A Primer for Monitoring Water Funds

GLOBAL FRESHWATER PROGRAM

The Nature Conservancy • June 2013

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Contents

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Source Information

Significant portions of this document were adapted from the following:

Federal Interagency Stream Restoration Working Group—Stream Corridor Restoration: Principles, Processes, and Practices (2001)
http://www.nrcs.usda.gov/wps/portal/nrcs/detailfull/national/water/manage/restoration/?cid=stelprdb1043244

University of Wyoming—Best Management Practices Monitoring Guide for Stream Systems (2011)
http://www.uwyo.edu/bmp-water/docs/bmp%20mon%20guide%20streams%20web.pdf

USDA Natural Resources Conservation Service—National Water Quality Handbook (2003)
http://www.wsi.nrcs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb1044775.pdf

Executive Summary

Introduction

Water Funds are governance and financial mechanisms organized around the central principle of watershed conservation. In an effort to secure a continuous supply of clean water, downstream users (e.g., municipalities, utilities, companies) and/or public agencies direct payments to upstream communities (e.g., farmers, ranchers) and land preservation organizations to finance long-term ecological restoration and protection efforts within the watersheds on which they rely for water. The Water Fund's managers, consisting of key stakeholders (or housed within an agency), typically determine the allocation of conservation investments in consultation with technical advisors. These investments are generally focused on maintaining existing intact natural areas, restoring lands throughout watersheds and along river corridors, and implementing management practices to minimize the impacts of land use activities on water quality and quantity. In addition, funds are often used for alternative livelihood strategies and other social initiatives in participating rural communities, ranging from organic gardens to education projects, which help protect water supplies and improve the living and economic conditions for upstream communities.

For more than a decade, The Nature Conservancy ("the Conservancy") has utilized a variety of strategies to protect water at its source—including investments in natural infrastructure and land use management practices, increase water efficiencies in agriculture, and engage corporate leaders and large water users in watershed stewardship. Water Funds are a critical tool to support this work.

Since the creation of our first Water Fund in Quito, Ecuador in 2000, the Conservancy has engaged more than 100 partners in developing and managing Water Funds. To date, we have initiated or launched more than 30 Water Funds throughout the world, in places as varied as Sao Paulo, Brazil; Lima, Peru; Nairobi, Kenya; Santa Fe, New Mexico; and Bogota, Colombia.

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Water Funds have the potential to provide significant benefits for investors, upstream and downstream communities, and the natural world. However, these benefits are not guaranteed by the creation of a Water Fund. They are engendered through science-based, sound investments founded on a clear set of goals and ongoing monitoring data. To ensure that investments are having their anticipated impacts and to enable corrections to management strategies, Water Funds must include robust monitoring programs to track the environmental, economic, and social impacts of their actions.

Given limited funding and capacities, monitoring resources must be targeted to capture the most relevant information to determine if the Fund is meeting its near-term milestones and long-term goals. Effective monitoring requires a clear understanding of the questions the data will address. These questions focus on judging progress toward meeting the Water Fund's environmental, social and economic goals; providing critical information to investors and participating programs and communities; and evaluating effectiveness, efficiency, and opportunities for adaptive management.

Why monitoring is necessary

Water Funds operate under the assumption that activities carried out to protect and restore watersheds will provide the intended benefits defined in the Fund's goals and objectives. The extent to which these goals and objectives are achieved depends on the efficacy of models that are used to estimate benefits, the effectiveness of the activities, the efficiency of implementation, the return on investments that occur, and the influence of environmental and socioeconomic factors external to Water Fund activities that can have significant impacts on results. The extent to which goals and objectives are met given these elements will determine the long-term sustainability of Water Funds.

Financial supporters require regular reports of status, progress, and returns on investments, participating communities require evidence of benefits, and Water Fund managers require information to strengthen models, and adaptive management based on results and changing environmental and socioeconomic conditions. All of these aspects require well-structured monitoring and reporting approaches.

What this document provides

This document is intended to assist people working on Water Funds to understand their information needs and become familiar with the strengths and weaknesses of various monitoring approaches. This primer is not intended to make people monitoring experts, but rather to help them become familiar with and conversant in the major issues so they can communicate effectively with experts to design a scientifically defensible monitoring program.

The document highlights the critical information needs common to Water Fund projects and summarizes issues and steps to address in developing a Water Fund monitoring program. It explains key concepts and challenges; suggests monitoring parameters and an array of sampling designs to consider as a starting-point; and provides suggestions for further reading, links to helpful resources, and an annotated bibliography of studies on the impacts that result from activities commonly implemented in Water Fund projects.

While this document highlights the importance of setting clear goals and objectives, which will guide a Water Fund and its activities and define what information should be tracked, it does not provide detailed information about how to develop goals and objectives (for more information on this process, see the Conservancy's primer on Water Fund creation and design, *Water Funds: Conserving green infrastructure: A guide for design, creation and operation*).

Goals and objectives

Goals define the overarching expectations of a Water Fund. Objectives are defined with specific, quantified, time-bound milestones that map pathways to success. Without clear goals and objectives, Water Funds cannot determine what they are trying to achieve or whether they have been successful in achieving it. Monitoring is conducted to assess progress towards achieving objectives and long-term goals, to identify obstacles and course corrections that may be needed, and to highlight successes.

Evaluating and strengthening models

Goals and objectives are commonly created using supporting information from models that estimate changes in environmental conditions from a suite of activities implemented in specific places throughout a watershed, and the ecosystem services and benefits expected to result. These models are often optimization models, illustrating best-case scenarios to achieve the most effective and efficient results. The ability of a Water Fund project to more accurately model expected results for setting goals and objectives, defining priorities on where and how to work, and identifying attributes to monitor depends upon empirical research. When necessary, monitoring data can give the models a more accurate basis for making estimations.

Implementation, impacts, and external factors

Tracking implementation is critical to understanding where and how watersheds are being protected and restored and provides the basis for evaluating progress in placing activities, understanding efficiencies and costs, and estimating impacts. It is essential to quantify the impacts that Water Fund activities are having within a watershed and to understand the role and influence of external factors beyond a Fund's control.

Implementation

Implementation monitoring tracks the outputs of the Water Fund, such as the number of families enrolled in a Payment for Ecosystem Services agreement, and the specific sites and spatial scope of different activities. Monitoring the implementation of Water Fund activities provides information necessary for modeling expected impacts based on progress made; evaluating relationships among activities and results observed via impact monitoring; guiding adaptive management; understanding the cost-effectiveness of specific management strategies; and reporting progress.

Impacts

Understanding changes in environmental, social and economic systems resulting from Water Fund activities requires data on those attributes over time. Monitoring these changes requires defining appropriate spatial scales and time-frames to address the specific information needs. The statistical resolution, accuracy, precision, and designs of monitoring approaches determine their strengths and weaknesses for measuring and communicating results.

External factors

Monitoring changes in climate, resource management, and land use and land cover is necessary to understand how the broader context is changing, and to account for the contributions that Water Fund activities are making through appropriate monitoring designs. In some cases, the condition of freshwater habitats, biota and services may decline over time due to factors external to the Water Fund, and it is critical to model the further degree of decline that would have occurred without Water Fund activities in place (the "business-as-usual" scenario).

Developing a monitoring program

The majority of monitoring and data collection elements may already be in place, and if not, portions of the Water Fund's information needs may already be filled. Monitoring of water supplies and their surrounding land is typically conducted on a regular basis by water utilities, hydropower companies, water resource management agencies, municipalities, and a range of government agencies, academic institutions and non-profit organizations.

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When Water Fund projects are being developed, defining information needs must be part of the initial dialogue and Water Fund design, rather than an afterthought. Identifying what information will be needed to determine the activities to be funded by the Water Fund and evaluate their impacts, what is already collected, and by whom are critical first steps. Creating a data-sharing and privacy Memorandum of Understanding or other formal agreement to allow access to data, or to allow specific analytical results to be provided at the initiation of the project, should be a priority for assessing data availability and filling information gaps.

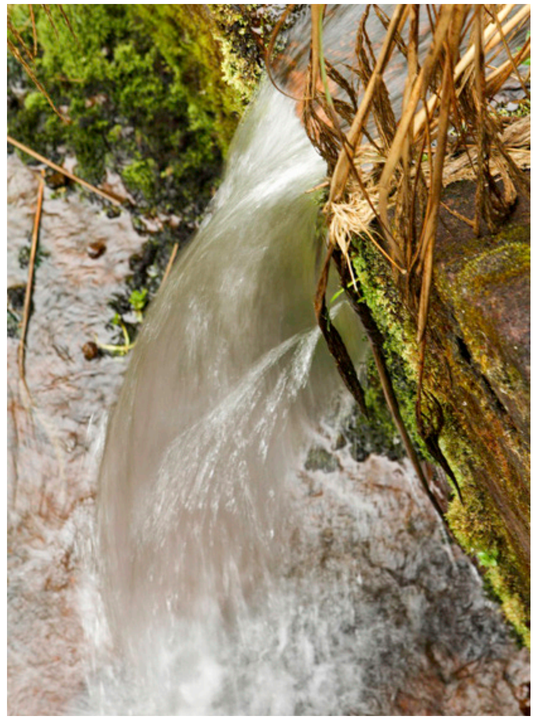
Experts in environmental, social and economic sciences, as well as statisticians, should be involved at all stages of monitoring design, implementation, and analysis. Monitoring should be seen as a cooperative process aimed at including the technical and scientific expertise to achieve sufficient rigor. Community-based data collection efforts may be included when appropriate, granted that the inherent limitations of these methods in terms of detection limits, accuracy, and precision are carefully considered. Community engagement may be critical for education, sustaining support for Water Fund projects, gaining political support, and illustrating first-hand the benefits of participants' efforts.

Audiences

Donors, investors, regulators, external stakeholders, partners, participating communities, and managers require different types and amounts of information reported in various timeframes. Defining information needs for each of these audiences informs monitoring needs and ways to communicate results to them. It is necessary to communicate information to each audience in the appropriate format and level of detail they require and are familiar with.

The most useful communications of results are those that are clear and direct. While some scientists will insist that a lack of detail in reports is "dumbing down" information, the most successful communications provide relevant information for key audiences through simple and intelligent presentation. Data are not useful if they cannot be understood or deemed relevant.

Introduction



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Water Funds are governance and financial mechanisms organized around the central principle of watershed conservation. For the promise of a continuous supply of clean water, downstream users (e.g., municipalities, utilities, companies) and/or public agencies direct payments to upstream communities (e.g., farmers, ranchers) and land preservation organizations, to finance long-term ecological restoration and protection efforts within their watersheds. The Water Fund's managers, consisting of key stakeholders (or housed within an agency), typically determine the allocation of conservation investments in consultation with technical advisors. These investments are typically focused on maintaining existing intact natural areas, restoring lands throughout watersheds and along river corridors, and implementing management practices to minimize the impacts of land use activities on water quality and quantity. In addition, funds are often used for alternative livelihood strategies and other social initiatives in participating rural communities, ranging from organic gardens to education projects.

Water Funds have the potential to provide significant benefits for investors, upstream and downstream communities, and nature. However, these benefits are not guaranteed by the creation of a Water Fund. They are engendered through science-based, sound investments founded on a clear set of goals and ongoing monitoring data. To ensure that investments are having their anticipated impacts and to enable corrections to management strategies, Water Funds must include robust monitoring programs to track the environmental, economic, and social impacts of their actions.

Since the creation our first Water Fund in Quito, Ecuador in 2000, the Conservancy has engaged more than 100 partners in developing and managing Water Funds. To date, we have initiated or launched more than 30 Water Funds throughout the world, in places as varied as Sao Paulo, Brazil; Lima, Peru; Nairobi, Kenya; Santa Fe, New Mexico; and Bogota, Colombia.

For example, site-level monitoring can be used to gauge whether management practices are providing their intended effects on ecosystem characteristics. At the watershed scale, monitoring can offer tangible evidence of the effects of upper watershed projects on ecosystem conditions and downstream water supply, and document trends toward meeting investors' goals for ecosystem services.[1] In addition, interviews, focus groups, and socioeconomic surveys and indicators can provide an understanding of the influence Water Fund projects are having on human communities, in terms of both economic and social impacts—factors essential to the long-term sustainability and success of a Water Fund.

Monitoring is the act of collecting information about something over time to provide data on status and change. Measures are a way to place monitoring information into context for communicating it to a specific audience. Too often, monitoring is conducted to collect a vast array of data and little if any of it is ever communicated or used to guide or evaluate management decisions. Although much of this document focuses on monitoring, the development and tracking of key indicators and addressing information gaps should form the basis for monitoring. It is critical that each Water Fund understand its specific requirements for measures and reporting before developing monitoring approaches.

This document is intended to assist people working on Water Funds to understand their information needs and become familiar with the strengths and weaknesses of various monitoring approaches. This primer is not intended to make people monitoring experts, but rather to help them become familiar with and conversant in the major issues so they can communicate effectively with experts to design a scientifically defensible monitoring program.

The document highlights the critical information needs common to Water Fund projects and summarizes issues and steps to address in developing a Water Fund monitoring program. It explains key concepts and challenges; suggests monitoring parameters and an array of sampling designs to consider as a starting-point; and provides suggestions for further reading, links to helpful resources, and an annotated bibliography of studies on the impacts that result from activities commonly implemented in Water Fund projects.

Each Water Fund will have unique goals, governance structure, activities, and constraints, as determined by the Fund's managers in the context of ecological, socioeconomic and political circumstances. The specific information needs for each Water Fund, including minimum monitoring requirements, will ultimately be defined by Fund managers in consultation with Water Fund partners. However, for a complete picture of Water Fund effectiveness, ideally all Water Funds will monitor the impacts occuring in each of the following themes: Ecosystem Functions, Services, and Benefits; Habitats and Biodiversity; and Communities (see Chapter 4).

For each of the major themes, this primer provides an overview of parameters and monitoring designs appropriate for particular goals, with a subset flagged as core measures that seem appropriate for inclusion in most Water Fund monitoring programs. In addition, issues regarding controls, external factors, lag-times for expected results, and aspects of statistical rigor are summarized to provide insights into components of monitoring that need to be evaluated while developing a monitoring program. Examples of monitoring approaches, as well as suggested further reading and links to additional information on sampling designs, protocols, technical guidance, and other resources are provided throughout the document.

A monitoring program that incorporates robust measures of the themes identified above and is designed to address the specific goals of the Water Fund will be equipped to advance those activities shown to accrue the most benefits and will also be able to demonstrate to local communities and investors that

1 The term *Ecosystem Services* is used throughout this document to denote services derived from ecosystems for human benefit. In Latin America, these are often referred to as *Environmental Services* or *Watershed Services*.

the project is a worthwhile endeavor to maintain in the future. This document puts forward a core set of steps for defining monitoring needs and priorities, and an initial set of parameters and designs to consider in order to fulfill these needs. Beyond these fundamental components, additional parameters and alternative approaches may be required so that each Water Fund can create a monitoring program that is complete, scientifically sound, ecologically meaningful, and locally relevant.

FURTHER READING

For a summary of the **needs for and approaches to monitoring watershed services projects**, see:

Porras, I., Alyward, B. and Dengel, J. (2013). Monitoring payments for watershed services schemes in developing countries, IIED, London
http://pubs.iied.org/16525IIED

Water Funds share a common overall vision of protecting and restoring natural systems to maintain and enhance water quality, regulate flows, and nurture biodiversity, while ensuring benefits for people. In short, the activities they fund help achieve The Nature Conservancy's mission to protect nature and preserve life.

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Section One: Preparation

Before developing a monitoring program, it is critical to define and prioritize information needs, understand the types and sources of information that already exist now and into the future (and who has it), critical information gaps, and opportunities to fill those gaps. It is also necessary to ensure that the appropriate expertise is engaged in designing monitoring programs to ensure that data collection and analysis is rigorous and scientifically-based.

Chapter 1: Priorities for Monitoring

What information do you need?

Given the often limited funding and capacities of Water Funds to implement monitoring programs, it is critical to target monitoring resources to capture the most relevant information. An effective monitoring program must be based on a clear understanding of the questions that the monitoring data will need to answer. These questions generally focus on judging progress toward meeting the Water Fund's goals; providing information to investors and participating programs and communities regarding this progress; and giving feedback to facilitate adaptive management.

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Water Fund goals

Water Funds should have specific long-term goals, developed in consultation with subject experts and stakeholders as part of the initial project planning stage, and relating to the ecosystem functions, services and benefits deemed most relevant to the Water Fund's success. A primary focus of monitoring should be to track progress toward achieving those goals as well as near-term objectives.

Priorities for impact monitoring generally concern ecosystem services, biodiversity conservation, and effects on people. Such monitoring should focus on measurable results (e.g., 55,000 hectares of vegetation restoration resulted in a 10% decrease in sediment concentrations at a water supply source, saving $100,000 annually and avoiding future infrastructure development and management costs that would have been passed on to utility customers), rather than the ecological mechanisms underlying results (e.g., deeper root structures and more dense coverage of shrubs results in higher sediment retention). This latter aspect of monitoring is often a focus of research scientists, but is not typically a priority for Water Fund monitoring, which is more focused on the outcomes desired by investors and participating communities (though this type of monitoring may be conducted if there is need, capacity and interest to do so).

The goals of the Water Fund should be clearly stated as part of the original planning process. These may be general statements such as:

- Maintain a regular supply of good quality water for water users
- Reduce need for investments in additional infrastructure and management actions for providing municipal water supply
- Sustain natural terrestrial and freshwater biodiversity in the watershed
- Enhance quality of life for participating communities

Project goals typically include more precise objectives for defining success. These objectives should be defined using the SMART framework, where objectives are *Specific, Measurable, Achievable, Realistic*, and *Time-bound*. For example:

- Reduce sediment loads at downstream water intake points by 15% within 10 years
- Restore aquatic invertebrate assemblage to reference conditions within 10 years
- Decrease incidence of water-borne illness in upstream communities by 50% within 5 years
- Decrease water treatment costs at downstream water intake points by 20% within 10 years
- Protect or restore 30% of riparian corridors within 10 years

Other information needs

In addition to tracking progress toward achieving project goals, other information needs must be addressed when considering monitoring priorities. It is helpful to consider what type of monitoring information is expected, by whom, within what time frame, and in what form.

Donors (individuals, foundations and organizations that provide funding) generally want to know as part of a regular reporting cycle:

- What progress is being made in implementation?
- What types and degrees of environmental, social and economic responses are occurring as a result of activities?

In addition, donors generally require reporting on progress toward any explicit objectives for activities or products that were included in funding proposals.

Investors (individuals and corporations that are expecting benefits in return for the financial support they provide) generally want to know:

- Are activities changing the situation in the right direction to achieve objectives (e.g., are sediment and nutrient loadings being reduced)?
- Are activities resulting in the intended benefits for investors? (e.g., have there been reductions in operating costs, or has the risk of water scarcity during the dry season been reduced?)
- What have activities cost, and what have been the returns on investments of those activities?

Participating Communities (those receiving payments or compensation from the Water Fund) often want to know:

- Are the activities they are being paid to carry out providing benefits to them and/or to the environment and to the project?

Managers (those making management decisions) and ***participating partners*** (those implementing management decisions and supporting activities) need more detailed information than any of the groups above, because they are involved in making strategic management and resource allocation decisions for the Water Fund. Monitoring can allow these managing entities to make changes to programs if anticipated benefits are not being achieved as expected. There are five basic areas of monitoring questions that address major issues for Water Fund management:

1. Questions focused on tracking the implementation of activities paid for by the Water Fund:
   - » Where and when have activities been implemented?
   - » How many hectares and/or kilometers of activities have been implemented in each area and in total?
   - » How much has each activity cost per hectare or kilometer?
   - » To what extent have activities been implemented in priority areas?
   - » How many households have contracted with the Water Fund as part of a Payment for EcosystemServices (PES) program?

   These types of questions are further discussed in Chapter 8 (Implementation Monitoring).

2. Questions focused on the site (or smaller) scale, to monitor the impacts of a specific type of management activity:
   - » What effects has installation of live fences had on sedimentation rates in adjacent stream reaches? How does this compare to the effect of fencing cattle out of streams?
   - » What effect has cattle fencing had on vegetative cover regeneration?
   - » What effect has restoration of natural vegetation had on water flow?

   Impact monitoring should include Ecosystem Functions, Services and Benefits (Chapter 10); Habitats and Biodiversity (Chapter 11); and Communities (Chapter 12).

3. Questions to evaluate the impact of Water Fund activities at the watershed scale:
   - » Are changes occurring in the directions and degrees of expectations due to Water Fund activities at large scales?
   - » Are activities collectively being implemented at sufficient scopes to achieve project goals?
   - » Are activities collectively achieving the impacts predicted by computer models?

   Monitoring to assess trends is discussed in Chapter 9, and Impact Monitoring is covered in Chapters 10, 11, and 12. The use of computer models to assist with evaluating impacts and refining management strategies is discussed in Chapter 2.

4. Questions concerning new or changing environmental conditions, to explore new challenges to and opportunities for meeting project goals:
   - » Are there existing conditions (e.g., agricultural activities, roads, sources of sedimentation such as bank instability) not previously accounted for, that should be addressed by Water Fund activities?

   These types of questions can be addressed via reconnaissance monitoring (Chapter 7) and ongoing monitoring of external factors (Chapter 5).

5. Questions designed to understand factors external to Water Fund activities that may affect project outcomes:
   - » What other projects that affect water quantity and quality are operating within watersheds participating in the Water Fund?
   - » What other socioeconomic programs are working with Water Fund communities?
   - » What other factors (e.g., climate change, forest destruction, new agricultural activities) are causing change within the Water Fund project boundaries?

   Many factors that can impact watersheds exist outside the control of the Water Fund. These background land use changes, climatic variations, and macro-economic forces can impact water quality, flow, biodiversity, and socioeconomic conditions. We must monitor external factors, along with progress made toward meeting project goals, to accurately understand the impact (or lack of impact) of Water Fund activities separate and apart from other factors. See Chapter 5 for a discussion of the relevance of external factors that might impact watershed conditions.

FURTHER READING

For more information on **formulating project goals and translating goals into monitoring objectives**, see:

Adaptive Management: The U.S. Department of the Interior Technical Guide (2011).
http://www.doi.gov/initiatives/AdaptiveManagement/TechGuide.pdf

The Nature Conservancy—Water Funds: Conserving green infrastructure: A guide for design, creation and operation (Chapter 6: Evaluation, Monitoring, and Adaptive Management)
http://conserveonline.org/library/water-funds-conserving-green-infrastructure.-a/view.html

USDA Natural Resources Conservation Service—National Water Quality Handbook (Part 614, Chapter 3: Objectives)
http://www.wsi.nrcs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb1044775.pdf

USGS Patuxent Wildlife Research Center—Managers' Monitoring Guide: Adaptive Management -Integrating Monitoring and Management
http://www.pwrc.usgs.gov/monmanual/management.htm

Wood, L. (2011). Global marine protection targets: How S.M.A.R.T are they? *Environmental Management*, 47*(4)*, pp. 525-535.
http://link.springer.com/article/10.1007%2Fs00267-011-9668-6?LI=true

Chapter 2: Model Inputs and Validation

Understanding the accuracy of model inputs and results

Many Water Funds use computer models to identify spatial priorities for implementing activities, and to define expected results and returns on investments. Used in conjunction with land use and land cover maps and specific types of monitoring information (as discussed in Section 4), models can help to answer sophisticated management questions such as:

- Are management activities being implemented in sufficient scope and in the most important places to achieve project goals?
- What is the expected dose/response relationship of specific management activities and their impacts?
- Does the variety of activities being implemented represent the ideal mix for achieving project goals?

Computer modeling may have been used to develop a Water Fund's initial management plan and expected results. If empirical data from literature on the types of activities included in the models are not available at project inception, default values or estimates derived from expert opinion are often used for defining some model inputs. Monitoring data can give the models a basis for generating more accurate estimations. In all cases, providing updated data for these models will allow for ongoing calibration (adjusting future estimations based on past results) and validation (confirming model accuracy by comparing expectations to observations), ultimately producing valuable regional and site-specific information for guiding activities and for defining expectations of Water Funds.

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FURTHER READING

For information about **model application** and information on specific models, see:

AGNPS: AGricultural Non-Point Source Pollution Model
http://www.ars.usda.gov/Research/docs.htm?docid=5199

BASINS: Better Assessment Science Integrating Point & Non-point Sources
http://water.epa.gov/scitech/datait/models/basins/index.cfm

FIESTA: Fog Interception for the Enhancement of Streamflow in Tropical Areas
http://ambiotek.com/website/content/view/25/49/

InVEST: Integrated Valuation of Environmental Services and Tradeoffs
http://www.naturalcapitalproject.org/InVEST.html

RIOS: Resource Investment Optimization System—A Tool for Water Funds Design
http://www.naturalcapitalproject.org/pubs/RIOS_brief.pdf

SWAT: Soil and Water Assessment Tool
http://swat.tamu.edu/

World Bank—Modeling for Watershed Management: A Practitioner's Guide
http://water.worldbank.org/publications/modeling-watershed-management-practitioners-guide

For information about **model calibration & validation**, see:

Abbaspour, K., Yang, J., Maximov, I., Siber, R., Bogner, K., Mieleitner, J., Zobrist, J. & Srinivasan, R. (2007). Modeling hydrology and water quality in the pre-alpine/alpine Thur watershed using SWAT. Journal of Hydrology, 333, 413-430.
http://ssl.tamu.edu/media/11471/karim-swis-swat-application.pdf

Duan, Q., Gupta, H. V., Sorooshian, S., Rousseau, A. N., & Turcotte, R. (Eds.) (2003). Calibration of Watershed Models. Water Science and Application, 6. doi:10.1029/WS006.

FitzHugh, T. W. & Mackay, D. S. (2000). Impacts of input parameter spatial aggregation on an agricultural nonpoint source pollution model. Journal of Hydrology, 236, 35-53.
http://water.geog.buffalo.edu/mackay/pubs/pdfs/fitzhugh_mackay_joh.pdf

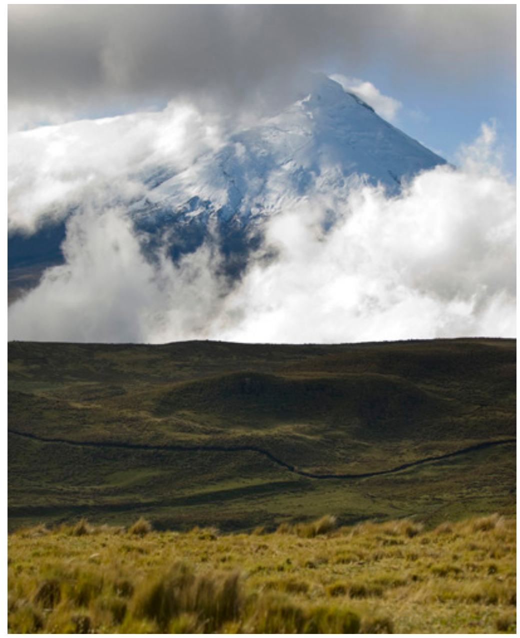
Srinivasan, R., Ramanarayanan, T.S., Arnold, J.G. & Bednarz, S.T. (1998). Large area hydrologic modeling and assessment part II: Model application. Journal of the American Water Resources Association, 34(1), 91-101.
http://www-ssl.tamu.edu/media/11971/large%20area%20dyrologic%20modeling%20and%20assessment%20part%202.pdf

White, K. & Chaubey, I. (2005, October). Sensitivity analysis, calibration, and validations for a multisite and multivariable SWAT model. Journal of the American Water Resources Association, 1007-1089.
https://engineering.purdue.edu/~ichaubey/Pubs/White_Chaubey_JAWRA_Oct05.pdf

Chapter 3: Evaluate Existing Evidence and Resources

Gathering relevant information

Some level of monitoring will always be required to assess status and trends in order to evaluate the impacts of Water Funds over time. However, prior to designing a monitoring approach to understand or verify impacts expected from a given project, it is useful to conduct a review of the body of evidence linking management activities and results.



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Assembling this evidence can provide valuable insights into the results expected, strengthen the process of goal-setting, and significantly reduce the burden of proof on monitoring for establishing causality. A review of previous work will often provide information on likely types and ranges of impacts, and the lag-times that can be expected between the time of implementation and detectable changes, among other useful information. Using information from existing research allows resources to be focused on those issues and places where monitoring is most needed; it is more important to monitor the impacts of activities that have not been extensively studied, than to collect data to illustrate cause and effect relationships that have been repeatedly documented.

A comprehensive analysis of previous studies will also allow for fine-tuning model inputs. For example, Water Fund projects often use spatial models to estimate runoff and retention results due to management activities. The model user may be required to enter parameter values that estimate the relative extent to which a given activity transitions a site, or to which a suite of activities transitions the ecosystem toward a desired state, such as natural vegetative cover. The model may also assess the relative extent to which that transition affects runoff and retention. Studies that have documented transitions in restoration projects elsewhere in similar ecosystems, given various restoration scenarios, can assist in honing these parameters, significantly improving the confidence that can be placed on both model inputs and results. Monitoring can fill gaps when sufficient information is not yet available.

Existing literature on relationships among activities, transitions, and impacts usually does not derive from the precise geographical area where the Water Fund is located. However, a review can begin by evaluating the literature based on studies conducted within the Water Fund's region and/or relevant ecosystem types, or those employing similar water management or community structures. Then, if more information is needed to proceed with monitoring design, the investigation moves "outward" from there (e.g., to Latin America, then to tropical rainforests or high-alpine grasslands and communities worldwide, then to wherever documented research has been done). Information from other areas may be limited in its applicability outside of a given region, but might be useful for establishing basic cause and effect relationships.

For example, studies done in the central Rocky Mountains of North America might be very useful for providing evidence that fencing cattle out of riparian areas improves water quality, but would be of little help for anticipating plant growth rates and lag-times for ecosystem recovery in a very different climate zone. Existing literature also provides examples of what to monitor and how, as well as analytical approaches for evaluating results. Specific information needs should be clear when reviewing literature, including data regarding the social and economic impacts of similar resource management approaches. Summaries of existing information and data gaps should be explicitly documented.

Formal systematic literature reviews and meta-analysis

A formal systematic review of available literature, including both academic and non-scholarly sources, is crucial for defining what is already known. Systematic reviews provide a synthesis of studies, information, and evidence, and commonly incorporate results from a meta-analysis. Meta-analysis is a statistical approach that combines and contrasts results from different studies to identify common patterns and differences among them, and often provides the types, weighted average, and range of effects of an activity based on a suite of studies. Systematic reviews and meta-analysis are useful for informing SMART goals (see Chapter 1), and to document the effects of activities that are under consideration in a Water Fund. A guide to conducting systematic literature reviews and meta-analysis is available from the Centre for Evidence Based Conservation (www.cebc.bangor.ac.uk), and a library of such reviews is maintained through the Collaboration of Environmental Evidence (www.environmentalevidence.org).

An initial literature review of ecological research concerning activities similar in type and scale to those being implemented by many Water Funds has been compiled, and is available in the Appendix, along with a link to a searchable list of the topics and content of cited studies. In addition, Chapter 12 contains a preliminary literature review concerning the impacts of payment or compensation for ecosystem services (PES/CES) programs, including links to resources that may be of assistance.

Local resources for research information should not be overlooked. Talk with university scientists in your area—especially those who are already working with Water Fund projects or have expressed interest in doing so—to see if you can obtain relevant information or referrals to other experts that might help address your needs. There is often a great deal of first-hand experience and knowledge that has not been published. Caution should be taken when relying on "expert opinion" that has not undergone peer review, but it can prove to be a valuable resource.

Identify and assess existing monitoring efforts

It is important to explore and understand potential uses of existing monitoring occurring in the project area. The majority of monitoring necessary for evaluating ecosystem services and benefits may already be in place, and if not, portions of the Water Fund's information needs may already be filled. Monitoring is typically conducted on a regular basis by water utilities, hydropower companies, water resource management agencies, municipalities, and a range of government agencies. Identify the regulatory requirements for monitoring and reporting in the area in which the Water Fund is active to understand what information may be readily available.

