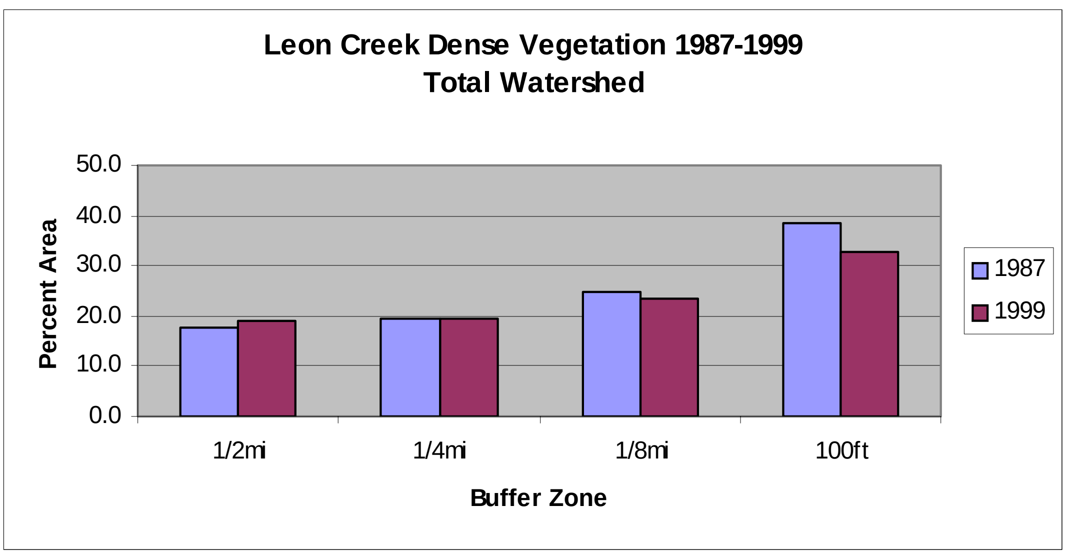


Figure 71. Total Watershed: Percent Area of Dense Vegetation. There was a gradual decline in percent area of dense vegetation as the buffer zones increased to 0.5 mi., with little difference between 1987 and 1999.

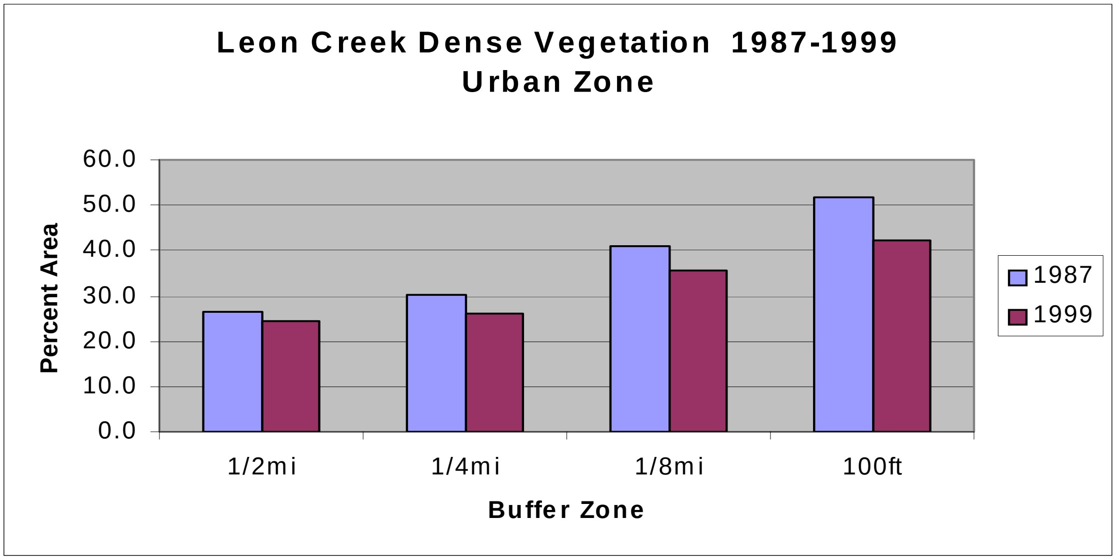


Figure 72. Urban Zone: Percent Area of Dense Vegetation. The urban zone exhibits the same pattern as the total watershed with a decline in percent vegetation areas as the buffer zones extend out, with the 0.5 mi. zone having the least vegetation (approximately 25%) and the 100 ft. zone (approximately 50%) as having the highest level.

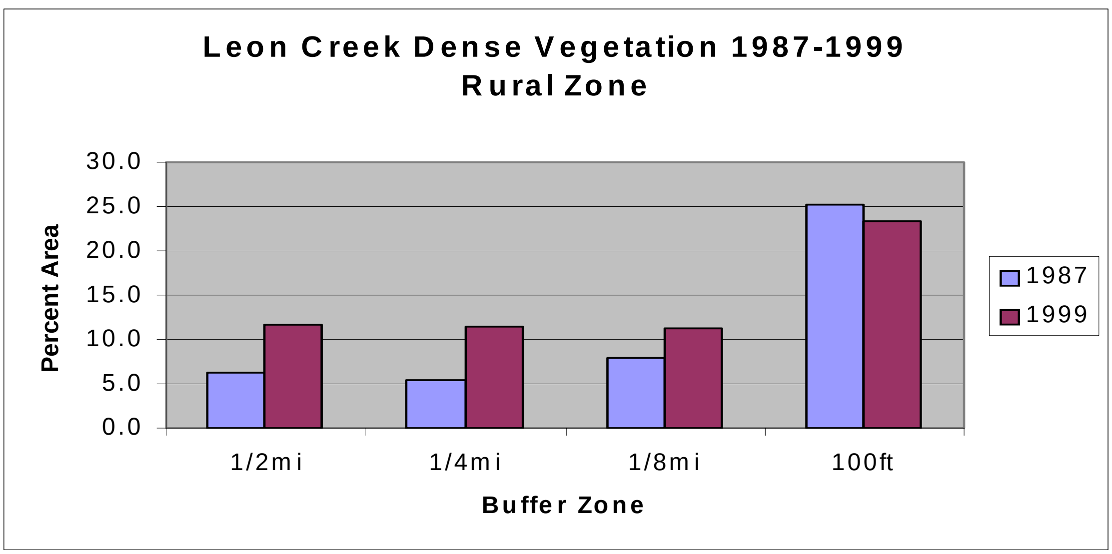


Figure 73. Rural Zone: Percent Area of Dense Vegetation. In the rural zone, there is a higher percent area in 1987 than 1999 but only slightly. All the other buffer zones (0.5 mi., 0.25 mi., 0.125 mi.) are at equal levels of percent area, all showing increases in 1999 (approximately 11%).

Landscape Indices Results

The Patch Analyst metrics of Number of Patches (NUMP), Mean Patch Size (MPS), and Patch Size Coefficient of Variation (PSCOV) were first analyzed for number of patches and patch size characteristics. When observing Number of Patches (NUMP) in both the 1987 and 1999 images, there is a slightly larger number in 1999 (428 to126 range) than 1987 (420 to 126 range) in all buffers (Figure 74). This is also true for the urban zone (Figure 75). This means that there are more dense vegetation patches in 1999 than 1987. In the rural zone, the NUMP increases in all buffer zones between 1987 and 1999, except for the 0.5 mi. buffer zone, where there is a decrease in patches (Figure 76). NUMP must be examined both with the Mean Patch Size (MPS) and the Patch Size Coefficient of Variation (PSCOV). Both MPS and PSCOV will tell us whether these larger quantities of patches in 1999 are smaller patches that vary in size.

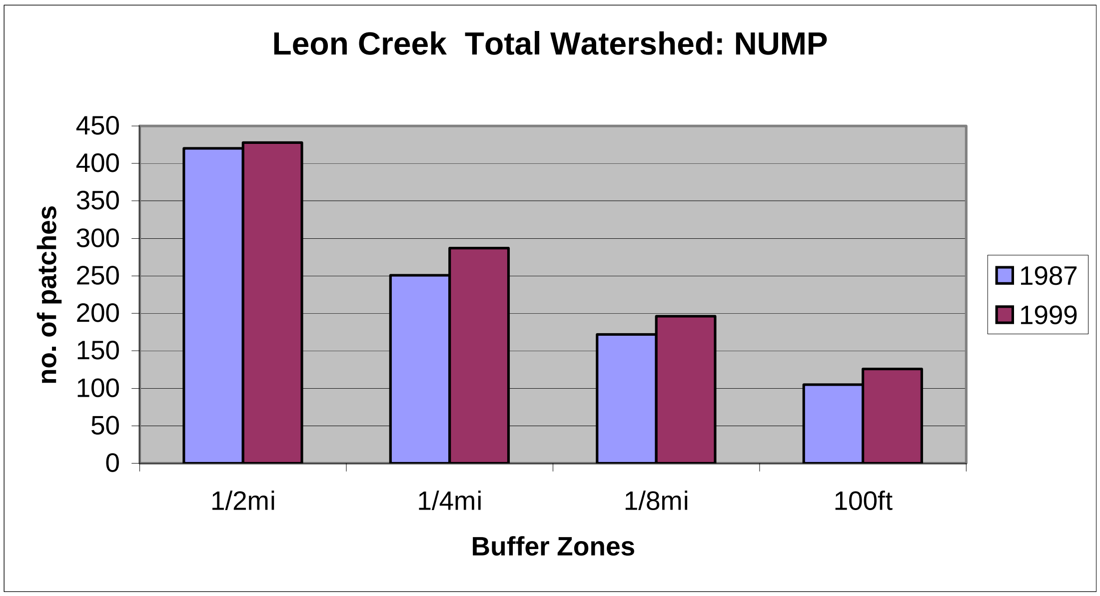


Figure 74. Number of Patches, Total Watershed Area. Graph depicts a slightly larger number in 1999 (428 to 126 range) than 1987 (420 to 126 range) in all buffer zones.

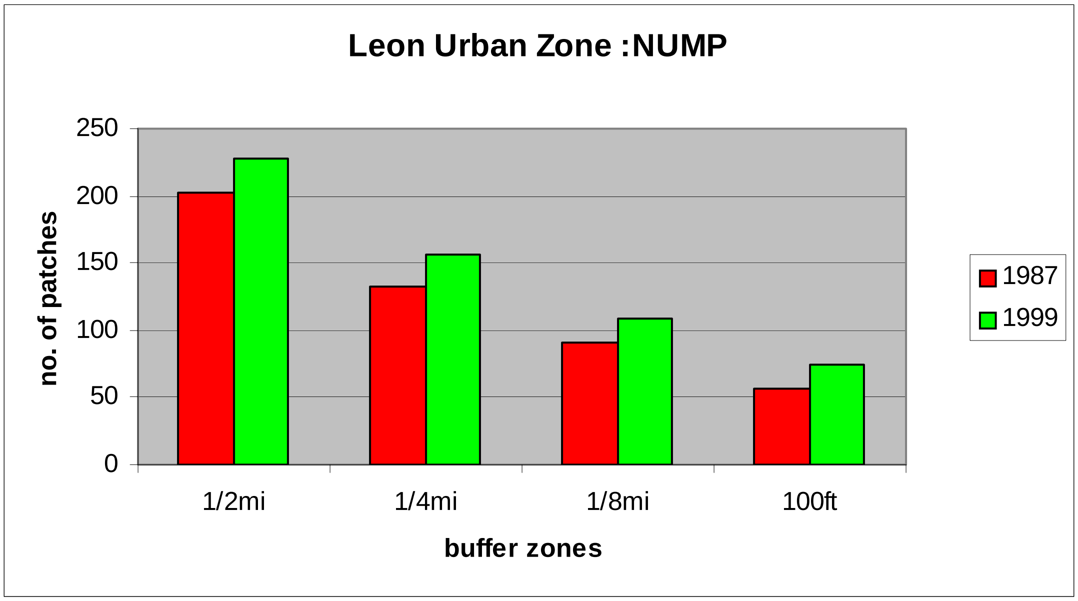


Figure 75. Number of Patches, Urban Zone. Graph depicts a slightly larger number in 1999 (225 to 75 range) than 1987 (200 to 55 range) in all buffer zones.

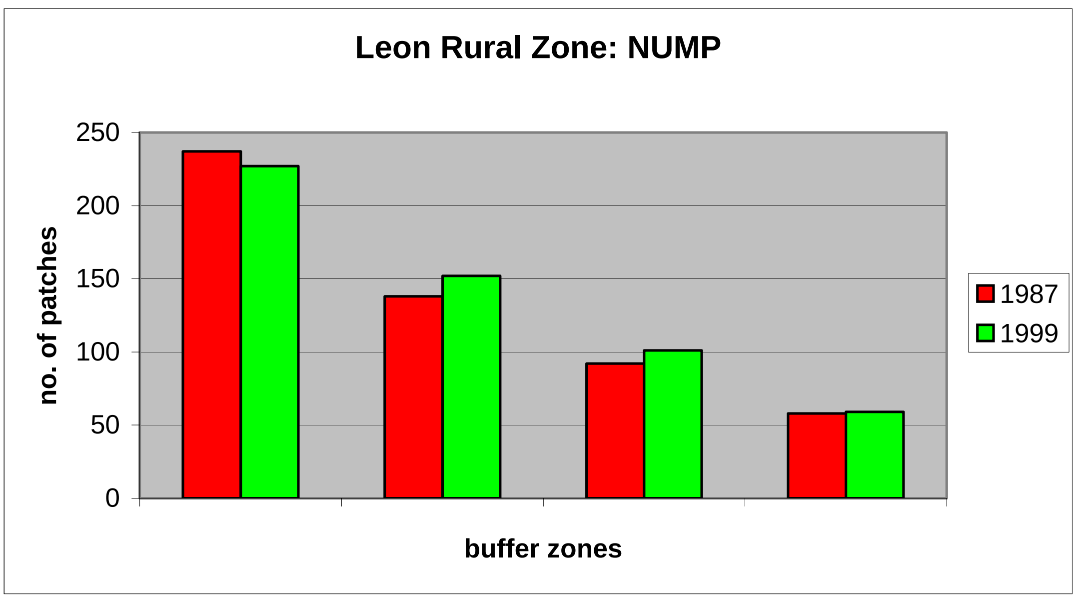


Figure 76. Number of Patches, Rural Zone. Graph depicts increases in all buffer zones between 1987 and 1999, except for the 0.5 mi. buffer zone, where there is a decrease in patches.

Within the total watershed in 1999, there were consistently smaller Mean Patch Size values in each buffer zone in 1999 (5.2 to 1.3 ha range) than in 1987 (5.0 to 1.9 ha range), except for 0.5 mi. zone (Figure 77). Smaller MPS indicates more fragmentation may have occurred in 1999. In the urban zone, the Mean Patch Size seems significantly less in 1999 (7.3 to 1.4 ha range) than 1987 (8.8 to 2.3 ha range) in all of the buffer zones (Figure 78) as opposed to the Mean Patch Size for the total watershed area. The rural zone shows MPS results that are similar to the NUMP, where the mean size decrease in all buffer zones by 1999, except for the 0.5 mi. zone (Figure 79).

The total watershed 1999 landscape also had a higher Patch Size Coefficient of Variation (PSCOV) indicating larger variation in patch size in 1999 (321.7 to 167.7 range) than 1987 (304.8 to 161.0 range) (Figure 80). The Patch Size Coefficient of Variation in the urban zone (Figure 81) did not show much change from the total watershed results, with 1999 patches showing more patch size variation. The PSCOV results in the rural zone indicate an increase in size variation in all buffer zones, except for the 100 ft zone (Figure 82). The Patch Size Coefficient of Variation "must be viewed with MPS or it can be misleading," (McGarigal and Marks, 1995).

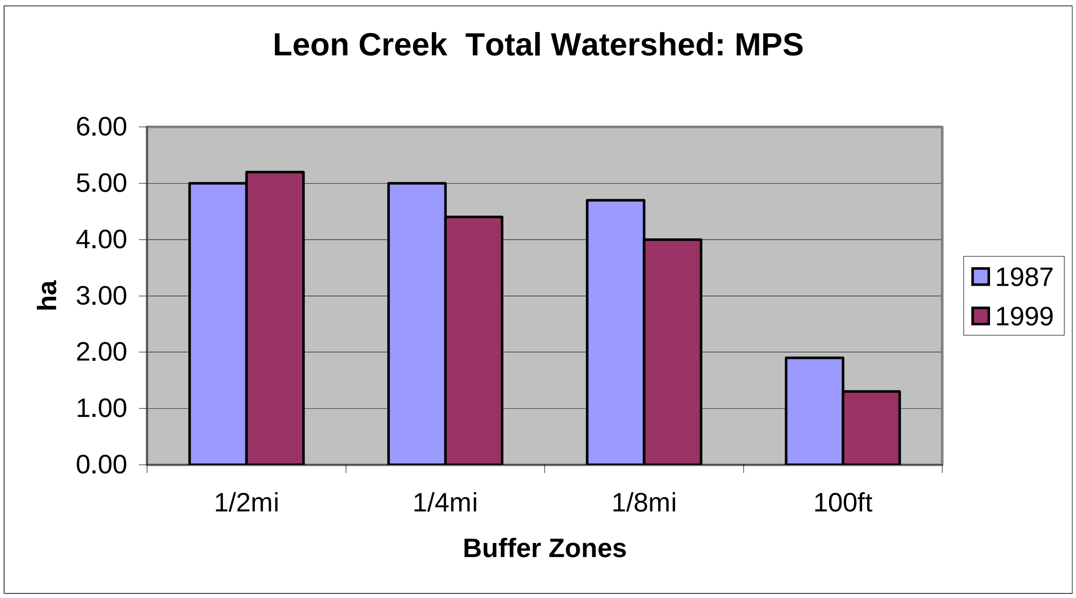


Figure 77. Mean Patch Size, Total Watershed Area. Graph depicts consistently smaller Mean Patch Size values in each buffer zone in 1999 (5.2-1.3 ha range) than in 1987 (5.0 to 1.9 ha range), except for 0.5mi zone

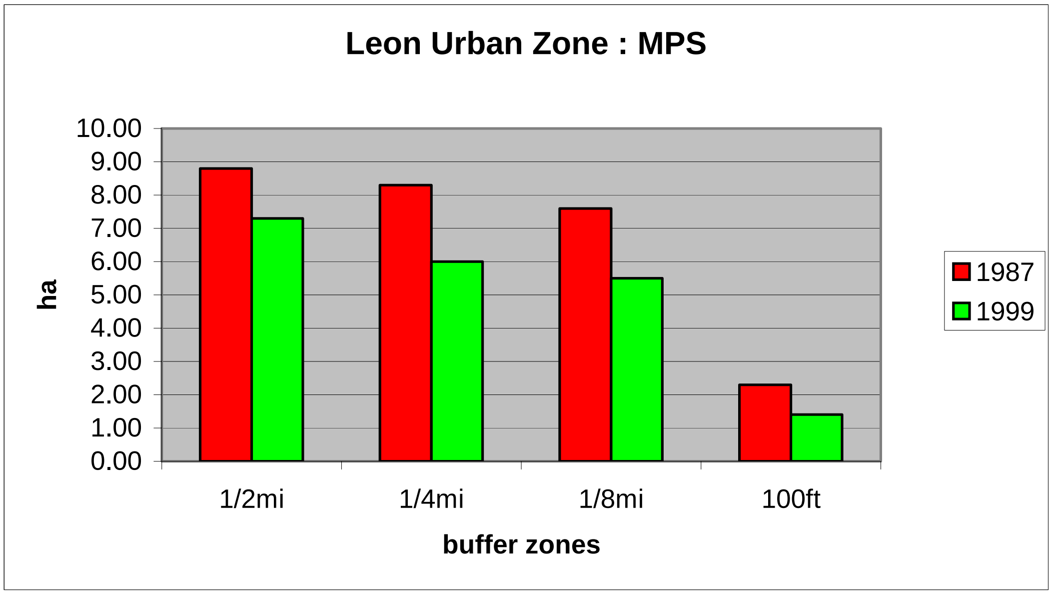


Figure 78. Mean Patch Size, Urban Zone. The MPS seems significantly less in 1999 (7.3 to 1.4 ha range) than 1987 (8.8 to 2.3 ha range) in all of the buffer zones.

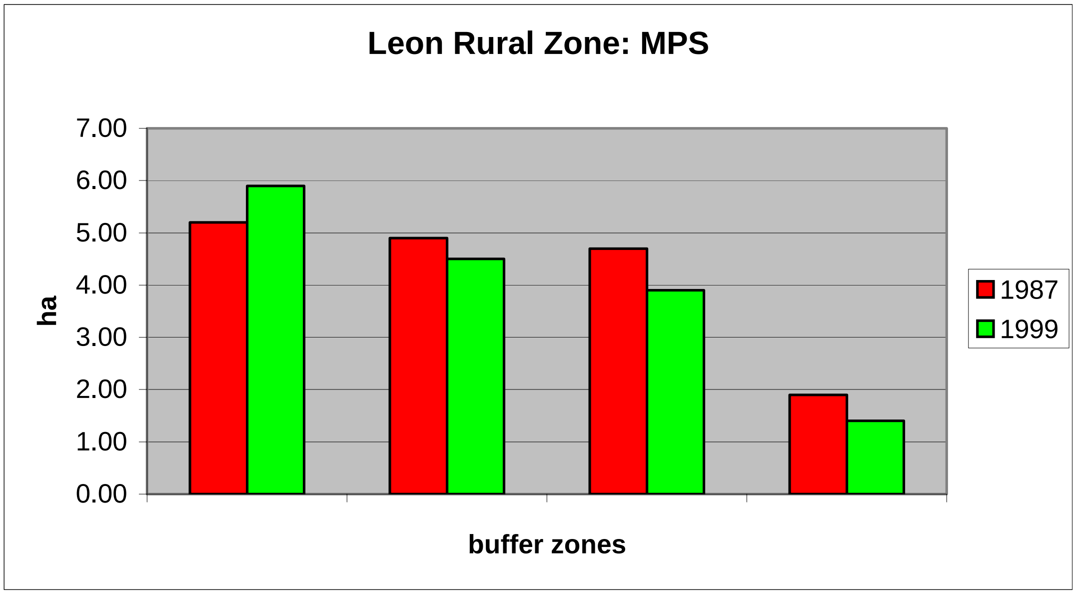


Figure 79. Mean Patch Size, Rural Zone. Graph depicts the mean size decrease in all buffer zones in 1999, except for the 0.5 mi. zone.

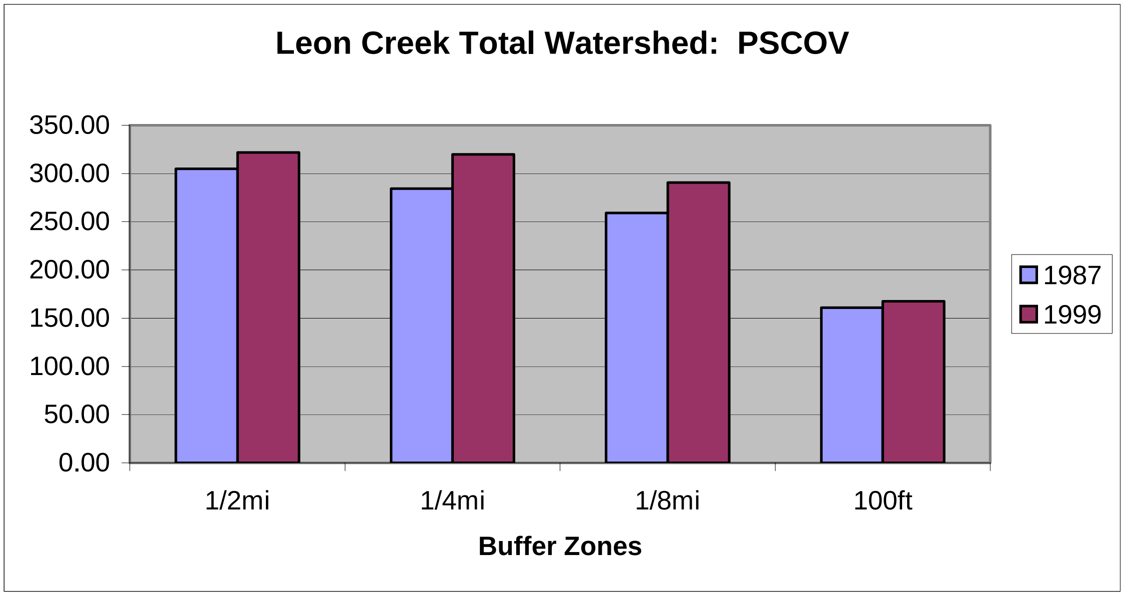


Figure 80. Patch Size Coefficient of Variation, Total Watershed Area. Graph depicts the 1999 landscape had a higher Patch Size Coefficient of Variation (PSCOV) indicating larger variation in patch size in 1999 (321.7 to 167.7 range) than 1987 (304.8 to 161.0) range.

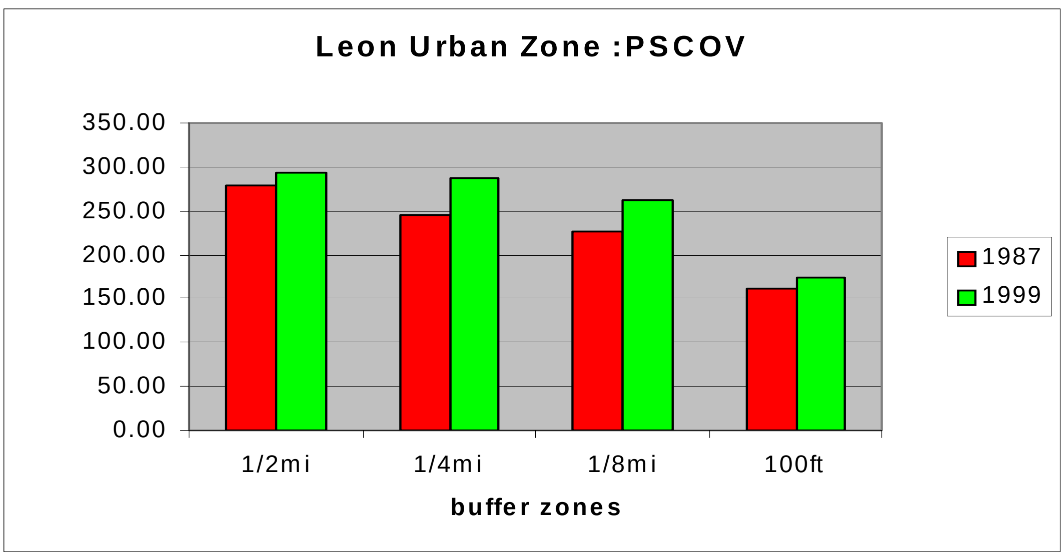


Figure 81. Patch Size Coefficient of Variation, Urban Zone. Graph depicts PSCOV higher in 1999, meaning 1999 patches having more patch size variation.

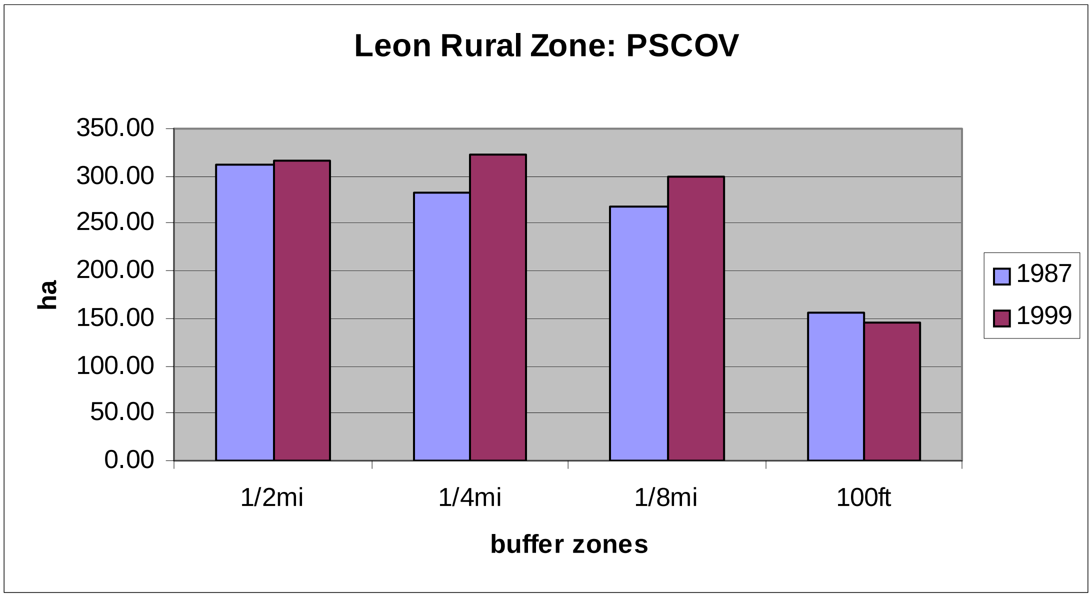


Figure 82. Patch Size Coefficient of Variation, Rural Zone. The PSCOV results in the rural zone indicate an increase in size variation in all buffer zones, except for the 100 ft zone.