Monitoring for research purposes is also conducted by academic, government agency, and non-profit organization scientists. While data from existing efforts may only partly address the questions and needs of a project, an assessment of what is available should be carried out in order to identify the gaps that need to be filled by the Water Fund's monitoring program. Knowing the parameters, locations, frequency, and methods used for existing monitoring efforts will also provide opportunities to explore partnerships to include additional parameters or data collection sites by providing financial support or other assistance.

When Water Fund projects are being developed, defining information needs must be part of the initial dialogue and Water Fund planning, rather than an afterthought. Often, public and private entities and non-profit organizations are collecting and assessing data that would address monitoring needs, but information is not readily available because of privacy or data-sharing issues. Creating a data- sharing and privacy Memorandum of Understanding or other formal agreement to allow access to data, or to allow specific analytical results to be provided at the initiation of the project, should be a priority for assessing data availability and filling information gaps.

Identify access to experts and capacity

Experts in environmental, social and economic sciences should be involved at all stages of monitoring design, implementation, and analysis. Local, regional and global experts can provide a range of perspectives and skills, as well as access to partnerships and information. Local experts are familiar with existing data collection, and opportunities to partner with others in the area to fill data gaps, manage information, design monitoring, conduct sampling, and conduct analyses. Monitoring should be seen as a cooperative process aimed at including the technical and scientific expertise required for sufficient rigor (see Chapter 6 for a discussion of statistical rigor). It cannot be stressed enough that statisticians be consulted *before* data collection, to ensure that selected monitoring designs allow for appropriate statistical analyses, and that sufficient sample replication is included to address expected background variations in parameter values.

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Section Two: Concepts and Considerations for Monitoring

Data collection and analyses should provide the information necessary to answer questions in a defensible way. This does not mean that more data are better. Data need to be collected and analyzed to take into account environmental and socioeconomic dynamics to clearly illustrate and demonstrate causality for results. There are a variety of basic concepts and considerations that need to be understood in order to develop appropriate monitoring and analytical approaches to address information needs.

Chapter 4: Lag-times and Spatial Scales

Results don't happen instantaneously

Environments respond to management activities at varying spatial and temporal scales. Understanding these differences is critical to creating appropriate expectations for results, and designing monitoring to deliver suitable information within desired timeframes. Watershed projects often fail to meet expectations because of unaccounted for lag-times –the time between the implementation of activities and measurable changes in the environment. Even when activities are well-designed and fully implemented, monitoring efforts may not show definitive results if monitoring approaches fail to account for the lag-time between activities and responses. The main components of lag-time include the time required for: a) an activity to produce an effect; b) the effect to be delivered to the environment; c) the environment to respond to the effect, and d) the effectiveness of the monitoring program to measure the response (Meals et al., 2010).

Lag-times are influenced by the type and scale of activities, the scale and magnitude of the issues being addressed, and the environments in which activities are implemented. Some activities that address local and point sources of pollution and habitat degradation, such as livestock exclusion or animal waste management taking place on small streams, are associated with relatively rapid environmental responses (less than one year to several years). However, it can take decades to detect the impacts of activities that address non-point and broader landscape sources of pollution and habitat degradation, such as large-scale land cover vegetation restoration and erosion control taking place in larger watersheds (Meals et al., 2010).

In general, limited point sources of water pollution can have relatively short residence times in stream systems (hours to weeks), dissolved nutrients from non-point sources such as agricultural runoff have intermediate residence times (hours to months for runoff, years to decades if associated with groundwater), and sediment and associated nutrients and other pollutants from non-point runoff can have residence times lasting decades, depending on location and watershed dynamics.

Environmental conditions also affect lag-time. For example, vegetation grows more slowly in colder environments, and changes in land cover and subsequent changes to the freshwater environment will take longer compared to warmer environments. The rate and magnitude at which water flushes stream systems also affect lag-times for effects on sediment and nutrient concentrations, because flushing magnitudes and rates affect loading, transport, and residence times.

These distinctions in expected lag-times are critical for defining monitoring approaches. Site-based monitoring may provide indications of the types and degrees of benefits of activities within a short timeframe, while watershed-scale monitoring, while necessary for evaluating Water Fund effects as a whole, will take longer to detect changes, and will be subject to more external factors influencing results.

FURTHER READING

For the original source of much of the information in this chapter, and a review of **lag-times in water quality responses to management practices**, see:

Meals, D. W., Dressing, S.A., & Davenport, T. A. (2010). Lag time in water quality response to best management practices: A review. Journal of Environmental Quality, 39, 85–96. doi:10.2134/jeq2009.0108
http://ebookbrowse.com/gdoc.php?id=73575476&url=99e90aac15f6873643d06e46ae1234b4

Chapter 5: Controls and External Factors

Understanding changes from sources other than managed activities

The inclusion of controls in study design is critical to isolating the effects of Water Fund activities and understanding the extent to which these activities contribute to any observed changes. A control is selected to reflect what would have happened in the absence of a management action, policy, or program. A control site, control watershed, or control group ideally starts with the same or very similar characteristics as the impact site, watershed or group (that is, the site, watershed, or group expected to be affected by Water Fund activities), and controls experience the same or very similar circumstances over time, with the one exception of being subject to Water Fund activities. In this way, researchers isolate causation from correlation and happenstance.

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Identifying appropriate controls at the watershed scale is often challenging, if not impossible. For example, in the two-watershed design, in which an activity or set of activities is being implemented in one of the watersheds, the watershed in which no activities are being implemented is often erroneously considered a control watershed. A watershed can only be considered a true control if it has been evaluated ***prior to implementing activities*** for similarity to the impact watershed in terms of characteristics such as watershed size, elevation, stream density and gradients, geology, climate, and patterns of land use/cover, water quality, and flow. Lacking such evaluation, there is no supporting evidence to define the watershed as a valid control. When an appropriate control watershed cannot be identified, alternative designs can be used, as discussed in the Impact Monitoring chapters.

Similarly, in order to select a control group to judge the impacts of Water Fund activities on human well-being, factors such as socioeconomic status and land ownership patterns (e.g., community/single-owner deed) might be considered. The control and impact communities should be as similar as possible in terms of these and other relevant characteristics before Water Fund activities are initiated. Otherwise, any differences found between the communities may be due to pre-existing differences, rather than to the impacts of the Water Fund. Chapter 12 offers an approach for comparing communities that differ in some characteristics prior to project implementation; however, it is essential to have baseline data for socioecomomic indicators from all control and impact communities.

If we can isolate the cause, and estimate the magnitude of the change caused by activities, we can demonstrate that Water Fund activities are making a difference, and describe the extent to which that difference is occurring over time. The latter is critical in light of changes in climate and land use patterns. The watersheds where Water Funds are operating will not necessarily remain static, even in areas where activities are not being implemented. Tracking external factors potentially influencing project outcomes is important for all watersheds that are part of Water Funds, as well as for those used as controls or references.

Tracking external factors is especially important where adequate controls cannot be defined and monitored. Such information can be used to model what would have occurred with "business as usual," and illustrate what contributions Water Funds have made. Changes in land use and land cover, human population size and distribution, and climate are basic attributes that should be monitored over time. Additional information, such as changes in management practices within land use classes, patterns of water extraction, fertilizer application rates, cattle densities, and resource extraction should also be monitored if data are available. For socioeconomic studies, it is similarly essential to monitor for factors other than Water Fund activities that may be impacting either participating or control communities, such as financial assistance programs or education projects external to Water Funds.

Monitoring external factors and comparing impact and control areas can both be critical to demonstrating project success. For example, it is possible that for some Water Funds, base-flow may decline over time due to factors beyond the control of the Water Fund. It is important to provide evidence that demonstrates the difference that Water Fund activities have made, and indicates how much worse the situation would have been if those activities had not been implemented. For instance, if base-flow decreases by five percent over ten years in a Water Fund project watershed, and if changes in climate, land use, land cover, and water extraction in the impact watershed had been tracked during that time period, we could model expected changes and use data on external factors to illustrate that in the absence of Water Fund activities (business as usual), an even lower base-flow would have existed. If additional monitoring data could be presented indicating that, for example, base-flow had decreased by 20% in control watersheds during the same time period, this would provide additional and stronger proof of the value of Water Fund activities.

Both careful selection and monitoring of control sites (and groups), and adequate tracking of external factors potentially affecting project outcomes, are important for demonstrating the effectiveness of Water Fund projects. These data are critical components of a monitoring design that can comprehensively assess Water Fund impacts, providing reliable feedback for adaptive management, and allowing communication of results to investors and others.

FURTHER READING

For information on **modeling potential outcomes in the absence of Water Fund activities**, see:

Morgan, S. L. & Winship, C. (2007). Counterfactuals and causal inference: Methods and principles for social research. Cambridge, United Kingdom: Cambridge University Press.
http://ebooks.cambridge.org/ebook.jsf?bid=CBO9780511804564

Chapter 6: Statistical Rigor

Ensuring defensible data

Considering the level of rigor required is critical when designing a monitoring strategy. Rigor takes into account the resolution, precision, and accuracy of monitoring techniques, as well as the strength of inference of the monitoring design. The ideal level of rigor depends on the questions that are being asked, the uses of the results, the natural background variation, and the magnitude to which monitored changes are expected to occur, as well as available capacity and funding. A statistician should always be consulted when monitoring parameters and approaches are considered, so that appropriate sampling design, replication, and analysis methods can be selected in advance. This will ensure that valuable time and resources are not wasted on a design that is not capable of detecting a significant effect.

Monitoring can be designed to make a variety of comparisons, resulting in different analytical strengths. Cottingham et al. (2005) provide eight alternative monitoring frameworks to evaluate environmental responses to environmental flow management, listed in increasing strength of inference. These designs are applicable to a broad range of monitoring needs.[2]

1. ***Impact-only design***. An activity has already been implemented (no before-activity data exist) and there are no spatial controls or reference sites, watersheds or groups being used for comparison; monitoring is limited to the site, watershed, or group where activities have been implemented. These responses can be evaluated against specific predictions based on the conceptual model. Causal links between temporal changes in responses are difficult if not impossible to determine because the changes might have occurred without the activities being implemented.
2. ***Reference–Impact design.*** A modification of (1) above, where there are no before-activity implementation data but the same parameters are monitored through time in a reference and impact site, watershed, or group, which represents the desired direction of change for the impact. This design provides slightly better evidence for a causal link between temporal change in response, because natural changes through time can be measured at reference sites as well. It is also possible to assess whether the trend of change at the impact location is towards the reference condition, if that is desired.
3. ***Control–Impact design***. Similar to (2) above except that comparison is with a control site, watershed, or group. This design provides stronger inference about causality because comparison with the spatial control reduces the likelihood that effects from activities are statistically confounded with natural change.
4. ***Control–Reference–Impact design***. This is a combination of (2) and (3) above. Statistical analyses test for divergence in temporal trends between the impact and the control, and for convergence in temporal trends between the impact and the reference site, watershed, or group. This design provides causal strength similar to (3), with the added advantage of assessing whether the trends are moving toward reference conditions, if that is desired.

2 The list from Cottingham et al. has been slightly modified for consistency with the terminology used in the current document.

5. ***Before–After–Impact design***. This is a standard "impact analysis" design comparing parameter values before versus after activities have been implemented. The "before" data provide baseline or temporal control conditions. Evidence for causal links is limited by lack of spatial controls, therefore it is unclear whether or not the change would have occurred independently of the activities being implemented. This design is also difficult to use if activities are implemented gradually, if there is a long lag-time for impacts to occur, or if the difference that occurs is not large.
6. ***Before–After, Reference–Impact (BARI) design***. This is similar to (5) but with a spatial component –a reference site, watershed or group that provides some measure of whether natural changes coincide with changes seen in the impact site. This design also allows assessment of whether the trend of a response is towards the reference condition. The test of interest is whether any before-after difference at the impact location is the same as at the reference location. The causal inference associated with this design is limited because the reference and impact sites, watersheds, or groups have different conditions prior to activity implementation. This makes it difficult to rule out a response to other factors coinciding with the start of the implementation of the activity.
7. ***Before–After Control–Impact (BACI) design***. Similar to (6), but using a spatial control instead of a reference. This design provides strong inference about causality because comparisons with spatial and temporal controls reduce the likelihood of confounding effects with natural spatial and temporal changes.
8. ***Before–After Control–Reference– Impact (BACRI) design***. A combination of (6) and (7) that provides strong evidence for causal links between activity and response, and also measures whether the change is towards reference condition, if that is desired.

Note that for all of these designs, inclusion of replicates improves the validity of control–impact contrasts.

Resolution refers to the scale with which measurements are taken; for example, a measurement to the nearest millimeter has higher resolution than a measurement to the nearest meter. Resolution is affected by choice and/or the monitoring technique. Precision refers to the ability to reproduce the result on the same sample. *Precision* is affected by the monitoring technique and sources of human error, and should not be confused with accuracy. *Accuracy* reflects the degree to which the measured observation reflects the true value, regardless of the precision of that measurement or bias of the technique or observer. Accuracy can be affected by the monitoring approach and technique, as well as human error. The selected levels of resolution and precision can affect the ability to measure parameters that require low detection limits, or to discern changes that are not abrupt or large. Precision and accuracy can strongly impact the results and confidence of scaled-up modeled expectations. The ramifications of different levels of resolution, precision, and accuracy should be weighed in consultation with appropriate experts when selecting monitoring approaches and techniques.

With adequate understanding of the natural variation associated with a given parameter spatially, seasonally, and over time, and an understanding of the probability distributions of alternative hypotheses, a statistician can design a monitoring approach that includes sufficient timing, spacing, and number of sample replicates, and appropriate statistical analysis, thereby reducing the potential for generating false assumptions through statistical errors. Parametric statistical methods (those that have assumptions about probability distributions) and non-parametric methods (those that do not make such assumptions) should be considered.

Projects may demand greater rigor in their monitoring strategy if management decisions require it, such as for model Water Funds being used to illustrate the benefits of an approach to broad audiences, or if participants, investors, or donors request quantitative, scientifically defensible monitoring results. Greater rigor may also be required for testing the efficacy of certain activities to obtain suitable support for implementation, or to provide higher confidence in input values for models used to estimate the basin-wide impacts of protection and restoration activities. Greater rigor is achieved through sampling designs that provide structure and replication for appropriate statistical analyses. It is imperative that a statistician is involved with monitoring design decisions, since statistical approaches and their requirements should be defined ***before*** designing a monitoring approach. Factors that must be weighed in making design decisions include natural variation, expected measured differences and variance in response attributes, and lag-times.

Less rigorous approaches are sometimes used to illustrate changes that are unequivocal, for results that are expressed categorically (impact or no impact, rather than degree of impact), and for initial surveillance monitoring where the focus is to identify significant issues that need to be addressed or that require further study (see Chapter 7). Photography is a common component of these approaches, and can communicate dramatic changes in a simple way. These approaches can also be used as part of a community-based monitoring program, wherein members of local communities are engaged in collecting and analyzing data. These programs can be indispensable for education, sustaining support for Water Fund projects, gaining political support, and illustrating first-hand the benefits of participants' efforts. However, consideration should be given to the inherent limitations of these methods in terms of detection limits, accuracy, and precision, and the corresponding potential restrictions on using these data. There are a variety of existing organizations and training modules that can be assessed for inclusion in a community-based component to a monitoring program. Links to resources can be found in the Impact Monitoring section.

FURTHER READING

For more information on designs and **statistical approaches** for monitoring, see:

Cottingham, P., et al. (2005). Environmental Flows Monitoring and Assessment Framework. Technical report. CRC for Freshwater Ecology, Canberra.
http://freshwater.canberra.edu.au/Publications.nsf/0/b217ed362dcbc90bca256fc7001cf693?OpenDocument

Hurlburt, S. H. (1984). Pseudoreplication and the design of ecological field experiments. Ecological Monographs, 54(2), 187-211.

Quinn, G. P., & Keough, M.J. (2002). Experimental design and analysis for biologists. Cambridge, U.K.: Cambridge University Press.

Sokal, R.R., & Rohlf, F.J. (1995). Biometry: The principles and practice of statistics in biological research (3rd ed.). New York: W.H. Freeman.

US EPA—Guidance for Choosing a Sampling Design for Environmental Data Collection
http://www.epa.gov/quality/qs-docs/g5s-final.pdf

USDA Natural Resources Conservation Service—National Water Quality Handbook (Part 615, Chapter 6: Hypothesis Testing)
http://www.wsi.nrcs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb1044775.pdf

USGS—National Analytical Methods Index
http://cida.usgs.gov/nemi/sams/search/

Zar, J.H. (2009). Biostatistical Analysis (5th ed.). Upper Saddle River, N.J.: Prentice Hall.

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Section Three: Basic Monitoring Approaches

Monitoring information helps project teams understand the types and sources of issues that need to be addressed through activities, and the degree to which activities are being implemented to address those issues. Tracking status and trends over time allows projects to evaluate whether changes are occurring in the direction and magnitude that are desired, and provide a way to assess progress towards achieving objectives.

Chapter 7: Reconnaissance Monitoring

Becoming familiar with the project area

Reconnaissance monitoring is commonly used to identify sources of factors affecting water quality and flow that are not easily assessed using remote land use/land cover data. These may include assessment of stream channel condition and bank erosion hazard, confined animal feedlot operations and other animal production facilities, point- and small-scale sources of pollution, and on-the-ground vegetation patterns and soil conditions, among others. It can also be used to obtain initial data on patterns of water quality, flow, habitat and biodiversity, to inform longer-term monitoring methods and designs. Reconnaissance monitoring is not an adequate replacement for a scientifically designed impact monitoring approach, but can provide baseline data for a monitoring program when appropriate methods are used, or serve as an initial assessment that would lead to more formal data collection efforts.

Reconnaissance monitoring is generally done as a one-time process using rapid assessment protocols that indicate qualitative and categorical status, or coarse quantitative results that are useful for characterizing initial status but are generally not sufficiently precise to thoroughly define baseline conditions or detect trends. However, well-documented rapid assessments can be used to evaluate the dynamics of very active sites.

When to use this approach:

- When monitoring data indicate that existing spatial data fail to identify issues and problematic areas, and reconnaissance methods can be used to identify and diagnose the problems
- When existing data are limited or do not exist, and a one-time monitoring survey can provide valuable information on the status of lands and waters

Assumptions:

- Monitoring approaches are adequate to identify patterns and sources of problems with rapid, one-time sampling.

Advantages:

- Reconnaissance monitoring is generally less expensive than long-term monitoring.
- Reconnaissance monitoring can identify problems and areas for treatment that are not identifiable using spatial data tools.
- Reconnaissance can be done at any time, depending on the issues and monitoring approaches taken.
- When there is little knowledge of a stream system or watershed, reconnaissance can be a tool to get people into the field to see the situations directly.
- Sampling design is flexible and can be customized according to the need.

Limitations:

- Reconnaissance monitoring is generally not sufficiently precise to define baseline conditions or use in trend monitoring.
- Reconnaissance monitoring may miss important information.
- Reconnaissance monitoring does not generally capture seasonal, annual, or longer-term information.
- Judgment and logistics may play a large role in selecting sample locations, which can bias the data collected, giving poor representation of actual conditions .
- Limited attention paid to sampling designs and replication can result in limitations to statistical analyses.

Statistical Approaches:

Because data are commonly observational, statistical evaluations can be limited or inappropriate, and general comparisons are often used. Statistical approaches will depend on sampling designs.

Methods for reconnaissance monitoring

Water quality reconnaissance monitoring is commonly accomplished using in-field test kits that have limited resolution and are only capable of measuring a few parameters. However, these can suffice to identify significant levels of contamination. Alternatively, sampling and analysis by certified laboratories is used to provide broader parameter analysis and increase resolution, accuracy, and precision. Common parameters that are sampled and evaluated include: pH, specific conductance, temperature, and concentrations of nitrogen and phosphorus compounds, major ions, organic carbon, suspended sediment, herbicides, and *E. coli.*

A basic guide to **water quality monitoring** that can be conducted with field-based test kits can be found at:

U.S. EPA Monitoring & Assessment—Water Quality Conditions
http://water.epa.gov/type/rsl/monitoring/vms50.cfm

Stream habitat reconnaissance monitoring commonly uses a combination of visual (qualitative) and quantitative assessment methods to evaluate riparian and stream habitat conditions and identify proximal and upstream land-based issues that are affecting stream channel stability, bank erosion, and in-stream habitat. Stream habitat reconnaissance monitoring is also needed to identify appropriate in-stream monitoring sites for water quality, flow patterns, and freshwater biota.

Several commonly used **stream reconnaissance** monitoring methods can be found at:

U.S. EPA Monitoring & Assessment—Stream Habitat Walk
http://water.epa.gov/type/rsl/monitoring/vms41.cfm

USDA Natural Resources Conservation Service—National Biology Handbook: Stream Visual Assessment Protocol Version 2
ftp://ftp-fc.sc.egov.usda.gov/ID/technical/svap.pdf

Stream biodiversity reconnaissance monitoring assesses invertebrate and fish assemblages and presence/absence of indicator species. The level of sampling rigor and taxonomic distinctions used influences the accuracy and resolution of the information. Examples of protocols with a range of levels of rigor can be found at:

Connecticut Department of Environmental Protection—Rapid Bioassessment in Wadeable Streams and Rivers by Volunteer Monitors
http://www.ct.gov/deep/cwp/view.asp?a=2719&q=325606&deepNav_GID=1654%20

Encalada, A.C., Rieradevall, J., Ríos-Touma, B., García, N. & Prat, N. (2011). Protocolo simplificado y guía de evaluación de la calidad ecológica de ríos andinos (CERA-S). Universidad San Francisco de Quito, Universidad de Barcelona, AECID, and Fondo para la Protección del Agua (FONAG), Quito, Ecuador.
http://www.libreroonline.com/ecuador/libros/29660/prat-fornells-narcis-garcia-katchor-natalia-rios-touma-blanca-encalada-romero-andrea-carolina-ri/protocolo-simplificado-y-guia-de-evaluacion-de-la-calidad-ecologica-de-rios-andinos-cera-s.html

U.S. EPA—Rapid Bioassessment Protocols for Use in Streams and Wadeable Rivers: Periphyton, Benthic Macroinvertebrates, and Fish
http://water.epa.gov/scitech/monitoring/rsl/bioassessment/index.cfm

U.S. EPA—Rapid Biological Assessment Protocols: An Introduction
http://cfpub.epa.gov/watertrain/pdf/modules/rapbioassess.pdf

A summary of different **stream habitat and biodiversity reconnaissance monitoring approaches** with links to resources can be found at:

U.S. EPA—Reviews of Representative Stream Assessment and Mitigation Protocols
http://water.epa.gov/lawsregs/guidance/wetlands/upload/Part2_Reviews.pdf

Terrestrial habitat reconnaissance monitoring evaluates patterns of vegetation and soil conditions to identify sources of runoff, and needs and opportunities for restoration. To assess environmental function it is important to know vegetation patchiness, patterns of exposed ground, and indicators of soil erosion and disturbance. For terrestrial vegetation condition, it is important to know species composition, and age/size structure and distribution. There are a variety of vegetation and terrestrial habitat assessment approaches used for reconnaissance monitoring, as well as for assessing status and trends. Rapid visual assessments to identify major issues and opportunities are recommended for reconnaissance monitoring. The characteristics that should be evaluated will be specific to the ecosystems, land uses, and stressors. Methods can be adapted to be more applicable to specific situations; for instance, a simple assessment of the proportion or locations of bare ground or signs of soil erosion can provide simple indications of potential sources of runoff.

A review of common **vegetation monitoring** methods to identify appropriate reconnaissance approaches can be obtained at:

Colorado State University—Vegetation Measurement Methods Comparison
http://www.cemml.colostate.edu/assets/pdf/methods.pdf

An example of approaches developed to assess **soil and site stability, hydrologic functions, and biotic integrity for rangelands** in the United States provides useful concepts for developing characteristics for rapid assessment that can be adapted for specific environments. It can be found at:

U.S. Bureau of Land Management—Interpreting Indicators of Rangeland Health
http://jornada.nmsu.edu/sites/jornada.nmsu.edu/files/IIRHv4.pdf

Chapter 8: Implementation Monitoring

Assessing the execution of Water Fund activities

Implementation monitoring records the outputs of the Water Fund, such as the number of families enrolled in a Payment for Ecosystem Services agreement, or the hectares of land that are being restored through a specific type of activity. Tracking Water Fund activities provides information necessary for modeling expected impacts based on progress made; evaluating relationships among activities and results observed via impact monitoring; guiding adaptive management; understanding the cost-effectiveness of specific management strategies; and reporting progress.

The Water Fund's implementation goals are often defined through the use of computer models that prioritize areas for project implementation. Spatially explicit models can provide solutions for optimizing impacts from a suite of activities, and quantify the expected collective impacts from those activities. While optimal solutions may not be implemented completely, it is important to know the specific places where activities have been implemented, both optimally and otherwise. This information is necessary to evaluate how efficiently and effectively activities are being implemented, and to estimate the impact of Water Fund activities over time.

In order to provide accurate inputs for these models, as well as to properly design impact monitoring approaches, it is critical that all Water Funds track the specific information about each activity implemented or planned. Spatial tracking of project outputs should be included in a Geographic Information System (GIS), so that actual implementation can be compared to initial spatial priorities, allowing projections of expected results to be updated, and providing a resource for adaptive management of Water Fund priorities.

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The data collected to track project implementation will be specific to each Water Fund's activities, but the following information should typically be included:

1. The spatially explicit and geo-referenced location of each current (and planned, to the extent possible) activity (polygon, linear or point location). Locations can be extracted using existing spatial data. Locations of some activities will require delineation by walking the perimeter of the activity using a GPS device and transferring data to a GIS.
2. The spatially explicit length of stream habitat that is within or proximal to the activity that is expected to benefit as a result of the activity
3. The type of activity in each location
4. The area (hectares) or length (meters) of implementation at each site
5. The date at which implementation was started and completed at each location, and record of any additional maintenance activities necessary
6. The cost for implementation at each location (cost/unit area), maintenance costs per activity, and total cost per watershed/Water Fund
7. The total area and length of implementation per activity in the watershed/Water Fund
8. The total length of stream habitat expected to directly benefit from land-based activities in the watershed/Water Fund
9. Land tenure for public lands and private lands where an activity has taken place or is planned.
10. Number of households participating in each type of project
11. Payments to participants
12. Data on the sources of implementation information

Data on the scope of activity implementation can be collected using a variety of approaches, ranging from self-monitoring and reporting by participants, participatory monitoring by groups of providers or interested parties, or through using remote sensing.

FURTHER READING

For more information on **implementation monitoring**, including suggested protocols for recording observations, see:

Porras, I., Alyward, B. and Dengel, J. (2013). Monitoring payments for watershed services schemes in developing countries, IIED, London
http://pubs.iied.org/16525IIED

USDA Natural Resources Conservation Service—National Water Quality Handbook (Part 614, Chapter 12: Land Use and Management Monitoring)
http://www.wsi.nrcs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb1044775.pdf

Chapter 9: Trend Monitoring

Assessing impacts over time

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Trend monitoring is a critical component of all Water Funds. Trend monitoring assesses *changes* in ecosystem functions, provision of services and benefits, and condition of habitat and biodiversity, as well as climate, land use, and other potential confounding external factors. Trend monitoring focuses on a core set of cost-effective indicators that can be monitored over time.

Trend monitoring requires a long-term dedication of capacity and funding for data collection. Data collection does not need to be continuous for all parameters, but needs to be scheduled to collect information at appropriate intervals. Appropriate intervals are determined based on factors such as expense, as well as the natural variability of and lag-time expected for the parameter of interest. Trend monitoring alone is a form of data collection, rather than an experimental approach, and is not designed to define the sources of changes; it is solely a measure of change over time. Trend monitoring can however form an integral component of sampling designs that seek to establish the causes of observed changes.

Trend monitoring is necessary to aid understanding of whether ecosystem functions are changing in the desired direction and magnitude in response to Water Fund activities, given other factors influencing the ecosystem such as climate and shifts in land use and land cover in the watershed. Long-term monitoring of water quality, flow, and habitat and biodiversity conditions, coupled with weather station data and computer modeling of future scenarios of climate, land use and water supply demands, is essential to adaptively managing Water Fund projects over time. Trend monitoring is not required when the reason for data collection is to assess two distinct points in time, such as once before and once after activity implementation.

When to use this approach:

- For tracking ecosystem characteristics that are expected to change gradually, such as sediment, nutrients, and flow
- For tracking ecosystem services and benefits, when appropriate (see Chapter 10)
- For tracking habitat and biodiversity characteristics that are expected to change gradually over time, such as changes in channel morphology, high-elevation vegetation, and fish assemblages
- As part of a sampling design focused on detecting changes that are expected to occur over long time periods

Assumptions:

- There is no assumption about cause and effect.
- Data collection interval is sufficient to reflect trends over the time period of interest, given inherent variability in the measurements.

Advantages:

- Monitoring for all parameters does not need to be continuous, but needs to provide a constant, uninterrupted dataset at regular intervals, as appropriate for monitoring the trend of interest. Seasonality should be taken into account.
- Since changes are expected to occur over extended periods of time, short-term intensive monitoring is not necessary, making trend monitoring generally less costly than experimental approaches.

Limitations:

- A long-term commitment must be made to provide funding and capacity for monitoring a constant set of indicators.
- Because trend monitoring does not represent an experimental approach, trend data alone do not provide evidence that Water Fund activities are the causes of observed changes.

Common Statistical Approaches

Trend monitoring can be used to plot changes over time to communicate simple results to a broad audience. There are a variety of statistical approaches for evaluating whether a trend exists at a single station, including least squares fit regression, comparison of annual means, cumulative distribution curves, quartile plots, double mass analysis, time-series analysis, and Seasonal Kendall test, among others. Paired regressions can be used to evaluate differences between an impact and a control (or reference) watershed.

FURTHER READING

For a detailed discussion of appropriate statistical approaches, see:

USDA Natural Resources Conservation Service—National water quality handbook
(Part 615, Chapter 12: Trend Analysis)
http://www.wsi.nrcs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb1044775.pdf

USGS Patuxent Wildlife Research Center—Managers' Monitoring Guide: Measuring Long-Term Population Change
http://www.pwrc.usgs.gov/monmanual/longtermchange.htm

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Section Four: Impact Monitoring

Given limited funding and capacities, it is critical to focus monitoring to assess changes resulting from activities to capture the most relevant information and conduct the right kind of analysis. A successful monitoring program is not necessarily one that collects a lot of information, but one that requires that the right kind of information is efficiently and effectively gathered and analyzed to address management needs and judge progress toward meeting near- and long-term Water Fund goals. As discussed previously, monitoring should focus on answering the questions posed by project goals and required for making management decisions, and must provide adequate and defensible results. Data needs should also be well-defined in terms of how the information will be analyzed, presented, and used. Any additional information under consideration for collection should be carefully evaluated in terms of what the data would be used for, and whether the benefits of collecting more information outweigh the costs.

The attributes selected for the Water Fund monitoring program should be well-defined and consistently monitored on a regular schedule by professionals using a sampling design and statistical procedures approved by a statistician. In most cases long-term monitoring is required, since there can be significant lag-times between project implementation and detection of change, especially in larger watersheds. In addition, it is important to recognize that the natural variation of flowing systems causes significant background "noise" and seasonal patterns in flow, sediment, and nutrient concentration data. Thus, finding statistically significant differences between baseline (or control site) data may take many years, particularly at the large watershed scale most relevant to Water Fund investors.

Given limited resources for monitoring in many Water Funds, ***attributes*** (the entity you are interested in studying, such as flow) and ***parameters*** (measureable properties whose values determine the characteristics of the attribute) should be selected based not only on relevance to project goals, but also on the likelihood of detecting a statistically significant change in a particular attribute, given its natural variability and the expected degree of impact of management actions. As discussed in Chapter 6, an appropriate sampling design can increase the likelihood of detecting change, but if an attribute is highly variable and management actions are not likely to have an impact that will exceed that background variability, it is usually advisable to select an alternative parameter (or set of parameters) for monitoring.

This section presents an overview of the types of data that may commonly be collected to assess the impacts of typical Water Fund activities. Individual chapters are provided for each theme, where you will find an overview of several conventional sampling designs, as well as a discussion and suggestions for attributes and parameters for Water Fund monitoring. These include both ***primary*** parameters, which directly reflect the changes in the attribute you are seeking to affect, as well as other data needed to accurately interpret results. Additional parameters may be selected by each Water Fund depending on more specific information needs and available resources for data collection and analysis.