Combining the results from these indices may indicate that in 1999 the number of dense vegetation patches increased because they have been more fragmented, therefore they are patches that are smaller in size with a high variation in their sizes. Similar results are also found in the urban zone, only with a more pronounced difference between 1987 and 1999. This may indicate that more fragmentation has occurred within the urban zone, as opposed to the entire watershed. All of these indices (Number of Patches, MPS, and PSCOV) show an overall decrease as the buffer zone decreases in size, as expected. The 1987 landscape had a more narrow range of patch sizes overall which may mean the patches are more uniform in texture as opposed to when there are a variety of patch sizes (Hargis et al, 1997).

The Mean Shape Index (MSI) and the Area Weighted Mean Shape Index (AWMSI) are indices that help determine the complexity of shape of the dense vegetation patches in 1987 and 1999. Human-induced fragmentation usually causes patches to become less complex in shape, while patches with high complexity are usually the least fragmented. In all buffer zones in the total watershed, the 1999 image shows a value closest to 1.0 for the MSI but only slightly (Figure 83). There is a pronounced difference between 1999 and 1987 when the AWMSI is examined, suggesting that the patch size is affecting the mean values (Figure 84).

When the index is area weighted according to patch size, then there is a clear difference between 1987 and 1999 in all buffer zones. This difference is even more pronounced in the urban zone, suggesting that the shape complexity of the patches has been even more affected between 1987 and 1999 in the urban zone (Figures 85 and 86). There are limitations to the MSI and the AWMSI in that they are both "subject to limitations of the first order statistics" or that the average shape value for a landscape could be misleading if there is a high distribution of complex patch shapes (McGarigal and Marks, 1995). In the rural zone, the MSI displays a decrease in shape complexity in all buffer zones, except for the 100 ft zone, where there is an increase in shape complexity (Figure 87). It is curious to note that with the AWMSI results, there is a decrease in shape complexity in all buffer zones (Figure 88).

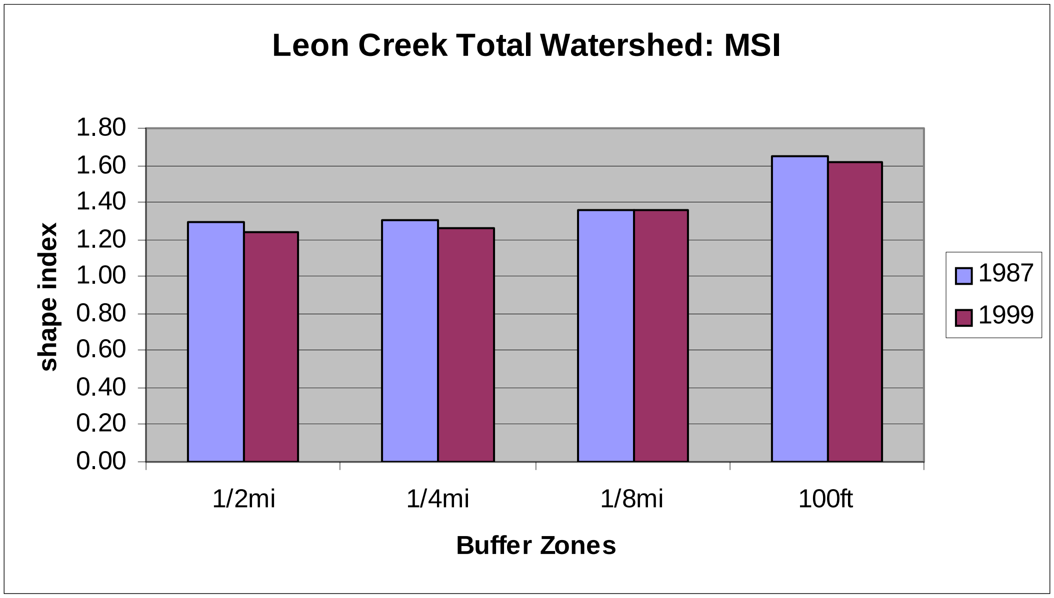


Figure 83. Mean Shape Index, Total Watershed Area. The 1999 vegetation patches show a value closest to 1.0 for the MSI but only slightly.

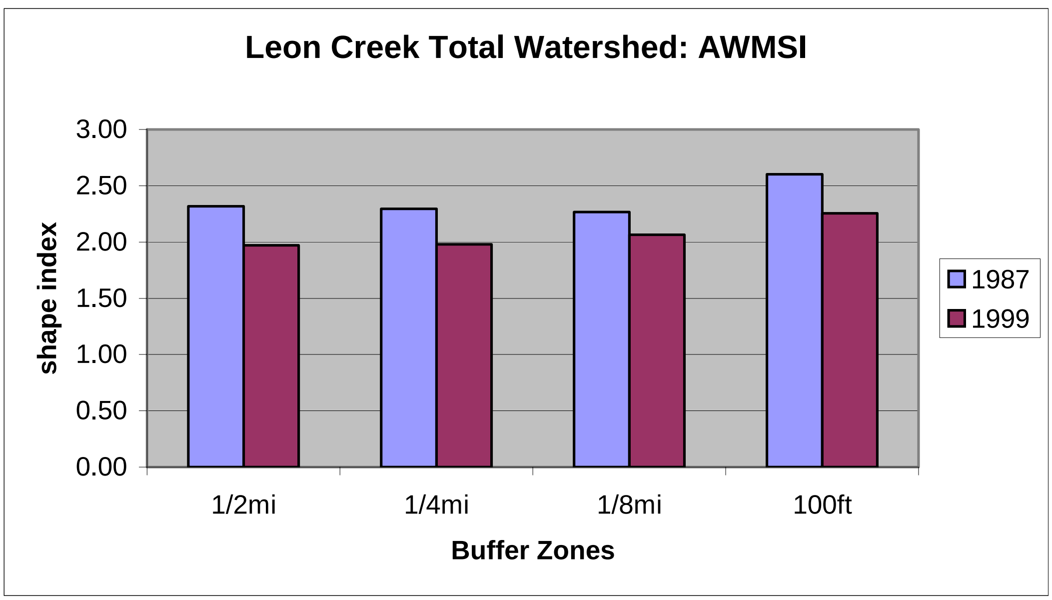


Figure 84. Area Weighted Mean Shape Index, Total Watershed Area. There is a pronounced difference between 1999 and 1987 when the AWMSI is examined, suggesting that the patch size is affecting the mean values. The graph depicts 1999 patches as closer to a value of 1.0.

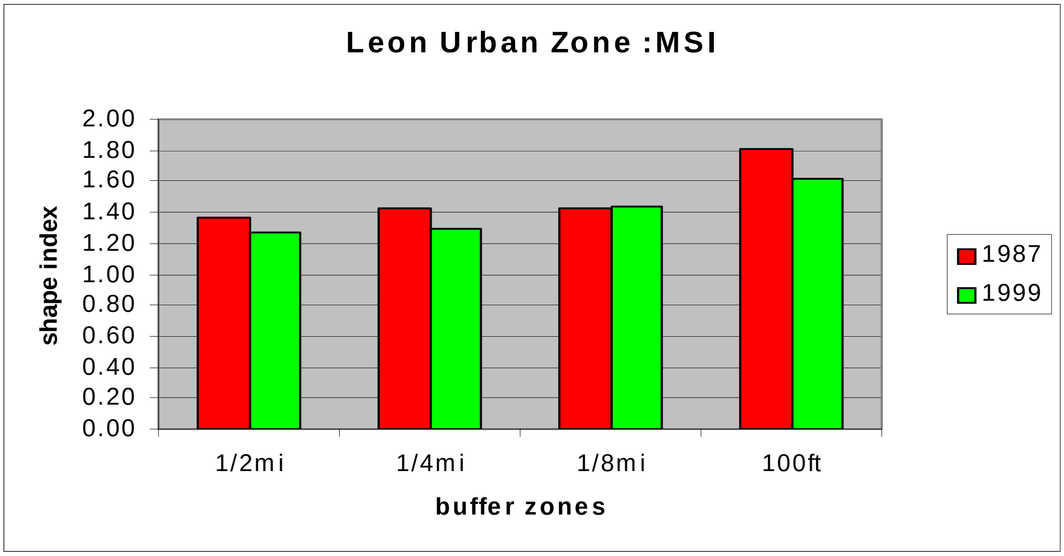


Figure 85. Mean Shape Index, Urban Zone. The graph shows only slight differences between 1987 and 1999, with 1999 patches closer to 1.0 value.

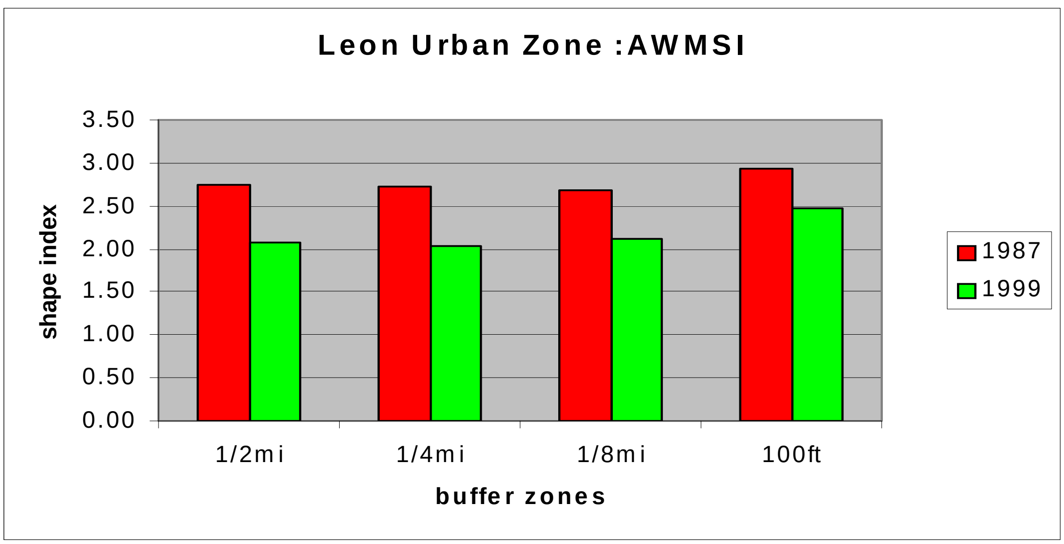


Figure 86. Area Weighted Mean Shape Index, Urban Zone. The difference between the AWMSI in 1987 and 1999 is greater in the urban zone. This suggests that the shape complexity of the patches has been even more affected between 1987 and 1999 in the urban zone

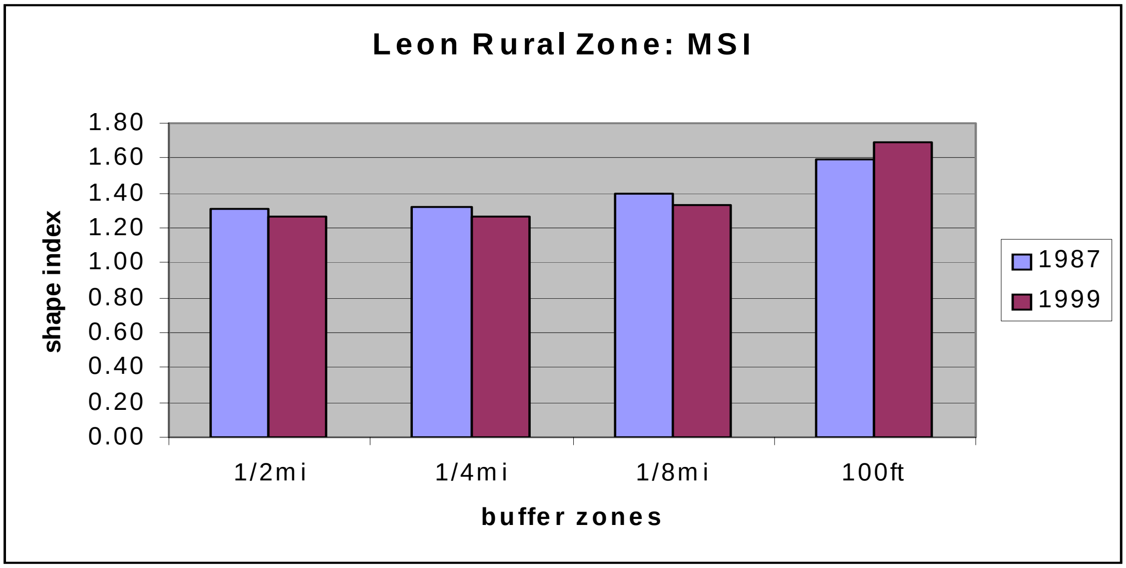


Figure 87. Mean Shape Index, Rural Zone. The graph shows only slight differences between 1987 and 1999, with 1999 patches closer to 1.0 value except for with the 100 ft. zone.

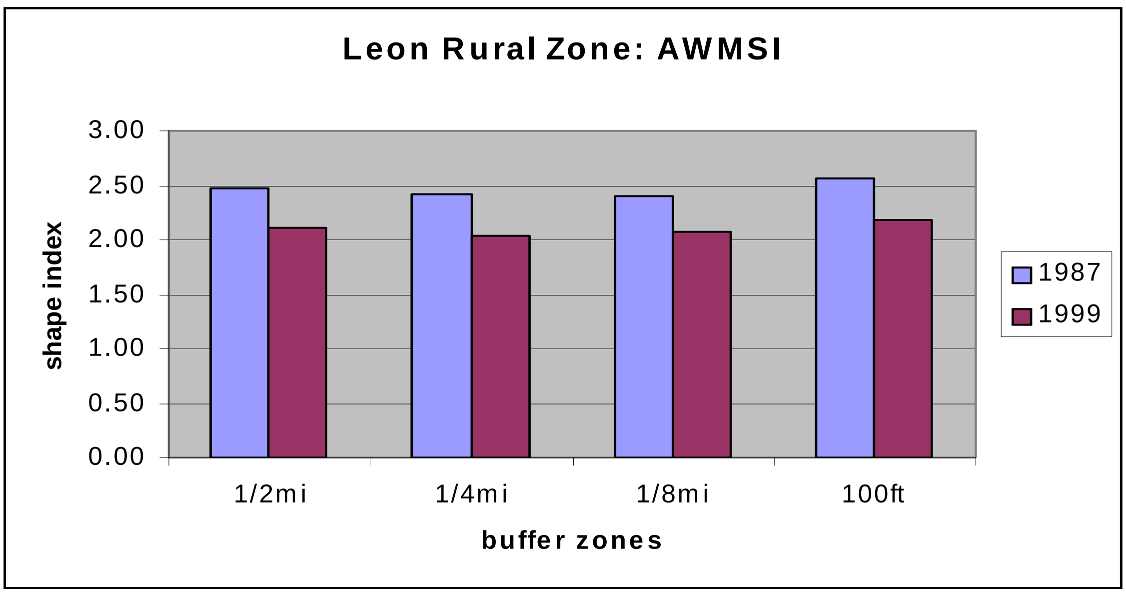


Figure 88. Area Weighted Mean Shape Index, Rural Zone. The AWMSI shows a more pronounced difference than the MSI, showing 1999 values closer to 1.0.

Conclusions

The preparation of the data is the first phase of the image analysis process, and it is also the foundation for the temporal mapping process. The two main uses for the 1987 and 1999 satellite images were for the Image Difference Calculation and the supervised classification. There are several different geo-referencing methods to be used such as Nearest Neighbor, Cubic Convolution, and Bilinear Interpolation. For this project, retaining the original data as much as possible was the main goal, not how sharp the image appeared. The Nearest Neighbor method fit that requirement the best. If the original data had not been retained as much as possible, this certainly would have resulted in false assessments for the Image Difference Calculation and the supervised classification.

Image Difference and Percent Area Calculation of dense vegetation patches were helpful in detecting rates of change between 1987 and 1999 (Table 31). The results for both the Image Difference and Percent Area were very similar for the total watershed and the urban zone, except for in the rural zone. This may be due to a calculation error or due to the configuration of the rural zone, which is not a cohesive unit. Both results for Image Difference Calculation and Percent Area Calculation suggest that for the total watershed, there are higher rates of decrease in vegetation occurring rather than increases. The greatest amount of change in the total watershed area, both in increases and decreases in vegetation, is occurring within the 100 ft. buffer zone, the zone closest to Leon Creek.

Less vegetation overall in the 0.5 mi. buffer zone in both the urban zone and total watershed areas, suggests that development activity may have affected productive vegetative growth especially in the urban zone. The 100 ft. buffer zone, closest to Leon Creek, has the greatest fluctuations between increases and decreases in vegetation suggesting that natural causation (flooding, change in stream morphology) may be affecting vegetative growth or non-growth rather than human-induced disturbances. These fluctuations of high decreases and increases may suggest this 100 ft zone may either have the highest resiliency of the zones or is the merely not developed as much as the outer buffer zones. "Many landscape disturbances, both natural and human-caused, are spatially heterogenous and set in motion a sequence of recovery events that profoundly influence the nature of stream ecosystems", (Jensen and Bourgeron, 1994).

The landscape patch size indices of MPS and NUMP both suggest fragmentation has occurred in both rural and urban areas by 1999, with the exception of the rural 0.5mi. buffer zone. Whereas, the PSCOV showed increased size variation in both urban and rural areas, except for the 100ft buffer zone. This also suggests fragmentation of riparian vegetation may have occurred by 1999 in both urban and rural zones, with the possible discrepancies of rural 0.5mi. and 100 ft. buffer zones. The landscape shape indices (AWMSI and AWMFD) are consistent in their results in both the rural and urban buffer zones, which show decreased complexity by 1999 indicating human-induced changes have occurred. Overall the shape indices seem to be indicating that there is an overall decrease in shape complexity in all buffer zones from 1987 to 1999, suggesting there has been more human-induced change.

These evaluations of the riparian vegetation of Leon Creek suggest a fragmentation trend from 1987 to 1999 indicated by the Image Difference Calculation, Percent Area Calculation, and the landscape indices (Table 31). There are questionable results with the rural zone results with both the patch size indices and the percent area results. There is an inconsistent result in the rural zone, with the percent area showing increases in dense vegetation by 1999, which would be inconsistent with increased fragmentation.

Both the Percent Area Calculations and Landscape Indices are consistent in the urban zone. Both the Image Difference Calculation and Landscape Indices are consistent in both the rural and urban zone, with the Image Difference Calculation showing decreases in vegetation, which would be consistent with fragmentation shown by the landscape indices.

When the Leon Watershed was divided into urban and rural zones this showed an increase in fragmentation level between 1987 and 1999, especially in the urban zone. The fragmentation of dense vegetation in urban zone may indeed be due to increased development, which would result in smaller patches. Urban pollution may also be a biotic stress on vegetated areas further decreasing production. So, there may be both the physical removal of dense vegetation due to development of roadways as well as the physiological stress on vegetation in an urban environment. In 1998, there was also a large flood that affected the waterways in eastern Bexar County, however the Leon Watershed also experienced moderate flooding that would have cleared away sections of riparian vegetation. The regional area has also experienced prolonged periods of drought, which would be a stressor on vegetation.

There are many suggestions to offer for improving this study, most of which would involve a thorough accuracy assessment on the dense vegetation class. For the temporal aspect of the project, the use of three or more images would be better for analyzing trends through time. Also, using images with greater temporal difference between each other would be more appropriate to see a significant difference. A time span of twenty years would be more reasonable; however there may be a sacrifice of good quality data. In terms of land use analysis with riparian vegetation, dividing rural and urban zones into more meaning land use divisions to analyze dense vegetation may provide more insight.

Table 31. Advantages and Disadvantages of Methods. The methods include Image Difference Calculation, Percent Area, and Landscape Indices.

	Advantage	Disadvantage
Image Difference Calculation	1. The temporal measure may be more accurate, since the 1987 & 1999 images calculated together. 2. There is one resultant image to assess instead of two, which is less time consuming. 3. May be a way to measure rate of vegetation change between two time periods.	1. The highlighted differences function may be too sensitive or not sensitive enough. 2. Provides no information on patterns. 3. Highlighted difference percent may be too subjective, depending on the research needs 4. May be difficult to comprehend meaning of the results in terms of pixel change and the difference percent.
Percent Area/Supervised Classification	1. Provides most basic assessment of vegetative cover. 2. Most easy to comprehend results	1. Provides no information on patch pattern. 2. Provides no information on average size per patch. 3. There are two images being compared, which is more time consuming.
Landscape Indices/Supervised Classification	1. Provides information on mean area of patches 2. Provides information on patch shape for a landscape.	1. Must use a number of indices to reach a conclusion. 2. May be the most difficult to comprehend, since multiple indices must be well understood. 3. There are two images being compared, which is more time-consuming.

WATER QUANTITY ISSUES—WATER RE-USE

Statistical Analysis of Scenarios

Box plots of the six scenarios depicting the treatment type (quantity of flow augmentation) as the predictor variable (X) and daily mean DO (mg/l) as the response variable (Y) are shown in Figures 89, 90 and 91 below.

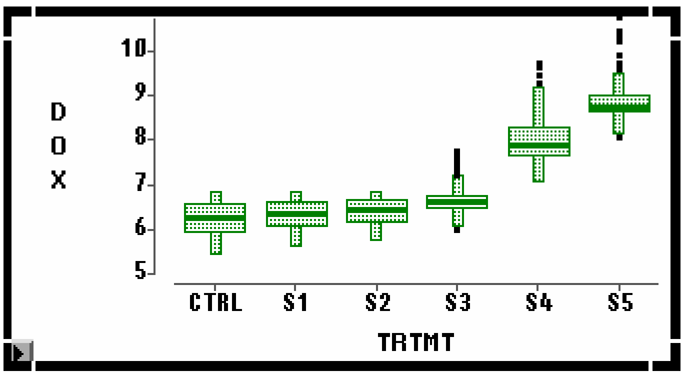


Figure 89. Box Plot for 1993 Scenario Analysis

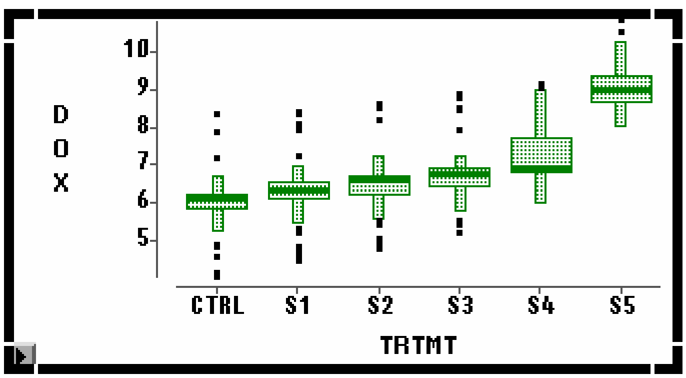


Figure 90. Box Plot for 1994 Scenario Analysis

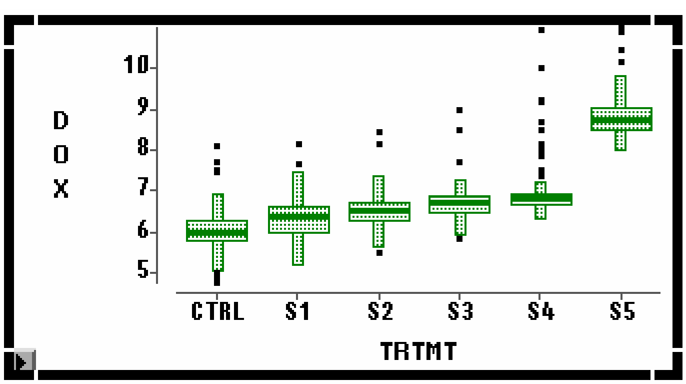


Figure 91. Box Plot for 1995 Scenario Analysis

It can be incurred from the box plots that the data from the sample populations is not normally distributed. Most of the sample populations are skewed and some of them are heavily tailed. A visual comparison of the box plots shows that there is a considerable overlapping between the first four populations (CTRL, S1, S2 and S3). However, sample population S5 shows a significantly different mean than the rest. This suggests that the null hypothesis (H_0 = all means are equal) is not true. A statistical basis for arriving at this conclusion is the one-way analysis of variance (one-way ANOVA) test. However, the one-way ANOVA test assumes that the data is normally distributed. Since the sample populations are not normally distributed, an alternative test called the Kruskal-Wallis Test was used. The Kruskal-Wallis Test is good for data that is highly skewed. Moreover it is a non-parametric test, i.e. it makes no assumption about population distribution. It tests the following two hypotheses:

H_0 (Null hypothesis) = All k populations have the same distribution

H_1 = Data from some populations tend to be larger than data from others.

From the SAS output of Kruskal-Wallis test, it was observed that with a 95% confidence level ($\alpha = 0.05$), the p-value (i.e. probability > Chi-square) was less than α for all three periods. Hence the null hypothesis was rejected and it was concluded that some means differ. To find which means differ, the Student-Newman-Keuls Test and the Tukey's Studentized Range Test were performed in SAS. A confidence level of 95% (α = 0.05) was set for the tests. Table 32 gives a summary of the statistical analysis.

Table 32. Summary of Statistical Analysis of Augmentation Scenarios

Scenario	Description	Rank	1993		1994		1995	
			SNK Grouping	Tukey Grouping	SNK Grouping	Tukey Grouping	SNK Grouping	Tukey Grouping
CONTROL	"As is"	6	E E	D D	F	E	F	E
S1	0.25 times base-flow (1.25 cfs*)	5	E D D	D D	E	D D	E	D D
S2	0.5 times base-flow (2.5cfs)	4	D	D	D	D	D	D
S3	1 times base-flow (5 cfs)	3	C	C	C	C	C	C
S4	2 times base-flow (10 cfs)	2	B	B	B	B	B	B
S5	4 times base-flow (20 cfs)	1	A	A	A	A	A	A

Note:
1. Average base flow for the study period during 1993, 1994, and 1995 was taken as 5 cfs.
2. Scenarios are ranked in descending order of mean daily DO for the sample populations.
3. Means with the same letter are not significantly different.

Student-Newman-Keuls Test

From the output the following observations were made:

1. The daily mean DO concentration for the sample population increases with increasing flows; i.e. S5 > S4 > S3 > S2 > S1 > CTRL.
2. 1993 scenario analysis showed no significant difference in mean DO for the Control, S1 and S2 scenarios. A significance difference in means is observed from scenario S3 onwards.
3. 1994 scenario analysis showed a significant difference in mean DO for all scenarios.
4. 1995 scenario analysis showed a significant difference in mean DO for all scenarios.

Tukey's Studentized Range Test

1. Daily mean DO concentration for the sample population increases with increasing flows; i.e. S5 > S4 > S3 > S2 > S1 > CTRL.
2. 1993 scenario analysis showed no significant difference in mean DO for the Control, S1 and S2 scenarios. A significance difference in means is observed from scenario S3 onwards.
3. 1994 scenario analysis showed no significant difference in mean DO for scenarios S1 and S2. All other scenario means were significantly different from each other.
4. 1995 scenario analysis showed no significant difference in mean DO for scenarios S1 and S2.
   All other scenario means were significantly different from each other.

Another important observation that was made from the box plots is that for 1994, scenario S1 and S2 as well as the control scenario had some data point that fell below the DO criteria of 5 mg/l. Similarly for 1995, some data point in the control scenario (CTRL) fell below 5mg/l where as some data point in scenario S1 were close to 5mg/l.

Interpretation of Results: Effects of Low-flow Augmentation

From the statistical analysis, the following conclusions can be made:

1. Increasing the flow in streams increases the mean daily DO concentration during low flow periods for the years 1993, 1994 and 1995
2. Scenarios S1 and S2 are not significantly different from each other and both show an increase in mean daily DO concentration as compared to the control scenario. However since some data points in S1 and S2 fall below DO criteria of 5 mg/l during at least one low-flow period (1994), they are not acceptable.
3. Scenario S3 shows a significantly different mean than scenarios S2 and S3 and also none of the data points in S3 fall below 5 mg/l. The same is true for S4 and S5. Hence it can be concluded that one can see a significant change (increase) in mean daily DO concentration from scenario S3 onwards.
4. Since the significance level of the tests was set to $\alpha = 0.05$, one can be 95% confident of the overall multiple comparison of means. This means that the confidence level for comparison of any two means in the ANOVA data set is greater than 95%. Hence, one can be more than 95% confident in saying that scenario S3 has a significantly greater

mean DO than scenarios CTRL, S1 and S2 compared one at a time. The same holds true for comparing S4 and S5 with CTRL, S1, S2 and S3 respectively, one at a time.