Basic information is also provided for possible data collection methods. As with the choice of which attributes and parameters to monitor, the selection of sampling methods will be dictated by the Water Fund's specific information needs, as well as on the monitoring budget and technical capacity. These cursory presentations of methods are intended only as examples. Appropriate sampling methods should be identified through research and consultation with subject experts.

Many types of data, such as water quality data that will be used in computer models, normally require collection by qualified technicians and analysis by certified laboratories. However, dependent upon the sampling design (discussed in the following chapters), it is likely that laboratory-led data collection will be necessary only in a subset of the watersheds where the Water Fund is working. In the remainder, and/or alongside more sophisticated monitoring, it may be helpful to include some level of community-based monitoring. These methods are often significantly less accurate than professional-level data collection, but they are also much less costly, allowing them to be conducted more frequently and at more sites.

The data from community-based monitoring also gives tangible, often immediate feedback on the benefits of Water Fund activities directly to the community. In addition, community members can be trained to assist in data collection efforts that may be led by laboratory technicians or university researchers, for example by downloading flow data from electronic loggers, or serving as field assistants for collecting biodiversity data or conducting surveys. In addition to its educational value, directly involving the community in monitoring can inculcate more interest in Water Fund activities and increase commitment to project goals. Comparison studies of community-based and professional monitoring results are suggested to elucidate any critical differences in the data obtained.

Chapter 10: Impact Monitoring for Ecosystem Functions, Services, and Benefits

Assessing changes resulting from activities

Ecosystem services are the primary focus of most Water Fund investments. These investments are intended to protect and improve the quality, quantity, and flow patterns of the freshwater supply, which in turn provides ecosystem services and benefits to water users (and for some Water Funds, reduced risk of flooding). This chapter focuses on monitoring the water supply throughout the watershed, and the downstream services and benefits accrued due to Water Fund activities.

For monitoring ecosystem services, it is helpful to differentiate three steps along the supply chain from nature to human beneficiaries: ecosystem ***functions, services,*** and ***benefits***. (See Figure 1 for a graphic depiction of these steps.)

Overview: Ecosystem functions, services, and benefits

Ecosystems provide important ***functions*** that yield services and associated benefits to people. In terms of freshwater, these functions can be thought of as either "provisioning" or "regulating." The effects of management actions on *provisioning* functions, such as increased stream base-flow, can be directly monitored in streams (assuming external factors are monitored as well). However, the effects of management actions on *regulating* functions, such as increased retention of flood waters and sediment, and assimilation of nutrients and bacteria on the landscape, present a challenge for monitoring because they cannot easily be directly observed and quantified. What is commonly monitored is not retention *per se*, but rather the opposite: runoff and erosion. Observing sediment levels in streams provides an indirect indication of the retention capacity in the watershed, but it does not tell us directly how much sediment has been retained on the landscape by vegetation, soils and management activities. Computer models are often used to estimate retention functions and the changes to these functions resulting from Water Fund activities. Monitoring ecosystem functions can provide information to calibrate and validate model outputs, strengthen confidence in model inputs (if necessary), gauge the relative effectiveness of specific management practices, and detect overall effects and trends in ecosystem functions resulting from Water Fund activities.

Ecosystem services *derive from ecosystem functions*, and are monitored at relevant points of use. Services concern the provision of resources that people use (e.g., water volume at drinking water extraction points), or the regulating effect of ecosystem functions on the condition of those resources (e.g., reduced sediment levels at reservoir inflows). Ecosystem ***benefits*** represent the contribution of these services to human well-being, expressed in monetary or non-monetary terms. Benefits from provisioning services, such as the volume of water for the municipal water supply, are directly consumed by people. The significance of a change in this service to people's well-being is determined by water scarcity and the ability for people in the area to obtain water from other sources. For example, if water is locally ample, then increasing the water supply will not accrue in much human benefit, since all needs are already met. As well, if water is locally scarce, but people can cheaply buy bottled water, increasing local water supply will not provide a large benefit to human well-being, and any monetary value associated with this benefit would be low. However, if water is scarce, and alternatives are either unavailable or expensive, then increasing local water supply will provide a large benefit, including a potentially high monetary value.

Similarly, a regulating service, such as retention leading to reduction of sediment at points of use, only provides significant direct benefits when a certain threshold is crossed. For instance, if forest restoration reduces the amount of sediment loading in the watershed, and sediment is decreased sufficiently downstream to reduce the need for water treatment, then there is a direct measureable benefit associated with the regulating service. Conversely, if sediment concentrations are already below the threshold requiring remedial management actions, then increased sediment retention would not provide a near-term measureable benefit.

Current or future avoided costs can be tallied as a benefit if analysis suggests that Water Fund activities are providing a buffer against changing conditions in the watershed (such as increased land clearing) that would otherwise have caused higher management costs (such as for reservoir dredging, water treatment, alternative water sourcing, etc.) or other losses (e.g., damages due to flooding). These analyses are essential for defining how Water Funds address potential gaps in future supply and demand. (See Table 1 for a listing of ecosystem functions and their associated potential services and benefits.)

Ecosystem function monitoring is helpful for illustrating important trends, such as whether management activities are having their intended effects (in the context of climatic and watershed land use data), and whether additional activities need to be implemented. However, for service users and beneficiaries, the most relevant information concerns the delivery of services and benefits, such as how much water is available for drinking or irrigation in a given season, and whether costs for providing water have increased or decreased. In terms of monitoring Water Fund success, then, monitoring information on services and benefits will generally be of the greatest interest and importance. Definition of the ecosystem functions, services and benefits relevant to the Water Fund's goals should be developed in consultation with subject experts and stakeholders as part of the initial project planning stage.

FIGURE 1: *Conceptual relationship between activity, ecosystem service gain, and associated benefits*

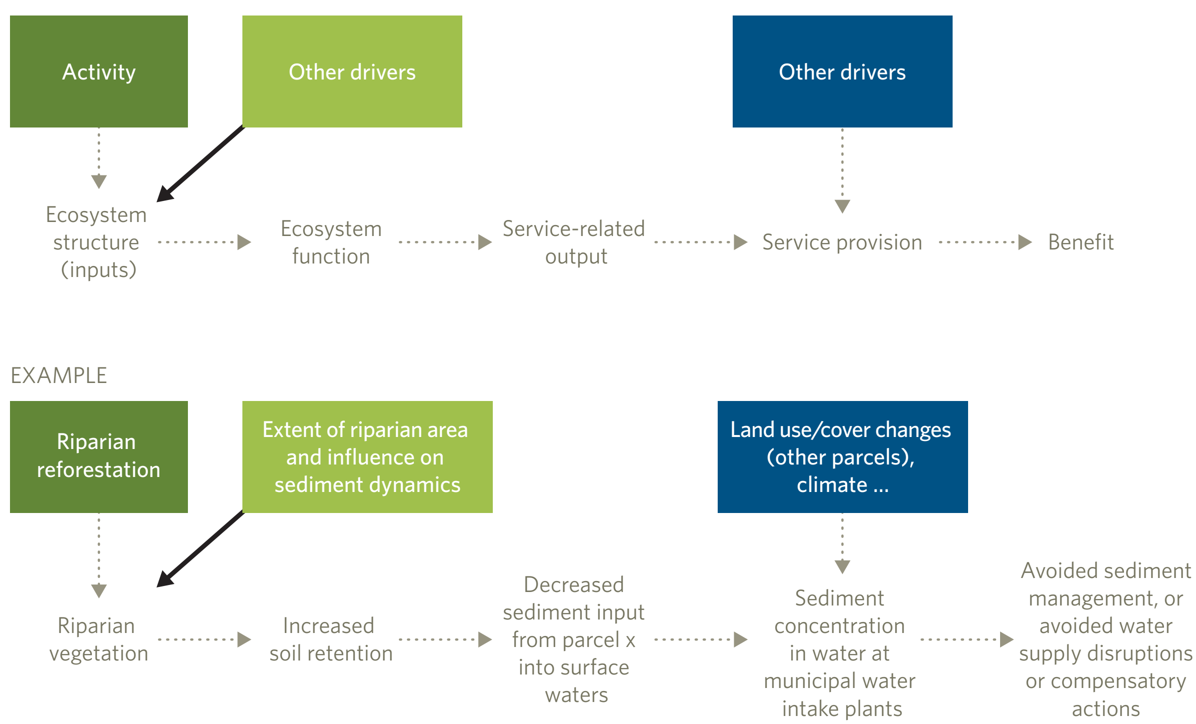


TABLE 1: ***Examples of common Water Fund ecosystem functions, and potential downstream services and benefits.*** *Potential services and benefits are realized if improvements to ecosystem function yield a change that is beneficial for human users.*

FUNCTIONS	POTENTIAL SERVICES *(current and future)*	POTENTIAL BENEFITS *(current and future)*
Nutrient retention in watershed	Reduced nutrient concentrations at points of use	Decreased nutrient management and associated treatment costs for utility
		Decreased revenue losses for utility due to disruptions of water supply
		Fewer water supply disruptions for utility customers
		Decreased water supply costs for utility customers
Sediment retention in watershed	Reduced sediment concentrations at points of use (e.g., for drinking water, hydropower)	Decreased sediment management and associated treatment costs, decreased infrastructure costs (e.g., turbine/dam replacement costs)
		Decreased revenue losses for utility due to disruptions of water supply
		Fewer water supply disruptions and associated costs for utility customers
		Decreased water supply costs for utility customers
Flow regulation	Adequate base-flow during low-flow seasons at points of use	Decreased alternative water supply costs for utility and customers (e.g., bottled water, storage tanks)
		Decreased revenue losses for utility due to supply disruptions
		Fewer supply disruptions and associated costs for municipal water supply users
	Reduction in magnitude and/or duration of peak flows	Decrease in area affected by flooding (per rainfall volume)
		Avoided human, property, and infrastructure losses due to flooding

Monitoring ecosystem function

The attributes of ecosystem function that are commonly of most concern to Water Funds are water quality (nutrients, bacteria, sediment), and water quantity (particularly low-flow conditions; some Water Funds are also concerned with reducing the magnitude of peak flows through watershed conservation activities).[3] Water quality is a concern because pollutant concentrations that exceed safe processing thresholds can increase water treatment costs, cause disruption of water supplies, and incur costs for additional infrastructure or remedial actions. Low base-flow conditions are of concern because of the potential they pose for sporadic disruptions in water availability, permanent gaps between supply and demand, and costs associated with disruptions and alternative water supplies. Higher peak flows can increase risk to people, property and infrastructure from flooding. Water quality and quantity concerns apply to municipal water and hydropower suppliers, utility customers, agriculture and industry.

Ecosystem function monitoring documents the status and trends of parameters associated with ecosystem services that are expected to respond to Water Fund activities. Although certain ecosystem functions, such as sediment and nutrient retention, cannot be directly monitored, they can be indirectly tracked using proxy indicators. These parameters provide information that reflects the assimilative capacity of the watershed, such as monitoring substrate embeddedness, suspended solids, or turbidity to infer changes in sediment retention and transport. Parameters related to flows can be directly monitored.

Ecosystem function monitoring can address a range of questions about changes in the environment. Some of these questions focus on illustrating the overall effectiveness of Water Fund activities:

- What is the evidence that can be used to document the impacts of Water Fund activities on ecosystem functions in the short term?
- Can we demonstrate in a defensible way that ecosystem function changes are due to Water Fund activities?

Other types of questions focus on progress made towards achieving specific goals and objectives for changes:

- What types and relative degrees of changes in ecosystem function result from a specific activity (one that has been poorly documented elsewhere or is new)?
- To what extent has the 7-day low-flow average been increased over time, and how close is it to closing the gap in supply/demand?

Additional questions related to ecosystem function focus on modeling estimated responses of the ecosystem to specific types of activities:

- Are there sources of impacts that are not currently being addressed by Water Fund activities, which need to be addressed to achieve objectives?
- What is the accuracy of attribute inputs and model outputs from RIOS or other spatial models estimating the relative changes that will result from activities in the watershed?
- Is the suite of activities in a watershed being implemented broadly enough and in the right places to result in the necessary types and relative degrees of change to ecosystem function?

3 For some Water Funds, groundwater recharge may also be a priority, but this is not covered in the current document because it is not a common concern for Water Funds.

Flows

The most common concern regarding water quantity in Water Funds is ***base-flow* during low-flow periods**. A secondary concern in some Water Funds is the ***flashiness* (rapid, high peaks)** caused by precipitation events. These two attributes of flow have been affected because of alterations to the natural capacity of watersheds to regulate flow. Land cover changes have resulted in water flowing more rapidly off the surface of the watershed during precipitation events, rather than moving more slowly over the surface and through sub-surface pathways and slowly returning to the stream. This not only results in higher and shorter-term peak flows, causing increased erosion and flooding frequency, but also decreases natural low-flow levels during dry seasons, which can lead to insufficient flows for water provision.

PARAMETERS FOR MONITORING FLOW

Scale: Watershed, Micro-watershed

Sample locations: Bottom/pour-point of watershed

Primary parameters:

- Base-flow
  - » 7-day low flow
  - » 7-day low flow as percent of annual average flow
- Peak flows
  - » rate of hydrograph rise or fall
  - » annual one-day high flow
  - » frequency of small floods
  - » peak-flow events: magnitude and duration

Other parameters to better assess and explain results:

- Climate (precipitation, humidity, temperature)
- Land use/land cover

Water extractions/inter-basin transfer

Flow is generally monitored at a "pour-point" of a watershed such as at a páramo stream outlet, or at the bottom of a watershed where Water Fund activities have been implemented. Flow volume rate is generally monitored using a constant flow meter or pressure sensor (for larger streams and rivers) at a stable place where the stream channel morphology has been measured and volume has been calculated by correlating water level and flow, and the stream width and depth associated with them. Small concrete weirs and flow meters can be installed in smaller streams. Fluid pressure sensors (piezometers) installed in protective pipes are commonly used as stream flow gauges. Continuous data loggers are available for access by remote computers through satellite transfer, or by plugging into the equipment on- site and downloading data that have been collected.

Hydrologic regimes reflect the results from a complete water balance, which is determined by water inputs (precipitation, glacial melt, groundwater, and inter-basin transfer if applicable), and withdrawals (water removed from the system through extraction or diversion). The larger the watershed, the greater the potential for factors outside of Water Fund activities to influence flow regimes. In addition to hydrologic regime monitoring, climate data (precipitation, humidity, and temperature) should be collected in all cases where it is necessary to demonstrate changes due to Water Fund activities. Changes in land use and land cover and water extractions (for human consumption, manufacturing, livestock, green houses, etc.) should also be monitored over time as part of the total calculation of water balance.

In order to detect the impact of Water Fund activities on base-flow conditions and the magnitude and duration of peak flows, flow should be monitored and compared through before/after assessments, ideally as part of a control/impact design (BACI). The number of years necessary for pre-impact and post-impact data will be dependent on the system's natural variability and the extent of changes resulting from Water Fund activities. Water Fund activities will generally affect flow in a gradual way, so abrupt significant differences are not expected to occur, and long-term trend data will be required. Illustrating significant differences in attributes of flow may require decades of data collection.

Hydrologic data can be analyzed using the *Index of Hydrologic Alteration* software (http://conserveonline.org/workspaces/iha) to assess differences in base-flow (e.g., 7-day low flow, or 7-day low flow as percent of annual average flow), and flashiness (e.g., rate of hydrograph rise or fall), and various flood flow characteristics (i.e., annual one-day high flow, frequency of small floods, etc.). The software provides statistical analyses for selected parameters. Given the natural variability of most systems, it will be challenging to show statistically significant differences in flashiness or flood patterns due to Water Fund actions within five to ten years, but longer-term monitoring data are invaluable for illustrating this.

FURTHER READING

For more information on why and how to measure **stream flow**, see:

North Carolina State University—Surface Water Flow Measurement for Water Quality Monitoring Projects
http://www.bae.ncsu.edu/programs/extension/wqg/319monitoring/TechNotes/technote3_surface_flow.pdf

The Engineering Toolbox—Weirs: Flow Rate Measure
http://www.engineeringtoolbox.com/weirs-flow-rate-d_592.html

U.S. Environmental Protection Agency—Monitoring & Assessment: Stream Flow
http://water.epa.gov/type/rsl/monitoring/vms51.cfm

U.S. Geological Survey—How Stream-flow is Measured
http://ga.water.usgs.gov/edu/measureflow.html

West Virginia University Extension Service—Estimating Flow in Streams
http://www.caf.wvu.edu/~forage/streamflow/estimat.htm

Instruments commonly used for evaluating flow can be viewed at:

Flow Meters
http://www.instrumart.com/Category.aspx?CategoryID=3049&gclid=CP-yIIvUt64CFSwDQAodgQK1qQ

Peizometers
http://www.slopeindicator.com/instruments/piezo-intro.html

Water Current Meter
http://www.globalw.com/support/current_meter.html

Water Quality

Monitoring water quality is a primary concern of most Water Funds because of the potential impact of sediments, nutrients, and bacteria on services and benefits for municipal water supplies, agriculture, and ranching, as well as for direct human consumption and contact. When considering water quality parameters, it is necessary to distinguish between loadings and concentrations. Loadings are the amount of pollutants entering the system, while concentrations are the amounts of pollutants within a given volume of water. Models often estimate changes in loadings, while monitoring typically focuses on concentrations.

PARAMETERS FOR MONITORING WATER QUALITY

Scale: Site, micro-watershed, watershed

Sample locations: above and below sites, bottom/pour-point of micro-watersheds or watershed

Primary parameters:

- Total nitrogen
- Total phosphorous
- Total suspended solids
- Total coliforms

Other parameters to consider:

- Orthophosphate or soluble reactive phosphorous (SRP)
- Turbidity
- Escherichia coli, fecal streptococci, and/or enterococci
- Sediments—total or specific fractions

Other parameters to better assess and explain results:

- Flow/volume (necessary to assess changes in concentrations)
- Climate (precipitation, temperature)

Nutrients:

Runoff from agricultural lands and cattle pastures often carries nutrients, such as nitrogen and phosphorous, into nearby streams. Nitrogen and phosphorous occur in several forms in water as a result of ongoing chemical transformation. Nitrogen is found as ammonia, nitrate and nitrite; total nitrogen is the sum of these. Total phosphorous is the sum of soluble reactive phosphorus (SRP), soluble unreactive or soluble organic phosphorus (SUP), and particulate phosphorus (PP). Orthophosphate is generally the major component of soluble reactive phosphorous, and is found in abundance in fertilizer and municipal and livestock waste. It is taken up readily by plants, including algae. At a minimum, most Water Funds should monitor total nitrogen and total phosphorous or orthophosphate (or SRP) concentrations in order to evaluate changes due to activities working to reduce agricultural or livestock waste runoff, or to prevent cattle from accessing streams. Monitoring external factors such as changes in fertilizer use, cattle densities, and municipal waste discharges may be necessary to adequately explain results.

Because nutrient runoff is strongly influenced by precipitation events, and concentrations are determined by water volume, time-matched water flow and volume data are necessary for accurate analysis of water chemistry data. Sampling should be continuous, or take place across several seasons with different flow patterns, and generally include storm events.

Monitoring can be conducted using continuous sampling devices that are permanently installed, and regularly checked and serviced, with periodic data downloads performed. As an alternative to continuous monitoring, samples can be taken at pre-determined intervals and (ideally) during rain events, while using a hand-held flow meter. Together with cross-section measurements (discussed in the above discussion on monitoring flow), these flow meter readings can be used to calculate flow volume and rate.

Chemical sampling and analysis should be conducted by a certified technicians and laboratories. Contamination can easily occur during sampling, handling and analysis. Appropriate detection limits, precision, and accuracy need to be defined based on the degree of change that must be detected. Sampling and analysis should include appropriate field and laboratory blanks and duplicates.

Bacteria

Many types of bacteria, viruses, and protozoa associated with fecal waste can cause human illness when they enter surface waters. Excess bacterial contamination in runoff is associated with pastures located adjacent to streams, direct access of livestock to water bodies, confined animal operations where large amounts of animal waste are produced, and limited availability or capacity of human waste management facilities. Because some of these pathogens are rare and difficult to sample, indicator organisms are often used to determine the extent of fecal contamination in waterways. Total coliforms are a group of bacteria that are widespread in nature. All members of the total coliform group can occur in human feces, but some can also be present in animal manure, soil, submerged wood, and in other places outside the human body. The usefulness of total coliforms as an indicator of fecal contamination depends on the extent to which the bacteria species found are fecal and human in origin. Fecal coliforms, a subset of total coliform bacteria, are more fecal-specific in origin. *E. coli* is a species of fecal coliform bacteria that is specific to fecal material from humans and other warm-blooded animals, and is a good indicator of health risk from water contact. Sampling can be conducted in areas and watersheds where the Water Fund is working to reduce the amount of fecal contamination in streams, such as where cattle fencing has been installed, or where human and/or livestock (e.g. cattle, sheep, chicken) waste management practices are being implemented.

It is important to sample for bacteria during and after storm events, as this is when contaminated runoff is most likely to enter streams in significant amounts. Analysis for bacteria requires collection of water samples in conjunction with flow and volume data (as for nutrient concentrations; see above). Sampling bacterial contamination levels in streams where cattle have been excluded is best conducted at times of low flow, so that changes can be observed at the site level. Sampling and analysis should be conducted by certified technicians and laboratories.

Certified labs should have field sampling and laboratory analysis Quality Assurance Project Plans (QAPPs). An example of a **QAPP for sediment monitoring** can be viewed at:

State of Washington Department of Ecology—Quality Assurance Project Plan: Suspended Sediment and Turbidity Total Maximum Daily Load Effectiveness-Monitoring Project in the Upper Yakima River Basin http://www.ecy.wa.gov/pubs/0910034.pdf

Sediment

Sediment enters streams from surface runoff and bank erosion. While a certain level of background sedimentation in waterways is normal and expected even where natural land cover is intact, land disturbance and certain land use activities can substantially increase the amount of sediment that reaches streams and rivers, along with whatever contaminants these sediments may carry. Suspended fine sediments are generally the major sources of turbidity in streams, are transported the furthest, and must be removed from municipal water supplies. Heavier components of sediment, which are suspended and transported primarily during high flow events, are of concern for municipal water supply and agriculture intakes, since gravel, sand and heavy silts need to be removed on a regular basis. These components are also of concern for hydropower and water suppliers dependent on reservoirs, since reservoir capacity is reduced over time due to filling from sediments.

The majority of sediment inputs to waterways and transport of sediment through the watershed occurs during storm events. Monitoring source and transport of all sediment components can be challenging. However, monitoring for expected results need not include all fractions of sediment. For example, total suspended solids (TSS) measurements assess those fractions of sediment which are in the water column, but not sediments that are moving along the bottom of a stream or river (bed load sediments, mostly composed of coarse particles of higher density). Accurate monitoring of suspended sediment load requires monitoring of both flow volume, and suspended sediments or turbidity. If turbidity (water clarity) is measured along with flow volume and suspended sediments, it can be correlated with suspended sediment concentrations and used as a surrogate. This relationship allows the use of continuous turbidity meters in conjunction with flow volume meters, to estimate sediment concentrations in lieu of directly measuring TSS. This approach requires calibration.

FURTHER READING

For more information on **nutrient sampling** and analysis methods, see:

USGS—National Field Manual for the Collection of Water-Quality Data
http://water.usgs.gov/owq/FieldManual/

For **bacteria sampling** guidance and methods, see:

US EPA—Monitoring & Assessment: Fecal Bacteria
http://water.epa.gov/type/rsl/monitoring/vms511.cfm

USGS—National Field Manual for the Collection of Water-Quality Data (Chapter 7: Fecal Indicator Bacteria)
http://water.usgs.gov/owq/FieldManual/Chapter7/7.1.html

For **sediment monitoring approaches**, designs and examples, see:

British Columbia Environmental Protection Division—Sampling Strategy for Turbidity, Suspended and Benthic Sediments
http://www.env.gov.bc.ca/wat/wq/BCguidelines/samp_strat/sampstrat.html

Federal Interagency Sedimentation Project
http://water.usgs.gov/fisp/

USGS and U.S. Army Corps of Engineers—*Sediment Concentrations, Loads, and Particle-Size Distributions in the Red River of the North and Selected Tributaries near Fargo, North Dakota, during the 2010 Spring High-Flow Event*
http://pubs.usgs.gov/sir/2011/5064/pdf/sir2011-5064.pdf

USGS—Guidelines and Procedures for Computing Time-Series Suspended-Sediment Concentrations and Loads from In-Stream Turbidity-Sensor and Streamflow Data
http://pubs.usgs.gov/tm/tm3c4/pdf/TM3C4.pdf

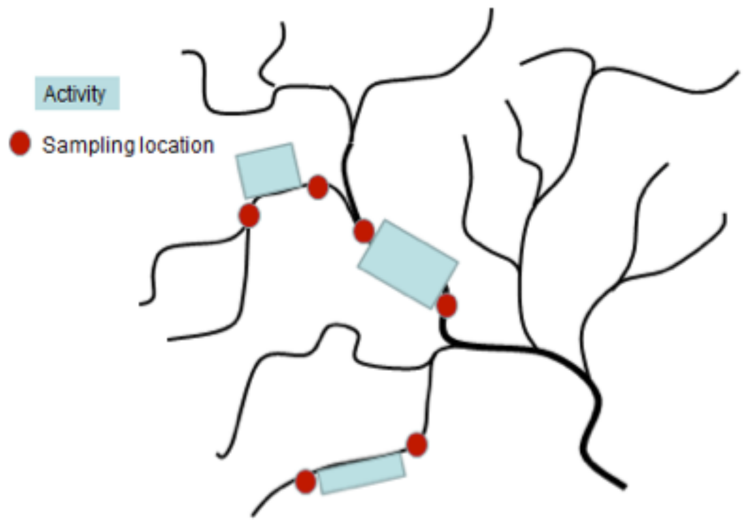
For more information about **community-based approaches** to monitoring water quality, see:

Global Water Watch
http://www.globalwaterwatch.org/GWW/GWWeng/GWWhomeEng.aspx

Riverkeepers—Red River Basin Water Quality Monitoring Volunteer Manual
http://www.riverkeepers.org/pdf/water_quality_manual01.pdf

US Environmental Protection Agency—Volunteer Stream Monitoring: Water Quality Conditions
http://water.epa.gov/type/rsl/monitoring/vms50.cfm

ABOVE/BELOW A SITE



This monitoring design is used to evaluate in-stream changes in nutrients, bacteria and sediments due to changes in runoff resulting from activities at a given site. If monitoring is conducted only after the implementation of an activity, this is a control/impact design and has inherent weaknesses. If monitoring includes before activity implementation data, this approach provides a before/after/control/impact (BACI) design at the site-level when implemented appropriately, supporting strong inference.

When to use this approach:

- Monitoring above and below a site, preferably before and after an activity, establishes a link between the activity being implemented and those in-stream parameters that respond directly to local changes, such as nutrients, bacteria, and sediment. This approach can be used:
  - » to gauge the impacts of activities that are new or have not been adequately evaluated and represented in the literature
  - » when there is low confidence in the attribute values being used in models
  - » to test the accuracy of models
  - » to collect information in the short term to provide evidence of changes occurring from Water Fund activities
  - » to illustrate to local communities the results of the management practices they are implementing

Assumptions:

- Any observed differences between the before and after data from upper and lower monitoring stations are due solely to Water Fund activity at the site.

Advantages:

- Directly measures the change in the stream between two points
- Typically involves a relatively short reach of river, simplifying data collection
- If there are no changes in flow between the upstream and downstream sites, then concentrations can be compared instead of loads. This means that flow measurement, which is sometimes difficult to obtain, would not be absolutely necessary. (However, if not difficult to obtain, flow should be monitored whenever possible to detect possible differences in flow between upstream and downstream sites.)
- Differences can be measured in a shorter period of time than at the watershed scale, and results can be used to illustrate the benefits of practices that are being implemented throughout a watershed.

Limitations:

- This approach generally works better when water is actively moving through the activity site and into the water body (i.e., during or following a rain event). For example, an above and below approach to monitoring will not generally detect chemical changes due to fencing cattle out of a stream during low-flow conditions, because at low flows, nutrient runoff levels are low.
- In some cases, such as for ephemeral stream systems, the best times to collect samples are when it is raining or when snowmelt is running off the land. Otherwise, the changes might be very subtle, if detected at all.
- If the upstream watershed or a site across the stream has highly degraded water quality, then a single activity that is not addressing a significant portion of the loading is unlikely to have a detectable impact at the "below" monitoring location.
- If no before data are collected, this design provides very weak inference of causality.

Common Statistical Approaches:

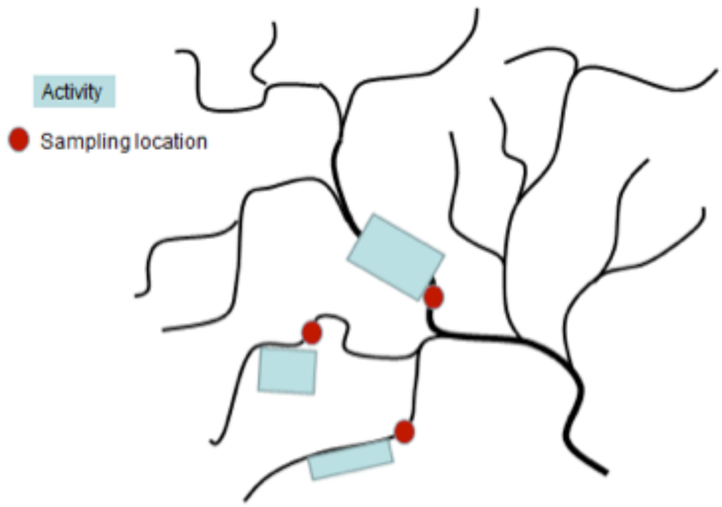
The above-and-below design is analyzed as a t-test of the differences between paired observations at the above and below stations both before and after activity implementation. Parametric and nonparametric (distribution free) t-test approaches are appropriate.

IN PRACTICE

For an example of **above/below site monitoring**, see:

*Line, D. E. (2003). Changes in stream's physical and biological conditions following livestock exclusion. Transactions of the American Society of Agricultural Engineers, 46(2), 283-295
http://www.pcwp.tamu.edu/docs/lshs/end-notes/changes%20in%20a%20stream's%20physical%20and%20biological%20conditio-1545513227/changes%20in%20a%20stream's%20physical%20and%20biological%20conditions%20following%20livestock%20exclusion.pdf

MONITORING DOWNSTREAM OF A SITE



With this design, a site is monitored for stream chemistry at a downstream location, preferably both before and after an activity is implemented. The before data are often referred to as "baseline" conditions. If monitoring is conducted with no before data, it is an impact only design, and is the weakest design. If before activity data are included, it becomes a before/after/impact design, which also has inherent weaknesses. This approach should be avoided unless adequate controls are available, because without measurements taken upstream of the activity ("above" data), it is difficult to attribute changes to the activity, since the monitoring is just tracking change over time, rather than controlling for external factors that may have impacted water quality.

When to use this approach:

- This is a weak monitoring approach and, when possible, generally should be avoided or used in conjunction with other monitoring information.
- This may be an acceptable approach under very restricted circumstances. For example, it may be useful if the activity is intended to be effective for only a very short time. Even in these cases, however, monitoring must be conducted under similar conditions (e.g., during a rain event) for both before and after data. Also, be aware that natural variability may complicate interpretation of the results.
- If the effects are expected to be clearly unequivocal after implementation and will remain over time, which is often difficult to know in advance

Assumptions:

- All conditions upstream (including land use, runoff, and flow) remain the same over time and therefore all changes are attributable to the activity.

Advantages:

- This approach requires only one monitoring station for monitoring in-stream chemistry.