One should note from the above discussion that scenario S3 defines a "threshold response", meaning a significant change in the system is observed from scenario S3 onwards (with a certain confidence level). A watershed level modeling of DO is a highly complex process fraught with uncertainty in data, parameter estimation (because of lack of knowledge of various processes) and model performance. Hence, in order to be more certain of observing a positive change in system response, scenarios beyond the threshold response scenario should be considered strongly. One can see that the system behaves linearly for scenarios S1, S2 and S3 and starts showing non-linear response from scenario S4 onwards. Scenario S5 shows a significantly different response for all three years whereas scenario S4 shows a significantly different response for the year 1994.

Feasibility of Scenario Implementation

The total capacity of Leon Creek Wastewater Treatment Plant is about 32mgd[7]. About 26mgd of this treated wastewater is earmarked for recycling purposes. The rest will be split up for downstream release and for delivery to Mitchell Lake. The recycling pipes are 42 inches in diameter and designed to deliver 26mgd of treated wastewater to various customers all year round. This translates to a flow of about 40.3 cfs. Hence, the flow augmentation scenarios of 5 cfs (S3), 10 cfs (S4) and 20 cfs (S5) for the three-month period can be implemented without any additional pumping or infrastructure costs to the City. As of now, no quantity of recycled water has been earmarked for in stream releases from the Leon Creek Water Recycling Center. Whether the City can allocate recycled water for flow augmentation depends on its commitments to other customers and is an issue that is beyond the scope of this study.

Conclusions and Recommendations

The hypothesis tested in this study was that increasing dry period base flow in streams by discharge from a wastewater treatment plant (with pre-defined water quality parameters) enhances water quality in terms of DO. This has been demonstrated to be the case. An increase in daily mean DO was observed for all treatment scenarios tested. With 95% confidence level, a significant change was observed with scenario S3 (one times base flow), i.e. doubling the base flow during low-flow periods. This constitutes a risk-based design approach to remedy DO problems by low-flow augmentation. In other words, we can say with 95% confidence that a minimum flow augmentation of one times base flow can remedy DO problems during dry, low-flow periods. This amounts to a discharge of 5 cfs during each three-month period. The various flow augmentation alternatives are well within the capacity of the plant.

Scenario S3 defines the "threshold response" of a complex system fraught with uncertainty. Hence, to increase the chances of seeing a significant improvement in the system, scenarios S4 or S5 are highly recommended. As is evident, these alternatives are also within the capacity of the plant.

The basis of the above hypothesis is that the treatment plant discharge has a pre-defined water quality. The above hypothesis might not hold true for any wastewater treatment plant discharge. In fact, in all probability, the hypothesis will fail if there is increasing organic and nutrient burden (higher BOD, nitrates and phosphates) from the discharge. Studying the diurnal swings of DO cycle can assess the effects of such burden.

From a design consideration standpoint, carrying wastewater in pipelines to the point of flow-augmentation will completely strip it of its DO content. Therefore a passive entrainment aeration system is highly recommended at the point of discharge so that plenty of oxygen is available to aquatic organisms before the water enters the stream. This can be accomplished using gravity powered entrainment devices such as cascade-reaeration. Potential odor problems must be given consideration before deciding on a reaeartion process. However, given the quality of the water (Type I, the highest grade effluent), odor problems are not likely to occur.

Presently, the model is loosely calibrated for hydrology as well as DO. Besides, A very simplistic built-up and wash-off process was simulated for BOD and nutrient loading from land surfaces. This might not represent true conditions. As such, for now the model is good for predicting trends and cannot accurately simulate diurnal swings. Lack of historic data for DO is also a factor that limited model calibration. A detailed knowledge of nitrogen and phosphorus cycle on land surfaces, and in stream processes such as reaeration, algal respiration rate, sediment oxygen demand and algal growth among others is required for better model performance. Availability of representative precipitation data is also very crucial since the model is driven by precipitation.

The project demonstrated that the Non Point Source Model (NPSM), is capable of simulating watershed processes as well as in-stream processes and is good for a screening level analysis of scenarios and trends. However the model is extensively parameterized and requires a big learning curve for detailed studies. Thus comprehensive data requirements, overparametrization, lack of detailed knowledge and time constraints have limited the model performance. Moreover, the DO simulation process using NPSM has not been documented in detail in literature or elsewhere.

As stated earlier, a screening level analysis was performed to evaluate scenarios and observe trends. It is recommended that a more detailed analysis be carried out using a steady-state in stream water quality model such as QUAL2E. DO conditions can then be studied in detailed segment by segment there by giving a longitudinal resolution to water quality that is not possible using NPSM. This way, specific lengths of the creek that have low-flow DO problems can be identified and scenarios evaluated.

Finally, there is uncertainty involved in any modeling process. This uncertainty gets translated in model predictions. Therefore, it is critical for decision-makers to be cognizant of this uncertainty before making decisions based on model performance.

INTEGRATED FINDINGS, RECOMMENDATIONS & IMPLICATIONS

WATERSHED MODELING

This project integrated three distinct watershed modeling activities to develop watershed restoration strategies: 1) BASINS—HSPF Model; 2) Ecophys.Fish Model; and 3) STELLA Watershed Model. In addition, a simulation of citizen behavior was developed.

BASINS—HSPF Model

This model was used to develop probable outcomes based on scenarios provided by the stakeholders. We used the U.S. EPA Better Assessment Science Integrating Point and Nonpoint Sources (BASINS, version 3-Beta) environmental modeling software for modeling hydrologic, physical, and chemical processes in the Salado and Leon Creek watersheds. BASINS integrates key data and analytical components into Arc View-Geographic Information System (GIS) to provide the user with a fully comprehensive, state of the art watershed management tool for developing Total Maximum Daily Loads (TMDLs) (Battin et al., 1999).

The BASINS suite was developed by the U.S. EPA's Office of Water to support environmental and ecological studies in a watershed context (USEPA, 2001). BASINS works with a GIS framework and comprises various components including (1) national databases (2) assessment tools (3) a watershed delineation tool (4) classification utilities (5) characterization reports (6) an in-stream water model, QUAL2E (7) watershed loading and transport models, HSPF and Soil and Water Assessment Tool, (SWAT); and (8) a simplified GIS based model, PLOAD that estimates annual average nonpoint source pollutant loads. The relationships of the subprograms employed in the current study are shown in Figure 92. HSPF is continuous hydrological modeling software that can be used to simulate a comprehensive range of hydrologic and water quality processes. HSPF has a modular structure and the watershed is divided into three groups, pervious land, impervious land and channels or reaches. The modules PERLND, IMPLND and RCHRES in HSPF simulate the processes that occur in a pervious land segment, an impervious land segment and a single reach respectively. However, spatial distribution of different land segments within a particular watershed is not considered in HSPF.

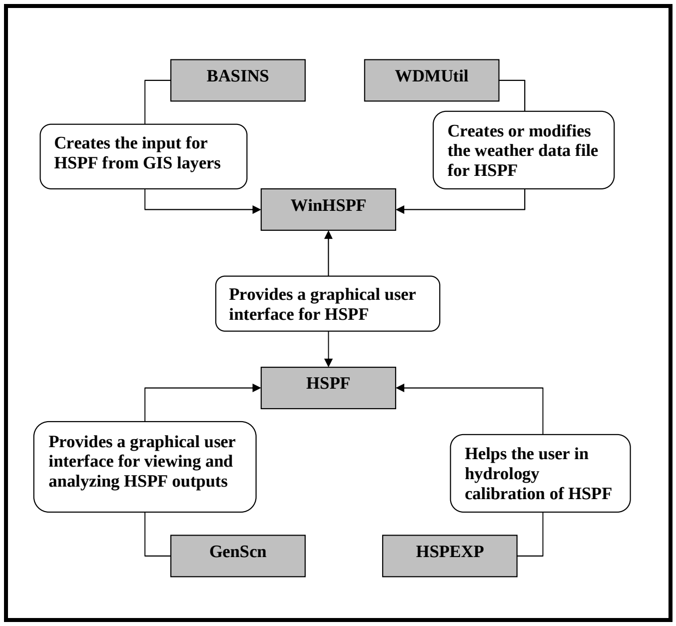
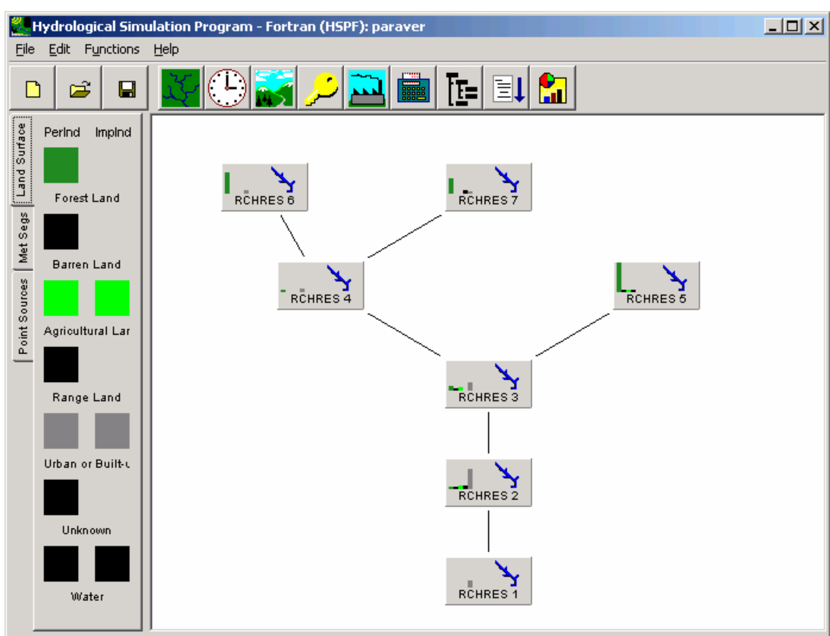


Figure 92. Different Computer Programs Employed in the Project



We used WinHSPF to develop scenarios based on the input provided by the stakeholders. WinHSPF (Windows interface to HSPF) provides an interactive graphical user interface (GUI) to HSPF. WinHSPF was created for BASINS 3.0 system, replacing the earlier program called Non Point Source Model (NPSM) used in combination with BASINS 1.0 and BASINS 2.0. WinHSPF helps the user to create a new HSPF input sequence or to modify an existing HSPF input sequence. WinHSPF also makes it easier to modify the input sequence for HSPEXP, an expert system developed by United States Geological Survey (USGS) for assisting users in calibrating HSPF. Figure 92 shows an HSPF project opened with the WinHSPF interface.

The model is a daily time-step continuous, lumped parameter model. WIN-HSPF resolution is based on the resolution of stream calibration data – the resolution we use was the sub-watershed, or approximately 500 ha, for pour points, and 1 ha for land use delineation. The model was implemented at the watershed scale. We used daily time steps for input for our scenarios, with monthly average water quality outputs. The outputs for BASINS-HSPF were flow, nutrient loading, and dissolved oxygen. BASINS-HSPF is an integrated hydrologic model.

The model was calibrated for hydrology using USGS gauging stations and historic data (Gholkar et al, 2002; Matlock & Gholkar, 2002). Very little data were available for calibration and validation for nutrients and DO. We collected field data for DO and nutrients over a 6-month period for calibration, and validated with field data we collected over a 12-month period following calibration. We conducted a sensitivity analysis to determine the variables of most concern for calibration and validation for fecal coliforms (Paul et al, 2002).

We used Monte Carlo simulation to characterize and quantify the uncertainties associated with critical input parameters. These analyses were informed by stakeholder deliberation and were iterative. We quantified the probability of specific outcomes of scenarios identified by stakeholders (Gohlkar et al, 2002; Matlock & Gohlkar, 2002).

Several obstacles were overcome. The model we used was data-intensive, and data rarely exist at the temporal and spatial scale necessary to implement models of this complexity. Hydrology is the most important parameter to calibrate in a watershed model. USGS gauging stations are critical for any hydrologic characterization, yet several had been decommissioned in this system. We had to simulate weather data across the watersheds due to the paucity of rainfall data, even in an urbanized area. Though water chemistry was available at higher resolution than most systems, the data were at inadequate temporal or spatial scale to inform calibration and validation.

The model results illustrated the critical impact of reduced base flow and loss of riparian zone on stream water quality, especially mean annual temperature and daily mean DO.

Ecophys.Fish Model

We developed a Stella-based hydrologic model linking watershed policy and management decisions to watershed hydrologic response.

This model had been developed for application to marine-fish-habitat assessment, with funding support from Sea Grant. Fortunately, we were able to adapt the model for bluegill and apply it to facilitate analysis and communication of the experimental results from this project. The model is a deterministic simulation model, developed in STELLA. Resolution of the model is individual spatial points through time. The temporal scale of the model is one-hour time-step. Model variables and sources of data to parameterize the model are time-series of environmental data (water temperature, dissolved oxygen concentration, conductivity, pH) and terminal fish metabolic-capacity assays via respirometry in lab. "Ground truth" from growth and mortality rates of caged fish in the streams. Processes and endpoints predicted by the model were fish growth and mortality rates.

Ecophys.Fish easily could be (but was not) linked with hydrologic models, so that environmental outputs from hydrologic models are inputs to Ecophys.Fish.

We compared model simulations with responses of fish caged in study streams. Model accounted for about 80% of variation in median observed growth rates. The main obstacle was that, when we started this project, the bluegill version of Ecophys.Fish did not exist! Our experience in this project taught us how to use respirometric bioassay to estimate a key parameter of Ecophys.Fish empirically. We also learned a valuable lesson about the utility of on-the-spot simulation as a tool for communication both with technical and non-technical stakeholders.

STELLA Watershed Model

We adapted this Stella-based model of fish physiological performance, to link watershed processes to ecological outcomes.

This model was newly-developed for this project to be used as a tool for facilitating group learning among stakeholders and to enable them to share their learning with other interested parties. This model was intended to illustrate how changes in watershed condition impact hydrologic response. The model featured a streamlined user interface allowing stakeholders to choose between three storms, two land uses, three channel conditions, and two upstream stormwater retention strategies.

The model assumes homogeneous conditions within subwatersheds. Model resolution of subwatersheds ranged in size from 33 to 6600 ha.

The model was developed for use on the Salado Creek and Leon Creek watersheds, which totaled 59,800 and 47,400 ha. Each watershed model used 18 subwatersheds.

The model used time steps of one hour for a total simulation period of two days.

The SCS Curve Number method was used to estimate runoff from each sub-watershed. This empirical procedure predicts runoff volume based on precipitation depth, land use, and soil type. Curve number values were selected from literature based on land use and soil type. Stakeholders were given three storms (2-year, 24-hour; 10-year, 24-hour; and a specific event from 1998) that they could select for modeling. Precipitation estimates for the 2- and 10-year storms were selected from literature and the 1998 storm depth was gathered from local precipitation data. Manning's equation was used to estimate flow routing through the channel system. As with the curve number values, Manning's roughness values were selected from literature. For purposes of model simplicity, channels were assumed to be rectangular.

Running the model yielded a runoff hydrograph for the chosen watershed conditions. By comparing hydrographs from different management scenarios, stakeholders were able to view the relationship between select strategies and runoff generation. The councils have used this tool to illustrate their watershed management recommendations to city and agency personnel.

This model was developed for use as an educational tool. It is not linked to other hydrologic or water quality models. Model results are intended for comparison of the impacts of various watershed management strategies. Hence, the model is severely limited for prediction use. Because of its intended use, there are no plans to validate model performance by comparison to observed data. While this model may be beneficial for watershed managers as they interact with stakeholder groups, the model is not appropriate for hydrologic prediction or for modeling specific watershed management plans.

Because of its simplicity, few obstacles were encountered in development or use of the model. However, the model's simplicity does preclude its use as a predictive tool. Although the model was severely over-simplified, it was still a useful learning tool.

The primary goal for this process was learning. Participation in mediated modeling enabled stakeholders to develop a systemic understanding of the Leon and Salado Creek watersheds. Grant et al. (1997) have identified four fundamental phases in the process of developing and using a quantitative systems simulation model: (1) conceptual model formulation; (2) quantitative model specification; (3) model evaluation and; (4) model use. We used Collaborative Learning (Daniels & Walker, 2001) to guide the stakeholder groups through these phases, spending the most time in the first phase.

The goal of the first phase is to develop a conceptual, or qualitative, model of the system of interest. Based on objectives of the project, all participants worked together to decide which components in the watershed could be included in our system of interest and how they should be related to one another. The goal of the second phase is to develop a quantitative model of the system of interest. This involved translating our conceptual model, which was represented diagrammatically and linguistically, into a series of mathematical equations that collectively formed the quantitative model. This translation, or quantification, was based on consideration of various types of information about the real system including theoretical concepts, empirical data, and expert opinion. The modelers performed this translation during the time between stakeholder meetings, then brought the results to the stakeholders for evaluation (phase 3). The participants then evaluated the usefulness of the model in meeting their objectives. The final phase involved designing and simulating the experiments with the model that we would have liked to conduct in the real system. During this phase, participants tried out different combinations of land use, channel condition, and rainfall to answer questions about how best to manage peak flows through the watersheds.

One of the main obstacles in moving toward sustainable, multiple use of resources is our inability to synthesize knowledge and perspectives from many distinct disciplines within a single problem-solving philosophy. Mediated modeling (Costanza & Ruth 1998, Van den Belt 2000, Vennix, 1996) provides a basis for sustainable, multiple resource use by facilitating multidisciplinary planning and by creating an effective communication interface between scientists, citizens, and policy makers, thus promoting identification and communication of policy options to decision makers. The four phases of systems analysis are highly interconnected (Grant et al., 1997), and we cycled through the phases repeatedly. The simulation model evolved into the communication interface between scientists and decision makers as team members developed a sense of joint ownership of the model, thus making the modeling process more important than the model itself.

This approach could be used in other planning efforts to aid the visualization of alternative schemes for watershed restoration and ecological processes at the watershed and floodplain scales.

Simulation of Citizen Behavior

To fulfill our proposal to integrate human behavioral models with watershed models, we pursued the following steps:

1. Estimated a set of models of watershed-impacting behavior of Bexar County Residents,
2. Produced simulations for the residents in areas adjacent to Salado and Leon Creeks based on what we find to be the most influential predictor variables from Step 1.

We included two batteries of questions in our survey of Bexar County Residents which asked them about their own watershed-impacting behavior. These questions were developed by members of our team who had previously worked with models of watershed quality. They anticipated the household behaviors that were most likely to have an impact on watershed quality and asked respondents one set of questions about their actual behavior and another set about how likely they would be to change their behavior if they thought that making this change would improve the quality of San Antonio's watersheds. This second set of questions was the most promising for developing an integrated set of models. From these questions, the two items that were most promising for directly affecting the quality of water in the local watersheds were questions 39 and 43. The answers to these questions from residents living near each watershed appear below in Table 33.

Table 33. Willingness to Change—Distribution of Responses for Watershed Area Residents

	Very Unlikely	Unlikely	Likely	Very Likely
…change the way you take care of your lawn?				
Salado Creek Residents	15.8%	13%	31.1%	40.1%
Leon Creek Residents	14.2%	10.3%	40.7%	34.8%
…change the way you take care of household wastes?				
Salado Creek Resident	6.6%	4.9%	29.5%	59%
Leon Creek Residents	7%	5.6%	41.1%	46.3%

From these initial results, we get our starting values for simulations of changes in human behavior relevant to the watersheds. We can say that if people held their current opinions and a public information campaign were launched to convince residents that they could improve the quality of watersheds by changing the way that they take care of their lawns 34.8% of the Leon Creek area residents would change their behavior and 40.1% of the Salado Creek area residents would change their behavior. If a public information campaign about the handling of household wastes were launched, 46.3% if the Leon Creek area residents would change their behavior and 59% of the Salado Creek area residents would change their behavior.

The next question to ask is what would happen if the factors that most affect watershed-area resident's willingness to change their behavior were to change? To examine a range of hypothetical changes, we first need to determine what are the factors that most influence respondent's willingness to change. The theoretical model that we developed to explain people's willingness to change is:

Willingness to Change = f(environmental attitudes + local efficacy + assessment of current conditions + length of time in neighborhood + income).

In other words, we expect that people's willingness to change their watershed-relevant behavior will be determined by their combination of the characteristics in the formula above. We expect that people with more environmentalist views will be more willing to change their behavior. In addition to general views about the environment, we expect people's views of their local community will also affect their willingness to change. To the extent that people feel locally politically empowered (local efficacy), they should be more willing to change. The worse people assess the conditions of their local watershed to be, the more willing they will be to change their behavior. We also expect that people who have resided in a neighborhood near the watershed for a longer period of time will be more likely to be willing to change their behavior. We want to estimate this affect while controlling for the effects of age. Thus, although we include age as an independent variable, we do not have a theoretical expectation for this variable. Finally, household income should be expected to be a powerful predictor of any household's behavior. With respect to these potential changes in household behavior, we have competing sets of expectations for the impact of household income. On the one hand, we might expect wealthier households to be more willing to change their behavior because they can more easily afford any economic impact that might result from such change. On the other hand, it may be the case that wealthier people are less willing to make changes. The results from our four model of willingness to change based on these theoretical propositions are presented in Table 34 below.

Table 34. Models of Willingness to Change

	Salado Creek Residents		**Leon Creek Residents**	
	Lawn Care	Household Wastes	Lawn Care	Household Wastes
Environmental Attitudes Index	.20	.33	N.S.	.23
Local Efficacy	.25	N.S.	N.S.	N.S.
Assessment of Local Conditions	N.S.	N.S.	N.S.	N.S.
Length of Time in Neighborhood	N.S.	N.S.	N.S.	N.S.
Age	-.22	-.24	N.S.	N.S.
Income	N.S.	.34	N.S.	N.S.
Model R-Squared	.28	.31	.10	.17

Notes: Cell values are the standardized betas for our OLS regression estimates. Only statistically significant effects are shown. Full models results are available from the authors upon request. "N.S." stands for "not significant" meaning that we failed to reject the null hypothesis that the relevant regression coefficient equaled zero using a two-tailed t-test with a p-value cut-off of .05.

Environmental Attitudes Index is the first factor score from a principal components analysis of respondent answers to 12 general questions about their environmental attitudes. Higher values indicate more environmentalist views. *Local Efficacy* is the respondent's answer to a question about whether or not people like them have an impact on local politics. Higher values indicate a greater feeling of being locally politically empowered. *Assessments of Local Conditions* is the first factor score from a principal components analysis of respondent answers to 4 questions about the quality of water in their local watershed. *Length of Time in Neighborhood* is the number of months that the respondent reported having lived in their neighborhood.

Age is age in years. *Income* is a standard 13 category measure of household income.

From the results in Table 34 it is apparent that the results vary substantially across dependent variable and watershed. For Leon Creek residents, it is quite hard to explain their willingness to change their behavior. This is due, in part, to their homogeneity on many of the predictor variables included in our models. Not surprisingly, environmental attitudes were a powerful predictor of the willingness of people to change their behavior in order to improve watershed quality. In the model of Leon Creek residents' willingness to change their lawn care, this effect just missed achieving statistical significance at conventionally accepted levels. The remaining results are somewhat varied.

For Salado Creek residents, willingness to change the way that they care for their lawns is greatest among those people who have environmentalist views and feel locally politically empowered. Holding these factors constant, older people are substantially less-willing to change their lawn care. Older people are also less willing to change their handling of household wastes. The effect for local efficacy is not present in this model, but there is a statistically significant effect for wealth, which indicates that wealthier Salado Creek residents are willing to change their current practices.

Given the lack of statistically significant results for Leon Creek residents, we concentrated our simulation efforts on Salado Creek. We know that with current resident characteristics, 40.1% of Salado residents say they would change their lawn care practices and 59% of Salado residents say they would change their handling of household wastes if they thought that these behaviors would improve the quality of their waterways. In order to simulate what would happen if the influential characteristics of Salado Creek residents were to change, we created a series of scenarios for change in resident's characteristics and then looked at model predictions for each scenario. The results of these scenarios are reported in Tables 35 and 36 below.

Table 35. Effects of Simulated Changes in Key Independent Variables on Percent of Salado Creek Residents Who Would Change their Lawn Care

	2 standard deviation decrease	1 standard deviation decrease	No change	1 standard deviation increase	2 standard deviation increase
Environmental Attitudes Index	36.7%	39.5%	40.1%	62.5%	63.5%
Local Efficacy	8.8%	27.5%	40.1%	63.7%	75.8%
Age	67%	57.1%	40.1%	31.9%	16.5%

Note: Cell contents contain the percentage of respondents who would be predicted to be "very likely" to change their behavior if the change listed at the top of each column took place.

Table 36. Effects of Simulated Changes in Key Independent Variables on Percent of Salado Creek Residents Who Would Change their Handling of Household Waste

	2 standard deviation decrease	1 standard deviation decrease	No change	1 standard deviation increase	2 standard deviation increase
Environmental Attitudes Index	8.8%	30.8%	59%	82.4%	91.2%
Income	8.8%	30.8%	59%	82.4%	91.2%
Age	84.7%	77%	59%	41.8%	23.1%

Notes: Cell contents contain the percentage of respondents who would be predicted to be "very likely" to change their behavior if the change listed at the top of each column took place.

LINKAGES AMONG THE ECOLOGICAL, PHYSICAL AND SOCIAL SCIENCE RESEARCH COMPONENTS ON THE GRANT

Our approach was to work with stakeholder groups from two watersheds—Leon and Salado Creeks—to develop a watershed model linked with an in-stream model to evaluate and optimize ecosystem management strategies. Our objective was to use this risk-based model to initiate discussions between stakeholders and scientists in an iterative process that would result in an informed and stakeholder-derived action plan for watershed rehabilitation and further refinements of the model.

INTEGRATION—TECHNICAL PERSPECTIVE

Collaborative Learning

We chose Collaborative Learning as the vehicle for integrating information from multiple disciplines because its emphasis on systems learning provided our stakeholders the opportunity to establish a common understanding of systems thinking, the modeling process, and the particular watershed, prior to developing a quantifiable model.

The CL meetings were designed to allow stakeholders to move through four phases of application. Each new phase incorporated activities and communication processes that had been developed during the previous phase. First, participants were informed about the CL process and received training in collaborative discussion and debate. Second, a common knowledge base on the major issues affecting water quality in the watershed was created for participants via formal presentations by experts, panel question/answer sessions, and informal small group discussions. Third, active learning exercises were used to help participants think systemically about the watershed ecosystem and enable them to identify key issues, concerns, and interrelationships between variables affecting water quality. During this phase, participants began to generate specific suggestions for improvement of the current situation (i.e., existing water quality in watershed) and then share and refine their ideas with other participants through a structured small group discussion process. Fourth, through collaborative debate with other stakeholders, a final set of improvement suggestions was organized, discussed, and refined.

The method of Collaborative Learning provided an effective framework for integrating the results from the three quantitative models in this section. A common knowledge base on basic hydrological principles, flood control strategies, and specific quantitative models such as BASINS-HSPF and ECOPHYS.FISH was achieved through formal presentations by Drs. Kenimer, Matlock, and Neill at CL workshop meetings beginning in April 2000. The iterative feedback cycle was initiated during this phase of the CL process by allowing stakeholders to question the ecological science team following their presentations and offer specific feedback about the results of their simulations. Discussion between members of the stakeholder groups and the ecological scientists ensued and alternative scenarios were suggested for simulation by these quantitative models. Drs. Matlock and Neill actually ran these suggested simulations and reported the results back to the stakeholder groups either at the same meeting, or at a subsequent stakeholder meeting. In October 2000, Dr. Kenimer, with the assistance of Dr. William Grant of the Wildlife and Fisheries Science Dept. at Texas A&M University, developed the STELLA Watershed model based on stakeholder feedback at previous CL meetings and the entire workshop agenda that month was devoted to allowing stakeholders to test various flood control options using various assumptions about storm severity and duration (see description under STELLA Watershed model). The "hands-on" experience gained by the stakeholder groups with this modeling activity had a significant impact on the thinking of participants, particularly about the issue of flood control in Salado and Leon Creeks. This impact was illustrated dramatically at the November 2000 meeting of the Leon Creek Restoration Council, in which the spokesperson presented the group's action plan to the Mayor of San Antonio, Howard Peak, and explicitly used results from their STELLA Watershed simulation runs to support their recommendations for watershed rehabilitation.

INTEGRATION—PHILOSOPHICAL PERSPECTIVE

The CL process intervention (Daniels & Walker, 2001) is grounded in theoretical work on soft systems methodology (Checkland & Scholes, 1990; Wilson & Morren, 1990) and alternative dispute resolution (Fisher, Ury, & Patton, 1991; Gray, 1989). The concept of "soft systems" represents an extension of theoretical work on systems analysis (Senge, 1990) and experiential learning (Kolb, 1986).

The basic assumption in soft systems methodology is that management of complex problem situations demands a different approach than the typical "hard" systems method used in engineering. Such hard systems methods focus on outcomes instead of processes, and consequently do not attach importance to learning. Management of an ecosystem is a situation where problems are often characterized by a high degree of uncertainty and equivocality. The soft systems approach focuses on "situation improvements" that can result from active learning and debate.