Limitations:

- The effects of activities cannot be separated from other confounding effects. It is often difficult to control for other activities upstream of the monitoring site; this approach will not be able to differentiate water quality changes resulting from Water Fund activities from any other changes upstream of the implementation. It is also difficult to control for changes that happen over time. For example, if the "before implementation" period happens to coincide with a drought and the "after implementation" monitoring occurs during a high-water period, it will be difficult to differentiate drought and other climatic impacts from changes due to Water Fund activities alone. Conversely, with inclusion of "above" and "below" data (discussed above), if "before" data collection occurs during a drought, both the above and below measurements will reflect drought conditions, and the difference in values will stay relatively constant for the paired t-test.

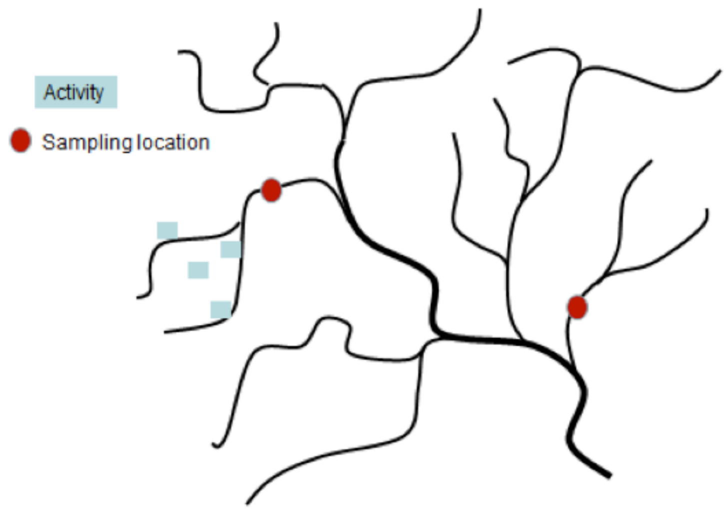
Common Statistical Approaches:

The difference in water quality caused by a practice is generally expressed as the difference between the means for the two periods. A t-test is most often used for this type of comparison.

For a detailed discussion of appropriate **statistical approaches**, see:

USDA Natural Resources Conservation Service—National water quality handbook (Part 615, Chapter 8: Single Watershed)
http://www.wsi.nrcs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb1044775.pdf

PAIRED WATERSHEDS



This design requires a minimum of two watersheds and two periods of study. Watersheds can be paired for control/impact analysis, where the control watershed starts off with the same characteristics as the impact watershed, and the impact watershed is expected to diverge from the control due to Water Fund activities. The two periods of study are referred to as calibration and treatment. Calibration is very similar to "before," but has stricter stipulations for enough "before" data to confirm that the paired watersheds behave in similar ways prior to implementation of activities. The control watershed serves as a check over year-to-year or seasonal climatic variations and receives no changes in management practices during the study. During the calibration period, activities are not conducted in either of the two watersheds. Since before data are used, this is a before/after/control/impact (BACI) design and provides strong inference.

This approach is used to study water quality and quantity. Watersheds may be defined at any scale, but are larger than what is commonly considered a site. This approach is often used when activities encompass a small headwater watershed, and there is no opportunity for sampling above the site.

Paired watersheds should be selected with the following criteria in mind:

- Similar climate, geology, soils, physiography, and drainage to control for natural differences
- The types and scope of activities in the impact watershed should be expected to be sufficient to make significant changes, well beyond the natural variation that exists within and among these watersheds.
- Monitoring can be conducted at the same sites over time, and the local conditions of those sites should not be expected to change in a way that would influence the monitoring data, beyond the changes caused by Water Fund activities. Alternatively or additionally, external factors can be monitored so that expected outcomes can be modeled.

When to use this approach:

- When it is necessary to demonstrate that a watershed is benefiting from Water Fund activities
- When it necessary to know if the scope of activities is sufficient to make a significant difference in water quality attributes

Advantages:

- Variations not associated with the treatment, such as climatic differences over years, are controlled for statistically.
- The control watershed eliminates the need to rely on measuring and understanding all the mechanisms generating the response. Be aware however that control watersheds account for many but not all mechanisms generating the response; they allow narrowing of causes toward activities generating change, but do not allow changes to be attributed solely to these activities.
- The water quality of runoff from the two watersheds need not be identical (though it should be similar, within an order of magnitude, for target parameters).

Limitations:

- The variances in water quality data are not likely to be equal between time periods (calibration and treatment), because the impact watershed is generally expected to change the variance of the response data, as compared to baseline data and to the control.
- Shortened calibration periods may increase the likelihood of serially correlated data; this is the case in most situations. This results in error terms from the calibration time period being carried over to the treatment period, and limited ability to discern either differences from initial conditions or causality of changes.
- The treatment effect may be gradual, or result in sudden changes after thresholds are exceeded; therefore, comparisons may not clearly illustrate distinctions, or may show an uneven response over time.
- The paired watershed experiment can be costly and time-consuming.
- Long-term changes in the soils or vegetation may occur in the control watershed due to activities outside the realm of Water Funds. Natural catastrophes, such as fires, dust storms, hurricanes, and insect infestations, may occur, which could have huge impacts on comparative power. (This limitation applies to all watershed designs.)
- This approach does not provide the exact difference caused by the activities, since other factors exist throughout the watersheds that are beyond the control of the Water Fund.

Common Statistical Approaches:

The primary statistical approach is regression analysis between the control and treatment watersheds during both the calibration and treatment periods. Regressions evaluate changes in water quality and quantity over time. These two regression relationships are then compared for identical slopes and intercepts using analysis of covariance. Further investigations of regressions among a dependent variable, such as a water quality parameter, and an independent variable, such as the total area of activities, are used to understand dose/response relationships resulting from Water Fund activities.

Notes on selecting the paired watershed: If a classic control/impact watershed design is being used, where the impacts are activities intended to restore ecosystem function, the expectation is that there will be a divergence of the impact watershed from the control in the direction of "improved' conditions. If however the impact watershed is primarily treated with a protection strategy intended to maintain existing good conditions, the expectation is that the treatment will remain relatively constant, while the control will diverge over time in the direction of a "degraded" condition, as a lack of protection results in continued changes in land use and land cover. Either way, the statistics are a test of significant differences between the control and impact watershed.

IN PRACTICE

For more information on design and analysis of a **paired watershed approach**, see:

U.S. EPA—Paired Watershed Study Design:
http://nepis.epa.gov/Exe/ZyNET.exe/20004PR6.txt?ZyActionD=ZyDocument&Client=EPA&Index=1991%20Thru%201994&Docs=&Query=&Time=&EndTime=&SearchMethod=1&TocRestrict=n&Toc=&TocEntry=&QField=&QFieldYear=&QFieldMonth=&QFieldDay=&UseQField=&IntQFieldOp=0&ExtQFieldOp=0&XmlQuery=&File=D%3A%5CZYFILES%5CINDEX%20DATA%5C91THRU94%5CTXT%5C00000008%5C20004PR6.txt&User=ANONYMOUS&Password=anonymous&SortMethod=h%7C-&MaximumDocuments=1&FuzzyDegree=0&ImageQuality=r75g8/r75g8/x150y150g16/i425&Display=p%7Cf&DefSeekPage=x&SearchBack=ZyActionL&Back=ZyActionS&BackDesc=Results%20page&MaximumPages=1&ZyEntry=1

USDA Natural Resources Conservation Service—National Water Quality Handbook (Part 615, Chapter 10: Paired Watersheds)
http://www.wsi.nrcs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb1044775.pdf

For examples of paired watersheds approaches, see:

Monitoring changes in flow
Fisher, M., Deboodt, T., Buckhouse, J. & Swanson, J. (2008). Lessons Learned in Calibrating and Monitoring a Paired Watershed Study in Oregon's High Desert. Third Interagency Conference on Research in the Watersheds, Estes Park, CO.
http://pubs.usgs.gov/sir/2009/5049/pdf/Fisher.pdf

Monitoring changes in sediments
Barber, T. (n.d.) Hetten and Tompkins Paired Watershed Study: Turbidity and SSC from paired managed and unmanaged watersheds.
http://www.watershed.org/?q=node/218

TWO WATERSHEDS

Two watersheds, one with an activity or set of activities and one without, have often been used incorrectly to evaluate the effects of activities on water quality and flow. The two watersheds design is not the same as the paired watershed design, because there is no calibration period for the two watersheds design, and the two selected watersheds may not fulfill proper criteria for control and impact watersheds –i.e., starting in the same conditions prior to activities. This approach is used to compare changes in water quality and flow between two watersheds. Since this design does not require before data, it has the same inherent weaknesses as other designs lacking before data. The absence of a control contributes to the weakness of this design.

When to use this approach:

- When it is necessary to infer that trends are occurring over time as a result of Water Fund activities and an appropriate control watershed cannot be identified and calibration period data are not collected
- To illustrate differences in water quality and quantity between watersheds where activities are being implemented and where they are not. Often, the situation is not clear-cut. Watersheds commonly referred to as "controls" may not actually be controls, but watersheds where activities are not being implemented that did not start with the same conditions as an impact watershed, or they may have some but fewer activities being implemented in them. In these cases, the approach does not represent a paired watershed design, but a two watersheds design.
- This approach is a weak design and is not recommended unless it is used for comparisons among watersheds using long-term monitoring to look for convergence or divergence of impact and non-impact watersheds (see below). One-time monitoring is not sufficient to make any statements about differences between watersheds resulting from treatments. This situation is not uncommon, but requires using additional monitoring information on climate and land use/cover changes to strengthen assumptions drawn from results.

Assumptions:

- Any differences are due to the causal factor being assessed. This is a difficult assumption to make without "before" data to check whether watersheds are actually similar in their response variables in terms of both mean values and variation, and is a weak assumption.

Advantages:

- Two watersheds with similar size, climate, geology, soils, and landform can sometimes be selected.

Limitations:

- The differences between the two watersheds may be caused by Water Fund activities, inherent differences in the watersheds, or an interaction between these two factors, and there is no way to distinguish among these causal factors with this basic design.

Common Statistical Approaches:

Although a statistical examination of changes associated with two watersheds may not be appropriate, comparisons of water quality and flow using a paired *t*-test or nonparametric *t*-test of treatment means could be used to analyze data. In some cases regressions between water quality and flow, considering land use and climate variables, could also be compared, to understand the influence of external factors on observed differences between the two watersheds.

MULTIPLE WATERSHEDS

The multiple watersheds design can be used in several ways:

1. To make comparisons among multiple watersheds. For instance, a watershed with no Water Fund activities that is degraded, a watershed with no activities, and a watershed that represents a desired condition for the impact watershed to achieve in the future.

This approach requires similarities among watersheds to control for external factors, which presents a significant challenge to properly using this design. For a watershed to be defined as a control watershed, it needs to fulfill the criteria discussed in previous examples. If appropriate control and reference watersheds are used, this is a control/reference/impact design, and can provide stronger inference than either a control/impact or reference/impact design. If before activity data are included with adequate control and reference watersheds, this results in a before/after/control/reference/impact design, providing the strongest inference of all these designs.

When to use this approach:

- To illustrate that an impact watershed is diverging in characteristics from a watershed with no treatment, and converging in characteristics towards a watershed that is in a more desired future condition
- Where long-term monitoring will be in place to observe trends over time

Assumptions:

- Differences are due to treatments alone.

Advantages:

- Distinctions among the control, impact, and reference (or desired condition) watersheds can be evaluated at the same time.

Limitations:

- Requires long-term monitoring in order to show trends of impact watersheds away from those of untreated and towards those of reference (or more desired condition) watersheds. Without long-term monitoring and trends, inferences for distinctions among watersheds are weak, especially without any calibration period data.
- Requires controlling for similarity of factors such as climate, geology, soils, and other factors (as discussed in the description of the paired watersheds approach in this chapter).

Common Statistical approaches:

There are a variety of statistical approaches that can be applied to this design depending on the monitoring questions. For a detailed summary, see:

USDA Natural Resources Conservation Service—National Water Quality Handbook (Part 615, Chapter 11: Multiple Watersheds)
http://www.wsi.nrcs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb1044775.pdf

2. To assess the accuracy of models by evaluating a range of watershed conditions across many watersheds that differ in the values of attributes being modeled, such as land use/cover changes due to ambient conditions, or due to activities that are in place.

This approach requires less demand for similarities among watersheds, since higher numbers of watersheds are used in the analyses. This approach generally uses data on spatial patterns (land cover/land use) and chemical and flow response data. This design is intended to provide data for regression analysis, and does not fit the standard design structures summarized in other examples.

When to use this approach:

- To assess the accuracy of models linking land use/cover and water quality and flow data

Assumptions:

- Differences that are monitored are due to relationships among the parameters that are evaluated.

Advantages:

- Results are transferable to the region that is monitored.
- Monitoring can be conducted as a snap-shot in time depending on the questions being asked.

Limitations:

- Spatial data must be available at appropriate resolution and time-series to track changes being accounted for through monitoring.

Common Statistical Approaches:

Regression analyses of watershed patterns and response variables is the most common and straightforward method, although other methods are available that provide more flexibility with regard to linearity between dependent and independent variables, as well as their distribution and variance structures (e.g. regression trees, neural networks).

MONITORING DOWNSTREAM OF A SINGLE WATERSHED

With this design, response variables are measured at a downstream location, preferably before and after the activity or group of activities are implemented. The before data are often referred to as representing "baseline" or calibration conditions. If before data are not collected, this is simply an impact design, and provides the weakest inference of causality. Including before data makes this a before/after/impact design, which also has inherent weaknesses. This approach should be avoided, as documented changes over time cannot be attributed to activities. Alternatively, as a way to make the best use of this approach, climate and land use data can be used in regressions to establish relationships among water quality and flow patterns, in order to help explain observed changes. This approach creates many constraints, and is most effective if using data from an existing monitoring site with a long record of data collection. It can be used to monitor both water quality and flow.

When to use this approach:

- When trends in water quality and quantity have been monitored for a long time, there is a desire to illustrate changes that may be linked to Water Fund activities, and scientific rigor is not required
- This may be a useful approach under very restricted circumstances, for example, if the activity is intended to be effective only for a very short time. Even in these cases, however, you must monitor under similar conditions (e.g., during a rain event) for before and after data, and be aware that natural variability may complicate interpretation of the results.
- When the effects will clearly be unequivocal after implementation and will remain over time, which is often difficult to know in advance

Assumptions:

- Conditions (including climate and land use) remain the same over time or are explicitly accounted for and therefore all other changes are attributable to the activity being implemented.

Advantages:

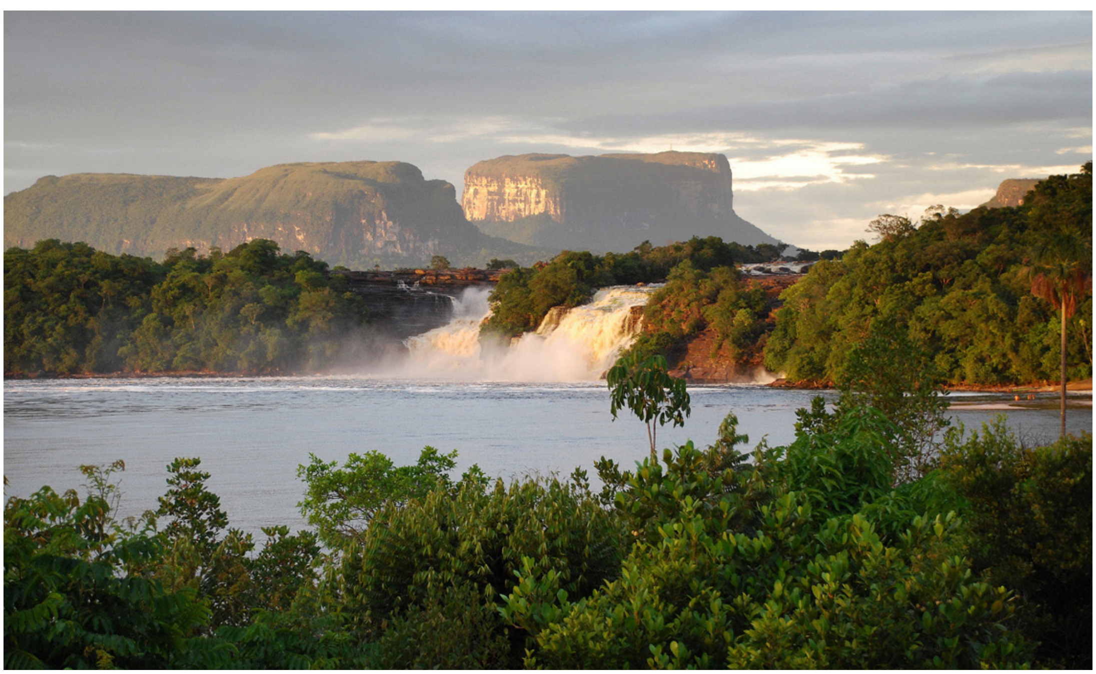
- This approach requires only one station for monitoring in-stream chemistry.

Limitations:

- Similar to monitoring only downstream of a site, the effects of activities cannot be separated from other confounding effects. As It is typically difficult to control for other activities upstream of the monitoring site, this approach will not be able to differentiate water quality changes resulting from Water Fund activities as opposed to those due to other changes occurring upstream of these activities. It is also difficult to control for changes that happen over time. For example, if the "before implementation" period happens to coincide with a drought, and the "after implementation" monitoring occurs during a high-water period, it will be difficult to differentiate the impacts of drought and other climatic factors from changes due to Water Fund activities alone.
- Interpreting the data from a single watershed approach is more complex and can result in inappropriate conclusions, requires a longer calibration period, and is less precise than a paired watershed design.
- Data collected using the single watershed design is not transferable to other areas.

Statistical Approaches:

The difference in water quality potentially attributed to an activity is generally expressed as the difference between the means for the two periods. A t-test is most often used for this type of comparison; this provides a limited sample number and degree of freedom (1). Regression analysis over time can be conducted as well.



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Monitoring ecosystem services and benefits

The provision of ecosystem services and benefits to downstream water users is typically the primary rationale for investor participation in Water Funds. Reliable documentation of those services and benefits is therefore essential to the financial and economic viability of Water Funds. In order to be financially sustainable, (1) Water Funds must provide target services and benefits to downstream investors at lower costs than alternative provision mechanisms, and (2) downstream benefits to Water Fund investors must exceed the investment costs. Economically, a Water Fund is justified if the total benefits to both investors and third parties exceed the total cost.

While careful attention should always be paid to the goals and benefits of all Fund participants and investors, in practice the primary constraint on a Water Fund's long-term viability is financial. Priority monitoring should therefore include financial benefits to fund investors and contributors. However, benefits to utility customers and other beneficiaries should also be monitored, as documenting these benefits may increase the number of supporters (e.g., other potential investors, political support for voter referenda) or their contributions, and thus ultimately affect the Water Fund's long-term financial viability.[4] These decisions should be made based on priorities of the Fund and data availability.

For project design and evaluation in general and for monitoring in particular, it is important to distinguish among: (1) ecosystem services, (2) the specific benefits those services support, and (3) the economic value of those benefits.

Monitoring downstream benefits resulting from Water Fund activities must focus on two distinct aspects: (1) quantification of the changes in *ecosystem services* that yield particular benefits, and (2) quantification of the changes in the *benefits* (monetary and non-monetary) that result from the changes in services. The latter should be considered a priority since it is the end result, while changes in services are the mechanism for securing those benefits. It must be recognized that the relationships among services and benefits may not be linear, so monitoring changes in services alone may not provide adequate information to estimate benefits. A separate component of monitoring benefits is the valuation of the changes in benefits. While necessary for assessing the overall financial or economic performance of a Water Fund, valuation of the target benefits provided by the Fund is not necessary for financial viability, as each Fund contributor presumably conducts—and will evaluate the decision to participate on the basis of—their own cost-benefit analysis.

Examples of common downstream ecosystem service and benefit parameters are provided in Table 2. Parameters and monitoring approaches should be developed with experts during the design of the Water Fund project in order to ensure that adequate information is made available and collected for analysis and reporting. Monitoring information for evaluating ecosystem services and benefits is commonly collected by utilities and other major water users and processors. Ideally, Water Fund design negotiations will include agreements for access to these data in a manner that protects the confidentiality of critical financial information and trade secrets.

4 Reducing the share of the benefits generated by the Water Fund that go uncompensated would allow the lowering of contribution rates for existing supporters, thus increasing the latter's net benefits from participation and strengthening the financial argument for the continuation—and possible expansion—of activities.

TABLE 2: *Selected parameters for monitoring common ecosystem services and associated benefits. Note that net services and benefits due to Water Fund activities must be interpreted in the context of precipitation patterns and other factors (such as watershed land use changes) that may affect parameters.*

ECOSYSTEM SERVICE	SERVICE PARAMETERS	BENEFIT PARAMETERS *(non-monetary)*	BENEFIT PARAMETERS *(monetary; $/yr)*
Reduced **nutrient concentrations** below management action threshold at **drinking water** points of use	***# hours** (or days)**/year** of disrupted water supply due to exceedance of nutrient thresholds*	***# hours** (days) of water supply shutdown/ **year** x # of customers affected*	***Revenue losses** for utility associated with supply disruptions*
			***Water treatment costs** for utility (operations & maintenance)*
			***Costs due to supply disruptions** for utility customers (revenue loss, defensive expenditures, future infrastructure investment)*
		***Million cbm/year** of water supply delivery*	*Future **infrastructure** investment **costs** for utility*
Reduced **sediment levels** below action threshold at **drinking water** point of use	*# **hours** (or days) of disrupted water supply/year due to exceedance of sediment thresholds*	***# of hours** (days) of water supply shutdown**/year** x # of customers affected*	***Revenue losses** for utility associated with supply disruptions*
			***Water treatment costs** for utility (operations & maintenance)*
			***Costs due to supply disruptions** for utility customers (revenue loss or defensive expenditures, future infrastructure investment)*
		***Million cbm/year** of water supply delivery*	*Future **infrastructure** investment **costs** for utility*
	*Amount sediment reduced **(metric tons/year)***	***# metric tons/year** of sediment removal needed*	*Future **dredging and infrastructure** investment **costs***
Reduced **sediment levels** in **hydropower reservoir**	*Reservoir capacity*	***Reservoir capacity** due to reduced sedimentation*	*Future **infrastructure** investment **costs** for utility*
	*Amount sediment reduced **(metric tons/year)***	***# metric tons/year** of sediment removal needed*	***Sediment removal costs***
		*Electricity production over extended lifetime of reservoir **(kWh/year)***	***Revenue losses** for utility associated with lower lifetime power production*
			***Revenue losses** for businesses associated with power supply interruptions*

Increased **base-flow** exceeding shutdown thresholds during dry season(s)	***# hours/year*** *water is unavailable due to insufficient flow*	***# of hours of shutdown*** *per year* x # of customers *affected*	***Revenue losses*** *for utility associated with supply disruptions*
		# person-days *of access to water*	***Defensive expenditures*** *(water tanks; bottled water)*
		Million cbm/year *of increased delivery*	***Costs due to supply disruptions*** *for utility customers (revenue losses, defensive expenditures, future infrastructure investment)*
			Future infrastructure/ alternative *water source* ***management investment costs*** *for utility*
Reduction in magnitude/ duration of **flood events**	***# instances/year*** *when municipal water supply is disrupted due to flooding*	# shutdown instances/ year	***Revenue losses*** *for utility associated with supply disruptions*
			Costs due to supply disruptions *for utility customers (revenue losses, defensive expenditures, future infrastructure investment)*
	# of instances/year *flood thresholds are exceeded*	***# people/homes/ businesses/****area of* ***croplands per year*** *impacted by flooding*	***Property/ infrastructure/ crop losses*** *due to flooding*

cbm = cubic board meters

Quantifying services and benefits

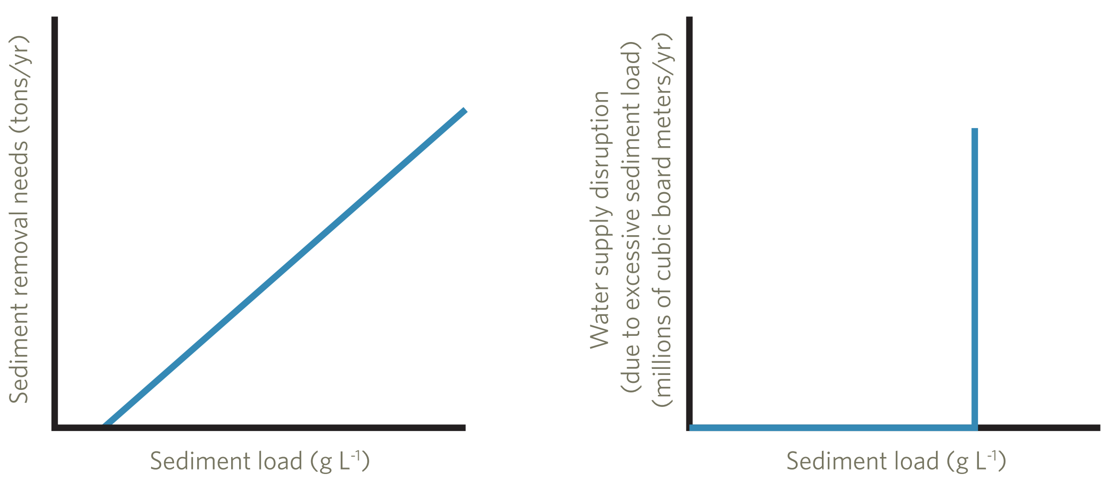
The benefit and value of a given change in a service often differs across space: the effect of a given reduction in sediment concentration may not be the same for similar users (e.g., municipal water suppliers with different water intake locations or treatment technologies), and may vary among different users (e.g., municipal water suppliers and agricultural producers).[5] Consequently, services should be monitored at their respective points of use by each beneficiary (e.g., the water intake of a municipal water supply, the irrigation intake of an agricultural producer).

To minimize monitoring costs for the Water Fund, downstream beneficiaries should be identified that are already monitoring services over time. For example, municipal water suppliers or industrial water users typically monitor water supply parameters of flow, quality, and sediment. And, because they are the beneficiaries of the services, their reports on changes in service benefit values are essential contributions to accurately quantifying the effects of Water Fund activities. In cases where users are not already monitoring these parameters, multiple users are operating in close proximity to each other, or the differences the service provided are not significant among users, using a single location to monitor the service for multiple users may be a reasonable approach. For instance, monitoring sediment concentrations above the first water intake of a series of industrial or municipal water users may be an option, provided that none of the users is itself producing water returns heavy in suspended solids which would impact other users downstream.

Estimating the increase in benefits that result from changes in a particular service requires information on benefit functions. A benefit function indicates how a change in service affects the associated benefit. This is shown in Figure 2 using two different functions. Both functions have thresholds below which changes in the service do not generate benefits. In the graph on the left, benefits increase linearly with changes in the service after a certain threshold is exceeded. In the graph on the right, benefits are zero up to the threshold, but once the threshold is exceeded, benefits immediately reach their maximum.

5 Different municipal water plants may employ different sediment removal technologies characterized by different cost functions. Even if they employ identical technology, different plants may be operating at different sediment removal capacity utilization rates and thus may be affected differently by a given reduction in sediment loads. For example, a plant faced with the need to expand removal capacity or with the opportunity to shut off a sediment removal unit if sediment is reduced by a given amount is affected differently than a(n identical or different) plant for which the same reduction would not lead to capacity or operational changes.

FIGURE 2: *Examples of benefit functions for two benefits produced by sediment reduction*



Beneficiaries themselves generally will have the best knowledge of their benefit functions and thus may be best positioned to estimate these benefits themselves or to assist in generating benefit functions for models to be used by others. The specific benefits potentially obtained from a change in a particular service are shown in non-monetized and monetized form, respectively, in columns three and four on the right side of Table 2.

The non-monetized benefit estimates can be used to assess the cost-effectiveness (dollar per unit of service gain) or non-monetized return on investment (ROI) expressed as the quantity of a particular benefit generated per dollar invested of the Water Fund for particular downstream beneficiaries. This ROI can then be compared to that of conventional ("gray" infrastructure) alternatives for providing the respective service. Similarly, the overall cost-effectiveness or non-monetized ROI of the Water Fund for a particular service can be assessed by dividing total costs by total gains in that service (i.e., across all beneficiaries).

For example, analysis may estimate that a Water Fund reduces sediment concentrations at the municipal water supply water intake by ten percent, reducing required sediment removal by 100 tons per year. If the municipal water supplier invests $1 million per year in the Water Fund, then its ROI in the Fund is 100 tons of sediment per $1 million, which results in a cost of $10,000 per ton of sediment avoided per year. This cost-effectiveness can be directly compared to, for example, the hypothetical $12,000 in annualized costs of the additional filtration capacity that would have been required to achieve those 100 tons of sediment removal.[6] Likewise, if overall investment in the Fund is $10 million per year and this reduces sediment management needs for all investors combined by a total of 20,000 tons, then the Water Fund's overall cost-effectiveness for avoided sediment management would be $5,000 per ton of sediment per year.

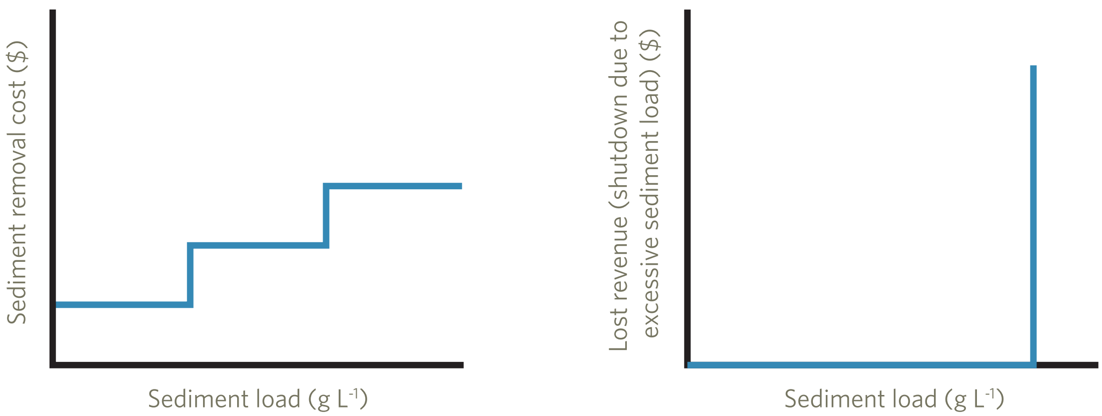
It should be noted that such single-benefit cost-effectiveness or ROI metrics may underestimate the competitiveness of the Water Fund with competing alternatives (in this case, the sediment filtration plant) if the Water Fund provides more than one service. For example, if the Water Fund also provides increased base-flow during critical low-flow periods, then some of the $1 million the water supplier invests in the Fund should be assigned to that service when calculating the cost-effectiveness and ROI for sediment removal/avoidance for the water supplier. In this example, the $1 million investment by the water supplier thus would yield 100 tons of reduced sediment removal needs per year and, say, 1 million cubic board meters of avoided water delivery shortfalls per year, with associated costs of, say, $2 million for the latter. Clearly, calculating the cost-effectiveness of the two services separately ($10,000 per ton of sediment avoided and, say, $ 1/cubic meter of additional base- flow in dry period), and then comparing these numbers to the individual cost-effectiveness of the two alternative "gray" infrastructure options would lead to misleading results, inaccurately representing the total cost-effectiveness of avoided water supplier "gray" investments. However, it is not obvious what portion of the investment should be assigned to each of the two benefits, since any apportionment would be arbitrary. This problem is avoided if the benefits can be expressed in terms of their monetary values.

6 This example illustrates why it is important to include the future flows of services and benefits in the analysis. If sediment removal plant capacity currently is adequate, reductions in sediment concentrations produce no benefit and no value for the water supplier in the current year. However, let's assume that increased development in the watershed would drive up sediment loads and would require installation and operation of additional filtration capacity of, say, 500 tons of sediment per year in, say, ten years from now. If the activities carried out by the Fund reduce this increase to 400 tons, they yield 100 tons of avoided sediment removal for the water supplier in year 10, and similar amounts in each year after that. Avoiding those future sediment removal needs has current value, equivalent to the present value of the stream of avoided annual future sediment management costs.

Valuing service gains

The economic value of particular benefits brought about by the Water Fund can be estimated using the parameters shown in column 4 of Table 2, as illustrated in Figure 3. This estimation requires value functions that indicate how a change in a service (e.g., avoided sediment load) translates into monetary benefits (avoided costs).

FIGURE 3: *Examples of cost functions needed to value benefits*



In most cases, beneficiaries themselves will be best positioned to quantify the value of benefits accruing to them, and indeed one would expect them to eventually conduct their own benefit-cost analyses of their investments in the Water Fund. Monetary valuation of the benefits derived from increases in services allows beneficiaries to estimate the total net benefit (total benefits from all services the beneficiary receives, minus fund contributions) or the benefit-cost ratio (i.e., monetized ROI, expressed as a unit-less ratio: dollars generated per dollar invested) of their investments in the Fund. For example, in the case of the water supplier above, the avoided needed investment and operating costs of $1 million per year for alternative removal capacity for 100 tons of sediment and costs of $2 million for 1 million cubic board meters of water supply translate into $3 million of annual benefits. Given the $1 million annual investment of the water supplier, the Water Fund delivers $2 million in net benefits to the supplier per year and reflects a benefit-cost ratio or monetized ROI of 3 for the supplier.

Information on the value of benefit gains across all beneficiaries combined with overall Fund costs would allow the estimation of the overall net benefit and overall ROI of the Fund (i.e., across all beneficiaries and services). This is essential information for developing a better overall understanding of the business case, as well as the public or societal case for investments in Water Funds, and thus beneficiaries should share these analyses with Water Fund research staff. Lack of quantitative information on the economic performance of Water Funds is likely to depress investment in watershed conservation and lead to missed opportunities in terms of both cost savings for private and public companies and co-benefits for the public at large.

It is important to point out that the cost of monitoring all services and estimating all service gains for all beneficiaries of a Water Fund is prohibitive in most cases. Rather, budget limits require monitoring to be restricted to major beneficiaries. This will result in a low-end estimate of the Water Fund's overall benefits or ROI. Depending on the share of total benefits accounted for by the beneficiaries included in the analysis, and the relative value of accounted- and unaccounted-for benefits, this bias may be substantial.

FURTHER READING

For additional information on assessing the **impacts of management practices on ecosystem benefits**, see:

Tallis, H., et al. (2011). New metrics for managing and sustaining the ocean's bounty. Marine Policy, 36, 303-306.
http://cmsp.noaa.gov/_pdf/New_Ocean_Metrics_MarPolicy_2011_online.pdf

Tallis, H., et al. (2012). A global system for monitoring ecosystem service change. BioScience, 62, 977-986.
http://woods.stanford.edu/sites/default/files/files/global-monitor-study.pdf

USDA Natural Resources Conservation Service—Tools for Economic Analysis
http://www.nrcs.usda.gov/wps/portal/nrcs/main/national/technical/econ/tools

Chapter 11: Impact Monitoring for Habitats and Biodiversity

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Assessing conservation impacts

In addition to the inherent value of protecting and restoring native ecosystems, the conservation and restoration of habitats and biodiversity is a primary mechanism to support ecosystem services. Because ecosystem services are often derived directly from healthy ecosystems, monitoring habitats and biodiversity can reveal much about the condition of ecosystems and the services we can expect them to provide.

Many Water Fund activities focus on protecting or restoring natural land cover, which in turn provides sources of food and habitat for native animals. Restoration takes two different basic approaches: unassisted –isolating areas from grazing, agriculture, or other disturbances and allowing them to re-vegetate on their own; and assisted, in which areas are actively seeded or replanted. These different approaches result in varying degrees and rates of native plant and animal restoration. These activities can also increase the capacity of the ecosystem to regulate surface runoff of water, sediment, nutrients and bacteria, which in turn impacts the condition of freshwater habitats and biodiversity. In addition to terrestrial habitat and biodiversity benefits, protection and restoration of riparian corridors can also provide direct benefits to freshwater habitats and biodiversity, by stabilizing stream banks, shading waterways, and providing organic matter and woody material, which are important sources of carbon inputs and habitat heterogeneity for freshwater systems. Riparian areas also serve directly as habitat for aquatic biodiversity throughout the year, such as for the adult stage of some aquatic invertebrates, and for fishes during high-flow periods.

Habitat monitoring provides data that directly link Water Fund activities to quantified biological responses. These responses involve a lag-time, which is influenced by the types of activities and the linkages between terrestrial and freshwater habitats and biodiversity. Most terrestrial habitat and biological responses occur directly in managed areas, and can respond in relatively short time periods: for example, recovery of soils and vegetation in an area that is isolated from cattle grazing. Freshwater habitat and biological responses may occur downstream, as well as in areas proximal to the managed site. In addition, the responses of freshwater habitats and biodiversity are affected by all other areas upstream in the watershed, since all land uses can affect aquatic ecosystem processes. Addressing these issues requires different sampling designs to track trends and test hypotheses.

Habitat and biodiversity monitoring can address questions such as:

- What is the condition of terrestrial and freshwater ecosystems where Water Funds are being implemented?
- How is terrestrial and freshwater ecosystem integrity changing in response to Water Fund activities?

PARAMETERS FOR MONITORING TERRESTRIAL HABITATS & BIODIVERSITY

Scales: Plot, Site, Watershed

Primary Parameters: Vegetation

Coverage: land cover classes, vegetation coverage

Conditions within land cover classes

- Plant Structure
- Diversity and composition (level of taxonomic resolution as determined by need and capacity)
- Relative abundance of taxonomic or physiognomic groups
- Age/size structure
- Indicator species presence/absence/abundance

Other Parameters to Consider:

- Soil moisture
- Soil drainage/porosity characteristics
- Soil temperature
- Soil horizon structure
- Soil organic matter content

Diversity, Composition, Abundance, Presence/Absence of Indicator Species for:

- Mammals
- Birds
- Amphibians
- Reptiles
- Invertebrates

Other parameters to better assess and explain results:

- Elevation
- Climate (precipitation, temperature)

Terrestrial monitoring

In terms of terrestrial conservation, the response of principal interest to most Water Funds is changes in vegetation. Terrestrial habitats are commonly described by their vegetation composition and structure. Although vegetation monitoring does not necessarily reflect diversity of animal species, vegetation is a major component of biodiversity, and an indicator of habitat quality. Vegetation characteristics also influence evapotranspiration, infiltration, and runoff rates, which are directly linked to the hydrological goals of Water Funds. Vegetative land cover is the primary variable in most computer models used by Water Funds to identify priority areas for implementation and to generate expectations of changes to runoff patterns in response to management activities.

Some Water Funds may also be interested in studying changes in soil attributes, such as soil temperature, moisture, and organic matter, to explore the underlying mechanistic responses of the terrestrial ecosystem to management activities. The approaches to study these factors will vary by project, management activity, and environment, and should be researched by project staff and partners if warranted based on project goals. Monitoring biodiversity beyond plants should also be conducted if necessary, to track progress towards a stated project goal, if resources allow.

Vegetation: Remote sensing

Since vegetation changes are a primary response to Water Fund activities, it is important to track these changes throughout the project area. Remote sensing imagery can provide information on all Water Fund sites as well as the rest of the watershed, and can be used in conjunction with on-the-ground monitoring to understand changes not well understood or not detectable using remote sensing data alone. Types of remote sensing data should be chosen with a clear understanding of their inherent capabilities and limitations. Appropriate spatial resolution to answer the monitoring questions is essential. As a rule of thumb, the area or width of the management activity should cover at two pixels of the digital product used to monitor it. For example, tracking changes in riparian areas that are 30m in width require resolution of at least 15m, if not finer. Coarser data may be acceptable for tracking changes in large land areas (and associated control or reference areas). Remote sensing products such as Ikonos, Geoeye, or aerial photography should be considered for higher resolution needs. See Tables 3 and 4 for an overview of several types of remote sensing data.

TABLE 3. *Remote sensing sensors and common uses.*

SENSOR	USE
MODIS NDVI/EVI	Large landscapes (basin, large forests, grasslands): measuring seasonal variation, rapid deforestation, droughts, floods, overgrazing, fire, coarse land cover.
LandSat/Aster/ SPOT MS	Medium-sized landscapes (basin/watershed, smaller forests, large rivers, wetlands, pastures/row crops, housing): baseline information on vegetation structure, condition, spatial extent—providing land cover and land use, and change information over time.
SPOT PAN	Small areas (reforestation, small rivers, housing): providing baseline land use information data delineation. Collected infrequently (see below).
IKONOS/GeoEye	Very small areas (reforestation implementation or monitoring, finding small changes or disturbances in landscape): Providing fine land use data. Collected infrequently due to cost, extent, use limitations and data computing requirements.

TABLE 4. *Remote sensing sensors, resolution, frequency of data updating, relative cost for data acquisition, and common frequency of data use for Water Fund projects.*

SENSOR	RESOLUTION	UPDATE	COST	COMMON FREQUENCY OF USE
MODIS NDVI/EVI	MS 250m	16 day	Free	Coarse—high frequency (daily)
LandSat	MS 30m	14 day	Free	Moderate Resolution—yearly
ASTER	MS 15m, 30m	14 day	Free	Moderate Resolution—yearly
SPOT MS	MS 10m	2-3 days	$$	Mod-High Resolution—yearly or less
SPOT PAN	Pan 2.5m, 5m	2-3 days	$$S	High Resolution—yearly or less
IKONOS/GeoEye	Pan~1m	~2 weeks	$$$$	V High Res—once or 5-10 year repeat

Remote sensing data are commonly provided as classification products, defining categories of land cover based on spectral analysis patterns. These categories can be used if distinct changes in plant structure and cover are expected to occur, such as shifts from agriculture to natural forest. However, using remote sensing to monitor changes of condition within a category, such as improving conditions of a natural forest, require analysis of spectral bands in relation to on-the-ground monitoring. Even with such analysis, it is generally challenging if not impossible to track changes in vegetation species composition or age/size structure using remote sensing data.

Changes within a defined category of land cover can sometimes be adequately inferred using sub-pixel monitoring with passive sensors, multi-spectral, or hyper-spectral data. Except for the recently launched Quickbird multi-spectral satellite and spectral airborne cameras, existing high resolution optical data are generally not suitable for this purpose and resolution. Medium-to-coarse resolution data are available with multi- or hyper-spectral attributes from passive sensors (e.g. Landsat, MODIS, SPOT, ASTER, etc.) or active sensors (Synthetic Aperture Radar). This approach to monitoring requires a known biologically meaningful signature of spectral responses. Such responses are identified and validated with on-the-ground monitoring.

For more information on using **remotely sensed data**, see:

Remote Sensing—Special Issue: "Remote Sensing of Biological Diversity"
http://www.mdpi.com/journal/remotesensing/special_issues/biological_diversity

Xie, Y., Zongyao, S., & Mei, Y. (2008). Remote sensing imagery in vegetation mapping: A review. Journal of Plant Ecology, 1(1), 9-23.
http://jpe.oxfordjournals.org/content/1/1/9.full

Vegetation: On-the Ground Monitoring

On-the-ground monitoring is necessary to investigate relationships between remote-sensing data and many vegetation changes (if this approach is taken), and to understand patterns of species composition and age/size structure that are critical to defining vegetation condition. Such monitoring is generally performed using randomly placed plots or transects to guide the collection of data.

One common way to sample vegetation is by estimating species composition and abundance by making counts at randomly selected points along transects. This information can be supplemented by adding square plots of varying sizes (ranging from one to several meters, depending on density), so that plant stem density and diameter may be monitored, abundance and age structure can be calculated, and the occurrence of rare species can be assessed. This method may focus on either woody or herbaceous species, or include both, depending on the habitat and goals of the project for conservation. When feasible, transect and plot information can also include measurements of plant height where recovery from disturbance is being monitored. Mature trees can be assessed along the same transects, for both trunk diameter and canopy coverage. Maps can be used to record the areas of sampling, and handheld geographic positioning system (GPS) units can provide accurate records of the sample locations, if resources allow.

Techniques vary for vegetation monitoring, depending on the environment, the vegetation, the area that needs to be assessed, the questions being addressed, and available capacity for monitoring.

Wildlife

The ultimate measure of the habitat quality of any ecosystem is the biodiversity it supports. For both forests and grasslands, wildlife monitoring can be relatively low-cost and easy to conduct, and can provide valuable information on the success of the Water Fund's protection and restoration efforts.

For details on various techniques for monitoring **mammals, birds, amphibians, reptiles, and invertebrates**, see:

Harris, R.R., Kocher, S.D., Gerstein, J.M., & Olson, C. (2005). Monitoring the effectiveness of riparian vegetation restoration. Berkeley, CA: University of California Center for Forestry.

Hodkinson, I.D., & Jackson, J.K. (2005). Terrestrial and aquatic invertebrates as bioindicators for environmental monitoring, with particular reference to mountain ecosystems. Environmental Management, 35(5), 649-666.

Morrision, M.L. (2002). Wildlife restoration: Techniques for habitat analysis and animal monitoring. Washington, DC: Island Press.

Nichols, J. D., & Williams, B. K. (2006). Monitoring for conservation. Trends in Ecology and Evolution, 22(12), 668-673. doi:10.1016/j.tree.2006.08.007

University of Montana—Evaluating Habitat Restoration at O'Dell Creek Using Bird Communities
http://avianscience.dbs.umt.edu/documents/projects/Odell_2007_finalreport.pdf

USGS Patuxent Wildlife Research Center—Managers' Monitoring Guide - How to Design a Wildlife Monitoring Program
http://www.pwrc.usgs.gov/monmanual/

Ward, D.F., & Larivière, M-C. (2004). Terrestrial invertebrate surveys and rapid biodiversity assessment in New Zealand: Lessons from Australia. New Zealand Journal of Ecology, 28(1), 151-159.
http://www.nzes.org.nz/nzje/free_issues/NZJEcol28_1_151.pdf

FURTHER READING

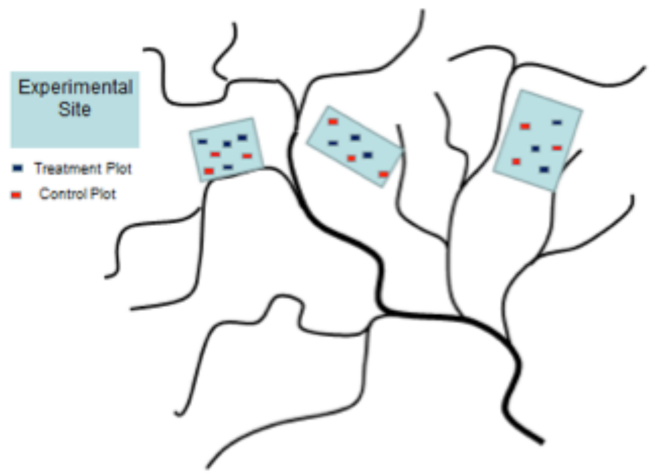
A summary of the strengths and weaknesses for a variety of **vegetation sampling techniques** can be found at:

Colorado State University—Vegetation Measurement Methods Comparison
www.cemml.colostate.edu/assets/pdf/methods.pdf

Sampling designs for terrestrial monitoring

Terrestrial sites can generally be evaluated for the parameters discussed above with designs independent of the treatment/control/reference watershed framework used for water quality and quantity monitoring, as the changes being assessed are site- and plot-based, and therefore not generally influenced by the broader watershed context, unless in direct contact with the water body, such as riparian corridors and floodplains.

PLOT DESIGN



Impact and control plots within experimental sites are commonly used to evaluate changes to plant and soil. Before/after data are recommended to strengthen assessment of differences due to treatment. Different monitoring approaches can result in a control/impact or a before/after/control/impact (BACI) design.

When to use this approach:

- When assessing changes in the characteristics of terrestrial vegetation or soil
- If warranted, for calibrating model attributes or understanding mechanisms underlying transitions of vegetation, animals, soils, etc. resulting from a specific activity
- If warranted, to compare the conditions of impact plots to reference or control plots

Assumptions:

- All plots/sites have the same environments and receive the same treatments, except for control plots (and reference plots if warranted for the assessment).

Advantages:

- Plots are replicated within a given experimental site.
- Plot designs generally allow for control of several variables, such as soils, slope, direction to sunlight (e.g., facing north or south on the side of a mountain), with generally uniform climate.
- A set of control plots can be selected in the same area as impact plots.
- Plot designs can be "blocked" with groups of plots in a single site receiving the same experimental treatment, to account for *a priori* differences and variations resulting from external factors affecting the experiment (see "Statistical Approaches" below).

Limitations:

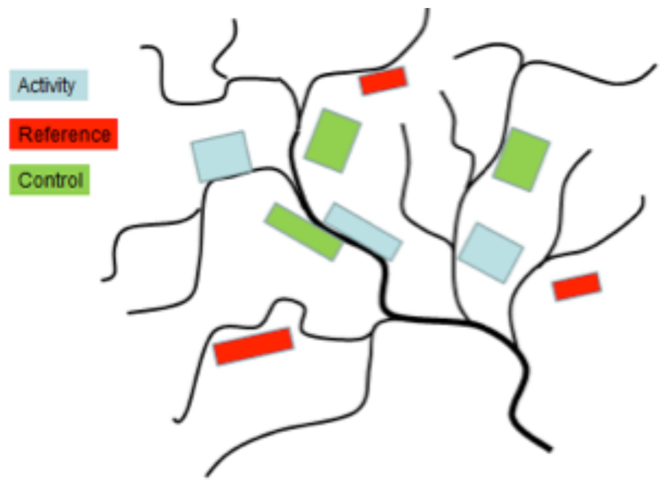
- Plots may be too small to adequately represent the dynamics of the larger ecosystem.
- Plots must be somehow physically separated from each other to prevent the effects of the treatment in one plot from affecting other plots.
- Suitable reference plots (if warranted for the assessment) may be difficult to identify.

Statistical Approaches:

The primary statistical approach is the analysis of variance (ANOVA) of a randomized complete block design. Blocks are assumed to be homogeneous sites that contain plots, and offer a method of assessing the extent to which variance is due to treatment or plot. The treatments are assigned to plots within the blocks randomly. Other more complicated designs are available, including the Latin square and split plot designs, or a factorial arrangement of treatments. For a detailed discussion of appropriate statistical approaches, see:

USDA Natural Resources Conservation Service—National Water Quality Handbook (Part 615, Chapter 7: Plot Designs)
http://www.wsi.nrcs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb1044775.pdf

MULTIPLE SITE DESIGN



A design using multiple sites as "treatments" can be used to evaluate changes in vegetation, soils, and other habitat characteristics. Before/after data are recommended to strengthen assessment of differences due to treatment. This design can also be used to only include control and impact sites, or reference and impact sites, each having less strength of inference than a control/reference/impact design. Different monitoring approaches can involve adding before data to these designs, resulting in four different designs with different levels of strength of inference. Replicates of each type of site are recommended to strengthen the inference power of the results.

When to use this approach:

- When assessing changes in the characteristics of vegetation and soils, due to management activities
- If warranted, for calibrating model attributes or understanding mechanisms underlying transitions of vegetation, animals, soils, etc. resulting from a specific activity, to compare the conditions of an impact site to a reference site

Assumptions:

- All sites have the same environments and receive the same treatments, except for control sites (and reference sites if warranted for the assessment).

Advantages:

- Sites for all "treatments" are replicated.
- Monitoring can be designed using existing patterns of treatments.

Limitations:

- Sites are generally located in environments that differ, and variation is difficult to control for statistically.
- Replication can be poorly designed and misunderstood, resulting in "pseudo-replication." The most common source of pseudo-replication is where multiple samples from one site are used to represent an activity, while multiple samples taken from a control or reference site are treated as replicates. However, these are not true replicates, but samples that can only be used to provide information on the mean and variance of a parameter at a given site. Lacking replication, there is no direct statistical approach to indicate whether results are due to differences in the treatments or to other variations among sites. Treating samples from a single site as independent and pooling data for statistical analysis is a common error. Replication must be achieved at the level of treatment, which is the site level in this case. Consultation with a statistician is highly recommended.

Statistical Approaches:

Analysis of variance (ANOVA), *t*-tests, and *U*-tests can be used to statistically assess differences among impact and control (and reference) sites. Other more complicated designs are available and should be evaluated with a statistician.

Stream habitat monitoring

Freshwater habitat monitoring and characterization is not only useful for tracking status and change in response to site-level activities, it is also necessary at the basin, stream, and reach scales to inform where and how to sample biota. Although the discussion below focuses on monitoring stream habitat at the reach-scale, a guide for characterizing habitat at larger scales can be found at: http://pubs.usgs.gov/wri/wri984052/pdf/wri98-4052.pdf

PARAMETERS FOR STREAM HABITAT MONITORING

Scales: Reach, Stream, Watershed

Primary Parameters:

- Stream channel morphology (cross-sectional depth)
- Stream substrate
- Riparian condition and cover
- Pool/riffle/run structure and ratio
- Stream Bank Erosion (degree of stability/instability)
- Amount & size distribution of woody material
- Riffle substrate embeddedness
- Dissolved oxygen
- Water temperature

Other parameters to better assess and explain results:

- Flow volume
- Climate data (precipitation, temperature)

Reach-scale

At the reach scale, characterization of stream channel, bank, and riparian conditions can be accomplished using a combination of visual assessments and quantitative methods. Each type of habitat (e.g., pool, riffle, run, stream bank, riparian area) has distinct geomorphic characteristics, which are affected in varying degrees by land-based alterations and management practices. Parameters chosen for monitoring should focus on those expected to undergo significant change in response to Water Fund activities, to increase the likelihood of detecting change given background fluctuations. Watersheds are assessed using reach-scale assessments at appropriate locations throughout in order to characterize status and changes in different portions of the watershed (i.e. headwaters, tributaries, mainstem). Multiple stream reaches must be monitored to provide replicates for any given study.

In watersheds were terrestrial habitats have been significantly altered, excess sediment runoff typically enters waterways. These sediments can become embedded in the stream substrate, filling pools and runs and negatively impacting the habitat suitability of these areas for fish and invertebrates. Losses of riparian vegetation are particularly detrimental to stream habitats and biodiversity, leading to less large woody material in streams, and resulting in changes to channel geomorphology and reduction in habitat complexity. Vegetation loss in riparian corridors also decreases inputs of other organic material, such as leaves and small woody material, which is an important energy input for stream systems, and provides less stream shade, resulting in greater solar radiation in streams, higher temperatures, lower dissolved oxygen, and increased algae growth. Water Fund activities are expected to positively affect stream habitat by decreasing the runoff of sediment, increasing shade, and providing inputs of woody material and other organic matter. Some best management practices, such as excluding cattle from streams, also eliminate direct physical disturbance in streambeds.

Habitat changes will occur in timeframes associated with the lag-times of Water Fund project impacts to processes that influence habitat characteristics. Flow dynamics affect the lag-time for flushing sediment, and moving substrate and woody materials in stream channels. The degrees to which these changes can be detected are influenced by upstream conditions and the natural flow dynamics of the watershed. Changes in riparian condition and cover, stream bank erosion, stream temperature, dissolved oxygen, organic debris, and the amount and size distribution of large woody material will respond in accordance with the rate of riparian vegetation growth and canopy coverage. Riparian vegetation growth and canopy coverage can be fairly rapid depending on the climate and vegetation types. Activities intended to abate sediment runoff generally have a shorter lag-time than do those working to increase woody material inputs, but they can both be quite lengthy.

Stream temperature and dissolved oxygen are measured using hand-held probes. The time of monitoring will affect these parameters, as stream temperature increases and dissolved oxygen decreases throughout the course of the day. These data can be collected coincident with other water quality monitoring, and/or be included as part of a community-based monitoring effort. In some cases, it will be desirable to collect dissolved oxygen and temperature data at the same time and sites that stream biodiversity monitoring is conducted. See Chapter 10 for links to water quality monitoring approaches designed for community volunteers.

FURTHER READING

For parameters and protocols for monitoring **stream habitats and riparian areas**, see:

National Marine Fisheries Service—Streams and Rivers Monitoring
http://www.habitat.noaa.gov/restoration/techniques/srmonitoring.html

Scholz, J.G., & Booth, D.B. (2001). Monitoring urban streams: Strategies and protocols for humid-region lowland systems. Environmental Monitoring & Assessment, 71(2), 143-169.

The Cosumnes River Experience and Recommendations for Restoration Monitoring—Floodplain Restoration Success Criteria and Monitoring
http://baydelta.ucdavis.edu/reports/final/chapter2

USDA Natural Resources Conservation Service—National Biology Handbook: Stream Visual Assessment Protocol Version 2
ftp://ftp-fc.sc.egov.usda.gov/ID/technical/svap.pdf

USDA Natural Resources Conservation Service—Stream Corridor Inventory and Assessment Techniques
http://www2.q-city.com/shelbayreports/streamassess/job2.pdf

USGS National Water Quality Assessment (NAWQA) Program—Laboratory and Field Sampling Protocols for Assessing Stream Biota and Habitats
http://water.usgs.gov/nawqa/protocols/bioprotocols.html

USGS—Revised Methods for Characterizing Stream Habitat in the National Water-Quality Assessment Program
http://pubs.usgs.gov/wri/wri984052/pdf/wri98-4052.pdf

For detailed information on assessing **stream substrate embeddedness**, see:

British Columbia Ministry of Water, Land and Air Protection—Guidelines for Monitoring Fine Sediment Deposition in Streams
http://www.geoscientific.com/technical/tech_references_pdf_files/BC%20RISC%20Guidelines%20for%20Monitoring%20Fine%20Sediment%20in%20Streams.pdf

Cawthron Institute—Sediment Assessment Methods: Protocols and Guidelines for Assessing the Effects of Deposited Fine Sediment on In-stream Values, Nelson, New Zealand.
http://www.cawthron.org.nz/coastal-freshwater-resources/downloads/sediment-assessment-methods.pdf

Journal of the American Water Resources Association—A Sampler for Measuring Deposited Fine Sediments in Streams
http://onlinelibrary.wiley.com/doi/10.1111/j.1752-1688.2011.00618.x/abstract

University of California—Monitoring the Effectiveness of Instream Substrate Restoration
http://forestry.berkeley.edu/reports/InstreamSubstrateRestorationMonitoringGersteinetal2005.pdf

U.S. Army Corps of Engineers—Techniques for Measuring Substrate Embeddedness
http://el.erdc.usace.army.mil/elpubs/pdf/sr36.pdf

For detailed information on assessing **bank erosion**, see:

Rosgen, D. (2001). A practical method of computing streambank erosion rate. Proceedings of the Seventh Federal Interagency Sedimentation Conference, Reno, NV
http://www.wildlandhydrology.com/assets/Streambank_erosion_paper.pdf

U.S. EPA—Bank Erosion Prediction
http://water.epa.gov/scitech/datait/tools/warsss/pla_box08.cfm

For detailed information on assessing **stream geomorphology**, see:

Florsheim, J. L., Mount, J. F., & Constantine, C. R. (2006). A geomorphic monitoring and adaptive assessment framework to assess the effect of lowland floodplain river restoration on channel-floodplain sediment continuity. River Research and Applications, 22, 353–375. doi:10.1002/rra.911.
http://onlinelibrary.wiley.com/doi/10.1002/rra.911/abstract

For detailed information on monitoring **large woody material**, see:

Washington Department of Natural Resources—Method Manual for the Large Woody Debris Survey
http://www.dnr.wa.gov/Publications/fp_tfw_am9_99_004.pdf

Washington State Department of Ecology—Standard Operating Procedure for Counting Large Woody Debris for the Extensive Riparian Status & Trends Monitoring Program
http://www.ecy.wa.gov/programs/eap/qa/docs/ECY_EAP_SOP_LargeWoodyDebris_v1_0EAP065.pdf

ABOVE/PROXIMAL/BELOW

This monitoring design is used to evaluate freshwater habitat changes due to an activity at a given site. It is designed to be conducted both before and after the activity is implemented.

The differences in this design and the above/below design for water quality monitoring include:

- For the upstream sample reach to be a true control, its watershed must have the same type of land use/cover as the activity site before implementation, and must remain in that land use/cover after the activity is implemented.
- Proximal sampling is included, to monitor in-stream and riparian habitat changes directly linked to adjacent Water Fund activities.
- Depending on the types and downstream extent of Water Fund impacts on habitat quality, downstream monitoring may or may not detect changes. Additional downstream reaches may need to be added to monitor the extent of changes downstream resulting from a site-based activity, if that is of interest.

When to use this approach:

- This approach allows assessment of links between an activity and those in-stream habitat parameters that respond directly to local changes in sediment loading and vegetation growth.

Assumptions:

- Any changes seen/measured are due to activities at the site.

Advantages:

- Typically this approach covers a set of relatively short reaches of river, facilitating sample collection.
- If there are no changes in flow between the upstream and downstream sites, then changes can be inferred to be due to activities at the site.
- Differences can be measured in a shorter period of time than at a watershed scale, and results can be used to demonstrate the benefits of practices that are being implemented throughout a watershed.

Limitations:

- Upstream impacts and changes over time can overwhelm changes due to site-based activities.

Statistical Approaches:

This design is analyzed as a t-test of the differences between paired observations from reaches before and after activities. Parametric and nonparametric (distribution-free) t-test approaches are available. For a detailed summary of statistical approaches appropriate for above/below designs, see:

USDA Natural Resources Conservation Service—National Water Quality Handbook (Part 615, Chapter 9: Above and Below Watersheds)
http://www.wsi.nrcs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb1044775.pdf

WATERSHED AND MICRO-WATERSHED MONITORING

In-stream habitat is monitored at a set of sites throughout the watershed to represent the range of natural variation of characteristics in reaches, and to capture the overall differences due to Water Fund activities. These sites can be sampled in a single effort or monitored over time to make comparisons within a single watershed or multiple watersheds, as summarized in the descriptions of Water Quality sampling designs. The distinction here is that multiple sites are monitored, as compared to a single pour-point for water quality monitoring. Care should be taken to categorize sample sites and organize analysis according to stream size and elevation, as habitat characteristics change naturally with changes in stream size and elevation. This monitoring design can incorporate before data or trend monitoring to strengthen inference. Guidance for selection of sites for one-time sampling (synoptic sites), and sites for long-term monitoring (fixed) can be found at:

USGS—Revised Methods for Characterizing Stream Habitat in the National Water-Quality Assessment Program
http://pubs.usgs.gov/wri/wri984052/pdf/wri98-4052.pdf

Stream Biodiversity Monitoring

The composition, abundance and distribution of macroinvertebrate and fish assemblages reflect the condition of stream processes and habitats, and are a direct measure of stream biodiversity. Macroinvertebrate and fish assemblages are also responsive to most of the types of land-based activities that Water Funds conduct to improve freshwater ecosystem services.

Macroinvertebrates

Macroinvertebrates are the insects and arthropods that live in streams. Monitoring is conducted on the larval stages of insects that emerge from the water as adults, as well as adult arthropods that are permanent inhabitants of stream habitats. Macroinvertebrate assemblages are excellent monitoring candidates for several reasons. They are particularly responsive to changes in water quality (nutrients, bacteria, and sediment), temperature and organic debris; are found in most if not all stream habitats, and are relatively easily sampled and preserved.

Macroinvertebrate monitoring is also well-suited to the purposes of most Water Funds, since many small high-altitude streams do not support fish, and macroinvertebrates represent the dominant animal biomass. Macroinvertebrates also respond relatively rapidly to local changes, making them an excellent choice for assessment of management practices at the site level. In addition, identification to the family level can usually be performed accurately by trained biologists, and community-based monitoring groups can be taught to sample and identify important components of invertebrate assemblages.

PARAMETERS FOR MONITORING MACROINVERTEBRATES

Primary Parameters*:

- Composition, relative abundance, and distribution of taxonomic and functional groups (e.g., shredders, scrapers, filter feeders, predators)
- Relative abundance and distribution of sensitive/intolerant taxa
- Other aspects of assemblage that are pertinent (may be regionally specific)

**See links below for more specific parameters used for various IBIs and the Biological Condition Gradient model*

Other parameters to better assess and explain results:

- Climate (precipitation, temperature)
- Flow
- Habitat parameters

Integrated macroinvertebrate indices, such as the EPT (*Ephemeroptera, Plecoptera, Trichoptera*), or the ICI (Invertebrate Community Index), are commonly used for assessing overall ecosystem integrity, but are often not sensitive to specific changes, such as sedimentation and individual water quality parameters. Such indices are developed regionally, and distinctions for different habitats and stream size or elevation are not provided. If macroinvertebrate assemblage responses due to Water Fund activities are a focus of monitoring, attention to changes in the abundance and distribution of specific sensitive taxa are often needed. For the most accurate data on macroinvertebrate assemblages, regional invertebrate experts should be contracted to identify sensitive taxa and to select sampling techniques that are most appropriate for particular stream types, sites, and sampling designs. These experts can be found at local universities, government agencies, and NGOs.