The CL process also promotes development and identification of stakeholder concerns and underlying interests. Recent theory and research on negotiation and mediation has adopted an interests-based approach. One viewpoint (e.g., Bazerman & Neale, 1992; Fisher et al., 1991) maintains that traditional positional approaches to negotiation may be inefficient because positions taken by parties in conflict are often extreme and obscure the underlying interests (i.e., needs, concerns, values) that parties seek to advance through negotiation. The structured set of CL activities is designed to move participants away from positional strategies and toward the identification of mutual interests and joint gains from collaboration. Moreover, the presence of outside facilitators during CL sessions permits use of effective principles from mediation theory and research (Gray, 1989).

To improve decision-making for sustainable development, new tools to facilitate common goal development and to test alternative scenarios are needed. These tools have to be able to communicate the complexity and uncertainties of the decisions and allow for broad stakeholder participation, while integrating different aspects of the situation. Integrated assessments use tools and inputs from multi-disciplinary, multi-scale and multi-social backgrounds to support decision making processes (van Asselt, 2000). Rather than the equilibrium-based view of decision and policy making, Collaborative Learning emphasizes a process oriented approach from a team learning perspective (Daniels & Walker, 2001).

This emphasis on process and learning from the method of Collaborative Learning dictated which components of our project to link together. The theme of systems thinking was emphasized throughout the entire 14-month CL process. It was relatively easy to decide which models to link together, once we established the learning objective of providing participants with a systemic view of their watershed. In short, we let basic ecological and physical processes drive the linkages that were most important, and used Collaborative Learning to help stakeholders visualize these interconnections in the context of their watershed.

PUBLICATIONS AND PRESENTATIONS

Theses

Cummins, K. (2000). The temporal mapping of riparian vegetation at Leon Creek in Bexar County, Texas from 1987-1999. MS Thesis, Texas A&M University.

Harris, H. (2000). Changes in ecosystem services and runoff due to land use change in the watersheds of San Antonio, Texas. MS Thesis, Texas A&M University.

Gholkar, T. (2000). Modeling the effects of low-flow augmentation by discharge from a wastewater treatment plant on dissolved oxygen concentration in Leon Creek, San Antonio, Texas. MS Thesis, Texas A&M University.

Kasprzak, K. (2001). Measuring and Modeling In Situ Sediment Oxygen Demand in the Arroyo Colorado River in South Texas. MS Thesis, Texas A&M University.

Rodriguez, A. (2001). Impact of Nutrient Loading from Point and Non-Point Sources on Water Quality and Lotic Ecosystem Health in the North-Bosque Watershed using a Bio-Indicator Response Approach. MS Thesis, Texas A&M University.

Murawski, M. (2001). Evaluation of Nutrient Diffusing Substrates and the Primary Importance of Light in Controlling Periphyton. MS Thesis, Texas A&M University.

Fontaine, L. P. (2001). Fish-performance ecoassay of urbanizing streams in the San Antonio river basin, Texas. Master of Science thesis, Texas A&M University, College Station, TX.

Gore, D.C. (2001). Adam Smith's rhetorical sympathy: A return of moral sentiments to public policy. Master of Science thesis, Texas A&M University, College Station, TX.

Journal Articles – Published

Brody, S. D., Highfield, W., & Alston, L. (2003). Does location matter? Measuring environmental perceptions of creeks in two San Antonio watersheds. *Journal of Environment and Behavior*. (See Appendix D.)

Kreuter, Urs P., Harris, H. G., Matlock, M. D., Lacey, R. E. (2001) Change in ecosystem service values in the San Antonio area, Texas. *Ecological Economics*. (See Appendix D.)

Journal Articles – In Process

Brody, S. D., Highfield, W., & Peck, B. M. (Manuscript submitted, December 2002). Exploring the mosaic of perception of water quality across watersheds in San Antonio, Texas. *Journal of Applied Geography*.

Grant, W. E., Peterson, T. R., & Peterson, M. J. (2002). A quantitative representation and application of Niklas Luhmann's social theory: Simulation of communication as it relates to ecological systems. *Ecological Modelling*.

Matlock, M., & Gholkar, T. (Manuscript submitted, 2003). A risk-based design of wastewater reuse for low flow augmentation to improve dissolved-oxygen concentrations in streams in San Antonio, Texas.

Matlock, M., & Murawski, M. (Manuscript submitted, 2003). The efficiency of nutrient delivery in Matlock periphytometers.

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ENDNOTES

[1] The Wilcoxon matched pairs signed-ranks test requires that all matched pairs with zero values for the difference score be dropped from the analysis, thus reducing the total N for the analysis. The resulting decrease in sample size has a negative effect on statistical power to detect significant differences caused by the treatment.

[2] The S statistic provided by the SAS program (PROC UNIVARIATE) is similar mathematically to the traditional T statistic obtained from the Wilcoxon matched pairs signed-ranks test (Siegel, 1956).

[3] We return to this apparent paradox later in this final section.

[4] N=59 because some participants listed more that one goal/reason for their participation.

[5] We did not find much evidence that council members participated for purely defensive reasons--that is, to prevent the Council from recommending actions that they would oppose. These findings are in contrast to the stated reasons for participation by stakeholders in other research studies. There are two possible reasons for the inconsistency in results. First, the questions that we asked our participants to elicit these reasons were different than those posed by other researchers. Second, the proportion of large landowners and/or resource users may have been smaller in our councils compared to other stakeholder groups.

[6]Some researchers would argue that the absence of a significant business/development interest group in the council meetings is a more serious problem for solving water quality problems in the watershed than the under-representation of a specific ethnic group. The assumption here is that changing the behavior of a specific target group is extremely difficult if that target group is not at the table during the council deliberations.

[*] cubic feet per second

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Appendix A

Salado Creek
Restoration Council (SCRC)
Meeting Agendas

♦SALADO CREEK RESTORATION COUNCIL♦

Workshop Agenda

Friday, November 12, 1999 • 3:00 pm – 7:00 pm
San Antonio River Authority Board Room • 100 E. Guenther Street

3:00	WELCOME	
	Introductions – Texas A&M University Team	Natasha Hudspeth
	Goals	Tarla Rai Peterson
	Introductions – SCRC Members	Susan Gilbertz
3:30	SYSTEMS ACTIVITY	Susan Gilbertz
4:00	BREAK	
4:10	WATERSHED AS A SYSTEM	Ann Kenimer
4:40	WATERSHED SYSTEM ACTIVITY	Ann Kenimer
5:10	BREAK	
5:20	QUESTIONNAIRE	Charles Samuelson
5:30	COLLABORATIVE LEARNING AS A SYSTEM	Charles Samuelson
5:40	CLOSING ACTIVITIES	Susan Gilbertz
	Announcements	
	Membership Recommendations	
6:10	SUPPER	
7:00	ADJOURN	

♦SALADO CREEK RESTORATION COUNCIL♦

Workshop Agenda

Saturday, December 11, 1999 • 9:00 am – 11:00 am
San Antonio River Authority Board Room • 100 E. Guenther Street

9:00	WELCOME & REVIEW	Tarla Rai Peterson
9:10	WATER QUALITY IN SALADO CREEK	Ann Kenimer
9:40	BREAK – WATER QUALITY EXERCISES	
10:10	WATERSHEDS AS COMPLEX SYSTEMS	Marty Matlock
10:40	RESEARCH STRATEGY TO ADDRESS QUESTIONS	Bill Neill
11:00	ADJOURN	

♦SALADO CREEK RESTORATION COUNCIL♦

Workshop Agenda

Saturday, January 22, 2000 • 10:00 am – 2:00 pm
San Antonio River Authority Board Room • 100 E. Guenther Street

10:00	WELCOME & REVIEW	Charlie Samuelson
10:10	WATER RECYCLING SYSTEM	Steve Clouse (SAWS) Jerry Berry (SAWS)
11:00	BREAK	
11:10	LISTENING	Tarla Peterson
11:30	COUNCIL DISCUSSION OF IDEAL WATERSHED	Susan Gilbertz
12:00	SMALL GROUP INSTRUCTIONS (short break after receiving instructions)	Susan Gilbertz
12:10	SMALL GROUP WORK SESSIONS	As Assigned
12:30	GROUP REPORTS	Tarla Peterson
1:00	LUNCH	
2:00	ADJOURN	

♦SALADO CREEK RESTORATION COUNCIL♦

Workshop Agenda

Friday, February 11, 2000 • 3:00 pm – 7:00 pm
San Antonio River Authority Board Room • 100 E. Guenther Street

3:00	WELCOME	Charlie Samuelson
3:10	SURVEY RESEARCH	Guy Whitten
3:55	BREAK	
4:10	MODELING PUBLIC OPINION AND BEHAVIOR	Guy Whitten
5:00	BREAK	
5:10	COUNCIL PLANNING AND SCHEDULING	Tarla Rai Peterson
5:20	DEFINING COMMUNITY AND COUNCIL RELATIONSHIPS	Tarla Rai Peterson
5:55	CLOSING	Tarla Rai Peterson
6:00	LIGHT SUPPER	

♦SALADO CREEK RESTORATION COUNCIL♦

Workshop Agenda

Saturday, March 25, 2000 • 9:00 am – 1:00 pm
San Antonio River Authority Board Room • 100 E. Guenther Street

9:00	WELCOME	Tarla Peterson (TAMU)
9:10	BASIC IDEAS OF HYDROLOGY AND FLOODING	Ann Kenimer (TAMU)
9:40	FLOOD CONTROL AND SARA	Steve Ramsey (SARA)
10:10	QUESTIONS AND ANSWERS	
10:40	BREAK	
11:00	INDIVIDUAL IDENTIFICATION ON CREEK	
11:30	LOCALE GROUPS GENERATE FLOOD CONTROL RECOMMENDATIONS	
11:50	REGROUP AS WATERSHED DISCUSSION GROUPS	
12:00	WATERSHED DISCUSSION GROUPS IDENTIFY KEY IMPLICATIONS OF RECOMMENDATIONS	
12:30	LUNCH SUBCOMMITTEE MEETS	
1:00	ADJOURN	

♦SALADO CREEK RESTORATION COUNCIL♦

Workshop Agenda

Friday, April 14, 2000 • 12:00 pm – 4:00 pm
San Antonio River Authority Board Room • 100 E. Guenther Street

12:00 LUNCH

12:30 WELCOME .... Tarla Rai Peterson

12:40 PLANNING PANEL – INTRODUCTIONS .... Tarla Rai Peterson
Mayor Howard Peak
Dixie Watkins
George Geis

1:10 QUESTION & ANSWER

1:40 AGENCY PANEL– INTRODUCTIONS .... Susan Gilbertz
Arthur Talley, Texas Natural Resource Conservation Commission
Mike Gonzales, San Antonio River Authority
Meg Conner, San Antonio Water System
Steve Uncapher, San Antonio Parks and Recreation Department
Judit Gowen, Texas Parks and Wildlife Department

2:30 BREAK

2:45 QUESTION & ANSWER

4:00 ADJOURN

♦Salado Creek Restoration Council♦

Workshop Agenda

Saturday, May 6, 2000 • 9:00 am – 1:00 pm
San Antonio River Authority Board Room • 100 E. Guenther Street

9:00	Welcome	David Gore
9:10	Texas A&M University	Marty Matlock
9:40	San Antonio River Authority	Mike Gonzales
10:10	Questions	
10:30	Break	
10:45	Developing a Systems Framework for Watershed Improvement Suggestions	
12:30	Lunch	
1:00	Adjourn	

♦Salado Creek Restoration Council♦

Workshop Agenda

Friday, June 9, 2000 • 12:00 pm – 4:00 pm
San Antonio River Authority Board Room • 100 E. Guenther Street

12:00 LUNCH AND ANNOUNCEMENTS
Committee Members Meet Over Lunch

12:30 WELCOME .... Charlie Samuelson, Texas A&M University

12:35 AQUATIC ECOLOGY .... Bill Neill, Texas A&M University

1:00 RIPARIAN WOODLAND SYSTEMS .... Mark Peterson, Texas Forest Service

1:30 TERRESTRIAL RIPARIAN ECOLOGY & WILDLIFE
.... Rufus Stephens, Texas Parks & Wildlife Depart.ment

2:00 QUESTION & ANSWER

2:15 BREAK

2:30 QUESTION AND ANSWER REGARDING SEWAGE SYSTEM REPAIRS AND MAINTENANCE
Cindy Kovacic: Engineering Director, San Antonio Water System
Dan Myers: Operations Director, San Antonio Water System

3:00 FLOW MAPS – UPDATED TO INCLUDE TODAY'S TOPICS

3:30 GROUP REPORTS (MAPS OF FLOWS)

4:00 ADJOURN

♦SALADO CREEK RESTORATION COUNCIL♦

Workshop Agenda

Saturday, July 15, 2000 • 9:00 am – 1:00 pm
San Antonio River Authority Board Room • 100 E. Guenther Street

9:00 WELCOME .... Tarla Peterson, Texas A&M University

9:05 LAND USE OPTIONS .... Paul Barwick & Paul Wagner, Nature Conservancy

9:35 ECOLOGICAL MEASURES IN STREAMS
.... Mike Gonzales, San Antonio Water Authority

10:05 QUESTION & ANSWER

10:15 BREAK

10:30 WATER QUALITY MODELING – WATERSHED APPROACH
.... Marty Matlock, Texas A&M University

11:00 MODELING EXERCISE AND WRAP-UP .... Marty Matlock, Texas A&M University

12:00 COMMITTEE MEETINGS

12:30 LUNCH AND ANNOUNCEMENTS

1:00 ADJOURN

♦SALADO CREEK RESTORATION COUNCIL♦

Workshop Agenda

Friday, September 8, 2000 • 12:00 pm – 4:00 pm
San Antonio River Authority Board Room • 100 E. Guenther Street

12:00 LUNCH

12:30 WELCOME AND ANNOUNCEMENTS .... Susan Gilbertz, Texas A&M University

12:45 COMMITTEE REPORTS .... Council Committees

1:00 DISCUSSION OF POSSIBLE FUTURES FOR COUNCIL .... Arthur Talley, TNRCC

1:15 WATER QUALITY MODELING – WATERSHED APPROACH .... Marty Matlock, Texas A&M University

2:00 QUESTIONS

2:15 BREAK

2:30 GROUP DISCUSSION OF IMPROVEMENTS & RECOMMENDATIONS

3:30 COMMITTEE MEETINGS

4:00 ADJOURN

♦SALADO CREEK RESTORATION COUNCIL♦

Workshop Agenda

Friday, October 6, 2000, 9:00 am–1:00 pm
San Antonio River Authority Board Room • 100 E. Guenther Street

9:00	WELCOME AND PREVIEW	Charlie Samuelson
9:15	MODELING THE SALADO CREEK WATERSHED SYSTEM	Bill Grant & Ann Kenimer
10:00	INTEGRATING AQUATIC LIFE INTO THE MODEL	Bill Neill
10:30	COMPLETE INDIVIDUAL INTEREST SHEET	Susan Gilbertz
10:40	BREAK	
10:55	DEBATE & MODEL TENTATIVE RECOMMENDATIONS	Council Members
11:45	SUMMARIZE MODELING RESULTS	Ann Kenimer
12:00	IDENTIFY TENTATIVE RECOMMENDATIONS	Tarla Peterson,
12:30	COMMITTEE REPORTS & LUNCH	
1:00	ADJOURN	

♦Salado Creek Restoration Council♦

Workshop Agenda

Saturday, November 4, 2000 • 9:00 a.m. – 1:00 p.m.
San Antonio River Authority Board Room • 100 E. Guenther Street

9:00	Welcome, Announcements, Review October Simulations	Tarla Rai Peterson, Texas A&M University
9:30	Nominating Committee Report	
10:00	Discussion	
10:30	Aquatic Life in Salado Creek	Bill Neill, Texas A&M University
11:00	Discussion	
11:15	Break	
11:30	Discussion & Planning -- Report for Mayor *(to be presented in December)*	
12:00	Lunch	
1:00	Adjourn	

♦SALADO CREEK RESTORATION COUNCIL♦

Workshop Agenda

Friday, December 1, 2000 3:00pm –7:00 pm
San Antonio River Authority Board Room • 100 E. Guenther Street

3:00	WELCOME AND PREVIEW	Susan Gilbertz
3:15	PRESENTATION OF COUNCIL RECOMMENDATIONS	Council Members
3:30	DISCUSSION OF RECOMMENDATIONS	Mayor Howard Peak
4:00	WATERSHED RESTORATION STRATEGIES	Nancy Beward, San Antonio Public Works
4:45	WATERSHED RESTORATION STRATEGIES	Karen Cummins, Texas A&M University
5:15	BREAK	
5:30	SURVEY/QUESTIONNAIRE	Charles Samuelson
6:00	NOMINATING COMMITTEE REPORT & DISCUSSION	Council Members
6:30	DINNER	
7:00	ADJOURN	

Leon Creek
Restoration Council (LCRC)
Meeting Agendas

♦ LEON CREEK RESTORATION COUNCIL ♦

Workshop Agenda

Saturday, November 13, 1999 • 10:00 am – 2:00 pm
San Antonio River Authority Board Room • 100 E. Guenther Street

10:00	WELCOME	
	Introductions – Texas A&M University Team	Natasha Hudspeth
	Introductions – LCRC Members	Susan Gilbertz
	Goals	Tarla Rai Peterson
10:30	SYSTEMS ACTIVITY	Susan Gilbertz
11:00	BREAK	
11:10	WATERSHED AS A SYSTEM	Ann Kenimer
11:40	WATERSHED SYSTEM ACTIVITY	Ann Kenimer
12:10	BREAK	
12:20	QUESTIONNAIRE	Charles Samuelson
12:30	COLLABORATIVE LEARNING AS A SYSTEM	Charles Samuelson
12:40	CLOSING ACTIVITIES	Susan Gilbertz
	Announcements	
	Membership Recommendations	
1:10	LUNCH	
2:00	ADJOURN	

♦LEON CREEK RESTORATION COUNCIL♦

Workshop Agenda

Saturday, December 11, 1999 • 2:00 pm – 4:00 pm
San Antonio River Authority Board Room • 100 E. Guenther Street

2:00	WELCOME & REVIEW	Tarla Rai Peterson
2:10	WATER QUALITY IN LEON CREEK	Ann Kenimer
2:40	BREAK – WATER QUALITY EXERCISES	
3:10	WATERSHEDS AS COMPLEX SYSTEMS	Marty Matlock
3:40	RESEARCH STRATEGY TO ADDRESS QUESTIONS	Bill Neill
4:00	ADJOURN	

♦LEON CREEK RESTORATION COUNCIL♦

Workshop Agenda

Friday, January 21, 2000 • 3:00 pm – 7:00 pm
San Antonio River Authority Board Room • 100 E. Guenther Street

3:00	WELCOME & REVIEW	Charlie Samuelson
3:10	WATER RECYCLING SYSTEM	Steve Clouse (SAWS) Jerry Berry (SAWS)
4:00	BREAK	
4:10	LISTENING	Tarla Peterson
4:30	COUNCIL DISCUSSION OF IDEAL WATERSHED	Susan Gilbertz
5:00	SMALL GROUP INSTRUCTIONS (short break after receiving instructions)	Susan Gilbertz
5:10	SMALL GROUP WORK SESSIONS	As Assigned
5:30	GROUP REPORTS	Tarla Peterson
6:00	LIGHT SUPPER	
7:00	ADJOURN	

♦ Leon Creek Restoration Council ♦

Workshop Agenda

Saturday, February 12, 2000 • 10:00 am – 2:00 pm
San Antonio River Authority Board Room • 100 E. Guenther Street

10:00	WELCOME	Charlie Samuelson
10:10	SURVEY RESEARCH	Guy Whitten
10:55	BREAK	
11:10	MODELING PUBLIC OPINION AND BEHAVIOR	Guy Whitten
12:00	BREAK	
12:10	COUNCIL PLANNING AND SCHEDULING	Tarla Rai Peterson
12:20	DEFINING COMMUNITY AND COUNCIL RELATIONSHIPS	Tarla Rai Peterson
12:55	CLOSING	Tarla Rai Peterson
1:00	LUNCH	

♦LEON CREEK RESTORATION COUNCIL♦

Workshop Agenda

Friday, March 24, 2000 • 12:00 pm – 4:00 pm
San Antonio River Authority Board Room • 100 E. Guenther Street

12:00	LUNCH	
12:30	WELCOME	Tarla Peterson (TAMU)
12:40	BASIC IDEAS OF HYDROLOGY AND FLOODING	Ann Kenimer (TAMU)
1:10	FLOOD CONTROL AND SARA	Steve Ramsey (SARA)
1:40	FLOODING AND PUBLIC WORKS	John German (City of San Antonio)
2:10	BREAK	
2:25	QUESTIONS AND ANSWERS	
3:00	INDIVIDUAL IDENTIFICATION ON CREEK	
3:20	LOCALE GROUPS GENERATE FLOOD CONTROL RECOMMENDATIONS	
3:45	PLENARY IMPLICATIONS	
4:00	ADJOURN	

♦ LEON CREEK RESTORATION COUNCIL ♦

Workshop Agenda

Saturday, April 15, 2000 • 10:00 am – 2:00 pm
San Antonio River Authority Board Room • 100 E. Guenther Street

10:00 WELCOME .... Susan Gilbertz

10:10 AGENCY PANEL– INTRODUCTIONS .... Susan Gilbertz

Scott Kinderwater, Texas Natural Resource Conservation Commission
Mike Gonzales, San Antonio River Authority
Meg Conner, San Antonio Water System
Lisa Morris, San Antonio Parks and Recreation Department
Rufus Stephens, Texas Parks and Wildlife Department
David Beales, City of San Antonio

11:00 QUESTION & ANSWER

11:45 BREAK

12:00 PLANNING PANEL– INTRODUCTIONS .... Tarla Rai Peterson

Mayor Howard Peak
Larry Clark

12:30 QUESTION & ANSWER

1:30 LUNCH

2:00 ADJOURN

♦Leon Creek Restoration Council♦

Workshop Agenda

Friday, May 5, 2000 • 12:00 pm – 4:00 pm
San Antonio River Authority Board Room • 100 E. Guenther Street

12:00	Lunch	
12:30	Visual Images of Leon Creek	Madeleine Pepin's Students
12:50	Welcome	David Gore
1:00	Texas A&M University	Marty Matlock
1:30	San Antonio River Authority	Mike Gonzales
2:00	Questions	
2:25	Break	
2:40	Developing a Systems Framework for Watershed Improvement Suggestions	
4:00:	Adjourn	

♦LEON CREEK RESTORATION COUNCIL♦

Workshop Agenda

Saturday, June 10, 2000 • 10:00 am – 2:00 pm
San Antonio River Authority Board Room • 100 E. Guenther Street

10:00 WELCOME .... Charlie Samuelson, Texas A&M University

10:05 RIPARIAN WOODLAND SYSTEMS .... Mark Peterson, Texas Forest Service

10:35 TERRESTRIAL RIPARIAN ECOLOGY & WILDLIFE
.... Rufus Stephens, Texas Parks & Wildlife Department

11:05 QUESTION & ANSWER

11:20 BREAK

11:35 AQUATIC ECOLOGY .... Bill Neill, Texas A&M University

12:00 QUESTION & ANSWER

12:15 QUESTION AND ANSWER REGARDING SEWAGE SYSTEM REPAIRS AND MAINTENANCE
Dan Myers: Operations Director, San Antonio Water System

12:40 FLOW MAPS – UPDATED TO INCLUDE TODAY'S TOPICS

1:10 GROUP REPORTS (MAPS OF FLOWS)

1:30 LUNCH AND ANNOUNCEMENTS
Committee Members Meet Over Lunch

2:00 ADJOURN

♦LEON CREEK RESTORATION COUNCIL♦

Workshop Agenda

Friday, July 14, 2000 • 12:00 pm – 4:00 pm
San Antonio River Authority Board Room • 100 E. Guenther Street

12:00	LUNCH AND ANNOUNCEMENTS	
12:30	WELCOME	Tarla Peterson, Texas A&M University
12:35	LAND USE OPTIONS	Paul Barwick & Paul Wagner, Nature Conservancy
1:05	ECOLOGICAL MEASURES IN STREAMS	Steve Lusk, San Antonio Water Authority
1:35	QUESTION & ANSWER	
1:45	BREAK	
2:00	WATER QUALITY MODELING – WATERSHED APPROACH	Marty Matlock, Texas A&M University
2:30	MODELING EXERCISE AND WRAP-UP	Marty Matlock, Texas A&M University
3:30	COMMITTEE MEETINGS	
3:55	ANNOUNCEMENTS	
4:00	ADJOURN	

♦LEON CREEK RESTORATION COUNCIL♦

Workshop Agenda

Saturday, September 9, 2000 • 9:00 am – 1:00 pm
San Antonio River Authority Board Room • 100 E. Guenther Street

9:00	WELCOME	Susan Gilbertz, Texas A&M University
9:15	COMMITTEE REPORTS	Council Committees
9:45	DISCUSSION OF POSSIBLE FUTURES FOR COUNCIL	
10:00	WATER QUALITY MODELING – WATERSHED APPROACH	Marty Matlock, Texas A&M University
10:30	QUESTION & ANSWER	
10:45	BREAK	
11:00	GROUP DISCUSSION OF IMPROVEMENTS & RECOMMENDATIONS	
12:00	COMMITTEE MEETINGS	
12:30	LUNCH	
1:00	ADJOURN	

♦ LEON CREEK RESTORATION COUNCIL ♦

Workshop Agenda

Friday, October 6, 2000 3:00–7:00 pm
San Antonio River Authority Board Room • 100 E. Guenther Street

3:00	WELCOME AND PREVIEW	Charlie Samuelson
3:15	MODELING THE LEON CREEK WATERSHED SYSTEM	Bill Grant & Ann Kenimer
4:00	INTEGRATING AQUATIC LIFE INTO THE MODEL	Bill Neill
4:30	COMPLETE INDIVIDUAL INTEREST SHEET	Susan Gilbertz
4:40	BREAK	
4:55	DEBATE & MODEL TENTATIVE RECOMMENDATIONS	Council Members
5:45	SUMMARIZE MODELING RESULTS	Ann Kenimer
6:00	IDENTIFY TENTATIVE RECOMMENDATIONS	Tarla Peterson
6:30	COMMITTEE REPORTS & DINNER	
7:00	ADJOURN	

♦ LEON CREEK RESTORATION COUNCIL ♦

Workshop Agenda

Friday, November 3, 2000 • 12:00 p.m – 4:00 p.m.
San Antonio River Authority Board Room • 100 E. Guenther Street

12:00 LUNCH

12:30 WELCOME

12:40 COUNCIL REPORT FOR MAYOR .... Council Members

1:00 DISCUSSION OF COUNCIL REPORT

1:15 MAYORS PRESENTATION .... Howard Peak

1:45 DISCUSSION OF MAYOR'S PRESENTATION

2:00 BREAK

2:15 AQUATIC LIFE IN LEON CREEK .... Bill Neill, Texas A&M University

2:45 DISCUSSION

4:00 ADJOURN

♦ Leon Creek Restoration Council ♦

Workshop Agenda

Friday, December 1, 2000 9:00am –1:00 pm
San Antonio River Authority Board Room • 100 E. Guenther Street

9:00	Welcome and Preview	Susan Gilbertz
9:15	Committee Reports	Council Members
9:30	Watershed Restoration Strategies	Karen Cummins, Texas A&M University
10:00	Watershed Restoration Strategies	Nancy Beward, San Antonio Public Works
10:45	Break	
11:00	Survey/Questionnaire	Charles Samuelson
11:30	Recommendations	Tarla Peterson
12:00	Lunch	
1:00	Adjourn	

Appendix B

PRETEST (WAVE 1) AND POSTTEST (WAVE 2) SURVEY QUESTIONS USED TO EVALUATE CL PROGRAM EFFECTIVENES

Questions below are ordered according to Table 20 and the discussion of results in the text of the Research Findings section. Minor wording changes made to several Wave 2 survey questions are denoted in ***bold italics*** within brackets.

1. **Q22**: "The sources of most contaminants in the Salado Creek watershed are easily identified by the local authorities responsible for monitoring water quality.
    - Strongly Agree
    - Agree
    - Neither Agree nor Disagree
    - Disagree
    - Strongly Disagree

2. **Q23**: "Water quantity can affect stream desirability as much as water quality.
    - Strongly Agree
    - Agree
    - Neither Agree or Disagree
    - Disagree
    - Strongly Disagree

3. **Q24**: "Water quality problems in Salado Creek are the result of a few major contaminant sources."
    - Strongly Agree
    - Agree
    - Neither Agree or Disagree
    - Disagree
    - Strongly Disagree

4. **Q25**: "It is easy to predict how water quality problems will affect living organisms in and near a stream."
    - Strongly Agree
    - Agree
    - Neither Agree or Disagree
    - Disagree
    - Strongly Disagree

5. **Q26**: "Water quality problems in Salado Creek change depending on factors such as weather and time of year.
    - Strongly Agree
    - Agree
    - Neither Agree or Disagree
    - Disagree
    - Strongly Disagree

6. **Q27**: "Water quality is easy to predict because pollutants and flow are fairly constant over time."
    - Strongly Agree
    - Agree
    - Neither Agree or Disagree
    - Disagree
    - Strongly Disagree

7. **Q28**: "I am familiar with the scientific information used to make decisions about water quality management in the Salado Creek watershed."
    - Strongly Agree
    - Agree
    - Neither Agree or Disagree
    - Disagree
    - Strongly Disagree

8. **Q29**: "I am familiar with the Total Maximum Daily Load (TMDL) program for ensuring water quality in Texas rivers and streams.
    - Strongly Agree
    - Agree
    - Neither Agree or Disagree
    - Disagree
    - Strongly Disagree

9. **Q36**: "My current understanding of systems thinking principles is:"
    - Poor
    - Fair
    - Good
    - Very Good
    - Excellent

10. **Q39**: "My ability to understand and use scientific information regarding water quality problems and related environmental issues is:"
   - Poor
   - Fair
   - Good
   - Very Good
   - Excellent

11. **Q30**: "Constructive communication regarding watershed management issues is occurring among affected parties in the Salado Creek watershed."
   - Strongly Agree
   - Agree
   - Neither Agree nor Disagree
   - Disagree
   - Strongly Disagree

12. **Q37**: "In general, my ability to listen to others is:"
   - Poor
   - Fair
   - Good
   - Very Good
   - Excellent

13. **Q38**: "In general, my ability to ask good questions is:"
   - Poor
   - Fair
   - Good
   - Very Good
   - Excellent

14. **Q31**: "Members of the Salado Creek Restoration Council get along well with each other."
   - Strongly Agree
   - Agree
   - Neither Agree nor Disagree
   - Disagree
   - Strongly Disagree

15. **Q32**: "I trust other members of the Salado Creek Restoration Council to do what is best for the long-term health and integrity of Salado Creek."
    - Strongly Agree
    - Agree
    - Neither Agree nor Disagree
    - Disagree
    - Strongly Disagree

16. **Q33:** "I have high personal regard for the other members of the Salado Creek Restoration Council."
    - Strongly Agree
    - Agree
    - Neither Agree nor Disagree
    - Disagree
    - Strongly Disagree

17. **Q34**: "I will be ***[have been]*** effective in negotiating with other members of the Salado Creek Restoration Council during [future] watershed policy discussions."
    - Strongly Agree
    - Agree
    - Neither Agree nor Disagree
    - Disagree
    - Strongly Disagree

18. **Q35**: "Overall, my past experience with collaborative planning/ decision making methods ***[experience with the Collaborative Learning format used in the Salado Creek Council meetings]*** has been:"
    - Very Positive
    - Positive
    - Neutral
    - Negative
    - Very Negative

19. **Q18**: "In your opinion, how safe is water from Salado Creek for drinking?"
    - Very safe
    - Safe
    - Not sure
    - Unsafe
    - Very unsafe

20. **Q19**: "How safe do you think Salado Creek is for swimming?"
   - Very safe
   - Safe
   - Not sure
   - Unsafe
   - Very unsafe

21. **Q20**: "How safe do you think it is to eat fish that come from Salado Creek?"
   - Very safe
   - Safe
   - Not sure
   - Unsafe
   - Very unsafe

22. **Q21**: "In your opinion, how safe is Salado Creek water for livestock to drink?
   - Very safe
   - Safe
   - Not sure
   - Unsafe
   - Very unsafe

23. **Q2**: "Please circle the response that best represents your opinion on this statement: People like me do not have any say about what the **national** government does."
   - Strongly agree
   - Somewhat agree
   - Neither agree nor disagree
   - Disagree somewhat
   - Disagree strongly

24. **Q3**: "Please circle the response that best represents your opinion on this statement: People like me do not have any say about what the **city** government does."
   - Strongly agree
   - Somewhat agree
   - Neither agree nor disagree
   - Disagree somewhat
   - Disagree strongly

Appendix C

Salado Creek Restoration Council (SCRC) Recommendations

SCRC Preliminary Recommendations
Develop a Master Plan for San Antonio Sub-Watersheds
From October & November 2000 Meetings

PLAN SHOULD INCLUDE THE FOLLOWING GOALS

I. Minimize damage from flooding and maintain sufficient flow to restore water quality

II. Restore and maintain water quality to a level sufficient to support a recreational fishery and wildlife habitat

SUGGESTED MEANS FOR ACHIEVING THESE GOALS

1. Increase citizen input in implementation of Proposition 3
   - Establish a watershed coordinator with responsive oversight – should work directly with citizens' groups
   - Use the Salado Creek foundation as a model/basis for citizen input
   - Develop a schedule for working with appropriate agencies toward analysis and possible implementation of recommendations 2 through 9
   - Provide regular water quality monitoring opportunities (share equipment)
   - Involve schools, bicyclists, hikers, riders, other recreational groups

2. Strengthen city and county ordinances (including enforcement procedures) to prevent inappropriate development, floodplain fills and dumping
   - Examine current "fee in lieu of" retention policy – increase fees to ensure covering cost of retention; ensure fees are used for retention
   - Eliminate grandfathering for floodplain development that has been planned, but not implemented
   - Control runoff in all new developments, including those planned before current regulations – respect needs of downstream areas

3. Assist grandfathered floodplain developments in working toward compliance with current regulations

4. Establish and maintain city-wide program requiring responsible clearing & cleaning of waterways

5. Eliminate hazards such as low water crossings (put in bridges as needed, ex. Roland Ave.)

6. Retention
    - Add retention facilities near Walzem Road and Beitel Creek
    - Explore placing retention facilities in NE sub basins (Pershing Creek)
    - Explore retention as part of a park at MLK and Rice Road
    - Develop retention facilities that provide multiple uses and require minimum costs for maintenance
7. Establish meaningful base flow
    - SAWS should consider augmenting base flow with additional discharge points for recycled water
    - Use water retained during floods to augment low flows
8. Solid Waste
    - Provide more frequent and convenient household hazardous waste collection
    - Establish permanent solid waste collection points
    - Offer incentives (pay a small sum) to people who bring solid waste to collection points rather than charging them a fee
    - Expand and refine definition of hazardous materials
9. Investigate impact of quarries on
    - Aquifer recharge
    - Water quality
    - Flow

Leon Creek
Restoration Council (SCRC)
Recommendations

LCRC Preliminary Recommendations
Develop a Master Plan for Leon Creek Watershed
From October and November 2000 Meetings

PLAN SHOULD

I. Include guidance for enhanced citizen involvement in watershed management

II. Maintain and Restore (as necessary) Leon Creek so that it is an Asset to Community

SPECIFIC SUGGESTIONS

I. Watershed Council led by citizens

1. Begin with existing stakeholder groups (LCRC)
2. Expand to include youth, specific interest groups, additional neighborhoods
3. Encourage strengthening of neighborhood coalitions
4. Include a public education and outreach component
5. Use agency expertise in its deliberations
6. Obtain resources (financial and other) to enable self-maintenance
7. Facilitate citizen oversight of agency performance
    - Use influence with public officials and media to hold agencies responsible for those actions
    - Coordinate organizations most directly involved in watershed management (San Antonio city offices, smaller cities, Bexar County, SAWS, SARA, TNRCC, TXDOT, TPWD,).
8. Hold regular (perhaps quarterly) meetings with relevant agency personnel
    - Provide agencies with specific charges and reasonable deadlines
    - Education and coordination with the media
    - Present requests and review agency performance of past requests
    - Facilitate common understanding of differing situations experienced by citizens and agency personnel
    - Encourage enforcement of existing rules
    - Request regular briefings on best management practices

Appendix D

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ANALYSIS

Change in ecosystem service values in the San Antonio area, Texas☆

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Abstract

San Antonio is one of the fastest growing metropolitan areas in the USA. Urban sprawl may significantly impact ecosystem services and functions but such effects are difficult to quantify and watershed-level estimates are seldom attempted. The objective of the study reported here was to determine whether LANDSAT MSS could be used to quantify changes in land-use and ecosystem services due to urban sprawl in Bexar County, TX, in which San Antonio is centered. The size of six land cover categories in the summer of 1976, 1985, and 1991 were estimated in the 141 671 ha of three watersheds in Bexar County. Coefficients published by Costanza and co-workers in 1997 [Nature 387 (1997) 253] were used to value changes in ecosystem services delivered by each land cover category, and a sensitivity analysis was conducted to determine the effect of manipulating these coefficients on the estimated values. Although we estimated that there was a 65% decrease in the area of rangeland and a 29% increase in the area of urbanized land use between 1976 and 1991, there appeared to be only a 4% net decline in the estimated annual value of ecosystem services in the study area (i.e. \$5.58 ha^{-1} per year, with a 15-year cumulative total value of \$6.24 million for the whole study area). This relatively small decline could be attributed to the neutralizing effect of the estimated 403% increase in the area of the woodlands, which were assigned the highest ecosystem value coefficient. When we assumed that the shift of rangelands to woodlands produced no net change in the value of ecosystem services per hectare, the estimated annual ecosystem service value declined by 15.4% (\$23.22 ha^{-1} per year) between 1976 and 1991. When conducting time-series studies of ecosystem services, it is important to identify parallel changes in land cover types in order to quantify the potentially neutralizing influence of positive land cover changes on the negative effects of urban sprawl on ecosystem services. © 2001 Elsevier Science B.V. All rights reserved.

Keywords: Urban sprawl; Land-use change; Remote sensing; Rangelands; Woodlands

☆ Additional information: this paper was in part prepared by Heather Harris for a graduate course in Ecological Economics offered in the Department of Rangeland Ecology and Management, Texas A&M University, College Station, TX, USA.

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1. Introduction

Between 1982 and 1997, the amount of urbanized land in the USA increased by 47% to 30.75 million ha, while the population grew by 17% (Fulton et al., 2001). During the same time period, the conversion of land for development was estimated to have increased from about 500 000 ha per year between 1982 and 1992 to 1.3 million ha per year between 1992 and 1997 (NRCS, 1999). In general, urban sprawl in the south has been exacerbated by a decline in population density in urban centers, though to a lesser extent in Texas where urban population densities have decreased less than in other southern metropolitan areas (Fulton et al., 2001).

Population growth has been especially rapid in the states along the USA–Mexico broader (USCB, 1993). In Texas, a border state, the human population is projected to increase from 19 to 33 million by 2030, with over 70% of the growth expected to occur along the central and southern portions of the I-35 highway corridor and in the Lower Rio Grande Valley (Conner and James, 1996). As a result of this growth, San Antonio has become one of the fastest growing metropolitan areas in the USA, experiencing a 25.2% increase in population from 1990 to 1998, reaching approximately 1 million in 1996, and now being the eighth largest city in the country (SAEDF, 1999). This growth can be largely attributed to a steady growth in employment in the San Antonio area during the latter half of the 1990s when several large manufacturers moved into the area in response to the North American Free Trade Agreement (Rylander, 1997).

This population growth is increasingly impacting rural areas, especially those close to major urban centers in the southern part of Texas, by accelerating land subdivision and reducing the average size of land parcels (Conner and James, 1996). In addition, increase in urban sprawl generally leads to greater traffic volumes, increased pressure on local resources, less open space (Holtzclaw, 1999), and such land-use changes often have a significant negative impact on the affected ecosystems and the goods and services that they provide. Ecosystem services represent the benefits that living organisms derive from ecosystem functions that maintain the Earth's life support system, and include nutrient cycling, carbon sequestration, air and water filtration, and flood amelioration, to name a few (Costanza et al., 1997).

While changes in land use may significantly affect ecosystem processes and services, monitoring and projecting the impacts of such land-use changes are difficult for several reasons. Monitoring changes at the regional scale (where the impact of land-use changes on ecosystems often become noticeable) is difficult because of the large volume of data and interpretation required. In addition, accurately quantifying the impacts of urban sprawl on changes in ecosystem services is difficult because of the lack of information about the contribution of alternate landscapes to these services. Finally, in order to facilitate informed decision-making by comparing the impact of anthropogenic land-use changes with the effect of 'natural' ecosystem changes requires more explicit measures than simple value indices.

The objectives of this study were: (1) to evaluate the efficacy of using LANDSAT multispectral scanner (MSS) data to quantify land-use change in Bexar County, TX, from 1976 to 1991; and (2) to determine if generalized coefficients can be used to evaluate changes in ecosystem services at the watershed scale.

1.1. Using LANDSAT MSS data to measure land-use change

Potentially adverse ecological impacts of urban sprawl have increasingly prompted attempts to map and characterize urban and suburban growth. The US Geological Survey is developing a geo-referenced database of urban land-use change in selected metropolitan regions by merging information from historical maps, census statistics, commerce records, remotely sensed data, and digital land-use data (Acevedo et al., 1997), but this database is incomplete. As historical satellite imagery has become more readily available and less expensive, LANDSAT imagery has become an important tool for acquiring environmental data at spatial, temporal, and spectral resolutions appropriate for assessing broad land-use changes (Verstraete et al., 1996).

While the relatively low 80 × 80-m spatial resolution of the LANDSAT MSS data limits the detail that can be extracted from these data, ancillary data, such as maps reflecting land-use at the time that a satellite image was taken, can facilitate classification of coarse-resolution images. If coarse-resolution data and classification levels provide sufficient explanatory power for a given purpose, their use may be advantageous because they are less data intensive and provide better broad-scale uniformity than finer resolution data and classification levels (Bourgeron et al., 1999). Moreover, because LANDSAT MSS images were initially produced as early as 1972, MSS data represent the most comprehensive data set for analyzing large-scale land-use changes during the last 25 years.

While in some instances it is desirable to use high-resolution data to conduct detailed land-use analyses, such data cannot be used to quantify long-term land-use changes. Aerial photographs have been used since the 1940s and thus predate LANDSAT MSS, but such images are generally not available for a specified area at regular intervals. In order to study temporal changes, a time series of images for the location in question must be available. Satellite-based imaging (e.g. LANDSAT MSS, LANDSAT TM, etc.) was the first technology to routinely produce images at regular intervals. Digital land-use maps (based on a wide variety of data including LANDSAT images) can also facilitate analyses of land-use patterns. However, because they are composed of data averaged over some time period, such maps do not represent time-specific data and, therefore, cannot be used for time-series analyses of land-use change. We used LANDSAT MSS data to classify land-use during a 15-year period in Bexar County because: (1) they provided readily available and affordable time-specific digital data obtained at regular intervals since the early 1970s; (2) an objective of this study was to quantify long-term changes in land-use; and (3) the resolution of the data was sufficient for classifying land-use patterns at the watershed scale.

1.2. Estimating the value of ecosystem services

Abramovitz (1998) pointed out that ecosystem services have extensive economic value but that they are not credited for the non-market values they provide until they become depleted. While economic tools can be used to identify trade-offs between known ecological values, it remains challenging to link technical measures of ecosystem services to attributes that can be effectively evaluated by untrained individuals (Schaberg et al., 1999). Despite this and other challenges, several attempts have been made to estimate the worth of natural resources. Most notably, Costanza et al. (1997; 1998) presented a model for placing an economic value on different biomes and the services that they provided. Based on their model, they estimated that the global biospheric value of 17 identifiable ecosystem services provided by the 16 dominant global biomes is $33 trillion per year, most of which is outside the market. However, because of uncertainties, they stated that this should be considered to be a minimum estimate.

While Costanza et al.'s article did focus debate on the importance of ecosystem services that are generally undervalued in standard economic analyses, their cross-sectional estimate based on average, often local, per-unit values, was widely criticized by economists for both theoretical and empirical reasons (Pimm, 1997; Toman, 1998; Masood and Garwin, 1998; Norgaard et al., 1998; Pearce, 1998). For example, because the last hectare of an ecosystem to disappear is likely to be worth much more than the first, simple multiplication of selected average values by all the units in the biosphere underestimated a potentially infinite social value of ecosystem services. Pearce's (1998) greatest concern was that Costanza et al.'s estimated $33 trillion 'value of everything' is larger than the world GNP which is around $18 trillion per year. Since 1997, additional studies conducted to quantify the value of ecosystem services have produced lower estimates. For example, Alexander et al. (1998) estimated that ecosystem services are 44–88% of global GNP and concluded that while this estimate is lower than Costanza et al.'s estimate, it nevertheless indicates that accounting for ecosystem service values would greatly alter current GNP estimates. In a regional study using locally derived data, Seidl and Moraes (2000) re-estimated the ecosystem contribution of the Pantanal sub-region Nhecolandia to global production and derived a value of $15.5 billion per year, approximately 50% of Costanza et al.'s corresponding estimate.

Although Costanza et al.'s estimates of the value of ecosystem services are imperfect, and we lay no claim to their veracity, they do represent the most comprehensive set of first-approximations available for quantifying the change in the value of services provided by a wide array of ecosystems. Since one objective of our study was to determine the effectiveness of using generalized value coefficients to estimate watershed-level changes in ecosystem services, and because the scope of our project did not allow us to obtain area specific value coefficients, we used Costanza et al. (1997) estimates in our study despite their limitations.

2. Study area and estimation approach

San Antonio is centered in Bexar County (29°27′ N, 98°31′ W) near the head of the San Antonio River Basin, which traverses the Edwards Plateau, the Texas Blackland Prairies, and the Western Gulf Coastal Plain eco-regions (Fig. 1). Bexar County was chosen for the study in order to maximize the probability of detecting changes in ecosystem services due to urban

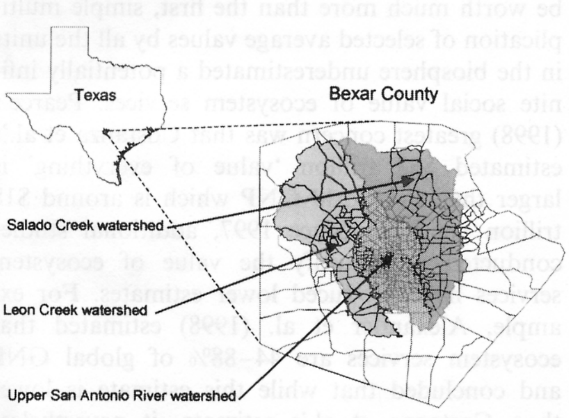

Fig. 1. Location of three watersheds in Bexar County, TX.

sprawl. The three major streams running through the county are Salado Creek, the Upper San Antonio River, and Leon Creek. Since watersheds often represent the minimum ecological management unit and the purpose of our study was to estimate changes in the value of ecosystem services in the wake of land use conversion, the watersheds associated with the three streams were used as the analytical units in Bexar County. The size of the three watersheds comprising the study area was 141 671 ha.

Satellite imagery and remote sensing analytical software were used to determine the area of six land-use classes described below in each of the three watersheds in Bexar County. These estimates were then incorporated into an economic valuation model to quantify changes in the value of ecosystem services in each watershed over time. In addition, a sensitivity analysis was conducted to determine how sensitive the estimates of the value of ecosystem services were to the applied valuation coefficients.

2.1. Land-use classification

Data sets incorporating Bexar County during relatively cloud-free days in August 1976 (the earliest available MSS data set of the study area), August 1985 (the next available cloud-free MSS data set in a year with rainfall similar to 1976), and June 1991 (the most recent available cloud-free MSS data set in a similar rainfall year) were obtained from the US Geological Survey EROS Data Center. The data sets were geo-referenced and Bexar County was extracted as a separate data set using the ERDAS Imagine® software (ERDAS, 1997), which incorporates functions for both image processing and the use of geographic information systems (GIS). Groupings of the spectral properties of the pixels in each of the three data sets of Bexar County were obtained through unsupervised classification using the Iterative Self-Organizing Data Analysis Technique (ISODATA) (Jensen, 1996) that is available in ERDAS Imagine®.

After the unsupervised classifications of the 1976, 1985, and 1991 data sets were completed, the resulting 100 spectral signature classes in each image were grouped into six land cover categories. This was accomplished using the US Geological Survey Land-use/Land Cover Classification System for Use with Remote Sensor Data, Level I (Jensen, 1996) in combination with a 1991 ground-truthed land-use map of Bexar County, developed for zoning purposes by the City of San Antonio Planning Department. The six land cover categories were: rangeland, woodland, bare soil, residential, commercial, and transportation. In Bexar County, the rangeland category corresponds to relatively open grasslands, while *Juniper*, *Quercus* (oak) and *Ulmus* (elm) species dominate the woodlands. During the time period covered by the LANDSAT MSS data used in this study, cultivated areas would have been largely bare and thus bare soils were assumed to reflect mainly post-harvest croplands. Residential (mainly suburban) and commercial (mainly urban) areas were differentiated by assuming that commercial areas have a reflective value more closely resembling paving materials, which occurs extensively in urban areas, while residential areas were assumed to be intermediate to the commercial and bare soil spectral signatures.

Once images of the six land categories were derived for each of the 3 years, vector files delineating the Leon Creek, Salado Creek, and the Upper San Antonio River Watersheds were projected onto these images in order to estimate the area of each of the six land cover types within each watershed. Since no true reference data were available for the 1976–1991 time periods of investigation, uncertainty associated with land-use classification change was not measured (see Congalton and Green, 1999).

2.2. Assignment of ecosystem service values

In order to obtain ecosystem services values for various ground cover types, the six land cover categories used to classify the LANDSAT MSS data sets were compared with the 16 biomes identified in Costanza et al.'s (1997) ecosystem services valuation model. The most representative biome was used as a proxy for each land cover category, including grass/rangelands for rangelands, temperate/boreal forest for woodland, cropland for bare soil, and urban for the commercial, residential, and transportation categories (which were not differentiated by Costanza et al.), as shown in Table 1. The total value of terrestrial ecosystem services in the study area in 1976, 1985, and 1991 was obtained as follows:

Table 1
Costanza et al. (1997) biome equivalents for the six land-use categories, and the corresponding ecosystem values

Land cover categories	Equivalent biome	Ecosystem service coefficient ($ ha^{-1} per year)
Rangeland	Grass/rangelands	232
Woodland	Temperate/boreal forest	302
Bare soil	Cropland	92
Residential	Urban	0
Commercial	Urban	0
Transportation	Urban	0

$$ESV = \sum(A_k \times VC_k) \quad (1)$$

where ESV is the estimated ecosystem service value, A_k is the area (ha) and VC_k the value coefficient ($ ha^{-1} per year) for land use category 'k'. The change in ecosystem service values was estimated by calculating the difference between the estimated values for each land cover category in 1976, 1985 and 1991.

The biomes used as proxies for the land cover categories are clearly not perfect matches in every case. Specifically, the juniper/mesquite/elm dominated woodlands in the study area may not be well represented by Costanza et al.'s temperate/boreal forest biome because of the different climatic conditions under which they occur, but in terms of the ecosystem services that they provide, the woodlands of Texas and temperate/boreal forests do have some similarities. For example, Texas woodlands can increase soil nutrient concentrations and contribute to gas regulation through their roles as carbon sinks (e.g. Boutton et al., 1999), and they can provide recreation opportunities. Due to the uncertainties about the representativeness of the proxies used for each land cover category as well as the veracity of Costanza et al.'s value coefficients, sensitivity analyses were conducted to determine the dependence of temporal changes in ecosystem service values on the applied valuation coefficients. The ecosystem value coefficients for rangeland, woodlands, and bare soil categories were each adjusted by 50%. Pervasive woody plant invasion throughout much of Texas, including Bexar County, resulted in a large portion of the area identified as rangeland in 1976 being classified as woodland in 1991. However, because of the uncertainty about the representatives of Costanza et al.'s temperate/boreal forest biome for woodlands, one additional sensitivity analysis was conducted by reducing the ecosystem value coefficient for woodlands (\$302 ha^{-1} per year) by 30% to that of rangelands (\$232 ha^{-1} per year), thereby assuming that woodlands are a variant of rangelands.

In each analysis, the coefficient of sensitivity (CS) was calculated using the standard economic concept of elasticity, i.e. the percentage change in the output for a given percentage change in an input (Mansfield, 1985):

$$CS = \frac{(ESV_j - ESV_i)/ESV_i}{(VC_{jk} - VC_{ik})/VC_{ik}} \quad (2)$$

where ESV is the estimated ecosystem service value, VC is the value coefficient, 'i' and 'j' represent the initial and adjusted values, respectively, and 'k' represents the land use category.

If the ratio of the percentage change in the estimated total ecosystem value (ESV) and the percentage change in the adjusted valuation coefficient (VC) is greater than unity, then the estimated ecosystem value is elastic with respect to that coefficient, but if the ratio is less than one, then the estimated ecosystem value is considered to be inelastic. The greater the proportional change in the ecosystem service value relative to the proportional change in the valuation coefficient, the more critical is the use of an accurate ecosystem value coefficient.

3. Estimated changes

3.1. Land-use change estimates

It is important to emphasize that, due to a lack of reference data, a limitation of retrospective land-use classification is uncertainty about the accuracy of the estimated size of land-use categories (Congalton and Green, 1999). Therefore, observation of changes in the size of land-use categories must be treated with caution. However, if the magnitude of the estimated changes in land use is substantial, it may still be possible to draw general inferences about the effect of perceived changes in land use patterns on ecosystem services.

In 1976, the rangeland category dominated all three watersheds within the study areas and totaled an estimated 80 497 ha (Fig. 2), but by 1991, the total area of this category was estimated to have decreased by about two-thirds to 27 896 ha. The effective annual rate of decrease appears to have accelerated from about 3% per year during the 1976–1985 period to 12% per year during the 1985–1991 period (Table 2). The woodland category, by contrast, increased substantially in all three watersheds and overall more than quadrupled from 8886 ha in 1976 to 44 654 ha in 1991 (Fig. 2), with an estimated annual growth rate of 10–12% per year (Table 2). The sizeable decrease in open rangelands and the concomitant increase in the size of the woodland category is consistent with the widespread and rapid encroachment of woody plants in the region (Smeins and Merrill, 1988; Smeins et al., 1997). Intensive livestock grazing and reduced fire frequency have resulted in a change in the grass–woody plant interaction on many rangelands in the USA (Scifres, 1980; Archer and Stokes, 2000), which has led to widespread transformation of grasslands to shrublands, and savannas and woodlands (Archer, 1989, 1994; Schlesinger et al., 1990). As a result of these changes and the associated decline in water yields in some areas (Thurow et al., 2000), brush control has become a dominant issue in rangeland management throughout the Edwards Plateau and the Edwards Aquifer Recharge Zone.

The bare soil category, which was assumed to predominantly represent post-harvest cropping areas, was estimated to have increased in all three watersheds from 6353 ha in 1976 to 13 057 ha in 1991 (Fig. 2). However, while the size of this land cover category increased by an estimated 113% between 1976 and 1985, it appeared to have decreased by about 3% during the subsequent 6 years (Table 2). The initial increase may have been

Table 2
Total estimated area (ha) of each land-use category in the San Antonio study area, and changes in land-use from 1976 to 1991

Land-use category	Total area (ha)			1976–1985			1985–1991			1976–1991		
	1976	1985	1991	ha	%	% per year	ha	%	% per year	ha	%	% per year
Rangeland	80 497	59 126	27 896	−21 371	−27	−3.4	−31 230	−53	−12.0	−52 601	−65	−6.8
Woodland	8886	25 336	44 654	16 450	185	12.0	19 319	76	9.9	35 769	403	11.0
Bare soil	6353	13 514	13 047	7161	113	8.7	−467	−3	−0.6	6694	105	4.9
Residential	11 499	10 087	16 655	−1412	−12	−1.4	6568	65	8.7	5156	45	2.5
Commercial	6116	10 457	15 362	4341	71	6.1	4905	47	6.6	9246	151	6.3
Transportation	25 748	23 060	23 857	−2687	−10	−1.2	797	3	0.6	−1891	−7	−0.5

associated with clearing of rangelands for crop production, as well as suburban development, but it may also have been partly due to misclassification because of seasonal variations in vegetative ground cover at the time that the three satellite data sets were captured. For example, more rapid spring growth due to above average spring rains in 1976 could have resulted in an underestimate of cropland due to misallocation of crop land (bare soil) to rangeland in the 1976 image (NCDC, 1994).

Although the estimated size of the residential land cover category increased in all three watersheds from 11 449 ha in 1976 to 16 655 ha in 1991 (Fig. 2), the rate of growth averaged only 3% per year during this 15-year period, and most of the growth appears to have occurred during the 1985–1991 period (9% per year) (Table 2). The estimated size of this category actually decreased slightly in the Salado Creek and Leon Creek watershed between 1976 and 1985. The apparently small change in the residential land cover categories may be an underestimate of the actual growth for two reasons. First, because asphalt shingles are widely used for roofing and residential area road networks are dense, some residential areas could have been misallocated to the urban and transportation categories. Secondly, the potentially increasing masking effect of turf and tree canopy cover in residential areas could have resulted in some misallocation of maturing residential areas to the rangeland or woodland land cover categories in 1985 and 1991.