Sampling for macroinvertebrates is generally focused in the riffles of the stream, as this is the habitat that generally contains the highest level of taxonomic diversity, and is where sensitive taxa are commonly found. If the stream lacks significant isolated riffle habitat, then other habitats and techniques will be necessary. The habitat and substrate will determine the appropriate types of sampling techniques and gear to be used. For instance, in very large rocky substrate, rocks will need to be sampled directly. In gravel and vegetated areas, kick-nets can be used. In soft substrate or substrate with small cobbles and pebbles, a Surber sampler can be used in streams with a depth of less than 50 cm. Samples are often taken for a specific length of habitat or for a subset of habitats within a stream reach, or sampling can be evaluated in terms of "time and effort." Methods should be selected by those contracted to do the sampling (or to design the sampling plan, for community-based monitoring), and must be used consistently across sites over time to allow for comparative analysis.

FURTHER READING

Macroinvertebrates have been commonly used to define ecological integrity indices in many areas around the world. See the text box below on the Biological Condition Gradient approach, which can include both macroinvertebrates and fish. Information on **macroinvertebrate index** development and examples can be found at:

Center for Ecology and Hydrology—RIVPACS (River Invertebrate Prediction and Classification System)
http://www.ceh.ac.uk/products/software/RIVPACS.html

Initial development of a multi-metric index based on aquatic macroinvertebrates to assess streams condition in the Upper Isiboro-Sécure Basin, Bolivian Amazon
http://link.springer.com/article/10.1007%2Fs10750-007-0725-3?LI=true

Macroinvertebrate-based index of biotic integrity for protection of streams in west-central Mexico
http://river.unu.edu/e-archive/16.pdf

Maryland Department of Natural Resources—Development of a Benthic Index of Biotic Integrity for Maryland Streams
http://www.dnr.state.md.us/irc/docs/00001535.pdf

Minnesota Pollution Control Agency—Development of a Macroinvertebrate Index of Biological Integrity (MIBI) for Rivers and Streams of the St. Croix River Basin in Minnesota
http://www.pca.state.mn.us/index.php/view-document.html?gid=6092

Protocolo simplificado y guía de evaluación de la calidad ecológica de ríos andinos (CERA-S)
http://www.libreroonline.com/ecuador/libros/29660/prat-fornells-narcis-garcia-katchor-natalia-rios-touma-blanca-encalada-romero-andrea-carolina-ri/protocolo-simplificado-y-guia-de-evaluacion-de-la-calidad-ecologica-de-rios-andinos-cera-s.html

Rapid Assessment of River Water Quality Using Macroinvertebrates: A Family Level Biotic Index for the Patagonic Andean Zone
http://ablimno.org.br/acta/pdf/acta_limnologica_contents1102E_files/Artigo%2011_11(2).pdf

University of Washington—Biological Integrity and the Index of Biological Integrity

U.S. EPA—Rapid Bioassessment Protocols for Use in Streams and Wadeable Rivers: Periphyton, Benthic Macroinvertebrates, and Fish
http://water.epa.gov/scitech/monitoring/rsl/bioassessment/index.cfm

A variety of macroinvertebrate **sampling methods** can be found at:

U.S. EPA—Monitoring & Assessment: Streamside Biosurvey
http://water.epa.gov/type/rsl/monitoring/vms42.cfm

U.S. EPA—Rapid Bioassessment Protocols for Use in Streams and Wadeable Rivers: Periphyton, Benthic Macroinvertebrates, and Fish
http://water.epa.gov/scitech/monitoring/rsl/bioassessment/index.cfm

USGS—Revised Protocols for Sampling Algal, Invertebrate, and Fish Communities as Part of the National Water-Quality Assessment Program
http://pubs.usgs.gov/of/2002/ofr-02-150/pdf/ofr02-150.pdf

Fish

Many Water Funds operate in high-elevation watersheds which often contain few, if any, native fish species. However, for larger, lower elevation streams, fish assemblages can be monitored as indicators of stream integrity. As with macroinvertebrate assemblages, fish assemblages are excellent monitoring candidates for several reasons: they integrate stream processes and habitat conditions over their life cycles; respond to changes in water quality (nutrients, bacteria, and sediment), temperature and organic debris; and are relatively easily sampled and preserved. In addition, fish assemblages can provide information on biodiversity that is not only relevant for assessing the impacts of Water Fund activities, but is also directly important to the public because of recreational fishing or commercial fisheries.

Fish assemblages have been used for decades to generate regional indices, such as the Index of Biotic Integrity (IBI) for warm-water streams in the US. Fish indices have also been developed and tested for many areas around the world (see examples in the *Further Reading* list below).

PARAMETERS FOR FISH MONITORING

Primary Parameters:

- Composition, relative abundance, and distribution of taxonomic and functional groups (e.g., benthic feeders, scrapers) trophic groups (e.g., insectivores, omnivores, top predators) and spawning groups (e.g., gravel spawners)
- Relative abundance and distribution of sensitive/intolerant taxa
- Specific indicator species
- Other pertinent aspects of assemblage (may be regionally specific)

**See links below for more specific parameters used for various IBIs and the Biological Condition Gradient model*

Other parameters to better assess and explain results:

- Climate (precipitation, temperature)
- Flow
- Habitat parameters

While the same set of metrics from a fish index generated for one region may not be applicable elsewhere, the general concept is transferable –a relative measure of stream integrity based on the response of the fish assemblage to a range of conditions. If a fish assemblage index does not exist for your area, sampling fish assemblages in the areas of interest and across ranges of conditions of similar habitats will allow for comparisons of assemblages across conditions, and initial metrics for an index can be developed. At a minimum, sampling in extreme ranges of conditions from highly degraded to pristine can provide valuable initial information on the characteristics of fish assemblages as they relate to stream conditions. See the text box below on the Biological Condition Gradient, an alternative approach which can provide more flexibility and sensitivity than a traditional IBI.

As with macroinvertebrates, an overall index may not be sensitive to specific changes resulting from Water Fund activities. Additionally, overall indices are generally developed regionally, and do not distinguish between stream sizes or elevation. Identification of taxa or groups of taxa known to be sensitive to changes resulting from specific activities may be selected working with regional experts.

FURTHER READING

For more information on **existing Fish IBI and other index approaches**, see:

A Preliminary Index of Biotic Integrity for Monitoring the Condition of the Rio Paraiba do Sul, Southeast Brazil
http://www.ufrrj.br/laboratorio/lep/pdfs/rios_e_riachos/2003%20A%20preliminar%20index%20of%20Biotic%20Integrity.pdf

An index of Biotic Integrity Based on Fish Assemblages for Subtropical Streams in Southern Brazil
http://link.springer.com/article/10.1007%2Fs10750-004-5738-6?LI=true#

Development and Application of a Predictive Model of Freshwater Fish Assemblage Composition to Evaluate River Health in Eastern Australia
http://link.springer.com/article/10.1007%2Fs10750-005-0993-8?LI=true

Development of a Preliminary Index of Biotic Integrity (IBI) Based on Fish Assemblages to Assess Ecosystem Condition in the Lakes of Central Mexico
http://link.springer.com/article/10.1023%2FA%3A1003888032756?LI=true

Development of an Index of Biotic Integrity Based on Fish Communities to Assess the Effects of Rural and Urban Land Use on a Stream in Southeastern Brazil
http://onlinelibrary.wiley.com/doi/10.1002/iroh.201111297/abstract

Evaluation of the Index of Biotic Integrity in the Sorocaba River Basin (Brazil, SP) Based on Fish Communities
http://www.ablimno.org.br/acta/pdf/acta_limnologica_contents1603E_files/Art2_16(3).pdf

The fish Community as an Indicator of Biotic Integrity of the Streams in the Sinos River Basin, Brazil
http://www.scielo.br/pdf/bjb/v70n4s0/v70n4s0a09.pdf

Examples of metrics and approaches for **developing a fish Index of Biotic Integrity** can be found at:

Big Darby Creek Watershed (Ohio) Index of Biological Integrity
http://www.darbywatershed.com/IBI.htm

Maryland Department of Natural Resources—Refinement and Validation of a Fish Index of Biotic Integrity for Maryland Streams
http://www.dnr.state.md.us/streams/pdfs/ea-00-2_fibi.pdf

U.S. EPA—Monitoring & Assessment: Fish Protocols
http://water.epa.gov/scitech/monitoring/rsl/bioassessment/ch08b.cfm

Although specific fish taxa are known to be sensitive to habitat quality change, making them good indicators of changing stream conditions, many fish are not responsive to interventions that address a specific problem, and their mobility means they are often not helpful for site-specific evaluations. Because they integrate long-term changes and reflect many factors operating in the aquatic system, fish assemblages can however be good indicators for assessing the effects of a Water Fund at a watershed scale.

The size of the stream and the types of habitats determine the appropriate sampling techniques to be used.For instance, seine nets, dipnets and electro-fishing can be used in small wadeable streams. Larger streams often require gill nets, hoop nets, beach and shoreline seining, and boat electro-fishing.Electro-fishing should only be conducted by trained professionals that have proper gear and emergency medical training.

For examples and approaches to develop fish and macroinvertebrate **multi-metric bio-indicators**, see:

Fish and Macroinvertebrates as Freshwater Ecosystem Bioindicators in Mexico: Current State and Perspectives
http://link.springer.com/chapter/10.1007%2F978-3-642-05432-7_19?LI=true

Examples of **fish sampling approaches** can be found at:

Minnesota Pollution Control Agency—Fish Community Sampling Protocol for Stream Monitoring Sites
http://www.pca.state.mn.us/index.php/view-document.html?gid=6087

U.S. EPA—Rapid Bioassessment Protocols (Chapter 8: Fish Protocols)
http://water.epa.gov/scitech/monitoring/rsl/bioassessment/ch08main.cfm

USGS—Revised Protocols for Sampling Algal, Invertebrate, and Fish Communities as Part of the National Water-Quality Assessment Program
http://pubs.usgs.gov/of/2002/ofr-02-150/pdf/ofr02-150.pdf

The Biological Condition Gradient (BCG) is a model developed to assess the integrity of stream ecosystems. It is based on ten attributes pertaining to fish and macroinvertebrates that reflect level of environmental stress, including community structure (e.g., sensitive/tolerant taxa), organism condition (e.g., fecundity), ecosystem function (e.g., rate of decomposition), and habitat connectivity. The model provides a standardized method of assessing the level of disturbance at the reach level, facilitating prioritization of areas for restoration and/or protection. The BCG can also be used for monitoring improvements in stream conditions over time.

For details on using the BCG, see:

Davies, S., & Jackson, S. (2006). The biological condition gradient: A descriptive model for interpreting change in aquatic ecosystems. Ecological Applications, 16(4), 1251–1266.

New Jersey Department of Environmental Protection—Biological Condition Gradient for Tiered Aquatic Life Use in New Jersey
http://www.state.nj.us/dep/wms/bwqsa/FINAL%20TALU%20NJ%20RPT_2.pdf

Sampling design for aquatic biodiversity

Sampling design for freshwater biodiversity should be coordinated with habitat sampling and can be conducted using the sampling designs presented for evaluating water quality, flow (Chapter 10), and changes due to site-based terrestrial activities. In addition, invertebrate sampling can be conducted in conjunction with the Above/Proximal/Below design presented for habitat monitoring earlier in this chapter, as invertebrates typically respond strongly to local changes.

Chapter 12: Impact Monitoring for Communities

Assessing impacts on human well-being

In focusing on water-related services important to human well-being, Water Funds inherently have joint social and ecological goals (Goldman-Benner et al., 2012). Beyond socioeconomic goals for improved downstream water supply, Water Funds may also have explicit primary or secondary objectives to improve livelihoods and contribute to poverty alleviation and human well-being in participating communities. These goals are often set in the context of payment or compensation for ecosystem (PES or CES) programs, which compensate landowners to manage their land in ways deemed to maintain or enhance targeted ecosystem services. Accordingly, in line with Water Fund goals, monitoring of impacts on communities and individuals is a critical component of evaluating outcomes.

Environmental goals echo wider debates surrounding the most effective way to link conservation and development objectives (Roe, Yassin Mohammed, Porras, & Giuliani, 2012). While the prioritization of social outcomes of PES varies, there is widespread interest in understanding the socioeconomic impacts of PES, both positive and negative (Landell-Mills & Porras, 2002; Miranda, Porras, & Moreno, 2003; Stefano Pagiola, Arcenas, & Platais, 2005; Wunder, 2008). This relates both to concerns over the equity of PES/CES, as well as to increased recognition that support for conservation and ecosystem services programs are unlikely to gain support without demonstrating the value of these programs to human welfare (Gockel & Gray, 2009; Luck, Chan, & Fay, 2009).

Although empirical research on PES/CES participation remains limited (Brockington, 2011), a number of studies have begun to evaluate and analyze PES/CES effects on poverty and human well-being, including analysis of PES participation patterns and human well-being outcomes (Kosoy, Martinez-Tuna, Muradian, & Martínez-Alier, 2007; Miranda et al., 2003; Wunder, 2005; Zbinden & Lee, 2005).

According to Wunder (2008), the potential of PES/CES to contribute to poverty alleviation depends upon 1) whether poor smallholders and communities participate in PES/CES, 2) how participation in PES affects participants' livelihoods, and 3) the effect of PES/CES on poor non-participants. While a Water Fund's social goals may be broader than poverty alleviation, this framework provides a way to understand upstream socioeconomic impacts of PES/CES.

In the context of Water Funds, the first component, "equity in access," or the extent to which programs will include the poor, depends of whether areas with high potential for ecosystem services production and high levels of poverty coincide and whether the poor have the ability and desire to participate (Brown & Corbera, 2003; Engel, Pagiola, & Wunder, 2008; Wunder, 2005). Outcomes of previous studies evaluating PES participation patterns range from those noting substantial participation among poor smallholders (Munoz-Pina, Guevara, Torres, & Brana, 2008; S Pagiola, Rios, & Arcenas, 2008; Wunder & Alban, 2008) to those finding that PES programs tend to favor wealthier larger landowners (Grieg-Gran, Porras, & Wunder, 2005; Kollmair & Rasul, 2010; Zbinden & Lee, 2005). Factors identified as important in influencing landowner ability to participate in PES include land tenure (Landell-Mills & Porras, 2002; Wunder, Engel, & Pagiola, 2008), the extent to which spatial targeting of ecosystem services overlaps with areas high in poverty (Luck et al., 2009); and access to social, financial, and human capital necessary to complete program requirements (Stefano Pagiola et al., 2005; Zbinden & Lee, 2005).

In turn, important factors related to landowner desire to participate in PES/CES include how conservation opportunity costs compare to incentive payments, perceived effects of participation on land and food security (Grieg-Gran et al., 2005; Miranda et al., 2003; Wunder et al., 2008; Zbinden & Lee, 2005), environmental attitudes (Muradian & Rival, 2012), and trust in government and NGO conservation and development programs. Where equity in access and/or poverty alleviation is a primary goal of Water Funds, it will be important to identify factors facilitating and constraining participation in order to improve access among upstream poor communities and smallholders.

Another key component of upstream socioeconomic impacts of Water Funds via PES/CES depends upon the extent to which the well-being of program participants changes as a result of program participation (Grieg-Gran et al., 2005; Landell-Mills & Porras, 2002; Miranda et al., 2003). Human well-being has been measured in a variety of ways through different methodologies, but there is general consensus that measurements of human well-being should include the domains of health, education, social cohesion, safety and security, and material living standards (Smith et al., 2012). These factors have similarly been evaluated in the context of PES/CES in multiple forms of capital beyond financial, including natural capital (shifts in land use or management that affect biodiversity and ecosystem goods and services), human capital (health and basic services, access to education and training), and social capital (land tenure, social organization, community institutions and associations, kinship ties) (Grieg-Gran et al., 2005; Zbinden & Lee, 2005).

Water Fund-related PES/CES can affect financial capital or material living standards through incentive payments or through land use or management change related to participation. Some researchers have pointed to the risk of a "PES trap" if incentive payments or compensation are lower than actual or potential earnings from productive land uses(Grieg-Gran et al., 2005). However, others argue that PES can be an important means of income diversification and are often more stable than existing or potential income sources, regardless of whether they strictly match opportunity costs (Grieg-Gran et al., 2005; Kollmair & Rasul, 2010; Wunder et al., 2008).

Researchers have noted both positive and negative impacts of PES on non-financial capital, including, for example, impacts on land tenure, social organization, and natural capital (Grieg-Gran et al. 2005; Miranda et al. 2003). Southgate and Wunder (2007) reported that impoverished families increased household income by up to 10% through engaging in the National Program for Hydrologic-Environmental Services in Mexico, whereas Ruiz-De-Ona-Plaza et al. (2011) found that there was no change in the economic status of Mayan households receiving PES for carbon sequestration in Chiapas. Wunder (2005, citing Muñoz 2004) reported on a PES scheme in Costa Rica's Osa Peninsula that raised half of the participants above the poverty line, and noted that PES became the primary source of income for 44% or those enrolled.

The opportunity costs for landowners receiving PES/CES are sometimes difficult to quantify, and in some cases may outweigh the value of payments or other compensation (Grieg-Gran et al. 2005; Pagiola et al. 2005). Some studies have found that when landowners remove land from agricultural production or protect it from development, they have been fully compensated for the opportunity cost (Postel et al. 2005), whereas in other cases the opportunity cost associated with crop or animal production has been much higher than the amount paid for ecosystem services (Kosoy et al. 2006; Corbera et al. 2006). For example, Corbera et al. (2006) estimate the opportunity cost for crop production in the San Pedro del Norte, Nicaragua, at US$126/ha/year, compared to PES received by participants of only $26/ha/year. However, when the PES is lower than the opportunity cost, these programs may still be attractive to landowners because payments are predictable and do not fluctuate with weather extremes and market conditions, as agricultural income does (Pagiola et al. 2008).

It should be noted that sustainable farming can bring about economic and social benefits even in the absence of PES/CES, and these benefits may not always be adequately quantified in calculating the overall balance between PES/CES benefits and opportunity costs. Researchers have identified multiple types of economic benefits for farmers engaged in various agroecological practices, such as increased crop yields (Altieri 2002; Ayarza et al. 2010; Pretty et al., 2003), increased rates of weight gain in cattle on silvopastoral land (Nair et al. 2011), reduced susceptibility of crops to drought or hurricane (Ayarza et al. 2010; Rosset et al., 2011), increased food security (Altieri 2002; Ayarza et al. 2010), and increased land value (Ayarza et al. 2010).

PES/CES programs can also create social capital as the decision-making processes connect stakeholders and require strong community relationships (Pagiola et al. 2005). The social benefits of adopting agroecological practices in Latin America have ranged from increased community organization and political power (especially in the CAC movement), increased employment opportunities, and shifts in gender roles where women lead more aspects of the farming operation (Rosset et al. 2011).

Monitoring Impacts in Communities

Water Funds should monitor the costs and benefits of participation in Water Fund programs, in order to assure that Fund activities are having positive impacts on human well-being, and that benefits are distributed equitably within and among local communities and individuals. Monitoring the human well-being impacts of Water Funds is also key to ensuring the long-term sustainability of these funds, as these programs are unlikely to be successful without support from local stakeholders. Potential human well-being benefits include higher living standards from conservation payments and non-monetary incentives, health benefits from cleaner water, increased environmental quality, and improvement in natural capital associated with Water Fund activities. There may also be cases where incentives, payments, and/or Water Fund activities have negative impacts for some stakeholders, which are equally important to monitor.

In addition, monitoring human well-being impacts can provide Water Fund managers with feedback on the relative value of their investments, including both the benefits accrued by participating communities and the level of involvement and support that has been engendered for Water Fund activities. This information will allow for a more equitable allocation of Water Fund resources, as managers adjust priorities and approaches in response to on-the-ground results.

When assessing the impact of a Water Fund, several key questions are worth exploring, concerning three general areas of concern: 1) impacts on human well-being of participating and non-participating communities and individuals; 2) equity in outcome; and 3) equity in access.

Area 1: Human Well-being

Water Funds can affect the well-being of participants and non-participants in three primary ways: 1) impacts related to Water Fund-related changes in land use and management; 2) impacts of Water Fund payments or incentives; and 3) changes in ecosystem services associated with Water Fund activities.

Addressing questions concerning human well-being related to payments or compensation incentives involves an analysis of financial impacts as well as effects on non-financial capital including human, natural, social, and physical capital. Because Water Funds may require that participants remove cattle and crops (or avoid further intensification) from some areas, and/or implement other best management practices, there can be a loss of income associated with program participation. In addition, nonparticipants may suffer reduced access to grazing or other productive lands with increased regulation of *de facto* commons. Such losses should be evaluated in the context of how incentive payments, compensation, or sustainable development projects offset these losses. Although Water Fund land use regulations may lower financial income in the short term, some activities may increase production in the long term.

Human well-being can also be conceived of as an endpoint measure of ecosystem services; conservation practices implemented by the Water Fund may, for example, alter water quality and quantity, with direct impacts for both upstream and downstream users. The evaluation of human well-being in terms of changes in ecosystem services provides a direct link between ecological and human well-being measurements. However, evaluating the impacts of changes in ecosystem services on human well-being is complex and requires information on supply (ecological information) of a defined attribute (e.g., water quality), demand (demographic information), and access (socioeconomic and political information). The Natural Capital Project at Stanford University (http://www.naturalcapitalproject.org/about.html) is currently working on creating a model to evaluate the human well-being impacts of changes in ecosystem services, which may be useful to Water Fund managers.

In addition to ethical considerations, understanding the Water Fund's effects on communities and individuals is important to gauging public support for Water Fund activities. Given that participation in Water Funds is voluntary, local perception of net benefits is essential for sustaining participation, and thus to the long-term sustainability and success of the Water Fund. Conservation opportunity costs will change with shifts in economic, political, and demographic opportunities and constraints. Monitoring these perceptions will allow Water Fund managers to anticipate shifts in perceived benefits or risks of participation, and adapt incentives accordingly.

Example Questions:

- What are the levels of fecal coliform in surface waters at drinking water access points? What is the incidence of water-borne diseases?
- How have perceptions of well-being changed for participating versus non-participating people?
- What is the financial status of participants and non-participants?
- How have land-use transitions due to Water Fund activities affected availability of natural resources (e.g., timber, firewood, medicinal plants)?

A framework for measuring human well-being impacts

Human well-being measurements need to focus on areas that are relevant to community members as well as on Water Fund goals. Owing to the unique mix of priorities and socioeconomic conditions in each region, and the range of activities that might be conducted by Water Funds in different areas (e.g., education projects, organic gardens), Water Funds should select human well-being focal areas and their nested indicators relevant to their site.

The framework presented here is based on Amartya Sen's exploration of how "capacities" and "freedoms" often determine human well-being across eight categories: material living standards; health; education; personal activities including work, political voice and governance; social connections and networks; environmental conditions; and insecurity. Sen partnered with another Nobel Prize-winning economist, Joseph Stiglitz, as well as Jean-Paul Fitoussi, to produce a framework for measuring human well-being that has been subsequently adopted by a number of organizations and countries (Stiglitz et al., 2009). Table 5 provides a summary of these focal areas, including two additions (based on Smith et al., 2012) which may be relevant for some Water Funds: spiritual and cultural fulfillment; and life satisfaction and happiness. This framework can be used to guide questions posed in interviews, focus groups, and surveys designed to assess impacts on human well-being for participants and non-participants.

TABLE 5. *Definitions and parameters for assessing human well-being.*

HUMAN WELL-BEING FOCAL AREAS	DEFINITION	POSSIBLE PARAMETERS
Material living standards	Life's physical circumstances; accessible goods and services and economic resources	• Income • Living conditions (locally defined metrics) • Consumption • Access to ecosystem goods (fodder, timber, clean water) • Perception of living standards • Distribution of wealth • Health of cattle (related to ecological monitoring impacts)
Health	Health outcome measures of human well-being including life expectancy and mental and physical health	• Life expectancy and mortality • Lifestyle and behavior • Food security (related to WF activities; e.g. organic gardens) • Access to clean water (link to ecological monitoring) • Access to sanitation • Access to health clinics and social services • Human health impacts of changes in other ecosystem services (e.g. clean air) related to Water Fund activities • Access to green space and nature (mental and physical health implications)
Education	Formal and informal transfer of knowledge and skills	• School attendance by gender and highest level of education • Informal and formal workshops and training opportunities associated with Water Fund activities or compensation. • Incorporation/valuation of local knowledge
Personal activities including work	Time use	• Hours of work versus hours of leisure • Time spent on formal versus informal work • Changes in the average time women spend on informal work
Political voice and governance	Influence of local leadership environment	• Perceived ability to influence local leadership environment • Disputes in community and local conflict resolution mechanisms • Frequency and quality of community meetings and work groups
Social connections and networks	Social cohesion; social capital	• Indicators of social trust, social isolation, informal support, work, community, and religious engagement • Number of and frequency of interaction in social associations
Environmental conditions	Quality of environment; related to human health, cultural and spiritual fulfillment, and material living standards	• Access and consumption of ecosystem goods and services • Human well-being metrics associated with environmental conditions and services
Insecurity	Freedom from harm and economic security	• Perceived environmental and personal security
Spiritual and cultural fulfillment	Opportunities to fulfill spiritual and cultural needs	• Religious or festival attendance • Programs promoting local cultural and language • Cultural services obtained from landscapes targeted by WF activities
Life satisfaction and happiness	Contentment with life	• Self-reported happiness and well-being • Relate to changes in ecosystem services and activities associated with WF

The parameters listed above are only a few of the many available. The World Bank has a database of indicators that may be helpful (http://data.worldbank.org/indicator). The many Demographic and Health Surveys are also a good source of indicators (http://www.measuredhs.com/data/Survey-Indicators.cfm). Using government indicators for areas such as health, education, and material living standards will reduce the work of developing viable local indicators, allow for the comparison of Water Fund site data with other sites, and provide impact assessment data using metrics familiar to many policymakers.

Area 2: Equity in outcome

For both communities enrolling communal land or individuals enrolling private land, it is important to evaluate how the risks and benefits of Water Fund activities are distributed among groups, including between men and women, and between poorer and wealthier people. This includes a consideration of how Water Fund activities affect wealth distribution and gender roles among participants and nonparticipants. Exploring equity in outcome (Brown & Corbera, 2003) requires interviews/surveys targeted at individuals (including both men and women) rather than household surveys, which tend to under-represent the opinions and experiences of women.

Example Questions:

- Are risks and benefits associated with Water Fund (PES/CES) activities distributed equitably within participating communities?
- What are the risks and benefits of participation among men and women? Young and old? Different socio-economic and/or cultural groups?

Sampling Design: Human Well-being & Equity in Outcome

A mixed-methods approach, combining qualitative and quantitative methods, facilitates a comprehensive understanding of the Water Fund's impact on human well-being for communities and individuals. In social science, qualitative methods are the most effective way to address process questions and gauge perceptions, while quantitative methods are best for answering questions about patterns, causation, and relations (Creswell & Clark, 2010; Sayer, 2000). An effective approach is to begin with qualitative methods, such as focus groups and interviews with key community members, to explore how and why Water Fund programs are impacting some community members or individuals. Qualitative methods can also help to identify local values, power dynamics, and well-being indicators critical to designing effective quantitative surveys. This research can be followed by quantitative surveys or interviews, which measure human well-being impacts. Quantitative surveys also allow for statistical comparisons between control and impact groups, producing data particularly useful for understanding which impacts were actually driven by Water Fund activities and which were not. This stage can, in turn, then be followed by qualitative interviews and focus groups to explore findings of the survey.

Design 1: Qualitative only

In cases where Water Fund activities 1) restrict access to natural resources, 2) displace people, or 3) pilot a new upstream approach, inclusion of a robust quantitative approach is highly recommended for assessing human well-being, as discussed below. When none of these characteristics apply, upstream human well-being impacts can typically be adequately assessed through a qualitative impact assessment such as a KAP (knowledge, attitudes, and practices) or Most Significant Change approach. Both use focus group discussions and key informant interviews to assess perceived changes in human well-being. Focus group discussions and key informant interviews can also be used as part of an adaptive management approach, with focus groups meeting every six months after activities begin to identify what aspects of project activities are working and what are not. In the later years of a Water Funds, these can be held less frequently. As with any form of qualitative data collection, it is important for researchers/facilitators to establish trust with the participants, and critical to examine one's own "positionality" as a researcher and how it affects the collection and interpretation of data.

Design 2: Difference in differences (DID), mixed qualitative and quantitative methods

When Water Fund activities 1) restrict access to natural resources, 2) displace people, or 3) pilot a new upstream approach, a difference in differences (DID) design, which combines qualitative and quantitative methods**,** is recommended to measure changes in the well-being of participants. This approach uses key informant interviews to design the protocols for focus group discussion, and the results of the focus group discussion to inform the design of a quantitative survey. The DID approach, a social science before-after, control-impact (BACI) design, analyzes differences between before and after and between control and impact groups. The pre-existing differences between control and impact groups are subtracted from the post-treatment differences to provide a projection of what would have happened in the absence of Water Fund activities (the "business as usual" scenario). This "counterfactual" data is then subtracted from the before-after differences for the impact group for the difference in differences result. Establishing control and impact baselines is critical for this approach. If there are no baseline data, this impact assessment approach is not possible.

A key challenge for the DID design is to identify a control group which can be compared with the impact (participant) group in terms of changes in well-being over time. Data from the control group is intended to reflect what conditions would have been for the impact group but for the influence of Water Fund activities. Control and impact groups should be similar as possible to each other prior to Water Fund activities in terms of material living standards, health, education, social cohesion, environmental quality, and security, and should be subject to similar socioeconomic, political, environmental, demographic, and cultural factors for the duration of the Water Fund program, with the exception of Water Fund activities. Matching of control and impact groups should be done first at the broad scale and then at the fine scale.

Although the DID design is recommended in Water Funds with the potential for significant human well-being impacts, resource constraints may preclude an expensive survey. In these cases, a qualitative KAP assessment (as described above) using key informant interviews and focus groups can provide anecdotal information on how and why Water Funds are impacting people. Without quantitative surveys, statistical comparisons cannot be made to indicate how extensive these impacts are, but results of qualitative studies nonetheless provide valuable input to Water Fund managers regarding perceived impacts and potential adaptations.

Previous research on human well-being and Water Funds suggests important caveats for evaluating impacts. First, it may be challenging to identify control groups that are similar to impact groups. Second, there can be a self-selection bias among participants in Water Funds. Third, it may be difficult to control for confounding factors or a "noisy" socioeconomic context, hence hindering our ability to isolate changes in human well-being due solely to Water Fund activities. Finally, issues of survey fatigue, confirmation bias, and a lack of trust of researchers by communities may compromise the validity of survey or interview results.

Following the same individuals in both control and impact groups over time (panel data) can help to overcome some of these limitations. Panel data involves gathering data on human well-being indicators for participating and non-participating individuals over time. Panel data reduces the importance of control and impact groups having the same initial conditions, because the important parameter is the relative change in well-being over time, rather than the absolute change versus pre-Water Fund conditions. However, this approach carries important challenges as well, including a need to aggregate time-series data to produce statistically robust results. Furthermore, baseline data collected at a single point in time will not necessarily capture divergent pre-project human well-being conditions and trends in control and impact groups; accordingly time-series data prior to program implementation is needed. As few Water Funds are likely to have pre-project time-series data on human well-being conditions of interest, secondary data sources are often the best alternative.

Where baseline data include national datasets, impact and control groups can also be compared with propensity-score matching (see Andam et al., 2010). Here the characteristics of participants, such as income level, education, and age, are used to calculate a propensity-to-participate score, and treatment and control units are matched by propensity score, thereby creating a control group with the same likelihood of participating as the impact group. This approach requires large datasets to identify a sufficient number of matches for statistical rigor, since unmatchable observations must be excluded. Projects that lack rigorous baseline data or a dataset large enough for propensity-score matching may be unsuitable for a quantitative approach to human well-being outcome assessment.