According to our analysis, there was a 151% increase in the area of the urban land use category from 1976 to 1991 (Table 2). By contrast, the transportation category appeared to decrease marginally during the 15-year period of our study, mainly from 1976 to 1985, but reduction in transportation is inconsistent with expanding urban sprawl. This anomaly may be explained by the fact that there are inevitable inaccuracies in delineations of urban and transportation land uses when using remotely sensed spectral reflectance data because asphalt in parking lots, roofs, and roads produces identical reflectance, and concrete has the same spectral signature regardless of its location. Thus it is better to consider changes in the urban and transportation categories together. In combination, these two land use categories increased 23% from an estimated 31 864 ha in 1976 to 39 219 ha in 1991, and the average annual rate of growth appeared to have accelerated from 0.6% per year during the 1976–1985 period to 3.5% per year during the 1985–1991 period.

3.2. Estimation of changes in ecosystem services

Using the estimated change in the size of each land cover category, together with the ecosystem service value coefficients reported by Costanza et al. (1997), we found that land-use changes in the

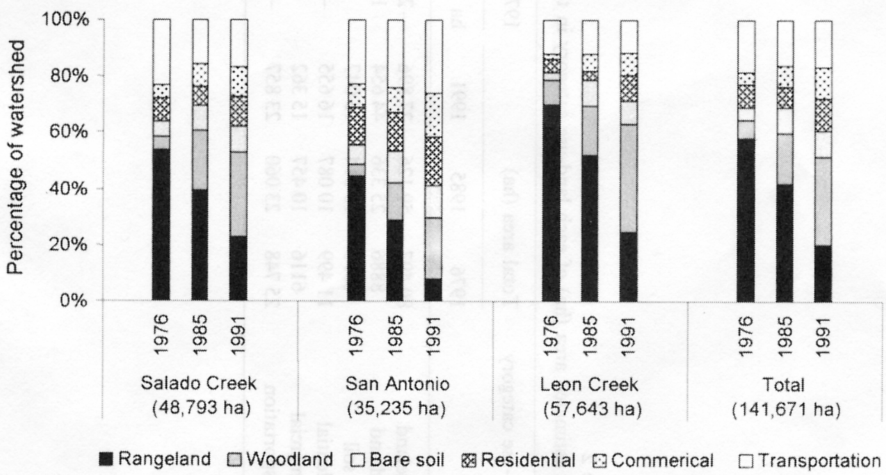

Fig. 2. Size and percentage contribution of each of six land cover categories in each watershed and the total study area.

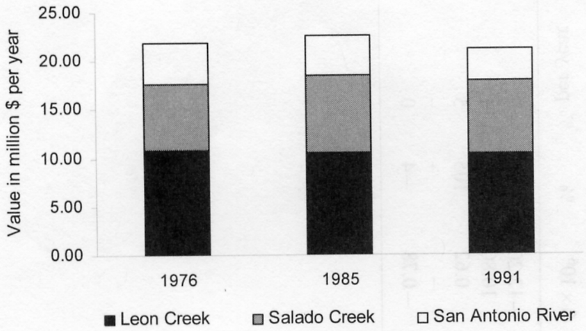

Fig. 3. Ecosystem service values estimated for each watershed using Costanza and co-author's 1997 coefficients.

141 671 ha of our study area resulted in a \$0.78 million per year (i.e. 4% or \$5.58 ha^{-1} per year) net decline in ecosystem services from 1976 to 1991 (Fig. 3, Table 3). Assuming a linear decline in ecosystem services from 1976 to 1991, this decline represents a cumulative loss of \$6.24 million in ecosystem services over the 15-year period of the study. However, this cumulative estimate must be regarded with caution because ecosystem services may fluctuate somewhat; our analysis detected a slightly higher ecosystem service value in 1985 than in 1976. In order to accurately estimate the cumulative loss of ecosystem services, it would be necessary to measure ecosystems' services at regular intervals, preferably annually.

The estimated change in the annual value of ecosystem services was small in all three watersheds between 1976 and 1991. However, while the change was negative in both the Leon Creek and San Antonio River watersheds (\$0.41 and 0.96 million per year, respectively), it was positive in the Salado Creek watershed (\$0.59 million per year). The primary reason for these small changes is that the effect of the estimated decline in rangelands, valued at \$232 ha^{-1} per year, was largely offset by the increase in woodlands that were valued at \$302 ha^{-1} per year, 30% higher than rangelands. Thus, while the value of ecosystem services provided by the rangeland category was estimated to have declined by \$12.0 million (65%) between 1976 and 1991, the value of the woodland land cover category was estimated to have increased by \$10.8 million (403%) during the same period (Table 3). The estimates also indicated a small increase (\$0.62 million per year) in the contribution of bare soil to the total ecosystem service value during this same time period.

Our estimated 15-year conversion of 9511 ha of rangeland, woodland and bare soils to urban and suburban land-uses (residential, commercial and transportation) and the associated loss in ecosystem services appeared to have a small net effect on the annual value of ecosystem services provided within the study area. This is a surprising result given that the residential, commercial and transportation land cover categories were assigned no ecosystem service value while the ecosystem services provided by rangelands, woodlands, and bare soils were assigned values of \$232, 302, and 92 ha^{-1} per year, respectively. One explanation for the apparent small net effect of land use conversion on the value of ecosystem services in the study area is that the loss of ecosystem services on land being developed was offset by the apparent conversion of ecologically 'less' valuable bare soil (cropland) and rangelands to ecologically 'more' valuable woodlands. This ecological conversion is consistent with the increase in distribution and density of woody plants throughout much of central Texas (Smeins et al., 1997).

3.3. Ecosystem services sensitivity analyses

The effects of using alternative coefficients to estimate total ecosystem service values in the study area in 1976 and 1991 are shown in Table 4. The CS of these analyses were less than unity in all cases. The estimated value of the ecosystem service value for the study area increased from a low of 0.03–0.06% for 1% increase in the value of the bare soil coefficient, to a high of 0.31–0.85% for a 1% increase in the value of the rangeland coefficient. This indicates that the total ecosystem values estimated for the study area are relatively inelastic with respect to the ecosystem service coefficients. While this implies that our estimates were robust, highly under or over valued coefficients can substantially affect the veracity of estimated changes in ecosystem service values over time even when the CS are less than unity. For this reason, we also report on the effect of large variations in coefficient values on the estimated value of land-use related changes in ecosystems services.

Table 3
Total ecosystem service values (ESV in US$ × 10^6 per year) estimated for each land cover category in the study area using Costanza et al. coefficients, and the overall change and rate of change between 1976 and 1991

Land-use category	ESV (US$ × 10^6 per year)			Difference in ESV between the first and last year of each time period and the annual change rate that this difference represents								
	1976	1985	1991	1976–1985			1985–1991			1976–1991		
				$ × 10^6	%	% per year	$ × 10^6	%	% per year	$ × 10^6	%	% per year
Rangeland	18.68	13.72	6.47	−4.96	−27	−3	−7.25	−53	−12	−12.20	−65	−7
Woodland	2.68	7.65	13.49	4.97	185	12	5.83	76	10	10.80	403	11
Bare soil	0.58	1.24	1.20	0.66	113	9	−0.04	−3	−1	0.62	105	5
Urban categories[a]	0.00	0.00	0.00	0.00	–	–	0.00	–	–	–	–	–
Total	21.94	22.61	21.16	0.67	3	0	−1.45	−6	−1	−0.78	−4	0

[a] Includes residential, commercial and transportation land cover categories.

When the rangelands coefficient was increased by 50%, the ecosystem services in the study area decreased in value by 22.0% (−$6.89 million per year) between 1976 and 1991, which was a substantially greater decrease than the initial 4% decline. A 50% decrease in the rangeland coefficient resulted in a 43.6% ($5.62 million per year) gain in ecosystem services during the same time period. Increasing or decreasing the rangeland coefficient by 50% affected the estimated 1976 ecosystem service value more (±42.6%) than the 1991 value (±15.3%). Conversely, when the coefficient for woodlands was increased by 50% the value of ecosystem services grew by 19.8% ($4.62 million per year) between 1976 and 1991, but a 50% decrease in the woodland coefficient led to a 28.10% (−$5.88 million per year) decrease in value. In contrast to rangelands, increasing the woodland coefficient by 50% affected the estimated 1976 ecosystem service value less (±6.1%) than the 1991 value (±31.9%). These differences between land cover types can be accounted for by the decrease in size of the ecologically 'less valuable' rangelands between 1976 and 1991 and a concomitant increase in size of the 'more valuable' woodlands.

Due to the uncertainty about the relative ecological value of rangelands and woodlands, perhaps most relevant was the result of the sensitivity analysis in which the woodland coefficient was equated to that of rangelands ($232 ha^{-1} per year), the underlying assumption being that the transformation of rangelands to woodlands results in no overall change in ecosystem services. In this scenario, the annual value of ecosystem services declined by 15.4% ($3.29 million per year in the 141 671 ha of our study area, i.e. $23.22 ha^{-1} per year) between 1976 and 1991. This is a substantially greater loss than the originally estimated 4% ($5.58 ha^{-1} per year) decline in annual ecosystem service values. Assuming a straight-line decline, this would equate to a cumulative loss of $26.32 million over the 15-year period of our study compared with the estimated cumulative loss of $6.24 million using the original coefficient values.

Increasing the value coefficient of bare soils by 50% resulted in a reduced loss of annual ecosystem services from 4 to 2.1% between 1976 and 1991, while decreasing the coefficient by 50% resulted in an increased loss of 5%. These comparatively small changes were mainly because of the small size of this land cover category.

4. Discussion

Remote sensing from satellites may be the only economically feasible way to regularly gather information with high spatial, spectral, and temporal

Table 4
Estimated total ecosystem service values (ESV in US$×10^6) in the San Antonio study area, after adjusting ecosystem service valuation coefficients (VC), the magnitude of changes in the ESV following the adjustments, and the coefficient of sensitivity (CS) associated with these adjustments

Change in valuation coefficient (VC)	ESV[a]		1976–1991 change		Effect of changing CV from original value[a]			
	1976	1991	$×10^6	%	1976		1991	
					%	CS	%	CS
Rangeland VC+50%	31.28	24.39	−6.89	−22.0	42.6	0.85	15.3	0.31
Rangeland VC−50%	12.61	17.92	5.62	43.6	−42.6		−15.3	
Woodland VC+50%	23.28	27.90	4.62	19.8	6.1	0.12	31.9	0.64
Woodland VC−50%	20.60	14.41	−5.88	−28.1	−6.1		−31.9	
Woodland VC=Rangeland VC	21.32	18.03	−3.29	−15.4	−2.8		−14.8	
Bare soil VC+50%	22.24	21.76	−0.48	−2.1	1.3	0.03	2.8	0.06
Bare soil VC−50%	21.65	20.56	1.09	−5.0	−1.3		−2.8	

[a] Total ecosystem service values before adjusting the ecosystem service coefficients were: $21.94 million and $21.16 million in 1976 and 1991, respectively.

resolution over large areas (Verstraete et al., 1996). This advantage will increase as the cost of obtaining such data declines and computational power to cope with larger data sets from higher resolution sensors increases. However, one limitation for conducting time series analyses of land-use changes using remotely sensed data is that satellite data from high-resolution detectors have a relatively short history. Even LANDSAT data cannot be used for analyzing land-use changes prior to 1972. Due to the lack of recorded historical land cover data, one further limitation of using remotely sensed data to determine changes in land-use patterns is the difficulty of measuring uncertainty about land-use classification. Congalton and Green (1999) state "To date, no standard accuracy assessment technique for change detection has been developed". They propose the use of an error matrix, but this assumes that some reference data exist. Without such reference data, the investigator is left with two choices: do no analysis, or conduct the analysis with the full knowledge that there is uncertainty associated with historical land use classification. Even with uncertainty, simple classification schemes, such as those used in this study, are reasonable because the distinction between gross land covers is relatively large (Congalton and Green, 1999). Thus, despite the uncertainty of classification, the relatively course spectral resolution of early sensor technology, such as LANDSAT MSS, can be useful for estimating broad scale land-use changes within watersheds.

One challenge for identifying land-use changes was that different land uses might produce similar spectral signatures. For example, it was impossible to determine whether changes in the extent of bare ground were due to changes in the area of cropland, denuded rangeland resulting from drought or overgrazing, or new construction. In Bexar County, most crops are harvested by August, the month in which two of the three images used in the study were taken. This resulted in a lack of distinction between post-harvest cropland and other bare soil surfaces. In addition, areas designated as rangelands in 1976 appear to have converted to woodland in the later images. Such increases in woody plant distributions have been associated with long-term overgrazing in many areas in Texas since the 1880s, as well as fire suppression, which is exacerbated in areas with rapid development (Smeins and Merrill, 1988). Nevertheless, there was some uncertainty about the true extent of the rangeland to woodland conversion. As satellite data with greater spectral and spatial resolution becomes more readily available such land-use ambiguities are likely to decrease.

While Costanza et al.'s ecosystem service values that we used in our analysis have been challenged on theoretical and empirical grounds, they represent the only set of valuation coefficients for a wide array of biomes, each of which encompass several related ecosystems. Also important is the realization that absolutely accurate coefficients are often less critical for time series than cross-sectional analyses because coefficients tend to affect estimates of directional change less than estimates of the magnitude of ecosystem values at specific points in time. We were primarily interested in changes in ecosystem services over time, and the scope of our project precluded us from deriving area-specific ecosystem value coefficients.

Based on the estimated size of six land cover categories and Costanza et al.'s ecosystem services values for related biomes, we determined that the total annual ecosystem service values in Bexar County declined from \$21.94 to 21.16 million per year from 1976 to 1991. Thus, while there appeared to be a 65% decrease in the size of rangelands and a 29% increase in the area of the urban land-use categories, we estimated there to have been only a 4% (\$5.58 ha^{-1} per year) loss in the annual value of ecosystem services. Assuming a straight-line decrease, this represents a \$6.24 million 15-year cumulative loss. This relatively small decline is largely attributable to the fact that the estimated \$12.2 million per year loss in ecosystem services delivered by rangelands was largely offset by the estimated \$10.8 million per year increase in ecosystem services from woodlands. When we assumed that the shift of rangelands to woodlands produced no net change in the value of ecosystem services per hectare, the loss in the estimate annual ecosystem service value between 1976 and 1991 grew to 15.4% (\$23.22 ha^{-1} per year), representing a \$26.32 million 15-year cumulative loss if the decline is linear.

While our study showed that urban sprawl in Bexar County resulted in a decline in the value of ecosystem services delivered by the affected land, it also showed that changes in the value of ecosystem services over time depend on the interaction of changes in various land cover types. Our study suggests that urban spread may not necessarily lead to a large net decline in ecosystem services if there is a concomitant increase in size of other land cover types that provide a greater level of ecosystem services. This is not to say that urban sprawl is beneficial for the delivery of ecological services, but that the negative impacts of the spread of urban and suburban land uses can be potentially offset by other mitigating changes in land cover. This is important because it is unlikely that with increasing human population pressure, the conversion of land to urban and suburban land use will cease or even dissipate in the near future.

Our study showed that LANDSAT data can be used to obtain coarse estimates of changes in ecosystem values at the watershed level. However, in order for this type of analysis to become valuable for policy formulation affecting land-use, it is imperative to obtain a wider array of value coefficients for ecosystem services that more accurately reflect local conditions. This is no easy or costless exercise. One approach could be to identify benchmark ecosystem service values for dominant ecosystem types within a region and then to evaluate the ecosystem services provided at specific locations relative to the representative benchmark. Since ecosystem services are not traded and, therefore, have no market-based price, indirect valuation techniques will be required to obtain these estimates. Such an approach would be analogous to the evaluation procedure developed by Dyksterhuis (1949) for determining the ecological condition of rangelands, and it would allow systematic estimation of changes in the ecological services delivered at specific locations over time. Such information could be extrapolated to larger watershed and regional scales though the use of remotely sensed data and GIS tools to classify land into representative ecosystems for which benchmark values have been established.

Tools for this sort of analysis are becoming increasingly available in common GIS systems. While we limited the use of GIS tools to quantifying land-use change, such tools can also be used to conduct a variety of additional analysis because the data are spatially explicit. Ecological economists could incorporate variables such as patch size, edge effect, contiguity, wildlife corridors, biodiversity potential, to name a few, in their analyses to better evaluate changes in land-use from an ecosystem service perspective. For example, in future watershed level studies, quantifying the location and sequence of different land cover types relative to watercourses could be used to facilitate ecosystem value estimates. Thus, increasingly sophisticated GIS tools could be used in conjunction with ecosystem benchmarking to efficiently and accurately estimate to impact of land use conversion on the services provided by the affected ecosystems.

From a policy standpoint it is also critical that the factors facilitating urban sprawl be clearly understood and that rural landowners be provided with incentives that minimize the negative impacts from such changing land use. For example, weak local government, developer pressure, high estate taxes for rural landowners, and greater urban affluence are major causes of rapid rural land fragmentation and urban sprawl west of San Antonio and Austin. Unless landowners have positive incentives to sustain rural land-uses, urban sprawl will increasingly impact ecosystem services in these areas even when we are able to accurately quantify the loss of ecosystem services associated with such changes in land-use.

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Does Location Matter?
Measuring Environmental Perceptions of Creeks in Two San Antonio Watersheds

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Abstract

In the past, researchers in the field of environmental psychology have explained environmental perceptions primarily through socioeconomic and demographic factors. However, it is reasonable to assume that knowledge of and support for protecting specific natural features of the landscape should also be influenced by one's location, setting, and proximity to such features. This article focuses on residents' familiarity with and concern for two creeks passing through San Antonio, TX. Using Geographic Information Systems (GIS) analytical techniques, we expand on previous studies by introducing driving distance from the creeks as a variable to precisely identify the effects of this location-based factor on environmental perceptions. Specifically, we test the degree to which the actual driving distance respondents live from two San Antonio, Texas creeks affects respondents' knowledge and perceptions of the natural resource. We show that when controlling for socioeconomic and geographic contextual variables, residential distance from the creeks remains a significant factor in explaining both familiarity with them and views on extent of water pollution in them. Based on the results, we discuss the implications of incorporating proximity factors in watershed planning and policy.

Introduction

The field of environmental psychology has a long tradition of explaining the factors influencing environmental attitudes and perceptions. These descriptive and explanatory studies usually pinpoint socio-demographic factors, such as party identification, age, income, and education as the drivers of familiarity (in this paper considered synonymous with awareness) with and concern for the natural environment. However, knowledge of and support for protecting even general features of the environment can also be influenced by location, place, and space. Proximity and exposure to natural features, such as wildlife habitat or water bodies, may be important factors in forming an individual's understanding and views toward maintaining the quality of the surrounding natural environment.

This article expands on previous conceptual models stemming from the environmental psychology and behavior literatures that rely primarily on socioeconomic and demographic factors to predict how the public perceives the natural environment by adding a spatial dimension. The study focuses on residents' familiarity with and concern for two creeks passing through San Antonio, TX. Salado and Leon Creeks stretch from Northern Bexar County southeast to the confluences of the San Antonio River and Medina River, respectively (Figure 1). Salado Creek runs for a total of 44 miles through the eastern portion of the city, while Leon Creek flows for approximately 57 miles through the western portions. Both watercourses traverse a variety of land uses, ranging from rural/agricultural to urban/commercial. Currently, the U.S. Environmental Protection Agency (EPA) recognizes both creeks as "impaired waters" because of their high levels of pollution.

Based on the results of a survey of residents in two San Antonio watersheds, we empirically test the degree to which the distance respondents live from creeks affects their knowledge and perceptions of the natural resource. We draw from a variety of previous distance-based studies demonstrating a relationship between general proximity and environmental awareness. Using Geographic Information Systems (GIS) analytical techniques, we introduce geographic variables, such as driving distance, land use, and land cover to precisely identify the effects of location-based factors on environmental perceptions. Results indicate that when controls are introduced for socioeconomic and geographic contextual variables, driving distance of residence remains a significant factor in explaining whether or not respondents are familiar with Leon and Salado Creeks, as well as their views on the safety of these watercourses for drinking, swimming, and other activities.

This study makes several important contributions to the literature on environmental psychology and understanding environmental perceptions. First, including proximity-based variables along with socioeconomic and demographic variables highlights the importance of location in explaining environmental perceptions and improves upon past environmental psychology models. Second, using GIS analytical techniques provides a high level of precision when measuring distance and setting that has rarely, if ever been used in the literature on environmental perceptions. We employ network analysis programming to determine the driving distance from a respondents' home to a creek taking into consideration the road network for San Antonio. Third, conducting multivariate analysis along with precise proximity measurements enables us to empirically test a longstanding conception that residents living closer to a nature resource will be more familiar and possibly more concerned with its environmental quality. Using quantitative analysis enables us to go beyond simply saying that location matters by demonstrating that on average for every unit increase in kilometers from one of the creeks, the likelihood that a person will view them as being more polluted decreases by a specific amount while controlling for income, education, party identification, and other factors. Finally, this study increases understanding of how residents in and around San Antonio perceive key water bodies. While the results of the study are generalizable to other regions, they provide important information on which factors cause an average resident to be familiar with or concerned about the water quality of Leon and Salado Creeks. This information has important ramifications for state or city environmental planners and policy makers focused on water management issues in the San Antonio region.

The following section reviews the past literature explaining environmental attitudes and perceptions primarily through socioeconomic variables. We then relate this research to place-based issues and the need to consider geographic factors when understanding individual views on the environment. In this section, we build on previous research using proximity to explain environmental views and preferences. Next, we describe sample selection, variable measurement using Geographic Information Systems (GIS), and data analysis procedures. Our findings are reported in two sections. First, we examine factors influencing familiarity with the two creeks using logit models. Second, we test the degree to which distance from these creeks affects views on water quality through multiple regression analysis. Based on the results, we discuss the policy implications of the importance of distance and other location-based factors in understanding environmental perceptions, particularly within the context of watershed planning.

Explaining Environmental Perceptions Through Socioeconomic Variables

Traditionally, researchers in the field of environmental psychology have relied upon socioeconomic and demographic variables, such as age, education, income, political orientation, and occupation to explain broad scale environmental perceptions such as attitudes, views, awareness, and concerns (Buttel, 1987). Generally, these studies conclude that young women with high levels of income and education and with liberal political views are the most likely to consider environmental protection a priority.

For example, in their summary of over a decade of previous research, Van Liere and Dunlap (1980) found that, "age, education and political ideology are consistently (albeit moderately) associated with environmental concern, and thus we have confidence in concluding that younger, well-educated, and politically liberal persons tend to be more concerned about environmental quality than their older, less educated and politically conservative counterparts" (p.192). Jones and Dunlap (1992) and Scott and Willets (1994) found the same results: young, highly educated, liberal-minded individuals demonstrate greater recognition of and concern for environmental problems. Other, more recent studies focusing on the role of socioeconomic factors find evidence that younger age (Fransson and Garling, 1999; Honnold, 1981; Nord, et al., 1998) and higher levels of education (Guagano and Markee, 1995; Howell and Laska, 1992; Raudsepp, 2001) are significant drivers of environmental attitudes and concern.

While not as pronounced in the literature as other socioeconomic factors, income is another variable shown to explain environmental perceptions and attitudes (Fransson and Garling, 1999; Van Liere and Dunlap, 1980). For example, Scott and Willets (1994) found that respondents with higher income levels were more likely to demonstrate pro-environmental concerns. Gender is also a variable that receives consistent attention by environmental psychology researchers. Raudsepp (2001) found women were "significantly more likely than men to be concerned with environmental problems" (p.363). Most research finds slight evidence that women are more environmentally concerned (Jones and Dunlap, 1992) or possess stronger environmental attitudes than men (Foster and McBeth, 1994). However, as acknowledged by Van Liere and Dunlap (1980), gender does not appear to be as significant a predictor of environmental concerns or attitudes as other socio-demographic variables.

Over two decades of research using socioeconomic and demographic variables to explain environmental perceptions have advanced our understanding of how people view, think about, and are aware of the natural environment (Samdahl and Robertson, 1989). However, the authors of these studies are quick to point out that socio-demographic variables alone are insufficient in their explanatory capabilities (Samdahl and Robertson, 1989). As stated by Van Liere (1980) in reference to years of research on the topic, “the foregoing review indicates that researchers have had limited success in explaining the social bases of environmental concern.” In a review of the socio-demographic correlates of environmentalism, Cantrill and Senecah (2001) concluded, “contradictory findings such as these indicate that perceptual processes beyond the mitigating influences of socio-demographic factors may drive perceptions of the environment” (p. 188). Statistical evidence of the difficulty in thoroughly explaining environmental perceptions can be found in the relative size of R^2 values and the overall amount of unexplained variances in models. Lowe and Pinhey’s (1992) research covering sixteen separate years of support for spending to protect the environment (categorized here as environmental attitudes), reports R^2 values ranging from 0.067 to 0.130 (all significant to $p < 0.001$). While it is certain that socio-demographic variables play an important role in shaping attitudes toward the environment, it is also clear that other factors also must be considered to more fully understand what shapes environmental perceptions, attitudes, and concerns.

Considering Proximity, Location, and Place

A second stream of literature measures of location, proximity, and setting to examine environmental perceptions. Work that emerged in the 1920s under the heading of human ecology was extremely varied in approach and emphasis but did consistently argue that human behavior was significantly influenced by spatial and geographic factors (McKenzie, 1925; Hawley, 1944). Research that emerged from this approach mapped phenomena such as crime (Longmoor and Young, 1930;Duffala, 1976; Farley and Hansel, 1981), mental disorders (Queen, 1940), and family disruption (Lind, 1930), correlating these with the proximity to amenities such as recreational facilities, good housing, and to declining city centers. Increasing sensitivity to environmental issues and the emergence of environmental sociology in the 1970s continued a line of research that focused on the correlates of environmental attitudes and values (Buttell, 1987). Proximity has played a small role in this research. However, the idea that living in a place or near particular environmental features influences knowledge and perception is central to the current emphasis on local knowledge as a legitimate source of useable information in the development of policy and solutions to environmental problems.

Recently, researchers have begun to consider geographic factors to explain the underpinnings of environmental perceptions in addition to the traditional socio demographic variables. The environmental psychology literature lays a strong foundation for understanding place-based perceptions by examining the difference in views among urban and rural residents, for example. Tremblay and Dunlap (1978) found that rural residents were less concerned with environmental problems than those living in urban settings, and that rural farmers were particularly uninterested in protecting the environment. Lowe and Pinhey (1982) confirmed these rural anti-environmental conclusions in a national study focusing on a respondent's place of socialization. These findings are consistent with other research indicating that those whose livelihood is based on extractive activities are less likely to be concerned with environmental conservation (Freudenburg, 1991). More recent empirical research disputes the rural anti-environment hypothesis and instead suggests increasing environmental concern in non-metropolitan areas (Alm and Witt, 1994; Fortmann and Kusel, 1990). For example, Foster and McBeth (1994) demonstrate that rural residents are more likely to be concerned with local environmental issues when they are defined in terms of quality of life features. In addition to general place of residence, the length of stay, or "tenure" has also been linked to building value-based awareness and connections with the physical environment (Cantrill, 1998).

Hannon (1994) and Norton and Hannon (1997) are among the first to directly link environmental attitudes to location and distance. These authors propose that the intensity of environmental valuation is discounted across time and space. In other words, proximity factors play a critical role in determining how individuals view physical place. Similarly, in a government survey of resident's environmental views of the South Shore of Long Island, it was found that general proximity to the waterfront is associated with greater interest in and support for protecting the estuary. Respondents living closer to the water assigned greater importance to the quality of the South Shore natural environment (Cornell Local Government Program, 1998). Gobster (1998) found that neighborhood residents living near the Chicago River were generally more aware of the water body and chiefly concerned with its water quality.

Brown et al. (2002) used straight-line distance to test a place-based theory of environmental evaluation. These authors admitted their measures of distance could be confounded by physical barriers between points of study, but their results showed "moderate support for the theory that community place attachment is related to distance and intensity of environmental valuation" (p 70). Examinations of the importance of proximity also include research into attitudes toward and decisions about environmental risk. For example, Gawande and Jenkins-Smith (2001) found that distance from transportation routes for nuclear waste drove perception of risk and influenced property values. Elliot, et al. (1999) found that proximity to adverse air quality locations affected community cohesiveness over air pollution issues, and Bush et al. (2001) found that in their study sample, residents of communities closest to industrial facilities expressed the strongest need for adequate information on air quality. In addition, literature on recreational site choice includes various studies focusing on the relationship between general proximity and awareness of metropolitan trails (Gobster, 1995), park use (Gobster, 2002), urban wildlife (Gilbert, 1982), and riparian landscapes (Zube et al., 1998).

Modeling Environmental Perceptions Through Distance

In most cases, measures of nearness or proximity in the literature have lacked precision. In order to test the importance of proximity, consistent and reliable measure of the variable are required. We draw upon geographic methods of analyzing distance variables to expand on past studies of association between distance from environmental elements and environmental perceptions (i.e. attitudes and concerns). Specifically, we build on previous findings to test the hypothesis that:

the driving distance from a respondent's residence to either Salado or Leon Creeks impacts the level of awareness of the creek's existence and perceptions of its water quality.

We use two suites of variables to control for the effects of driving distance on 1) familiarity (or awareness) with the creeks, and 2) the degree to which respondents believe the watercourses are polluted or unsafe to use (environmental concern). First, we employ a series of socioeconomic contextual control variables selected on the basis of previous studies that modeled environmental perception. These include party identification (PID), age, education, income, gender, population density, and environmental views. Second, we add geographic variables to better account for place and setting when isolating the influence of distance on the dependent variable. Since residency variables are meant to capture place and setting issues, we advance this approach by directly measuring land cover and land use using GIS analysis of remote sensing data. Land cover types include forest, rangeland, agriculture, and built-out. Land use is defined as urban, rural, and residential. This conceptual model not only helps explain the influence of place-based factors on environmental awareness and concern, but also provides a more complete approach to understanding environmental perceptions in general.

(Insert Figure 1 about here)

Research Methods and Data Analysis

Sample Selection

We selected the sample of respondents from a random household telephone survey of residents in San Antonio, Texas that over-sampled the area within the two major watersheds for Salado and Leon Creeks. The sample was stratified into three groups: Salado Creek watershed, Leon Creek watershed, and Bexar County as a whole. In order to make sure the households were in the targeted areas, we used only listed numbers that have addresses. This approach made certain that the correct strata could be determined for every household. A random selection of 4,000 listed households within each stratum was conducted. A sample of 2,400 households was sampled from the Bexar County and 800 from each of the over-sampled areas. Telephone interviews took an average of 19.5 minutes; the shortest took 10 minutes and four took over an hour. 7.3% of the initial sample required return phone calls. The overall response rate was 25.4%, which generated a sample of 1017 for analysis. Of the 1,017 respondents selected, 1,005 were geocoded (placed in their true location on earth using X and Y coordinates) by tying their reported addresses to a 2000 U.S. Census Bureau TIGER line file. Once each respondent was located in space, we could effectively employ geographic factors to examine environmental perceptions within the study area.

Measurement of Variables

Respondents' perceptions of the environment, the dependent variable for the study, were measured through the survey in two formats (Appendix A). First, familiarity with Salado and Leon Creeks was measured as a dichotomous "yes" or "no" variable. Second, environmental concern was measured based on views of the creeks' safety for drinking, swimming, consumption of fish, and drinking for livestock were measured on a scale from 1-4, where 1 is very safe and 4 very unsafe.[i] These two measures of environmental perception enabled us to be systematic in our investigation by examining the dependent variable in increasing levels of detail.

Socioeconomic contextual variables were measured primarily through the survey instrument. PID, age, income, education, and gender were registered based on the methods used most widely throughout the environmental psychology the literature (see Appendix A for more detail). Environmental views were measured based on questions initially used by Van Liere and Dunlap (1980). Responses were summed and ranged from 1 (strongly agreeing humans are abusing the natural environment) to 16 (strongly disagreeing that humans are abusing the environment).[ii] Finally, population density was measured using GIS along with census data to determine the population per square mile.

The majority of geographic variables in the model were measured through GIS analysis techniques. Using GIS to analyze the data enabled us to derive more accurate measurements of geographic factors than rough approximations of distance or general land use settings. Driving distance was measured by tying the geocoded survey respondents to a 2000 U.S. Census Bureau TIGER line file product, which contains the road network for San Antonio.[iii] Through the program Network Analysis, we determined the shortest driving distance in meters from a respondent's residence to the nearest intersection with Salado and Leon Creeks. In other words, based on the street network of San Antonio, the computer program was able to find the driving distance from each geocoded resident to the closest intersection of each creek.

Land cover for each respondent's location was measured using the Texas Natural Resource Conservation Commission (TNRCC) Land Use/Land Cover GIS coverage.[iv] From this data layer, we formed the following four major land cover categories: forest, agriculture, built, and rangeland (the baseline category). Using actual zoning and land use information, we were also able to interpret land use for each respondent using three categories: urban, rural, and residential (the baseline category). Each respondent fell into only one land cover and land class category. All land use and land cover categories were entered into the model as dichotomous variables.

Data Analysis

Analysis of the data proceeded in two phases. First, the dichotomous familiarity variable was examined using a logit model for each creek. From these results, we calculated predicted probabilities of minimum/maximum distance on familiarity in order to understand the effect of distance in more detail (see Table 3). Second, the water safety variable was analyzed for each creek using Ordinary Least Squares (OLS) multiple regression to explain the impact of driving distance on environmental perceptions. We paid close attention to adjusted r-squared values to see if the addition of geographic variables improved model fit over previous research relying on socioeconomic and demographic factors alone. Several statistical tests for reliability were conducted to ensure the OLS estimators were Best Linear Unbiased Estimates (BLUE). Tests for model specification, multicollinearity (particularly between built and urban variables), heteroskedasticity, and autocorrelation revealed no violation of regression assumptions.

Results and Analysis

The first phase of analysis used logit models to examine the relationship between respondent distance from creeks and familiarity with the creeks in question. We used familiarity or recognition that a natural resource exists as an initial step in investigating overall environmental perceptions. As illustrated in Table 1, driving distance to Salado Creek is a significant factor in explaining whether residents are familiar with its existence ($p < .01$). As the residential distance from Salado Creek increases, familiarity with the creek decreases. That is, respondents are significantly more likely to be aware of the creek the closer they live to it. Education is also a significant factor in explaining familiarity with Salado Creek ($p < .01$). Respondents with higher levels of education (based on the amount of formal schooling they have completed) are more likely to be familiar with the creek. A higher level of income is another socioeconomic factor associated with increased familiarity with the creek ($p < .01$). In terms of gender effects, our study found that males are significantly more likely to be familiar with the creek ($p < .05$).

(Insert Table 1 about here)

Overall, tenure, or length of residence, is the most powerful predictor of a respondent's awareness of the creek's presence. The longer a resident lives in San Antonio, the more familiar that person is likely to be with the creek. While there is little research or data on the long-term effects of tenure on environmental perceptions, this finding may support the argument that environmental familiarity is enhanced when individuals have the opportunity (through either time or location) to increase their knowledge of place and the natural surroundings (Cantrill, 1998). Land use or land cover does not play a significant role in predicting familiarity with Salado Creek. We believe that these geographic variables are dictated largely by increasing distance from the city center where the creeks are situated. The presence of human constructed, or built elements, forested tracts, or agricultural lands are functions of distance. As one moves away from the urban core into the surrounding suburbs and less populated areas, forested and agricultural areas become more common.

Results regarding familiarity with Leon Creek are, for the most part, identical to the results for Salado Creek, with the exception that education is not a statistically significant factor (Table 2). On average, men with higher incomes living close to Leon Creek for longer periods of time are more familiar with the water body. It is important to note, however, that the effect of driving distance from Leon Creek is almost double that of Salado Creek. Residents living farther away from Leon Creek are far less likely to be aware of its presence. This result may be due in part to the higher profile of Salado Creek throughout the San Antonio region, but more likely it is because residents in the sample live farther away from Leon Creek and therefore are less likely to be familiar with it. The average driving distance from Leon is approximately 12.5 kilometers compared to 9.6 km for Salado Creek. Most residents live to the west of the Leon Creek in suburban neighborhoods. This contrast in proximity and location seems to account for the difference in the effect of driving distance on the dependent variable for Leon and Salado Creeks.

(Insert Table 2 about here)

To better understand the impact of distance on environmental perceptions, we calculated the predicted probabilities that respondents would be familiar with the two creeks for minimum and maximum distances (Table 3). The probability that a respondent living at the minimum distance from Salado Creek will be familiar with it, holding all other variables in the model constant, is .93 compared to .73 for Leon Creek. At the maximum driving distance, the probability of a respondent being familiar with Salado Creek is .42, while it is only .08 for Leon Creek. Table 3 reveals that, overall, respondents are more familiar with Salado Creek in part because they live closer to the watercourse. Those living at the outreaches of the sampling area are, in contrast, unlikely to be aware of Leon Creek. The difference in the impact of distance on creek awareness appears in the size of the coefficients in the logit models (Tables 1 and 2).

(Insert Table 3 about here)

Familiarity with or awareness of a natural resource is only part of understanding environmental perceptions. In the second phase of analysis, we probed further in our investigation of perception by examining residents' levels of environmental concern based on how they perceived a creek's overall state of health. The same independent variables used in the logit models were analyzed again to explain level of concern over the safety of a creek's water. Opinions on the safety of the creek's water for drinking, swimming, consuming fish from the creek's waters, and water consumption by livestock indicated the degree to which respondents believed them to be polluted.

Driving distance again is a significant factor in predicting perceptions of the safety of a creek's water. Residents living closer to Salado Creek are more likely to believe it is unsafe for human use and consumption by livestock ($p < .01$) (Table 4). The impact of contextual variables is much the same as with the logit models. Men with high incomes living close to the creek for long periods of time were more likely to consider the watercourse to be polluted. However, in the multiple regression model, population density is statistically significant at the .05 level. Respondents in areas with higher population levels believe Salado Creek is unsafe compared to more sparsely populated locations. This result is expected since a large urban cluster of respondents lives in close proximity to the creek. Finally, respondents with stronger views on protecting the environment are significantly more likely to believe the creek is polluted.

(Insert Table 4 about here)

Multiple regression results for Leon Creek parallel those for Salado (Table 5). Again, males with high incomes living close to the creek for longer periods of time are significantly more likely to respond that the creek is polluted. However, population density is not a strong factor in this model. This result is consistent with the demographic characteristics of the San Antonio area, where most residents near Leon Creek are located to the west in suburban neighborhoods with relatively lower population densities.

When looking at the results as a whole, we find that residents who are more familiar with the creeks (i.e. those living closer to them) are significantly more likely to believe the water is polluted. A larger percentage of respondents live near Salado Creek, and correspondingly, respondents believe that Salado Creek is more polluted than Leon Creek. These perceptions are consistent with objective indicators of the health of the creeks. Data show that, historically, both creeks have been polluted, which helps explain the strong statistical relationship between tenure and views on water safety. Based on a recent study, however, the Texas Natural Resource Conservation Commission (TNRCC) found significantly greater portions of Salado Creek (over four times) contain high bacterial levels and other water borne pollutants compared to Leon Creek (State of Texas List of Impaired Water Bodies 303(d), 1996, 1998, 1999, 2000).

Proximity, in this instance, not only influences environmental perceptions and general sense of place, but is also associated with more accurate information on the health of the surrounding natural environment. Thus, location does not simply drive perception, but may help residents of a community understand its environmental realities.

(Insert Table 5 about here)

An important element of the multiple regression analysis is the issue of model fit. In both regression models, distance accounts for approximately half of the variance explained by all of the independent variables. Thus, when distance is taken out of the equations, adjusted r-squared values are reduced by approximately 50 percent. The significant improvement that distance brings to bear on model fit is one indication that proximity and location play a critical role in predicting environmental perceptions and subsequent behavior.[v] However, it is important to note that while adjusted r-squared values are consistent, and in most cases higher than related studies (the adjusted r-squared values for Salado and Leon creeks are .14 and .08 respectively), there is still a large amount of unexplained variance that needs to be addressed in future studies. Although this study demonstrates the importance of the geographic dimension, it is clear that more work is needed to improve model fit and adequately predict environmental perceptions.

Conclusions and Policy Implications

Based on the results of our study, driving distance has a significant effect on determining familiarity with and views on the safety of Salado and Leon Creeks. Respondents within closer driving distance to the watercourses are far more likely both to be familiar with the creeks and believe they are polluted. In addition, we observe a consistent profile of the San Antonio resident who is aware and knowledgeable about the creeks; generally, males with high incomes, living for long periods of time in close driving proximity to the creeks are more likely to be familiar with the creeks and cognizant of their pollution levels.

The significant impact of proximity when predicting how residents perceive critical natural resources has important implications for policy and public education. If residents are knowledgeable and concerned with a particular creek, they may be more likely to support watershed planning initiatives or efforts to improve water quality. For example, public sector planners who wish to start community-based watershed projects may be more effective if they can identify neighborhoods, based on location, that will take more interest in the issues and therefore be more willing to participate in a planning process. Political candidates who wish to gain support for a particular environmental issue may be more persuasive for their positions if they can use proximity to predict which constituents will be more aware of a critical natural resource or be in favor of cleaning up water pollution in a local creek.
Directing watershed policy at locations that match socioeconomic and demographic profiles of known pockets of potential support, however, should not be the only planning strategy initiated by public officials. In the San Antonio case, designing campaigns for support that target high income, well-educated males in relatively stable neighborhoods could marginalize important sectors of the population. There is general evidence that lower income segments of the population are about as supportive of strong environmental controls as higher income segments (Morrison, 1986; Van Liere and Dunlap, 1980,1981; Mitchell, 1979; Freudenburg, 1991).

Specific locations may provide a starting point for initiatives, but efforts must be made to develop an inclusive socio-economic and demographically broad based environmental planning process that will gain support from a variety of interests.

While this study demonstrates the importance of proximity (defined as potential for access) for awareness and information on an environmental feature, further research is needed to fully understand its impact on environmental perceptions. Case study analysis involving interviews with respondents may provide additional insight into the role of proximity in forming environmental views. Further investigation will be done to search for neighborhoods within the study area that may contain patches of similar responses. Clustered responses would indicate that environmental perceptions are not spatially independent, but related to each other across watersheds or other natural landscapes. Understanding why similar responses occur in specific locations will not only enhance statistical modeling of environmental perceptions, but also have implications for policy development and public participation. Examining how the mosaic of interaction among residents in specific locations or neighborhoods contributes to collective environmental awareness may further increase our ability to explain environmental perceptions in general.

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Appendix

(Insert Appendix)

Table 1:

Explaining Familiarity of Salado Creek

Variable	Coefficient	Standard Error	z-value	Significance
Driving Distance				
To Salado	0.537	0.112	-4.78	0.000
		Socioeconomic Variables		
Population Density	0.000	0.000	-1.06	0.288
Political Identification	0.007	0.066	0.12	0.908
Education	0.083	0.027	3.07	0.002
Tenure	0.003	0.000	8.04	0.000
Age	0.001	0.004	0.40	0.692
Income	0.077	0.019	3.93	0.000
Gender	-0.371	0.160	-2.31	0.021
Environmental Views	-0.021	0.019	-1.10	0.273
		Geographic Variables		
Forest	-0.241	0.376	-0.64	0.521
Agriculture	0.348	0.389	0.89	0.371
Built	-0.077	0.497	-0.16	0.876
Urban	-0.048	0.320	-0.15	0.879
Rural	0.252	0.607	0.42	0.677
Constant	-0.690	0.801	-0.86	0.389

N	=	1005
LR chi2(14)	=	156.50
Prob. > chi2	=	0.000
Log Likelihood	=	-500.476
Pseudo R2	=	0.135

Table 2:

Explaining Familiarity of Leon Creek

Variable	Coefficient	Standard Error	z-value	Significance
Driving Distance				
To Leon	0.672	0.086	-7.78	0.000
Socioeconomic Variables				
Population Density	0.000	0.000	-0.53	0.594
Political Identification	0.079	0.060	1.32	0.186
Education	0.031	0.024	1.29	0.197
Tenure	0.001	0.000	3.79	0.000
Age	0.002	0.004	0.56	0.577
Income	0.041	0.017	2.33	0.020
Gender	-0.585	0.138	-4.23	0.000
Environmental Views	0.022	0.016	1.33	0.182
Geographic Variables				
Forest	- 0.146	0.3456522	-0.42	0.672
Agriculture	0.427	0.3475617	1.23	0.219
Built	-0.318	0.4545263	-0.70	0.483
Urban	0.200	0.3019675	0.66	0.508
Rural	-0.242	0.5536304	-0.44	0.662
Constant	-0.734	0.7077858	-1.04	0.299

Number of Obs. = 1005

LR chi2(14) = 123.15

Prob > chi2 = 0.000

Log likelihood = -622.725

Pseudo R2 = 0.090

Table 3:

Predicted Probabilities of Familiarity With Creeks Based on the Minimum/Maximum Distance (km), Holding All Other Variables Constant

	Minimum Distance	Probability	Maximum Distance	Probability
Leon Creek	.14	.73	38.82	.08
Salado Creek	.03	.93	43.97	.42

Table 4:

Explaining Views of Water Safety In Salado Creek

	Coefficient	Standard Error	t-value	Significance
Driving Distance To Salado	0..147	0.024	-6.14	0.000
Socioeconomic Variables				
Population Density	0.000	0.000	-2.75	0.006
Political Identification	0.028	0.142	0.20	0.840
Education	0.039	0.057	0.69	0.493
Tenure	0.007	0.000	8.50	0.000
Age	-0.003	0.010	-0.33	0.744
Income	0.155	0.042	3.70	0.000
Gender	-0.988	0.331	-2.98	0.003
Environmental Views	-0.094	0.039	-2.38	0.017
Geographic Variables				
Forest	-1.438	0.808	-1.78	0.075
Agriculture	-.350	0.814	-0.43	0.667
Built	-.706	1.081	-0.65	0.514
Urban	-.026	0.719	-0.04	0.970
Rural	.361	1.311	0.28	0.783
Constant	7.841	1.705	4.60	0.000

Number of Obs. = 1005

F-Ratio (14, 990) = 12.67

Prob > F = 0.000

Adj. R-squared = 0.140

Table 5:

Explaining Views of Water Safety In Leon Creek

	Coefficient	Standard Error	t-value	Significance
Driving Distance To Leon	0.126	0.019	-6.52	0.000
Socioeconomic Variables				
Population Density	0.000	0.000	-0.61	0.545
Political Identification	0.180	0.140	1.28	0.200
Education	0.055	0.056	0.98	0.326
Tenure	0.004	0.000	4.66	0.000
Age	-0.006	0.010	-0.60	0.550
Income	0.113	0.041	2.73	0.006
Gender	-1.295	0.328	-3.95	0.000
Environmental Views	-0.014	0.039	0.36	0.720
Geographic Variables				
Forest	-0.823	0.806	-1.02	0.308
Agriculture	0.811	0.806	1.01	0.315
Built	-1.706	1.069	-1.06	0.288
Urban	0.411	0.711	0.58	0.563
Rural	-0.746	1.294	0.58	0.564
_cons	4.182	1.671	2.50	0.012

Number of Obs. = 1005

F-Ratio (14, 990) = 7.52

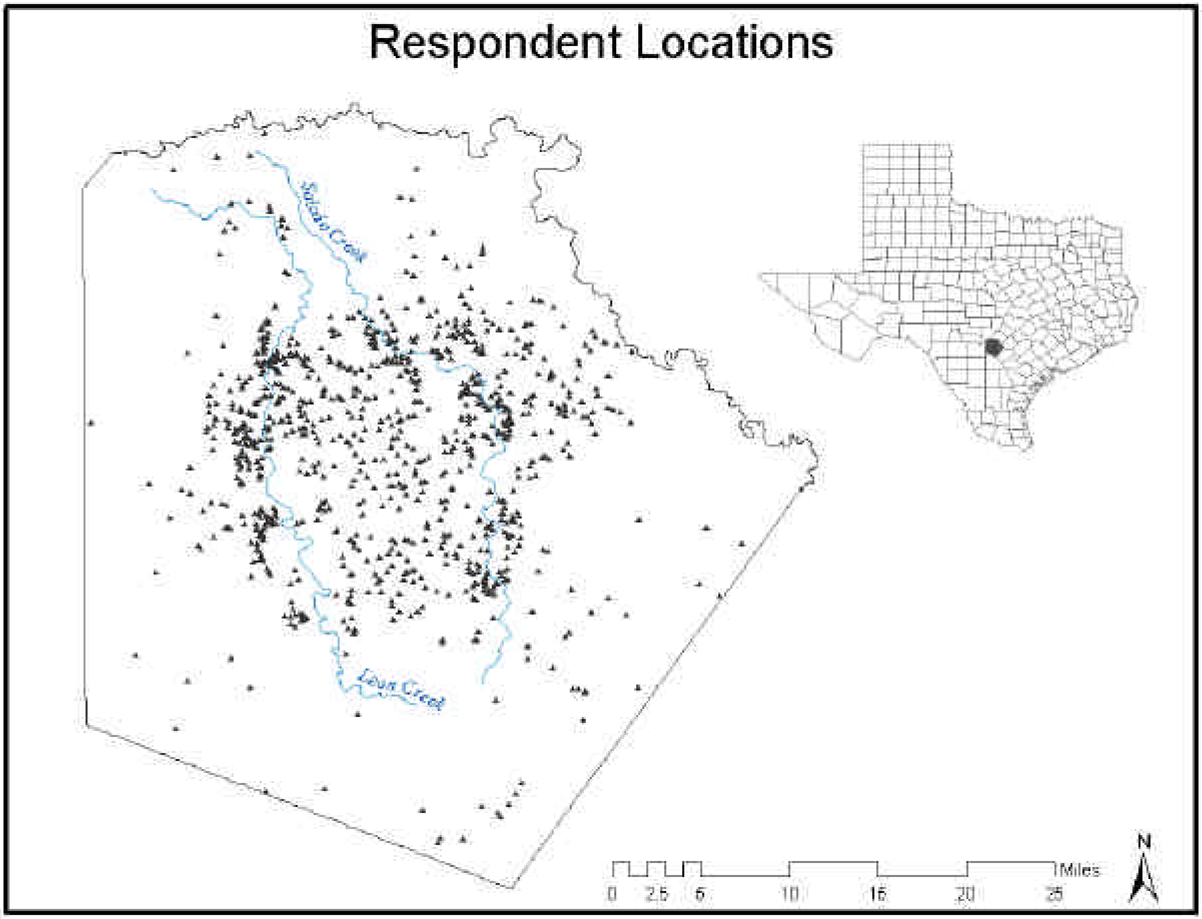
Prob > F = 0.000

Adj. R-squared = 0.083

Appendix A: Concept Measurement

Name	***Type***	***Measurement***	***Scale***	***Source***	***Mean*** *Leon Salado*	***Std.Dev.*** *Leon Salado*
Familiarity	Dependent	Awareness of creeks based on "yes" or "no" answer	Dichotomous; 0-1	Survey	1.61 1.27	.69 .48
Water Safety	Dependent	Views of water pollution in creeks from very safe to very unsafe	1-16	Survey	4.01 7.00	5.31 5.55
Driving Distance (km)	Independent	Distance in meters from residence to nearest intersection with creeks	Continuous	GIS Analysis	12.50 9.58	8.74 7.12
Population Density	Independent	Population per square mile in study area	Continuous	U.S. Census	3520.40	2149.20
Party ID	Independent	Association with specific party	1-4	Survey	2.24	1.19
Education	Independent	Number of school grades completed	Continuous	Survey	14.16	3.16
Age	Independent	Reported age in years	Continuous	Survey	47.24	17.92
Income	Independent	Reported annual income	Continuous	Survey	7.4	4.19
Gender	Independent	Reported gender	Dichotomous	Survey	.529	.429
Environmental Views	Independent	Range of views on the level of human impacts on the natural environment	1-32	Survey	15.19	4.23
Tenure	Independent	Number of months living in San Antonio area	Continuous	Survey	303.67	219.7
Land Cover	Independent	Type of dominant land cover for residence	Dichotomous	GIS Analysis	Forest .18 Range .05 Agriculture .16 Built .56	.39 .23 .37 .47
Land Use	Independent	Type of land use for residence	Dichotomous	GIS Analysis	Urban .09 Rural .40 Residential .50	.29 .49 .50

Figure 1: Respondent Locations



[i] Four separate questions regarding the safety of the Salado and Leon Creeks (for drinking, swimming, eating fish, and drinking for livestock) on a scale from 1-4 were combined into a single variable. This variable was thus measured on a scale of 1 to 16. Cronbach's Alpha for the final scale is .91 and .95 respectively.

[ii] Eight separate questions regarding the degree to which humans are impacting the environment on a scale from 1-4 were combined into a single variable. Cronbach's Alpha for the final scale is .96.

[iii] Previous studies correlating distance with perceptions use Euclidian (straight line) measures. We improved upon these methods using the most recent GIS technology for the following reasons: a) people tend to perceive distance not "as the crow flies" but how they gain access to natural resources, which is usually by automobiles. Driving distance is therefore a more accurate measure since it takes into account a respondent who lives close to a creek, but must drive a comparatively long distance to gain access; b) as noted by a recent study examining the relationship between distance and environmental values (Brown et al., 2002), "barriers" are important issues when considering a person's location in relation to a natural resource. By using driving distance, we take into account urban barriers, such as buildings, watercourses, or neighborhood districts that a respondent would need to traverse to access Salado or Leon Creeks.

[iv] The TNRCC data layer was originally generated through the EPA by interpreting a series of Landsat satellite images into a raster format. This raster-based layer was then converted to a vectors format to analyze with respondent data.

[v] We used a nested model approach to test the overall impact of adding the distance variable. Distance was found to be statistically significant ($F<.01$) for both Salado and Leon Creeks.

Appendix E

SWIMMING UPSTREAM:
COLLABORATIVE APPROACHES TO WATERSHED MANAGEMENT

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Swimming Upstream: Collaborative Approaches to Watershed Management

Manuscript Outline

Part I: Watershed Management Approaches in the U.S.

1 *Collaborative Approaches to Watershed Management*
Paul Sabatier, Will Focht, Mark Lubell, Zev Trachtenberg, Arnold Vedlitz and Marty Matlock

This chapter outlines the policy issues addressed by collaborative watershed approaches, the contributions and deficiencies in the existing literature, and our strategies for developing a new generation of interdisciplinary research on the legitimacy and effectiveness of collaborative approaches.

Part II: Collaborative Engagement Processes in Watersheds

2 *Eras Of Water Management In The U.S.: Implications For Collaborative Watershed Approaches*
Paul Sabatier, Chris Weible and Jared Ficker

Sets the stage for the ones that follow by providing an historical context of several eras of water management institutions found in the U.S. since the late 19th century. The water management eras are characterized by the types of participants involved (public vs. private, federal vs. state and local), the geographic scope of water management (watershed vs. other), the principal substantive purpose (preservation, conservation, development), and the principal water goal (supply vs. quality). The analysis suggests that the collaborative approaches we are studying—while sharing some characteristics with predecessor institutions in different eras—really do represent a rather novel form of public participation and overall governance within water management in the U.S.

3 *Legitimacy and Watershed Collaborations: The Role of Public Participation*
Zev Trachtenberg and Will Focht

This chapter develops a set of criteria that can be applied to watershed decision-making processes to assess their legitimacy. It is particularly concerned with the question of how legitimacy can be enhanced by the participation of a wide range of stakeholders. The criteria address both the procedures by which decisions are made, and the actual impacts of watershed policy on the well-being of watershed residents and others. The criteria are based on a general conception of legitimacy that rests on the fundamental values of personal autonomy, welfare enhancement, and justice. This chapter provides an excellent analysis of the moral issues which permeate collaborative watershed management.

Part II: Collaborative Engagement Processes in Watersheds

4 *A Trust-Based Guide to Stakeholder Participation*
Will Focht and Zev Trachtenberg

Sets the context for much of the analysis in Part II by hypothesizing that the selection of stakeholder participation methods should be based upon contextual factors, particularly the level of trust among policy actors. Trust is conceptualized as the willingness of stakeholders to defer to the competence and discretion of others, specifically fellow "private" stakeholders and "official" policy actors (government decision-makers and subject matter experts), to manage risk on their behalf. Assigning high/low scores to these two trust dimensions produces four decision contexts, for which tailored stakeholder involvement strategies are prescribed. For example, when stakeholders trust both public officials and each other, information exchange is recommended as an appropriate strategy. However, when stakeholders distrust both officials and each other, a deliberative strategy (e.g., policy dialogue) is recommended as more appropriate. The authors test these hypotheses using data obtained from face-to-face interviews with 144 stakeholders residing in the Illinois River Basin in Eastern Oklahoma and find strong empirical support.

5 *Citizen Participation and Representation in Collaborative Engagement Processes*
Charles D. Samuelson, Arnold Vedlitz, Guy Whitten, Marty Matlock, Letitia Alston, Tarla Rai Peterson and Susan Gilbertz

Long-term, collaborative stakeholder participation supposedly enhances the legitimacy of watershed management decision-making due to the complexity of decisions and the competition among various actors involved. Such prescriptions beg the question, however, of whether citizens actually are willing to engage in participatory activities of this type. This chapter systematically examines the correlates of the willingness of stakeholders in San Antonio, Texas to participate in a three-year collaborative process to learn about impacts on several watersheds, to frame policy analyses, and to deliberate on impact management policies. Utilizing a public opinion survey of more than a thousand citizen respondents in the city of San Antonio, three types of correlates are examined—general outlook variables (efficacy, trust, optimism); specific environmental attitudes (concern about resources, environmental problems, human-nature interface); and SES/demographic factors. The subsequent discussion focuses on questions of participant representativeness, the policy efficacy of these processes, and the relevance of these processes for legitimate, democratic decision-making.