Designing the quantitative sample size, survey instrument, and sample frame are specialized skills, and it is usually better to hire survey experts to lead this work. If any of these elements are not correctly chosen, the results of the study may be unusable.

Determining sample size requires knowing the size of the total population to be sampled. As a rule of thumb, for Water Funds with less than 100 people in upstream areas affected by payment or compensation programs, a quantitative survey is rarely cost-effective and qualitative methods are usually more suitable. For Water Funds with between 100 and 400 participating households, a quantitative survey of all the population should be used. For Water Funds with more than 400 people upstream, a random sample of the population should be selected. For most Water Funds, study populations are likely to be smaller than 100,000, so the sample size can be determined based on the target confidence interval (usually 95%). Rough guidelines on sample size are n = 300 for populations up to 10,000 and n = 350 for populations up to 100,000. Oversampling by 10% is advisable due to the likelihood of some data recording errors and incomplete surveys.

For quantitative study results to be valid, participants in control and impact groups must be randomly selected. Survey participants can be selected using a numbered list of community residents and a random number generator (http://www.random.org). See http://www.povertyactionlab.org/methodology/what-randomization for more information on random assignment for socioeconomic studies.

Area 3: Equity in Access

Although the majority of Water Funds spatially target activities and participants based on ecosystem services provision, in some cases poverty alleviation may also be an important goal. Where this is the case, monitoring of participation patterns is important to understand the extent to which Water Funds are including small landholders and the poor. Monitoring for "equity in access" (Brown & Corbera, 2003) should include identifying the factors facilitating or constraining the ability and desire to participate in Water Fund projects, so that program design and outreach can be adapted accordingly.

Example Questions:

- Who is participating in Water Fund (payment/compensation for ecosystem services; PES/CES) activities?
- Is participation in Water fund PES/CES programs both accessible and desirable to poor people and communities?

Sampling Design: Equity in Access

Water Funds often favor large landholders because the return on investment is higher and the transaction costs lower than working with small landholders. Yet if only large landholders participate in a Water Fund, local support for the Water Fund outside the small group of beneficiaries may be lacking. Without widespread upstream support for a Water Fund, water quality and quantity goals may be undermined by a few actors who oppose the Water Fund. If Water Fund benefits are accessible to a wide group, the ability to reach a critical mass of participation and support is more likely.

Water Funds should, first, decide the scale at which equity in access is important. For example, it may be important that Water Fund activities are accessible and desirable to a wide range of individuals within the watersheds targeted by the Water Fund.

Equity in access questions will primarily be addressed by analyzing baseline socioeconomic, demographic, political, cultural, and environmental characteristics of participants versus non-participants. Potential factors such as income, livelihood source, farm and family size, land tenure, slope, etc., can be obtained either through secondary sources (e.g., census data) or through quantitative surveys and qualitative interviews. Depending on the scale of interest described above, a control group should be selected. However, unlike equity in human well-being impacts (discussed above), the control group should not be chosen to match the participant group in socioeconomic, demographic, environmental, and political factors. Rather, the control group should be randomly selected based on the area/scale of interest. Where the area of interest is small, nearest neighbors may be sufficient to choose a control group. In essence, the idea is to understand how participants compare to non-participants of interest, which depends on the scale at which the Water Fund chooses to focus.

Mixed qualitative and quantitative methods are an effective approach to answer equity in access questions with similar sample sizes as suggested above for human well-being questions. However, unlike human well-being impacts, equity in access studies do not require monitoring over time, but rather a comparison of baseline data, plus qualitative and quantitative questions targeted at participants and non-participants that seek to understand landowners' reasons for participating or not participating. While some factors facilitating participation may be illuminated through quantitative methods (e.g., time on farm, family size, farm size, etc.) some reasons for participating or not participating may be better understood through open-ended questions which encourage individuals to explain, in their own words, the factors constraining or facilitating their ability or desire to participate.

Where the upstream impacts of Water Fund activities are considered limited in size or influence, an alternative approach involves qualitative interviews only, combining focus groups and key informant interviews for both participants and non-participants.

Potential factors facilitating or constraining participation in Water Fund activities include:

Demographic:	Environmental:	Political:	Socio-economic:	Cultural:
Gender	Land productivity (soils etc.)	Land tenure	Education	Environmental attitudes
Family size	Slope	Pre-existing protected areas or environmental laws affecting area	Community organization	Spiritual/religious values
Farm size	Elevation	Trust of government/NGO programs	Social networks	Cultural environmental values
Time on farm	Land cover		NGO interaction	
	Climate		Income	
			Livelihood source (farm or non-farm based)	

Equity of access can be increased by targeting a proportion of Water Fund activities specifically at those less likely to participate ("set-asides") such as women, small landholders, or the landless. For the social sustainability of upstream activities, and to engender self-policing by local people of compliance with agreed land-use changes, broad-based local support is critical. Such support depends on a quorum of upstream people perceiving net benefits from the Water Fund.

FURTHER READING

For more information on **KAP** and other survey approaches, see:

A Study and Development on Knowledge, Attitude and Practice in Forest Conservation and Reforestration of Youth in Ban Khao Phra Community Nakhon Nayok Province (Thailand)
http://www.medwelljournals.com/abstract/?doi=sscience.2010.554.558

FAO—Participatory Survey Methods for Gathering Information
http://www.fao.org/docrep/W8016E/w8016e01.htm#knowledge,%20attitude%20and%20practice%20surveys

Knowledge, Attitudes and Practices Concerning Community Conservation in the Group Ranches around Amboseli National Park (Kenya)
http://www.awf.org/content/document/detail/3262

Whole Village Project in Tanzania
http://wholevillage.umn.edu/

For more information on the **Most Significant Change** technique, see:

Measuring social impacts in conservation: Experience of using the Most Significant Change Method
http://journals.cambridge.org/action/displayAbstract?fromPage=online&aid=2450648

The 'Most Significant Change' (MSC) Technique: A Guide to Its Use
http://www.mande.co.uk/docs/MSCGuide.pdf

For more information on **interviewing** techniques, other **qualitative approaches**, and **mixed qualitative/quantitative methods** for conducting social research, see:

Creswell, J., & Clark, V. (2010). *Designing and Conducting Mixed Methods Research*. Sage Publications. Retrieved from http://onlinelibrary.wiley.com/doi/10.1111/j.1753-6405.2007.00097.x/full

Dunn, K. (2005). Interviewing. In L. Hay (Ed.), *Qualitative Research Methods in Human Geography* (Vol. 2nd, pp. 79–105). Oxford: Oxford University Press.

McDowell, L. (2010). Interviewing: Fear and Liking in the Field. In D. DeLyser, S. Herbert, S. Aitken, M. Crang, & L. McDowell (Eds.), *The SAGE Handbook of Qualitative Geography* (pp. 156–171). London: Sage Publications, Ltd.

Sayer, A. R. (2000). *Realism and Social Science*. London: Sage Publications.

For information on **designing PES programs**, see:

Greenwalt, T., & McGrath, D. (2009). Protecting the City's Water: Designing a Payment for Ecosystem Services Program. *Natural Resources & Environment, 24*(1).

Kelsey Jack, B., Kousky, C., & Sims, K.R.E. (2007). Designing payments for ecosystem services: Lessons from previous experience with incentive-based mechanisms. *Proceedings of the National Academy of Sciences, 105*(28), 9465-9470.
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Appendix

Additional Watershed Monitoring and Conservation Planning Resources

Sources of information on a variety of types and impacts of land-based management practices that address water quality and quantity:

Natural England
http://www.naturalengland.org.uk/ourwork/farming/csf/casestudies.aspx

North Carolina State University—Watershedss: A Decision Support System for Nonpoint Source Pollution Control
http://www.water.ncsu.edu/watershedss/

Stream Restoration Databases and Case Studies: A Guide to Information Resources and Their Utility in Advancing the Science and Practice of Restoration
http://www-personal.umich.edu/~dallan/pdfs/Jenkinson_2006.pdf

USDA Natural Resources Conservation Service—Conservation Effects Assessment Project (CEAP)
http://www.nrcs.usda.gov/wps/portal/nrcs/main/national/technical/nra/ceap

Watershed Connect—Resources
http://www.watershedconnect.com/resources/?view=recent

World Overview of Conservation Approaches and Technologies
https://www.wocat.net/

Sources of additional information on monitoring approaches:

Project Monitoring: A Guide for Sponsors in the Upper Columbia Basin
http://www.nwcouncil.org/dropbox/2008amend/cbfwa/Section_2/Sec2.1.5RM&E/Hillman_2005.pdf

The Western Center for Monitoring and Assessment of Freshwater Resources
http://www.cnr.usu.edu/wmc/

U.S. Army Corps of Engineers—Evaluating Cumulative Ecosystem Response to Restoration Projects in the Columbia River Estuary (Annual Report 2004)

U.S. Army Corps of Engineers—Protocols for Monitoring Habitat Restoration Projects in the Lower Columbia River and Estuary.
http://www.pnl.gov/main/publications/external/technical_reports/PNNL-15793.pdf

US EPA—Monitoring Guidance for the National Estuary Program
http://water.epa.gov/type/oceb/nep/upload/2009_03_13_estuaries_wholeguidance.pdf

USGS: Links to Monitoring Resources
http://search.usgs.gov/results.html?cx=005083607223377578371%3Ab5ixbbpqpx0&cof=FORID%3A11&q=monitoring+guidance&sa=Search#1010

Annotated Bibliography: Impacts of Restoration and Conservation Activities

Researchers: Willow Batista, Melanie Jonas, Vanessa Perkins, Pamela Krone-Davis, and Adrian Vogl

This review includes research addressing activities similar in type and scale to those being conducted by many Water Funds. It does not provide a formal systematic review or meta-analyses, but can serve as a starting point for conducting a comprehensive literature review. A searchable database of findings from all of these studies, as well as links to full abstracts, can be accessed at:
http://ncp-dev.stanford.edu/~dataportal/Watershed_Outcomes_Lit_Review_database/

Table of Contents:

Introduction

The following bibliography is the result of a focused literature review to compile information on watershed conservation activities (protected areas, restoration, improved agricultural management, etc.) and their impacts on watershed health, community well-being, and hydrologic ecosystem services. The literature review covered four areas where human activities strongly influence the environment and where management practices or land use decisions can result in either degradation or improvement of ecosystems and the hydrologic services they provide for water quality and conservation. These areas of focus are: a) protection or restoration of the environment as compared with land use change/degradation; b) agricultural production including traditional crop production, ranching, agroecology and silvopastoral operations; 3) prevention of bacterial contamination of drinking water supplies, and 4) the social and economic influence of ecosystem service provision on people in Latin America. The emphasis of the review with regard to the first three areas was to find numeric measures that could differentiate the influence of specific activities on water quality, water yield, biodiversity, groundwater recharge, erosion control, flood control, nitrogen retention, phosphorus retention and bacteria degradation. Journal articles, reports and scholarly papers relevant to small- or medium-scale activities were included that are applicable to implementation in a Latin American Water Fund context. Reported values for any changes in hydrologic conditions brought about by the adoption of these activities or other land use decisions are included in the database of results, in order to show the range of outcomes achieved by such efforts. There are many factors that relate to the relative success of such efforts, and it is beyond the scope of this annotated bibliography to convey the full amount of detail and information included. Full citations and links (when available) are included for each article, so that readers may refer to the original publications for more details on practices, measurement methods, and outcomes.

The literature search for quantitative impacts (focus areas 1 to 3) utilized scholarly databases and websites with the use of Boolean search terms to find relevant articles, reports, and scholarly papers. The databases included in the searches were Google Scholar, Agricola, Environmental Sciences and Pollution Management, Science Direct, Biosis, and the Stanford library. We also searched specific websites for information related to best management practices, including the U.S. Environmental Protection Agency, the Wild Farm Alliance, and the International Stormwater BMP Database. The search terms used included various combinations of the following phrases: Latin America (as well as Central America/ South America); watershed management; restoration; water yield; silvopasture; ecosystem services; erosion; flood control; groundwater; livestock exclusion; rotational grazing; riparian buffers; watershed protection; hydrological impact; runoff; flow; bacteria; coliform; vegetation cover; species diversity; stream restoration; revegetation; water quality; wetland; buffer strips; economic; social. Specific Latin American country names were also used in combination with the above search terms.

The fourth area of the focused literature review pertained to the social and economic impacts of ecosystem service provision on the people of Latin America, in areas where these practices have been implemented.

Agroforestry

Altieri, M. (2000). Multifunctional dimensions of ecologically-based agriculture in Latin America. *International Journal of Sustainable Development & World Ecology, 7***(1), 62-75.**

includes qualitative data with good references

This paper contains qualitative data about traditional agroecological systems in Latin America, emphasizing that agro-biodiverse traditional systems represent a strategy to ensure diverse diets, income, efficient use of land resources, and enhanced ecological integrity. Advocates study of traditional agroecosystems to achieve food security and environmental conservation. Highlights case of Quezungal agroforestry farms enduring through intense climate changes of El Niño, drought, and Hurricane Mitch in 1998 in Honduras. 84 communities practicing Quezungal agroforestry systems did not experience the damage sustained by slash and burn farms in the region from Hurricane Mitch and severe drought. Agroforestry systems only lost 10% of crops from drought and experienced a grain surplus of 5-6 million pounds in the wake of the hurricane.

Michel, G., Nair, V., & Nair , K. (2007). Silvopasture for reducing phosphorus loss from subtropical sandy soils. *Plant Soil, 297,* **267-276.**

Phosphorus loss from sandy soils is a cause of water quality problems in 1.4 million ha. of pastureland in Florida. Researchers hypothesized that phosphate loss could be reduced by silvopastoral practices since trees take up P and have a deeper root structure than grass alone. Paired comparisons of water soluble phosphorus on silvopasture (slash pines and bahia grass) and on treeless pastures (bahia grass) were made at different soil depths for two soil types (Ultisols and Spodosols). Build-up of P concentrations in the soil profile were higher at all depths in the soil profile for treeless pasture compared with the paired location with silvopasture. Water soluble phosphorus in soil profile at 0-5 cm (median) ranged from 36% to 60% lower under silvopasture as compared with treeless pasture. In addition the researchers computed the soil P storage capacity (SPSC) of the soil at a range of depths. SPSC is a measure of the amount of P that can be added to a soil without creating an environmental concern. The silvopasture sites had a greater SPSC than the treeless sites. Silvopasture portrays greater capacity to protect water quality in terms of P retention when compared with treeless pasture.

Nyakatawa E., Mays, D., Naka, K., & Bukenya, J. (2010). Carbon, nitrogen, phosphorus dynamis in loblolly pine-goat silvopasture system in the southeast USA. ***Agroforestry Systems, 86*****, 129-140.**

Nyakatawa E., Mays, D., Naka, K., & Bukenya, J. (2010). Carbon, nitrogen, phosphorus dynamis in loblolly pine-goat silvopasture system in the southeast USA. Agroforestry Systems, 86, 129-140.

This study investigated the opportunity to establish an animal production (goats) on a loblolly tree plantation for the purpose of increasing income from combined animal production and forestry. Over a four year period from 2006 to 2010, a series of measurements of soil carbon, nitrogen and phosphorus were taken at different soil depths for three treatments: low density (10 goats per ha), high density (20 goats per ha) and a control (no goat) site. Each treatment was replicated 3 times. Silvopasture sites were prepared by thinning pines from 1480 to 370 trees/ha in 2006, clearing the understory, and planting forage (red clover, white clover, lepedeza) for goats. Goats were pastured from 2007—2010 and soil chemistry was measured annually each June. Results indicated that the addition of animals to the plantation increased soil nitrate, soil ammonia and soil phosphorus compared with plantation sites without goats. The study concluded that grazing improved soil fertility through nutrient recycling; and silvopasture systems could provide improved economic and environmental benefit over tree production alone.

Paris, T. (2002). Crop–animal systems in Asia: Socio-economic benefits and impacts on rural livelihoods. *Agricultural Systems, 71*(1), 147-168.

** includes qualitative data**

Contains case studies of agricultural systems integrated with crop and animal enterprises. To increase crop and animal productivity, incomes, and maintain ecological balance, several options have been developed through farm studies and international research organizations. This review focuses on socioeconomic impacts of the new systems and strategies on poor farmers in South Asia. Contains relevant cost-benefit analysis information pertaining to silvopastoral and agroforestry systems with implied watershed improvements or qualitatively stated improvements.

Scarsbrook, M.R., & Halliday, J. (2010). Transition from pasture to native forest land-use along stream continua: Effects on stream ecosystems and implications for restoration. *New Zealand Journal of Marine and Freshwater Research, 33*, 293-310.

This New Zealand study examined the effects of patches of late-succession riparian forest on water quality, epilithon, stream morphology and aquatic macro-invertebrates. Nine 50-m stream reaches were used within two catchments, half within native forest and half within pasture. All three transition zones sites within forest had decreased pH and increased turbidity, dissolved reactive phosphorus, ammonium nitrogen, dissolved inorganic nitrogen and dissolved organic carbon relative to the upstream pasture site. Fine sediment was highest at two early transition zone sites in forest and lowest at the forest reference site. Levels decreased at the site further in the forest. Invertebrate density and biomass was greater in all pasture sites as compared to each paired forested site.

Staley, T., Gonzalez, J., & Neel, J. (2008). Conversion of deciduous forest to silvopasture produces soil properties indicative of rapid transition to improved pasture. *Agroforestry Systems, 74*, 267-277.

Conversion of mixed hardwood forest to silvopasture was accomplished in 2001 by thinning trees and planting forage (clover, ryegrass and orchardgrass) as well as adding starter fertilizer in the spring of 2002 (34 kg/ha as 19-19-19) and summer of 2003 (34 kg/ha as 34-0-0). Soil chemistry of the original hardwood forest, the silvopasture and a previously established pasture were compared over the 2003 growing season. The traditional pasture had been established for 40 years and received annual fertilizer addition (56 kg/ha as 10-20-20) and occasional lime treatment (~6 years apart). Sheep were grazed in the silvopasture and pasture, however not in the forest. Measurement in August 2003 of total organic phosphorus in the three ecosystems compared were 300, 380 and 400 mg/kg soil for forest, silvopasture and pasture respectively. Total inorganic phosphorus measurements were 50, 160 and 80 mg/kg soil for forest, silvopasture and pasture respectively. The high inorganic phosphorus measurement in silvopasture was likely due the recent addition of phosphorus fertilizer.

Udawatta, R., Garrett, H., & Kallenbach, R. (2010). Agroforestry and grass buffer effects on water quality in grazed pastures. *Agroforestry Systems, 79*(1), 81-87.

Agroforestry and grass buffers can reduce pollution from pastured watersheds. Six treatment areas of two agroforestry buffers, two grass buffers, and two control (no buffer) treatments were set up to test the effects of buffers on water quality in a pastured area in New Franklin, Missouri (mostly silt-loam soils). Agroforestry buffers contained grass with cottonwood trees, grass buffers contained the same species of grass, and all six areas were divided into 6 paddocks with 3 cows rotated in each paddock 3.5 days each for a total of 215 days during the years of 2005-2008. Agroforestry and grass buffer treatments had lower runoff volumes compared to the control, lower sediment loss, and lower total N loss. Most sediment and nutrients were retained 4-7.5 m of the buffer strip. The difference was most significant during the years with above average precipitation.

Udawatta, R., Krstansky, J. J., Henderson, G. S., & Garrett, H. E. (2002). Agroforestry practices, runoff, and nutrient loss: A paired watershed comparison. *Journal of Environmental Quality, 31*(4), 1214-1225.

Paired watershed study to test the effects of agroforestry with grass contour strips with corn-soybean rotation, grass contour strips only with corn-soybean rotation, and control corn-soybean watersheds on runoff and nutrient loss. From 1991-1997 calibration took place and calibration equations were developed to predict runoff, sediment, and nutrient losses, which explained 66-97% of the variability between watersheds. The treatment period three years after resulted in reduced runoff (10 and 1% for contour strips and agroforestry watersheds, 19% reduction in erosion in 1999 only by contour strip, and total P loss reduction 8 and 17% by contour strips and agroforestry over the treatment period compared to predicted values of loss. Sediment losses not reduced because grass and tree components in these two watersheds were not well enough established to be effective in controlling sediment loss. During the 3rd year, nitrate N was reduced 24 and 37% by contour and agroforestry treatments (21 and 20% during a large precipitation event in the 3rd year). Most reductions happened second and third years after treatment establishment.

Verbist, B., Putra, A.E.D., Budidarsono, S. (2005). Factors driving land use change: Effects on watershed functions in a coffee agroforestry system in Lampung, Sumatra. *Agricultural Systems, 85*(3), 254–270.

In Sumberjaya, Sumatra, an important Indonesian coffee-growing region, forest cover declined from 60% to 10% over three decades and became smallholder coffee fields, rice paddies, and vegetable operations. With deforestation, farmers have simultaneously been "re-treeing" the landscape so that monoculture coffee gardens have become mixed systems with shade trees (agroforestry). Researchers investigated the land uses that replaced forests and their impact on the watershed, perceptions of policy makers, farmers, and engineers (stakeholders) about watershed functions in the region, and measured discharge rates for the site's hydropower dam. Farmers had been evicted in the 1990s because of claims that agriculture would reduce hydropower production by the dam in the deforested region. However, results here show the hydropower dam operates better now than 30 years ago with the incorporation of agroforestry systems beginning in the 1980s. Land use change impacts on the watershed had not been properly evaluated and results here show compatibility of agroforestry with hydropower production. This analysis concludes that deforestation is only the first phase of land-use change, supports better planning for the land that has already been deforested, and shows the potential compatibility of agroforestry coffee systems in areas that rely on hydropower.

Sustainable Agriculture

Altieri, M., Nicholls, C. (2005). Agroecology and the search for a truly sustainable agriculture. United Nations Environmental Programme, Environmental Training Network for Latin America and the Caribbean.

includes qualitative data with good references

This book contains basic information about sustainable agriculture techniques (agroecology) and referenced case studies in Latin America. Case studies had qualitative data about watershed services after sustainable practices were implemented. Campesino á Campesino surveyed impacts of Hurricane Mitch on both sustainable and conventional paired farms in 360 communities across Nicaragua, Honduras, and Guatemala—sustainable plots had 20-40% more topsoil left, greater soil moisture, less erosion, and lower economic losses than the conventional partners. Plan Sierra in the Dominican Republic promoted sustainable farming methods on highly erosive concuos (family farms), which led to much lower accumulated erosion rates than on original concuo and monocrops systems in the region (mainly because continuous soil cover maintained with agroforestry). In Brazil in Santa Catarina, EPAGRI government extension and research service focused on microwatershed management with contour grass barriers, contour ploughing, green manures, and cover crop species testing. Impacts on the farm were improved yields, soil quality, moisture retention, and less labor demand. Since 1991, 60 microwatersheds with 38,000 farmers are involved and found cover crops are more important to prevent erosion than terraces or conservation barriers and cheaper to sustain. On the Brazilian Cerrados, soybean monoculture was dominant so NGOs working with the government designed a crop rotation and minimum tillage system for farms. This increased yields, slowed erosion, and decreased pest and disease and weed problems.

Andraski, T.W., Bundy, L.G., Kilian, K.C. (2003). Manure history and long-term tillage effects on soil properties and phosphorus losses in runoff. *Journal of Environmental Quality, 32,* 1782-1789.

Surface residue management in subsequent years of manure applications can influence the long-term risk of P losses as the manure-supplied organic matter decomposes. Researchers tested the manure history and long-term tillage (chisel plow and no-till) effects on phosphorus losses in runoff in corn systems in Wisconsin. The Madison site had chisel-plowed corn plots with 3 types of manure application histories: no history, applications to plots in 1995 and 1998, applications to plots in 1996 and 1999, and annual applications from 1994 through 1999). The site in Lancaster had both chisel-plowed and no-till plots of corn with either no history of manure applications or once-a-year manure applications from 1993 through 1997. P loads/concentrations of chisel-plowed corn plots in Madison were not affected by frequency of manure application or time since the last application. However, sediment load was reduced 60% where manure was applied annually and 30% where it was applied in both 1996 and 1999 compared to the no history control. This is probably because manure containing high OM content increased SOM, stabilized aggregates, and reduced erosion to minimize total P loss after the application. Runoff was 60% lower for no till plots with a manure history than in no till systems without manure applications. Runoff strongly correlated with surface cover rather than soil organic matter. The effect of the manure applications on runoff water quality depends on the type of tillage: In no till systems, manure history had no effect on P or sediment losses in the runoff. All no till systems (with or without application histories) had P losses lower than all plowed systems.

Bassi, L. (2002). Valuation of land use and management impacts on water resources in the Lajeado São José micro-watershed Chapecó, Santa Catarina State, Brazil. Prepared for e-workshop on Land-Water Linkages in Rural Watersheds: Case Study Series. Food and Agriculture Organization of the United Nations (FAO), Rome.

** see notes for full explanation of calculations**

Best management practices of zero and minimum tillage, contour-tillage, crop rotation, cover crops, green and organic manure, level terracing, and forestation were implemented by farmers in the Lajeado São José micro-watershed and monitored from 1988-1997. The watershed studied above the water treatment station (CASAN) was 6,348 square meters, and infiltration, soil erosion, water pollution, and water treatment cost changes were measured. Turbidity, fecal coliform bacteria, and sediment concentrations and the amount of aluminum sulfate and calcium hydroxide consumed to treat water were measured to quantify cost savings and pollution reduction. Across different tillage systems with Ferralsol soils (80% of the soil in the region), water infiltration was 93% higher under zero tillage compared to conventional tillage systems. Turbidity decreased 61%, sediment concentration decreased 69.5%, and there was a downward trend in bacteria concentration at two sampling points from 1996 to 1997. Aluminum sulfate and calcium hydroxide treatment chemicals decreased almost 50%, translating to a savings of $2,445/month. Results support that investment in watershed protection with better management practices implemented by the farmers here, especially zero-tillage systems, result in economic benefits beginning in the first year as reduced labor on-site, reduced erosion, and water treatment savings off-site and eventually increased crop production.

Cestti, Rita, Jitendra Srivastava, and Samira Jung. 2003. *Agriculture non-point source pollution control: good management practices--the Chesapeake Bay experience*. No. 7. World Bank Publications.

Contains table with Chesapeake Bay averages for N and P for some land uses, some agricultural BMPs, others conventional.

Chow, T.L., Rees, H.W., & Daigle, J.L. (1999). Effectiveness of terraces grassed waterway systems for soil and water conservation: A field evaluation. *Journal of Soil and Water Conservation, 54*(3), 577-583.

In Canada, on a site on the St John River Valley in New Brunswick, potato is an important crop and produced on rolling topography. Erosion caused by excessive water runoff is a major problem, and soil losses of 20-30 tons/ha/year are common. Variable grade diversions and grassed waterways are often used to reduce the erosion (ditching/terracing). This study quantified the benefits of this and found contour planting of potatoes associated with terracing reduced runoff by 150mm of rainfall equivalent (max). Soil losses decreased from 20 to 1 ton/ha/year. The drainage basin also became less prone to ditch and stream flooding because of the terracing.

Clausen, J. C., et al. (1996). Paired watershed comparison of tillage effects on runoff, sediment, and pesticide losses. *Journal of Environmental Quality, 25*(5), 1000-1007.

Paired watershed study in Vermont. Calibration period took place for 13 months in 1986 and then treatment period where two watersheds designated conventional tillage and reduced tillage. Treatment period was 30 months from 1987-1989. Reduced tillage had significant reductions in runoff and sediment loss, which in turn affected pesticide losses even though pesticide sediment bound concentrations were higher in the reduced tillage treatment watershed. Reduced tillage reduced runoff by 64% and sediment export by 99%. Dissolved pesticide concentrations didn't change while sediment-ound pesticide concentrations increased and 4 times The reduction in

sediment made up for this. Extreme rainfall events important to sediment export, and one rainfall accounted for 80% of total sediment losses during treatment period.

Dorren, L., & Rey, F. (2004). A review of the effect of terracing on erosion. *Soil Conservation and Protection for Europe,* 97-108.

This review paper covered studies about the effects of terracing on soil erosion as well as the proper implementation and potential disadvantages. One study in Malaysia was mentioned where pepper plants were grown on sloped land with and without terraces. Terraces drastically reduced soil loss by about 98%.

Ehigiator, O.A. Anyata, B.U. (2011). Effects of land clearing techniques and tillage systems on runoff and soil erosion in a tropical rain forest in Nigeria. *Journal of Environmental Management, 92*(11), 2875–2880.

In southwestern Nigeria in a secondary rainforest, effects of land clearing and soil tillage were tested on runoff and soil erosion in 14 sub-watersheds. Deforestation of primary vegetation had occurred in between 1978-1979 with secondary vegetation cleared 18 years before data was collected in 2002-2004. Each sub-watershed received a treatment of either manual, mechanical tree-pusher/root-rake or shear blade, or traditional land clearing. These plots were managed with either no-tillage, traditional farming, plowing with terraces, or plowing without terraces and had three-year, uniform crop rotation planted in different sequences. Runoff rate and soil erosion were measured from each sub-watershed with a flume. The most soil erosion occurred under mechanically cleared and conventionally tilled sub-watersheds. High soil-erosion with graded-channel terraces indicated that plowing with terraces did not minimize erosion. Less soil erosion happened on manually rather than mechanically cleared sub-watersheds and under no-tillage and traditional farming rather than conventionally plowed systems. Sediment concentration in runoff was lowest in manually cleared and untilled sub watersheds. Soil erosion was highest in mechanically cleared sub watersheds, higher in the tree-pusher/root-rake method than shear blade. Although manually clearing land best for environment, it is slower and more labor-intensive.

Holt-Giménez, E. (2002). Measuring farmers' agroecological resistance after Hurricane Mitch in Nicaragua: A case study in participatory, sustainable land management impact monitoring. *Agriculture, Ecosystems & Environment, 93*(1), 87-105.

MCAC (farmer-to-farmer movement of multiple NGOs) has promoted sustainable agriculture techniques on farms across Nicaragua since 1987. Sustainable land management practices (SLM) observed in the study were a mix of structural (contour plowing, terraces, etc.), agronomic (intercropping, composting, integrated pest management, manure applications, intensive in-row tillage), and agroforestry (woodlots, alley cropping, vegetative strips, live fences) practices. Conventional practices included structural and agronomic practices (chemical inputs, slash/burn). 880 plots of agroecological (implemented at least one SLM practice) and conventional land management were paired (under similar topographical conditions), and impacts of the hurricane were measured in terms of topsoil loss, landslide occurrence, erosion loss (rill and gully), percent vegetation cover, net profit/loss, depth to moist soil. On average, agroecological plots had more topsoil, higher field moisture, more vegetation, less erosion, and lower economic losses after the hurricane than control plots. Differences increased with storm intensity, steeper slope, and years of implementation, although thresholds of stress existed. Effects in topsoil were highly significant when examined the 19 different agroecological practices in storm conditions (rock bunds, green manure, crop rotation, incorporation of stubble). Green manures had some effect on differences in rill erosion and gullies.

Maass, J.M., Jordan , C.F. , & Sarukhan, J. (1988). Soil erosion and nutrient losses is seasonal tropical agroecosystems under various management techniques. *Journal of Applied Ecology, 25*, 595-607.

Erosion and nutrient retention management techniques were tested in a sloped agricultural region of Mexico, where population growth is forcing agricultural operations into the hills. A forested area was selected, and plots of maize (a common crop), maize treated with mulch, guinea grass, buffer grass, maize treated with 3 meter, and maize treated with 10 meter anti-erosion grass strips were made. From 1983 to 1984, rainfall, crop development, soil coverage changes, erosion losses, soil losses, and nutrient losses dissolved in runoff water were measured. Maize with mulch had the least soil loss (besides control forest plots) while maize with no mulch had the highest. Mulching reduced erosion by more than 90% and increased crop productivity when compared to maize-only plots. Grass strips would need further testing because of poor initial establishment. Supports low cost combination of practices with mulching to reduce erosion.