Part III: Measuring and Explaining the Success of Watershed Partnerships

6 *Frameworks Explaining Partnership Success*
Paul Sabatier, William Leach, Mark Lubell, and Neil Pelkey

This chapter lays out the basic principles from five general theoretical frameworks of collective decision-making, and then suggests several hypotheses specifically related to participants' willingness to engage in watershed partnerships and to the success of those partnerships. The frameworks are (a) Institutional Analysis and Development (Ostrom, 1990; Ostrom et al, 1993); (b) Transaction Cost Economics (Williamson, 1975; Lubell et al, 1998); (c) Social Capital (Putnam, 1993; 2000); (d) Advocacy Coalition Framework (Sabatier and Jenkins-Smith, 1993; 1999); and (e) Alternative Dispute Resolution (Bingham, 1986; Carpenter and Kennedy, 1988; Susskind et al, 1999). The analysis reveals important similarities across IAD and TCE, on the one hand, and ACF and ADR, on the other. And all the frameworks use the concept of trust, the key variable in Social Capital.

7 *Do Watershed Partnerships Enhance Beliefs Conducive to Collective Action?*
Mark Lubell

This chapter examines whether or not watershed partnerships change stakeholder beliefs in ways that facilitate cooperation. To test hypotheses drawn from the IAD and ACF frameworks discussed in Chapter 7, the author compares stakeholder beliefs in 20 estuaries that were part of the National Estuary Program with 10 estuaries that were not part of the NEP. The analysis shows that NEP stakeholders have beliefs more conducive to cooperation and are less likely to disagree about the process of decision-making in watersheds.

8 *Are Trust and Social Capital the Keys to Success? Watershed Partnerships in California and Washington*
William D. Leach and Paul A. Sabatier

This chapter first develops a set of six indicators for measuring the success of watershed partnerships, and then uses data from random samples of 44 partnerships in California and Washington to operationalize success and to test hypotheses from several of the frameworks in Chapter 6 regarding the factors affecting success. The analysis indicates trust is by far the single most consistent predictor of success, but that several of the variables hypothesized as critical in the existing literature (e.g. presence of a facilitator, amount of grant funds) are not important—once other factors have been controlled for.

Part IV: Conclusions

9 *Conclusions and Recommendations*
Mark Lubell, Paul Sabatier, Arnold Vedlitz, Will Focht, Zev Trachtenberg and Marty Matlock

This chapter reviews important conclusions from throughout the book for both practitioners and academics.

SWIMMING UPSTREAM:
COLLABORATIVE APPROACHES TO WATERSHED MANAGEMENT

OUTLINE

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SWIMMING UPSTREAM:
COLLABORATIVE APPROACHES TO WATERSHED MANAGEMENT

A quiet revolution is occurring in water management institutions in the U.S. (John, 1994; Kenney, 1999; Born and Grenshaw, 1999; 2001).

For the past century, water quality, flood control, fish and wildlife habitat, and, to a lesser extent, water supply have been managed by single-function federal and state agencies, each pursing its own legal mandate. For the most part, decision-making has been quite technocratic, with public involvement usually relegated to public hearings and comment periods that serve to fine-tune agency proposals. The scope of decision-making has usually been specific types of pollution sources or specific areas within a watershed (such as the coastal wetlands), rather than the watershed as a whole.

A New Approach to Water Management

Over the past 10-15 years, however, a new approach to water management has been emerging. Its scope has been explicitly focused on all sources of a pollutant within the watershed as a whole (or distinct subwatersheds), rather than on types of sources (e.g. point sources) or on the rather arbitrary political boundaries represented by states and counties. The new decision-making process has been collaborative, meaning that it involves face-to-face negotiations among a variety of stakeholders—including federal, state, and local agencies, landowners, environmentalists, industries, researchers—involving relatively consensual decision rules. This is in contrast to the traditional approach in which each agency jealously guarded its decision-making prerogatives, with private stakeholders and other agencies largely confined to commenting on proposed regulations. The new approach has sought to find win-win solutions to an interrelated set of social, economic, and environmental issues confronting the watershed, rather than have each agency single-mindedly pursue its legal mandate.[1] The end result is that a collaborative process

[1]The New Webster's Dictionary of the English Language (1981) defines "collaborate" as "to work together with others; cooperate with the enemy." Following the 1996 National Research Council (NRC) report, we define

involves interrelated negotiations over a number of years, rather than a series of standardized rule-making decisions in which an agency proposes a rule, receives comments, amends the rule, and then awaits litigation by one or more dissatisfied stakeholders

The new approach emerged because of dissatisfaction with the traditional strategy's inability to deal with a variety of problems, including non-point source pollution, the protection of coastal estuaries, water quality planning under the Total Maximum Daily Load (TMDL) provisions of the Clean Water Act, protection of aquatic species under the Endangered Species Act, and the development of management plans under the National Forest Management Act.[2] All of these problems require detailed knowledge of local situations and/or the coordination of multiple agencies, which proved extraordinarily difficult to accomplish under the old top-down, dominant-agency strategy. The traditional strategy also tended to result in policy decisions that left many actors dissatisfied, and they, in turn, looked to the courts for redress. In addition to being ineffective for handling many types of water management issues, the old paradigm suffered from problems of legitimacy because decisions were perceived as being made by far-off, faceless bureaucrats with little knowledge or concern for how their decisions were affecting local citizens. The emergence of the "wise use" movement in the West is the most visible sign of this legitimacy crisis (Gottlieb, 1989).

In contrast, the collaborative approach involves face-to-face information exchange and problem-solving among all the relevant stakeholders, usually under fairly strict civility guidelines and some form of consensus rule. The emphasis is on finding creative, win-win solutions to a variety of problems facing different stakeholders. Admittedly, this takes time. However, the argument is that such a process is likely

"stakeholder" as "any interested and affected party." They can include private citizens, constituency groups, agencies, researchers, etc. In this volume, we often distinguish between "public" and "private" stakeholders.

[2] Total Maximum Daily Loads (TMDLs) arise from Section 303 of the Clean Water Act designed to deal with "residual" pollution sources, i.e. those not subject to technology-based standards. Most, but not all, deal with non-point sources. States must first determine reaches of water bodies that are impaired (i.e. do not meet water quality standards). For impaired water bodies and their associated pollutants, the state must issue a TMDL for pollutant and waterbody that includes an assessment of all pollution sources, a total load for the waterbody, and an allocation of pollution reduction among the sources. TMDLs provide a substantial incentive for pollution sources to negotiate a TMDL, rather than wait for the state or EPA to impose one on them. Leach et al (2001) found that TMDLs were by far the most important stimulus to the formation of partnerships in California.

to generate mutual understanding and trust, rather than animosity and suspicion, among stakeholders. This should facilitate decision-making over time and give collaborative efforts the ability to deal with a whole series of interrelated problems. In addition, since any solutions developed will have the support (or at least the acquiescence) of all parties, implementation should be less problematic than under the traditional approach and less plagued by endless litigation. Finally, because collaborative partnerships seek to involve all the major affected stakeholders and usually operate under some form of consensus rule, they should have greater legitimacy than traditional approaches that rely upon the legal authority provided bureaucracies by legislatures (Bingham, 1986; John, 1994; Born and Genshaw, 2000).

The collaborative watershed management approach is not a detailed blueprint, but rather a broad strategy for solving very complex sets of interrelated problems. It includes several variants:

> *1) Collaborative engagement processes*. These are techniques for conflict resolution among diverse stakeholders that are developed by outside actors and then applied to specific planning exercises of relatively limited duration (but still longer than the public participation procedures under the old approach). The techniques include environmental mediation (Bingham, 1986), collaborative learning (Daniels and Walker, 2001), and the analysis and deliberation framework developed by the National Research Council (NRC, 1996).
>
> *2) Collaborative watershed partnerships*. These are relatively informal organizations involving a wide variety of public and private stakeholders who seek to develop a management plan for the watershed and then implement it via specific restoration projects, changes in land use practices, water quality regulations, etc. Partnerships are intended to be rather long-term affairs (5-10 years). The partnership itself tends to have little formal legal authority to carry out projects or prescribe regulations. Rather, the partnership provides a forum in which management plans and implementing actions are negotiated, and then turned over to member agencies for formal legal actions. Thus partnerships tend to complement and hopefully transform—rather than replace—traditional agencies. Such partnerships are quite widespread. In 1999, for example, Leach and Pelkey (2001) counted about 150 in California and 60 in Washington. Multi-stakeholder partnerships receive financial support from state agencies in at least three states: Massachusetts, Oregon, and Washington.
>
> *3) Collaborative superagencies*. These are partnerships that have become formalized so that they both negotiate management plans and administer the implementing actions. These are still quite rare, with the foremost example probably being the CALFED superagency in California. It began as a stakeholder partnership to recommend provisions for implementing the 1994 water quality standards for the San Francisco Bay/Delta and has evolved into an agency with a $900 million annual budget and charged with overall water management and restoration throughout the Sacramento and San Joaquin River watersheds of the Central Valley. Its Policy Council includes representatives of twenty-one federal and state agencies, as well as several private (or quasi-private) stakeholder groups.

As an indicator of the popularity of the new approach, nine federal agencies and departments have endorsed the "Clean Water Action Plan" (1998) calling for "a new cooperative approach to watershed protection in which state, tribal, federal, and local governments, and the public first identify the watersheds with the most critical water quality problems and then work together to focus resources and implement effective strategies to solve those problems."[3]

Despite its recent popularity, the collaborative watershed approach is not without its critics. One set of criticisms focuses on problems of procedural and substantive representation within partnerships. The leaders of several national environmental groups are concerned that they cannot possibly place representatives in most of the partnerships and planning processes throughout the country (McCloskey, 1996; Kenney, 2000). This procedural criticism is then linked to substantive concerns that non-local environmental interests will be sacrificed (Sagoff, 1999). Similarly, landowner and property-rights groups who take the position that only landowners should be involved in decisions affecting their land criticize the inclusiveness of partnerships and worry about the threats to property rights they entail (Gottlieb, 1989). Others worry that stakeholder processes take enormous amounts of time and thus effectively exclude the general public and/or all but the most interested actors with substantial resources (Coglianese, 2001). They certainly include very few elected officials, and thus one can properly worry about the extent to which they represent the distribution of views within the watershed.[4]

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[3] Several federal agencies have issued guidebooks that advocate decentralized, consensus-oriented policymaking. Examples include the U.S. Department of Housing and Urban Development's Community-Building Coming of Age (Kingsley et al, 1997), the U.S. Environmental Protection Agency's Community-Based Environmental Protection, the National Oceanic and Atmospheric Administration's Watershed Restoration (Kier, 1995), and the U.S Forest Service's Guidelines for a Watershed Approach (Federal Register, Feb. 22, 2000, pp.8834-8840). State and federal programs have recently channeled millions of dollars through watershed partnerships. Examples include EPA's Section 104(b)(3), 205, and 319 grants; the Department of Agriculture's EQIP program; California's CALFED program and Propositions 12, 13, and 204; Washington state's Watershed Planning Act of 1998 (about $4.5 million annually); and the Massachusetts Watershed Initiative, which funds full-time partnership coordinators in each of the state's 27 watersheds.

Following Sommarstrom (1998) and Cook (1999), we think it important to distinguish "watershed partnerships," which are composed of actors with diverse views, from "watershed groups," which are composed of like-minded individuals (usually either landowners or environmentalists).

[4] In their survey of 44 watershed partnerships in California and Washington, Sabatier et al (2001) found that, while 25% of partnership members were from local governments and agencies, only 3% were elected officials from cities and counties.

A second set of concerns involves questions regarding the effectiveness of collaborative processes and partnerships in developing and implementing solutions to watershed problems. Clearly, consensus-based decision rules offer a potential recipe for delay and for avoiding the most complex and difficult problems (Coglianese, 1999). Many collaborative efforts create policies that rely on voluntary cooperation without any formal legal enforcement mechanisms, which often creates considerable doubt about the likelihood of policy implementation. Because agencies and private interests find it necessary to resort to traditional enforcement mechanisms (e.g., the Endangered Species Act), the evidence is mixed about whether collaborative processes are more successful in avoiding litigation than more traditional single-agency rulemaking (Bingham, 1986; Coglianese, 1997).

A final set of threats to the viability of the new approach involves the ability of collaborative processes and partnerships to find a permanent niche in a landscape dominated by agencies that are usually either single-function and/or dependent upon legal boundaries that have little relationship to watersheds. Collaborative processes may threaten agency autonomy and thus create an incentive for traditional agencies to usurp control of collaborative outcomes. For example, there is some evidence that Law 2514 in Washington State puts multi-stakeholder partnerships clearly under the thumb of general-purpose local governments, whose boundaries typically split watersheds into conflicting fiefdoms. To the extent a single agency manages to control a given collaborative project, other agencies and private interests will look for alternative venues to pursue their policy goals.

In order for the new approach to succeed, it will have to demonstrate that it is superior to traditional approaches on both procedural and effectiveness criteria, as well as win its fair share of the "turf" battles with traditional agencies. None of this is given. Like the salmon swimming upstream to spawn in their home waters, only the fittest collaborative projects are likely to survive in the institutionally complex American system. Hence the title of this volume.

The Existing Literature

Unfortunately, the literature on collaborative watershed planning is dominated by studies of one or two cases that rely on very subjective methods of data acquisition and analysis and are usually not grounded in any body of theory (Leach and Pelkey, 2001). Such studies are useful for gathering descriptive information about partnerships and for generating hypotheses about the factors contributing to partnership success, but the latter contribution is now fully realized. As of 2000, such studies had produced a set of 210 "lessons learned," which can then be grouped into over 20 general themes.[5] This literature has a number of important limitations:

1) *Interpretive methods*. Studies relying upon subjective methods of data acquisition and analysis may provide intriguing insights, but an outsider has virtually no way to know if they are valid or not. Different people observing the same phenomena can "see" different things unless they are given relatively detailed categories in which to standardize perceptions. The fundamental dictum of science is that methods be "public" enough that they can be replicated (King et al, 1994).

2) *Sampling bias*. Since different people in a partnership see different things, one needs to make sure to gather data from a variety of participants and even knowledgeable outsiders. For example, there is now solid evidence that partnership coordinators perceive partnerships as more trust-generating and more successful than do most of their members (Leach, 2001). This renders problematic the data from several surveys of large numbers of collaborative groups that rely heavily for information upon the views of one to two people (often the coordinator) per partnership (Yaffee et al, 1996; Kenney et al, 2000).

3) *Confounding variables*. Almost all the literature dealing with factors affecting partnership success focuses on variables largely, or at least partially, under the control of participants. But, if there are 25 such variables and the study involves only one to two partnerships, the author cannot ascertain the *relative* importance of each of the variables. Determining the relative importance of each of the 25 requires at least 150 cases. In addition to this statistical problem, the literature tends to neglect whole sets of factors that are *not* under the control of participants.

4) *Generalizability of conclusions*. Studies of collaborative processes that involve data collected systematically from numerous participants potentially have high internal validity. However, if they only involve one or two partnerships/processes, their results will not be generalizable to

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[5]Following are the themes relating to partnership success (Leach and Pelkey, 2001: Appendix): funding, 2) broad/inclusive vs. limited membership, 3) cooperative, committed participants, 4) effective leader/coordinator/facilitator, 5) bottom-up initiation/leadership, vs. balanced local/state/federal participants, 6) trust, 7) low or medium level of conflict vs. conflict unimportant, 8) geographic scope, 9) limited vs. broad scope of activities, 10) adequate time, 11) well-defined process rules, vs. flexibility/informality, 12) consensus rules: yes vs. no, 13) formal enforcement mechanisms, 14) effective communication, 15) adequate scientific and technical information, 16) monitoring outcomes, 17) training in collaborative skills, 18) agency support and participation, 19) legislative encouragement, 20) community resources, and 21) Other.

cases in different types of settings. For example, if the case studied involves relatively low conflict, the results may not be generalizable to situations involving high conflict.[6]

5) *Explicit theory(s).* Theories are very useful for guiding empirical research because they clarify concepts and relationships, remind the researcher of possibly interesting relationships that have not occurred to participants and provide a means for relating findings from different studies that share the same theoretical framework.[7] Ideally, researchers should compare different theories so as not to become wedded to any of them (Platt, 1964; Allison, 1969). In the literature on collaborative watershed processes, only about 25% make reference to any body of theory and only one or two try to test multiple theories.

In Riechenbach's (1938) terms, studies of collaborative watershed approaches have been largely conducted in the "context of discovery," where the generation of hypotheses is the focus and the use of rigorous scientific methods is secondary. The emphasis is on generating intriguing ideas. In the "context of verification," however, the emphasis is on determining the internal and external validity of posited relationships (hypotheses). This requires classic scientific methods, including intersubjectively reliable methods of data acquisition and analysis, explicit attention to the uncertainty arising from measurement and sampling error, and testing propositions developed from one or more bodies of theory (King et al, 1994).

Objectives and Audiences of the Book

In response to these shortcomings in the existing literature, the authors of this book have adopted three principal objectives:

The first is to focus attention on, and bring some understanding concerning, the field of collaborative watershed management. In particular, we distinguish among (a) collaborative engagement processes, (b) collaborative partnerships, and (c) collaborative superagencies. As we shall see, different

[6]Given limited resources, there is an inherent conflict between internal and external validity. Getting high scores on both requires both large numbers of systematic observations within each collaborative process (internal validity) and large numbers of collaborative processes (external validity). In the existing literature, very few studies combine both—Susskind et al (2000) being a major exception. In contrast, two of the four empirical studies in this volume (Chapters 7 and 8) have this desirable combination.

[7]As we shall see in Chapter 7, two of the most important factors explaining the success of watershed partnerships are (a) the density of social networks in the community and (b) (lack of) an alternative decision-making venue. The former was taken from Putnam's social capital framework (Putnam, 1993; 2000) and the latter from the

types of actors—not just watershed partnerships—use different types of collaborative engagement processes. Conversely, some partnerships do not use any formalized process but, instead, treat process in a relatively informal and *ad hoc* fashion.

The second, and perhaps most important objective, is to bring some analytical rigor to a field that, in its first generation, has been dominated by practitioners anxious to share insights ("lessons learned") from their particular situations. In the process, they have not always articulated their underlying premises and reasoning, nor have they generally employed scientific methods of investigation. As authors of perhaps the initial "second generation" book, we hope to (a) clearly articulate a set of normative/value-based arguments dealing with the procedural and substantive legitimacy of collaborative approaches; (b) provide some detail on the logic behind two collaborative engagement processes, the collaborative learning framework and the analysis and deliberation framework; (c) articulate several different theoretical frameworks concerning the factors affecting the success of watershed partnerships; and (d) use intersubjectively reliable methods of data acquisition and analysis to investigate propositions drawn from (b) and (c). Note that we seek to bring some analytical clarity to both the normative justifications for collaborative endeavors and to the factors affecting the success of both engagement processes and partnerships.

Our third, and perhaps most challenging objective, is to make our analyses and results of real use to practitioners. This requires, first, that we make our writing intelligible by avoiding jargon and by providing non-technical interpretation of our statistical analyses. Second, we need to deal both with factors under the control of practitioners and those beyond their control. The latter may be unsettling to practitioners, but efficient strategies for changing the world begin with an accurate assessment of the goals, resources, and constraints of one's own organization and other critical actors. Third, we seek to provide practitioners with concrete methods for estimating the success of both collaborative engagement processes and multi-stakeholder partnerships.

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literature on alternative dispute resolution (Ury, 1993; Carpenter and Kennedy, 1988). Neither is barely mentioned in the 210 "lessons learned" identified by Leach and Pelkey (2001).

It is hoped that this volume will provide practitioners with explicit information on some of the best practices in the area of legitimate and effective stakeholder involvement in water and other natural resource planning. It is also hoped that, by outlining the normative and empirical theoretical frameworks relevant to collaborative watershed approaches, we will provide resource managers with the intellectual tools necessary to adapt those approaches to new situations and to evaluating their outcomes along multiple dimensions.

The academic audience hopefully will find in the theoretical and empirical analyses integrative, high-quality research that addresses important topics in a rigorous, but insightful, manner. In addition, the interdisciplinary approach that informs these discussions—including philosophers, environmental scientists, communications researchers, and political scientists—is still relatively rare in the field of water policy, dominated as it is by hydrologists, civil engineers, and, to a lesser extent, economists. In fact, academic research on water policy is rather schizophrenically divided between, on the one hand, very technical water management models that rely upon simple models of the individual (largely, *homo economicus)* and assume that technical analyses dominate the decision process and, on the other hand, empirical studies of decision-making processes that are rich in insightful detail but rely upon very subjective methods of data acquisition and analysis. We see this book as an attempt to integrate and advance the field by subjecting hypotheses produced in the first generation of case studies to more systematic normative and empirical analysis.

Organization of the Book

The book is divided into four parts. Part I: *Watershed Management Approaches in the U.S.* presents an overview of watershed management in the U.S that sets the stage for the chapters that follow. This part includes three chapters: an introduction that frames the book, a history of water management in the U.S., and the presentation of a normative theoretical framework that proposes criteria for the evaluation of procedural and substantive policy legitimacies relevant to collaborative approaches.

Part II: *Collaborative Engagement Processes in Watersheds* attempts to integrate scientific and social knowledge into a better understanding of the interrelated problems and possible solutions articulated by a wide variety of stakeholders. This part extends the arguments presented in Part I in two chapters that consider, both theoretically and empirically, the application of collaborative stakeholder involvement strategies in watersheds in Oklahoma and Texas. Based on the level of trust that private stakeholders have of other policy actors, Chapter 4 tests a prescriptive framework for the selection of stakeholder participation processes that seeks to satisfy the policy legitimacy criteria that were discussed in Chapter 3. Chapter 5 considers the elements that affect the willingness of stakeholders to participate in the sorts of collaborations that are envisioned in high distrust contexts and examines both the representativeness of these collaborations and the motivations of participants.

Part III: *Measuring and Explaining the Success of Watershed Partnerships* deals with the factors affecting the success of collaborative watershed *partnerships.* Chapter 6 first outlines several theories of collective action and policy-making relevant to such partnerships. These theories are then tested in Chapter 7 using a national sample of participants in National Estuary Program sites and comparable non-NEP sites. Chapter 8 then tests hypotheses developed from several of those theories on a sample of 44 collaborative watershed partnerships in California and Washington.

Part IV: Chapter 9 presents an evaluation of the framework we posited in Chapter 1 utilizing the empirical research we present in the other chapters and then summarizes our empirically grounded "lessons learned" for both practitioners and academics.

Part I: Watershed Management Approaches in the U.S.

Chapter 1, "Overview of the Book," by the six principal authors. This chapter outlines the policy issues addressed by collaborative watershed approaches, the contributions and deficiencies in the existing literature, and presents a comprehensive framework of the relationships and concepts we believe are essential for developing a new generation of interdisciplinary research on the legitimacy and effectiveness of collaborative approaches. It is an expanded version of the discussion that introduces this outline with a detailed presentation and explanation of our conceptual framework.

Chapter 2, "History of Watershed Management in the U.S.," by William Leach, Neil Pelkey, and Paul Sabatier (UC Davis), sets the stage for the ones that follow by providing an historical context of several eras of water management institutions found in the U.S. since the late 19th century. The different water management eras are characterized by the types of participants involved (public vs. private, federal vs. state and local), the geographic scope of water management (watershed vs. other), the principal substantive purpose (preservation, conservation, development), and the principal water goal (supply vs. quality). The analysis suggests that the collaborative approaches we are studying—while sharing some characteristics with predecessor institutions in different eras—really do represent a rather novel form of public participation and overall governance within water management in the U.S.

Chapter 3, "Criteria of Legitimacy for Evaluating Stakeholder Participation in Watershed Policy Processes " by Zev Trachtenberg (University of Oklahoma) and Will Focht (Oklahoma State University). This chapter develops a set of criteria that can be applied to watershed decision-making processes to assess their legitimacy. It is particularly concerned with the question of how legitimacy can be enhanced by the participation of a wide range of stakeholders. The criteria address both the procedures by which decisions are made, and the actual impacts of watershed policy on the well-being of watershed residents and others. The criteria are based on a general conception of legitimacy that rests on the fundamental values of personal autonomy, welfare enhancement, and justice. This chapter provides an excellent analysis of the moral issues that permeate collaborative watershed management.

Part II: Collaborative Engagement Processes in Watersheds

Chapter 4, "A Trust-Dependent Guide to Stakeholder Participation," by Will Focht (Oklahoma State University) and Zev Trachtenberg (University of Oklahoma). It sets the context for much of the analysis in Part II by hypothesizing that the selection of stakeholder participation methods should be based upon contextual factors, particularly the level of trust among policy actors. Trust is conceptualized as the willingness of stakeholders to defer to the competence and discretion of others, specifically fellow "private" stakeholders and "official" policy actors (government decision-makers and subject matter experts), to manage risk on their behalf. Assigning high/low scores to these two trust dimensions produces four decision contexts, for which tailored stakeholder involvement strategies are prescribed. For example, when stakeholders trust both public officials and each other, information exchange is recommended as an appropriate strategy. However, when stakeholders distrust both officials and each other, a deliberative strategy (e.g., policy dialogue) is recommended as more appropriate. The authors test these hypotheses using data obtained from face-to-face interviews with 144 stakeholders residing in the Illinois River Basin in Eastern Oklahoma and find strong empirical support.

Chapter 5, "Citizen Willingness to Participate in Collaborative Engagement Processes" by Charles Samuelson, Arnold Vedlitz, Guy Whitten, Letitia Alston, (all at Texas A&M University) Marty Matlock (University of Arkansas), Tarla Peterson (University of Utah) and Susan Gilbertz (University of Montana). Long-term, collaborative stakeholder participation supposedly enhances the legitimacy of watershed management decision-making because of the complexity of decisions and the competition among various actors involved. However, such prescriptions beg the question of whether citizens actually are willing to engage in participatory activities of this type. This chapter systematically examines the correlates of the willingness of stakeholders in San Antonio, Texas to participate in a three-year collaborative process to learn about impacts on several watersheds, to frame policy analyses, and to deliberate on impact management policies. Utilizing a public opinion survey of more than a thousand citizen respondents in the city of San Antonio, three types of correlates are examined—general outlook variables (efficacy, trust, optimism); specific environmental attitudes (concern about resources, environmental problems, human-nature interface); and SES/demographic factors. The subsequent discussion focuses on questions of participant representativeness, the policy efficacy of these processes, and the relevance of these processes to legitimate, democratic decision-making.

Part III: Measuring and Explaining the Success of Watershed Partnerships

Chapter 6, "Theoretical Frameworks Explaining Partnership Success," by Paul Sabatier, Bill Leach, and Neil Pelkey (UC Davis). This chapter lays out the basic principles from five general theoretical frameworks of collective decision-making, and then suggests several hypotheses specifically related to participants' willingness to engage in watershed partnerships and to the success of those partnerships. The frameworks are (a) Institutional Analysis and Development (Ostrom, 1990; Ostrom et al, 1993); (b) Transaction Cost Economics (Williamson, 1975; Lubell et al, 1998); (c) Social Capital (Putnam, 1993; 2000); (d) Advocacy Coalition Framework (Sabatier and Jenkins-Smith, 1993; 1999); and (e) Alternative Dispute Resolution (Bingham, 1986; Carpenter and Kennedy, 1988; Susskind et al, 1999). The analysis reveals important similarities across IAD and TCE, on the one hand, and ACF and ADR, on the other. All the frameworks use the concept of trust, the key variable in Social Capital.

Chapter 7, "Do Watershed Partnerships Enhance Beliefs Conducive to Collective Action?" by Mark Lubell (UC Davis). This chapter examines whether or not watershed partnerships change stakeholder beliefs in ways that facilitate cooperation. To test hypotheses drawn from the IAD and ACF frameworks discussed in Chapter 7, the author compares stakeholder beliefs in 20 estuaries that were part of the National Estuary Program with 10 estuaries that were not part of the NEP. The analysis shows that NEP stakeholders have beliefs more conducive to cooperation and are less likely to disagree about the process of decision-making in watersheds.

Chapter 8, "Conceptualizing, Measuring, and Explaining Success in Watershed Partnerships," by Paul Sabatier, William Leach, Neil Pelkey, and Chris Weible (UC Davis). This chapter first develops a set of six indicators for measuring the success of watershed partnerships, and then uses data from random samples of 44 partnerships in California and Washington to operationalize success and to test hypotheses from several of the frameworks in Chapter 7 regarding the factors affecting success. The analysis indicates trust is by far the single most consistent predictor of success, but that several of the variables hypothesized as critical in the existing literature (e.g. presence of a facilitator, amount of grant funds) are not important—once other factors have been controlled for.

Part IV: Conclusions

Chapter 9, "Lessons Learned," will be developed collaboratively by the six principal authors. This chapter reviews the important conclusions from throughout the book for both practitioners and academics and relates the empirical findings of the study to the concepts and relationships outlined in the framework presented in Chapter 1.

As of June 2003, drafts of all chapters, except chapter 6, are complete and ready for submission to MIT Press. Chapter 6 is in development and will be finished by July 15, 2003. The submission of the completed draft to MIT Press will be on August 15, 2003.

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