Silvopastoral/Farm Animal Management

Agouridis, C. T., Workman, S. R., Warner, R. C., & Jennings, G. D. (2007). Livestock grazing management impacts on stream water quality: A review. *Journal of the American Water Resources Association, 41*(3), 591-606.

Examined multiple studies of best management practices for livestock and their impacts on water quality in riparian areas of the southern humid regions of the United States (Alabama, Florida, Georgia, Kentucky, Louisiana, Mississippi, North Carolina, South Carolina, Tennessee, Virginia, West Virginia). Many studies document negative impacts of grazing on stream health while few examine success of BMPs for mitigating negative effects. 21 BMP studies examined that effectively reduced grazing impacts to streams : 6 documented water quality effects from implementing BMPs, 14 assessed geomorphic changes to stream, 6 examined habitat changes. Most literature examines use of exclusion fencing and riparian buffers as a grazing BMP. Five studies summarized and included in our spreadsheet: Sheffield et al 1997, Lyons et al 2000, Brannan et al 2000, Edwards et al 1997, Line et al 2000. (*See all below.*)

Boyer D.G. & Neel, J.P. (2010). Nitrate and fecal coliform concentration differences at the soil/bedrock interface in Appalachian silvopasture, pasture and forest. *Agroforestry Systems, 79*, 89-96.

A research farm operated by the Appalachian Farming Systems Research Center was studied for the effect of sheep grazing on deciduous forest, conventional pasture and silvopasture. The conventional pasture had been established previously and was well covered with orchard grass, bluegrass and white clover. The silvopasture was created by cutting trees in a deciduous forest to allow for 70-80% full sunlight and by planting orchard grass. The existing deciduous forest was primarily white oak, with some poplar and sugar maple. Piezometers were installed at variable depths in all three treatments to collect water samples after storms. The mean concentrations for inorganic nitrogen was similar across all three treatments and although the geometric mean for fecal coliform (18 CFU/100 mL) in the silvopasture plot was nearly three times that in the hardwood forest (5.6 CFU/100 mL) or conventional pasture (7.5 CFU/100 mL) , it was still far below the regulatory limit of 200 CFU/100 mL.

Brannan, K.M., Mostaghimi, S., McClellan, P.W. , & Inamdar. S. (2000). Animal waste BMP impacts on sediment and nutrient losses in runoff from the Owl Run Watershed. *Transactions of the American Society of Civil Engineers, 43*(5), 1155-1166.

Multiple BMPs were installed in the Owl Run watershed in Virginia (1,153 ha of dairy farms and some crop production). Pre-BMP conditions (July 1986—July 1989) were compared to post BMP conditions (monitored July 1989-June 1996). From 4 sites, surface runoff was collected. Station A collected from the whole watershed, B from 45 hectares without BMPs implemented, C collected agricultural runoff to demonstrate cropland BMP effectiveness (462 ha), and D collected from 331 ha of 5 dairy operations to demonstrate animal waste BMP effectiveness. All run off samples were generated by storm events. The objective was to measure the effect of a system of multiple BMPs on watershed quality: waste storage facilities, nutrient management based on crop needs, exclusion fencing along streams, water troughs, stream crossings, winter cover crops, field strip cropping, grassed waterways across whole watershed. Sediment loads and concentration, compared to before BMPs installed in July 1989 were 19 and 35% lower, soluble organic N concentrations were reduced 62%, nitrate N concentration 35%particulate P concentration 78%, and soluble P concentration 39%. The combination of BMPs effectively improved watershed quality.

Burt, C., Bachoon, D.S., Manoylov, K., Smith, M. (2012). The impact of cattle best management practices on surface water nutrient faecal bacteria and algal dominance in the Lake Oconee watershed. *Water and Environment Journal*, 1-9.

The Lake Oconee watershed in Georgia has high land coverage of cattle and chicken farming, and a concern developed among residents for fecal contamination of the lake and for the growth of algae due to nutrient addition from manure. Farms in this watershed practice a variety of BMPS with fencing, riparian buffers and unrestricted cattle access to the rivers or lake. There are also creeks where cattle are not grazed in the watershed. Water quality monitoring over a 7 month period at different sites with all four conditions (fencing, riparian buffers, unrestricted and no cattle) were taken for bacteria and nutrients as well as algae enumeration. The report provides bar charts of the water quality results (nutrients and bacteria) for these sites (unrestricted access—2 sites, riparian buffers—2 sites, fencing—1 site, no cattle 3 sites) seasonally for the spring and for 2 summers.

Byers, H.L., Cabrera, M.L., Matthews, M.K., Franklin, D.H., Andrae, J.G., Radcliffe, D.E., McCann M.A., Kuykendall, H.A., Hoveland, C.S., Calvert , V.H. (2005). Phosphorus, sediment, *Escherichia coli* loads on unfenced streams of the Georgia Piedmont, USA. *Journal of Environmental Quality, 34*(6), 2293-2300.

This study monitored water quality parameters in streams in 2 pastures at the University of Georgia, comparing results when water troughs were available and not available to cattle. The percent of time cattle spent in the riparian area was 7% when troughs were available as an alternative drinking source to stream water and 17% when water troughs were cordoned off by electric fencing. The base flow chemical loads of dissolved reactive phosphorus (g/day), total phosphorus (g/day), and total suspended solids (kg/day) were reduced by 80%, 50% and 97% respectively when cattle had drinking troughs. The *E. coli* load when cattle did not have troughs was 1.2E+08 compared with 1.3E+10 when troughs were not available. The addition of a water trough for cattle drinking can improve water quality and may be a good management practice when fencing the stream from cattle access is cost prohibitive.

Chára, J. & Murgueitio, E. (2005). The role of silvopastoral systems in the rehabilitation of Andean stream habitats. *Livestock Research for Rural Development, 17*(2).
http://www.lrrd.cipav.org.co/lrrd17/2/char17020.htm

Contains quantitative information from a handful of studies in Colombia looking at physical stream characteristics and water quality metrics in pasture catchments with and without silvipastoral systems. Most of the information is from a study comparing a forested area (classified as protection) to a nearby pasture area with no silvipastoral amendments. This study found that across the board water quality and stream stability metrics were better in the forested area. A small amount of info is given from a study comparing streams in a pasture with restored riparian corridors to streams in a pasture with no restoration. This study found higher macroinvertebrate abundance and better macroinvertebrate habitat in the stream with restored corridors.

Collins, R., McLeod, M., Hedley, M., Donnison, A., Close, M., Hanly, J., Horne, D., Ross, C., Davies-Colley, R., Bagshaw, C. , Matthews, L. (2007). Best management practices to mitigate faecal contamination by livestock of New Zealand waters. *New Zealand Journal of Agricultural Research, 50*, 267-278.

This article summarizes findings from the Pathogen Transmission Routes Research Program in New Zealand and advises best management practices (BMP) for reducing faecal contamination in runoff. A number of BMP recommendations are included in the review: avoid grazing on poorly drained soils, fencing stock from streams, providing alternative water source to draw cattle from streams, riparian buffer strips, limiting irrigation rate to the capacity of the soil to infiltrate and stock rotation between pastures. The details of summarized studies were not provided such as time scale and location. Good qualitative discussion but poor detail of quantitative comparisons from practices.

Doran, J.W., & Linn, D.M. (1979). Bacteriological quality of runoff water from pastureland. *Applied and Environmental Microbiology, 37*(5), 985-991.

This study compared runoff from grazed and ungrazed cow-calf pastureland on the US Meat Animal Research Center in Clay Center, Nebraska from 1976-1978. The bacteria concentration in runoff for total coliform and fecal streptococci were similar for ungrazed and grazed lands. The fecal coliform in runoff from grazed lands were 5 to 10 times that on ungrazed land, representing approximately an 80% removal rate.

Edwards, D. R., Daniel, T. C., Scott, H. D., Murdoch, J. F., Habiger, M. J., & Burks, H. M. (1996). Stream quality impacts of best management practice in a northwestern Arkansas basin. *Journal of the American Water Resources Association*, *32*(3), 499-509.

Best management practices of nutrient management, waste utilization, pasture and hayland management, dead poultry composting, and waste storage structure construction were implemented across the 3,240 ha Lincoln Lake basin in Northwest Arkansas (34% forest and 56% pasture). Stream flow samples representing base flow conditions were collected from 1991-1994 and analyzed for quality. Samples were collected from the two main tributaries of the lake: Moores Creek and Beatty Branch. Moores Creek was monitored at three sites, and Beatty Branch was monitored at two sites. One site per tributary was located as close to the lake as possible while the others were upstream on the tributaries. Regression analyses of stream flow concentration data showed decreasing trends in NH3-N, TKN, and chemical oxygen demand decreased significantly with time for all sub-basins while other parameters were stable. TP and TSS showed decreasing trends only in isolated instances.

Edwards, D.R., Daniel, T. C., Scott, H. D., Moore, Jr., P.A., Murdoch J.F., & Vendrell , P.F. (1997). Effect of BMP implementation on storm water flow quality of two northwestern Arkansas streams. *Transactions of the American Society of Civil Engineers, 40*(5), 1311-1319.

Best management practices of nutrient management, waste utilization, pasture and hayland management, dead poultry composting, and waste storage structure construction were implemented across the 3,240 ha Lincoln Lake basin in Northwest Arkansas (34% forest and 56% pasture). Water quality was monitored from 1991-1994 concurrent with the agricultural BMP implementation on two main tributaries, each sampled at a site close to the lake. Storm flow concentrations were measured, and N, NH3, TKN, and COD all exhibited significant decreasing trends over time at both sites. Nutrient management is likely responsible for these decreases.

Ibrahim, M., Schlonvoigt, A., Camargo, J. C., & Souza, M. (2001). Multi-strata silvipastoral systems for increasing productivity and conservation of natural resources in Central America. *Proceedings of the International Grassland Congress*.

http://www.internationalgrasslands.org/files/igc/publications/2001/tema18-1.pdf

Review of the effects of silviculture systems on water availability, soil fertility, carbon sequestration, and conservation of biodiversity. Includes much information on the conditions and circumstances in Central America but not very much in the way of quantitative research findings. In general they conclude that silviculture is a good tool to improve all of these services but that it must be managed correctly to maximize benefits.

Jones, A. (2000). Effects of cattle grazing on North American arid ecosystems: A quantitative review. *Western North American Naturalist, 60*(2), 155–164.

Extensive review of research exploring various effects of livestock grazing in arid regions of the western US. Found that overall the 3 broad categories of variables (soil-related, vegetation-related, and animal related variables) showed a varied response to grazing influences, with soil-related variables in particular reflecting detrimental effects of grazing.

Kauffman, J. B. & Krueger, W. C. (1984). Livestock impacts on riparian ecosystems and streamside management implications: A review. *Journal of Range Management, 37*(5), 430-438.

Review of effects of livestock grazing on various hydrological metrics including erosion, runoff, and infiltration rates. Results are especially varied in this review, with many studies cited showing no detrimental effects of grazing. Various methods of livestock/ riparian management (exclusions, alternative grazing schemes, in-stream structures, etc) examined at the end of the paper.

Knox, A.K., Tate, K.W., Dahlgren, R.A., & Atwill, E.R. (2007). Management reduces *E. coli* in irrigated pasture runoff. *California Agriculture, 61*(4), 159.

A 0.5 acre constructed wetland was placed to receive irrigation water runoff travelling from a 12 acre irrigated cattle farm in California. The number of cattle ranged between 56 and 102 head. The residence time of water in the wetland varied between 40 to 120 minutes, and a higher *E. coli* removal rate was observed at the longer residence times than at the short residence time. The timing of grazing and irrigation was also managed to observe its effect, and researchers found that when grazing and irrigation were concurrent there was a higher *E. coli* level in runoff. The wetland inlet *E. coli* level ranged between 420 and 157,800 cfu/100ml with a median of 5400 cfu/100ml and the outlet ranged from 10 to 74,600 cfu/100ml with a median of 1283 cfu/100ml.

Kundargi, D. (2005). Effects of bovine enclosure fencing on water quality and vegetative conditions, Bluewater Creek, New Mexico. Thesis submitted to University of Mexico.

This study evaluated of the effect of cattle exclusion and riparian restoration in arid lands of New Mexico. The USDA acquired privately owned land in the Bluewater Creek in 1947, which had been degraded by logging as well as sheep and cattle grazing. Restoration of the riparian zone in the 1980s and in 2003 included planting on a total of 5.9 km of stream reach. The study included comparison of water chemistry and vegetative characteristics in 3 restored cattle exclusion sections (in a recent 2003 restoration section), 4 grazed degraded sections and a reference section (chosen as the first restored section from 1980s, as no pristine site was available). The study did not find significant differences in water quality parameters (turbidity, dissolved oxygen, nitrate, conductivity, temperature and fecal coliform) between degraded, reference and restored sites on Bluewater Creek. The author hypothesized plausible explanations for no significant difference as 1) not enough time since restoration for water quality improvements to take place, and 2) transport of contaminants to restored area from upstream degraded area and inability to accurately separate treatment effects.

Lewis, D.J., Atwill, E.R., Lennox, M.S., Pereira, M., Miller, W.A., Conrad, P.A., &, Tate, K.W. (2010). Management of microbial contamination in storm runoff from California coastal dairy pastures. *Journal of Environmental Quality, 39*, 1782-1789.

Researchers studied fecal coliform bacteria (FCB) in runoff during storm events following application of dairy farm effluent to pastures at 34 study sites over a 2 year period. This was an observational study comparing the application of different manure management practices already in place on different farms: no management, grass strips or vegetated buffers, as well as the time lag between manure application and storm event. Dairy farm manure is collected into primary and secondary lagoons and used in late spring, early summer and fall for pasture irrigation. FCB in runoff was reduced by 2.7% per meter of grassed strip or vegetated buffer, with a cumulative removal of 24% for a 10 m. buffer. FCB was reduced by 90% when the manure application was at least 2 weeks prior to the rain event. Researchers also noted that slope of the hillside and maintenance of the buffer can influence the effectiveness of this practice. Timing of irrigation with dairy farm effluent and use of buffers can reduce FCB contamination to receiving waterways.

Line, D. (2003). Changes in a stream's physical and biological conditions following livestock exclusion. *American Society of Agricultural Engineers, 46*(2), 287-293.
http://www.pcwp.tamu.edu/docs/lshs/end-notes/changes%20in%20a%20stream's%20physical%20and%20biological%20conditio-1545513227/changes%20in%20a%20stream's%20physical%20and%20biological%20conditions%20following%20livestock%20exclusion.pdf

Weekly grab samples were taken from small stream draining dairy cow pasture (56.7 ha) in North Carolina before and after installation of cattle exclusion fencing and tree planting within 3 meters from stream bank. Pasture was divided into upstream and downstream regions (site D was a monitoring station upstream and site E was a monitoring station downstream). Fecal coliform and enterococci levels in samples 2.25 years before fence installation were 300% greater at the downstream monitoring station compared to the upstream. However, after fencing was installed, levels decreased to 65.9 and 57%, respectively, in the downstream cattle-excluded area. Comparing upstream and downstream sites, decreases in turbidity and suspended sediment levels before and after fencing were significantly different (percent change decreased after fencing). An alternative water supply was also installed in both pastures, but there were no significant differences at the upstream monitoring site following the installation. Therefore, an alternate water supply for the cattle without fencing did not improve water quality in the upper pasture.

Line, D. E., Harman, W. A., Jennings, G. D., Thompson, E. J., & Osmond, D. L. (2000). Nonpoint-source pollutant load reductions associated with livestock exclusion. *Journal of Environmental Quality, 29*(6), 1882-1890.

Part of the Long Creek 319 National Monitoring Program, which is evaluating a variety of BMPs, scientists tested the before and after effects of excluding dairy cows from, and planting trees in, a 335 meter long riparian corridor along a stream that runs through a pasture in North Carolina. Trees were planted within 3 meters of the stream bank. The entire watershed was divided into upper and lower pasture, and an alternate water source was also installed in each pasture. 81 weeks of pre-exclusion and 137 weeks of post-exclusion caused 33%, 78%, 76%, and 82% reductions in weekly NO2 and NO3, total Kjeldahl nitrogen, total phosphorus, and sediment loads from the pasture with the fencing (lower pasture). All were significant decreases except for the nitrate and nitrite. Decreases in discharge, NO2 and NO3, total suspended sediment, total solids, and the increase in total Kjeldahl nitrogen were not significant, so an alternate fence alone may not be enough to improve water quality.

Lyons, J., Weigel, B. M., Paine, L. K., & Undersander, D. J. (2000). Influence of intensive rotational grazing on bank erosion, fish habitat quality, and fish communities in southwestern Wisconsin trout streams. *Journal of Soil and Water Conservation*, *55*(3), 271-276.

Riparian buffer strips are can improve streams degraded by continuous livestock grazing, but they can be costly to farmers. Therefore, farmers compared continuous grazing land, intensive rotational grazing, grassy buffers, and woody buffer treatments along 23 trout streams in Wisconsin. Monitored from 1996-1997, the study showed that, after factoring out watershed effects, stations with rotational grazing or grassy buffers had the least bank erosion and fine substrate in the channel of the stream it was adjacent to. Continuous grazing stations had significantly more erosion and combined with woody buffers, they had more substrate, too.

Macklin K. (2011). Effectiveness of best management practices on cattle farms in Central Amherst County. Department of Environmental Studies, Sweet Briar College.

This paper reviewed the effect of riparian buffers and fencing to keep cattle out of streams in Amherst, Virginia. It was an observational study based on testing water quality and stream conditions at 3 BMP sites where cattle had been excluded from 3-6 years previously by fencing, 2 sites where cattle were not restrained, and 1 site where there were no cattle in the subwatershed. The observations included water quality parameters for total nitrogen, total phosphorus, turbidity, total coliform and Ecoli. An assessment of stream condition to quantify cattle impacts on the floodplain slope was also conducted using survey methods. Additionally, habitat quality was evaluated using EPA Rapid Bioassessment. The bioassessment concluded that the BMP site with the longest period of implementation scored similarly to the site with no cattle as "optimal", the other two BMP sites scored "marginal", and the 2 cattle unrestrained sites scored one each, as suboptimal and marginal. On average the BMPs improved total coliform and *E. coli* counts -22% and -87% respectively. Discussion regarding provision of drinking water to excluded cattle was not mentioned. Implementation of fencing and riparian buffers improved water quality, prevented erosion of stream banks and increased the biota compared with sites where cattle were not restrained from entering the stream.

Meals, D.W. & Braun, D.C. (2006). Demonstration of methods to reduce *E. coli* runoff from dairy manure application sites. *Journal of Environmental Quality*. *35*, 1088-1100.

Researchers reviewed *E. coli* counts in aged manure dairy manure to assess the reduction from die-off. Manure from a dairy farm in Vermont was transported and stored at corn and hay field sites for a period of either 30 days or 90 days prior to field application. *E. coli* die-off from the aging process significantly reduced the *E. coli* count (t-test $P < 0.001$) in the manure slurry, indicating tarped manure storage prior to distribution of manure on a field could greatly reduce *E. coli* in the fertilizer application and thereby diminish *E. coli* in runoff. New manure (< 3 days old) had *E. coli* counts ranging from 326,500 to 753,500 organisms/gm of dairy cow manure slurry. After 30 days of storage, *E. coli* die-off reduced counts to a range between 8600 and 14500 organisms/gm (98.4% and after 90 days to a range between 100 and 1000 organisms/gm (99.8%). The die-off rate constant (k) for

E. coli ranged from 0.028 to 0.072 day-1. This study also investigated whether incorporating the manure into the soil versus leaving it on top of the soil made a difference in the *E. coli* concentration in runoff from a simulated rainfall event. Manure application rate on corn plots was 58.9 m3/ha. Simulated rainfall was applied within 1 day of manure application and *E. coli* concentration in runoff was measured. There were several different comparisons of 1 and 3 day delays to rainfall and of application of manure of varying ages. For application of 30 day old manure with simulated rainfall occurring within 1 day, the manure unincorporated into the solid had a mean *E. coli* runoff concentration was 52,900 #/ 100 mL and the runoff from soil with incorporated manure had a mean *E. coli* concentration of 12,100 #/ 100 ml., representing a reduction of 77% achieved by incorporation into the soil. However, other time periods did not display equivalent results and no significant difference was observed overall between incorporation vs non-incorporation; however load reduction was likely to occur due to decreased runoff. Additionally, researchers observed a 50% reduction in *E. coli* concentration in runoff when rain was delayed to 3 days subsequent to manure application compared with 1 day. The aging of manure prior to field application and the incorporation of manure into the soil shows potential to reduce *E. coli* concentration in runoff.

Miller, J. J., et al. (2010). Influence of streambank fencing on the environmental quality of cattle-excluded pastures. *Journal of Environmental Quality, 39*(3), 991-1000.

In pastures along the fenced reach of the Lower Little Bow River in southern Alberta, Canada, streambank fencing was installed to create a cattle-excluded pasture (that would cause the fenced pasture to act as a buffer/filter strip) and compared to a grazed pasture along the stream. Rainfall simulation runoff was measured from 2005-2007. Water quality samples from runoff were analyzed. Under cattle exclusion via streambank fencing, surface runoff depth was reduced 21-32% all three years and loads of total N fractions were reduced 21-52% (in 2006-2007), which suggests the fenced pasture acted as a buffer for runoff variables. Turbidity, TSS, and certain N and P fractions were not improved or only were for a single year. Overall, streambank fencing improved certain watershed variables within the cattle-excluded pasture.

Nair , V.D., Nair, P.K., Kalmbacher, R.S., & Ezenwa, I.V. (2007). Reducing nutrient loss from farms through silvopastoral practices in coarse-textured soils of Florida. *U.S. Ecological Engineering, 29*, 192-199.

This study of nutrient loss from three different pasture vegetation schemes evaluated concentrations of nitrate, ammonia, and water soluble phosphorus in the soil profile at a variety of soil depths and provided insight into patterns of nutrient use in tree versus no-tree landscapes. One pasture was covered by bahia grass only (treeless pasture), a second by bahia grass and slash pines (silvopastoral), and a third by native plants and pines (native silvopastoral). At a soil depth of 75-100 m. (the deepest soil profile measured) the percent difference in water soluble phosphorus in the treeless pasture compared with the silvopastoral and native silvopastoral pastures was -78% and -26% respectively. Greater removal of phosphorus may occur with the deeper root system of trees compared with just grass. The same relationship was observed with nitrate-N concentrations in the soil profile. At a soil depth of 75-100 m. the percent difference in ammonia-N in the treeless pasture compared with the silvopastoral and native silvopastoral pastures was 10% and -9% respectively. Ammonia changes form with other nitrogen compounds through soil microbial activity, making comparisons difficult. Based on these results, silvopastoral practices can enhance nutrient retention over grass pastureland and may reduce nutrient transport to surface and groundwater.

Owens, L. B., Edwards, W. M., & Van Keuren. R. W. (1996). Sediment losses from a pastured watershed before and after stream fencing. *Journal of Soil and Water Conservation, 51*(1), 90-94.

Near Coshocton, Ohio, a 26-ha unimproved pasture watershed was grazed year-around, and no fertilizer was applied. A beef cow herd had access to the entire watershed study area (including the small stream). Sediment loss through the stream was measured at the base of the watershed. Following 7 years of this management practice, the stream and the wooded areas on the sides of the stream were fenced so that the cattle no longer had access to them. During the next 5 years, with the cattle fenced out of the stream, the annual sediment concentration decreased by more than 50% and the amount of soil lost decreased by 40%. Average annual soil losses were reduced from 2.5 to 1.4 Mg/ha while annual precipitation averages were similar during each management period.

Sheffield, R.E., Mostaghimi, S., Vaughan, D.H., Collins, E.R., & Allen, V.G. (1997). Off-stream water sources of grazing cattle as a stream bank stabilization and water quality BMP. *American Society of Agricultural Engineers, 40*(3), 595-604.

This study evaluated the effectiveness of providing water troughs as off-stream water sources for cattle on two commercial cattle operations in Virginia. Cows chose to drink from the trough 92% of the time and diminished their total activity in the stream by 51%. The change resulted in a reduction in stream bank erosion of 77%, reduced total suspended sediment of 90% and reduced total nitrogen and total phosphorus by 54% and 81% respectively. Total coliform was reduced by 82%. The relatively short time frame of this study (14 months) is not sufficient to conclusively determine that the addition of water troughs has the sustained substantial effects observed.

Sovell, L. A., et al. (2000). Impacts of rotational grazing and riparian buffers on physicochemical and biological characteristics of southeastern Minnesota, USA streams. *Environmental Management, 26*(6), 629-641.

Riparian management effects on stream quality along five southeastern Minnesota streams. The impact of rotational grazing and continuous grazing on water quality and the effectiveness of different buffer types were tested. Three streams had a longitudinal design and two had paired watershed design. Rotational vs. continuous grazing were compared along one longitudinal study stream and at the paired watershed. Treatments of riparian buffer management were fenced trees (wood buffer), fenced grass, or unfenced rotationally grazed areas and were tested along the two remaining longitudinal study streams. Only fecal coliform and turbidity had consistent differences among land-use and buffer conditions. Fecal coliform and turbidity were consistently higher at continuously grazed than rotationally grazed sites. Mean turbidity was generally lower along grass buffers sites than wooded sites. Results imply more studies of buffer types are needed and that rotational grazing prevents stream degradation.

Sullivan , T.J., Moore, J.A., Thomas , D.R., Mallery, E., Snyder , K.U., Wustenberg, J., Mackey, S.D., Moore, D.L. (2007). Efficacy of vegetated buffers in preventing transport of fecal coliform bacteria from pasturelands. *Environmental Management, 40*, 958-965.

Cow manure was spread on a hillslope above vegetated strips, and fecal coliform was measured in runoff from storm events. A total of 17 treatment cells, each 14 m by 30 m from top to bottom were planted with vegetated strips of native plants on the downhill side of a slope (4% to 7% grade). Planted buffer widths varied (0, 1, 3, 8, 15, 30 m. wide). Fresh manure from a nearby dairy was placed on the hillslope prior to each precipitation event. Runoff was collected every 24 hours following rain events below the buffered area. Much of the storm water infiltrated into the soil and was not measured, i.e. only runoff that travelled over filter strips was collected. Fecal coliform bacteria (FCB) was measured in high concentrations on the plots with 0 m. buffer with a median coliform concentration of 16,500 cfu/ 100 mL. The plots with vegetated strips had a much lower concentration of fecal coliform, with a median of 6 cfu/ 100 mL. Whereas 90% of samples from the zero buffer had FCB>200 cfu/ 100 mL. , only 26% of the samples collected from vegetated buffer cells had FCB>200 cfu/ 100 mL. Storm to storm performance of vegetated buffer strips for FCB removal varied, however a relationship between buffer strip size and removal efficiency was not observed. Other studies found buffer size mattered for FCB removal, however those studies pertained to irrigated land and this study related to storm runoff.

Sunohara, M. D., et al. (2012). Impact of riparian zone protection from cattle on nutrient, bacteria, f-coliphage, and loading of an intermittent stream. *Journal of Environmental Quality, 41*(4), 1301-1314.

includes quantitative data

In Ontario, Canada on a 285 ha watershed, pasture was divided into 1.8 ha upstream and treated with restricted cattle access to stream with 3-5m riparian buffer (RCA pasture)and 4.8 ha downstream and treat with unrestricted cattle access (URCA pasture). Upstream to downstream comparisons were made from 2005-2009 while categorizing flow as "no flow", "low flow," "high flow," and "all flow" (both low and high flow conditions included in this category). Greater mean load reductions were associated exclusively with restricting cattle access: 88% of all mean LRIs were positive (a reduction was made) for the RCA treatment while only 38% were positive for URCA. Under "all flow" conditions, mean LRIs for the RCA were significantly higher than those of URCA for ammonium-N, dissolved reactive P, total coliform, E coli, and Enterococcus. Mean LRIs were significantly higher for RCAs in high flow conditions in terms of dissolved reactive P, total coliform, and Enterococcus. For low flow conditions, LRIs were significantly higher under RCA for DRP, total P, total coliforms, fecal coliforms, E coli, Enterococcus. There was less reduction in parasites for RCA pasture than URCA, and this may be because the RCA pasture contained a wildlife habitat.

U.S. EPA (2010). Water recreation use restored in alpine waterbody. US EPA Office of Water, Section 319 Nonpoint Source Program Success Story. EPA 841-F-10-001U.

Fecal coliform bacteria levels in the Upper Truckee River and Big Meadow Creek monitoring sites violated water quality standards (log mean 20 FC units/ 100 ml over 30 day period and no samples > 40 FC units/100 ml) prompting the Lahontan Regional Water Quality Control Board to list these streams as CWA Section 303(d) water bodies. A grazing ban was placed on these watershed in 1999 following a decade of unsuccessful attempts to mitigate grazing impacts by installing BMPs, including exclusion fencing, stream revegetation and erosion control cloth. The USFS-LTBMU determined in 1999 that a viable grazing strategy to protect California state water quality standards could not be developed and cattle were banned. Cattle removal allowed recovery of the creeks and enabled achievement of water quality objectives. These sites were removed from the 303(d) list in 2010.

U.S EPA (2009). Applying agriculture best management practices reduces bacteria. US EPA Office of Water, Section 319 Nonpoint Source Program Success Story. EPA 841-F-09-001V.

Georgia's Broxton Creek violated fecal coliform water quality standards for using the creek for fishing and a six mile segment was listed as impaired under CWA Section 303(d). The Seven Rivers Resource Conservation and Development Council and local landowners worked collaboratively to improve cattle and poultry operations and install BMPs to mitigate water quality. The BMPs installed totaled $69000 and included riparian restoration, planting vegetation, grade stabilization, construction of a poultry litter house, waste management systems, and provision of alternative drinking water sources for cattle. Broxton Creek fecal coliform geometric mean value reached a high of 5386 cfu/100 ml in Feb. 1994 and was down to 30 cfu/100 ml in Feb 2003. This achievement allowed the Georgia EPD to remove Broxton Creek from the state's list of impaired waterbodies, designating it as safe for fishing.

NOTE: There are many similar case studies on the EPA website, however the implementation costs are is very high. Also, changes due to specific practice are not defined separately, making it difficult to individually examine approaches to lowering costs.

Wilcock R.J., Betteridge, K., Shearmand, D., Fowles, C.R., Scarsbrook, M.R., Thorrold, B.S., & Costall, D. (2012). Riparian protection and on-farm best management practices for restoration of a lowland stream in an intensive dairy farming catchment: a case study. *New Zealand Journal of Marine and Freshwater Research, 43*, 808-818.

The land use of the Wiokura Stream watershed in New Zealand is predominantly dairy farm and had serious water quality problems. The Taranaki Regional Council met in Nov 2003 and approached the 17 farmers in the watershed, encouraging them to plant riparian areas and fence along the stream to protect and improve water quality. Before 2001 it was estimated 40% of the stream was already protected. An additional 5 km of stream was protected by 2008 as a result of this effort, protecting 61% of the stream. This long term study showed improvements in water quality for sediment, turbidity and E coli; however there were higher nitrogen compound concentrations. The higher N was probably due to an increase in fertilizer application. Additionally, there was a 21% increase in dairy production over the study period from 2001-2008. A 25% reduction in dairyshed effluent (DSE) was achieved through the conversion from pond storage to land irrigation systems. Initially the DSE was discharged through 8 ponds, 6 land-irrigation systems and 3 mixed systems (pond + irrigation). By 2008 this shifted to 6 ponds, 10 land-irrigation systems & 1 mixed.

Williamson, R. B., Smith, C., & Cooper, A. B. (1996). Watershed riparian management and its benefits to a eutrophic lake. *Journal of Water Resources Planning and Management, 122*(1), 24-32.

Sediment and nutrient loads were measured before and after the implementation of livestock exclusion measures near eutrophic Lake Rotorua on the Ngongotaha Watershed in New Zealand. Cattle, sheep and deer were excluded from banks of most streams, erosion-prone hillslopes, and native forest pockets. Loads were reduced 85% for sediment, 27% for particulate P, 26% for soluble P, 40% for particulate N, and increase 26% for dissolved N. Applied to the whole Lake Rotorua watershed, researchers predicted TP loads were reduced 20%. These measures could help shift the lake from eutrophic to mesotrophic.

Riparian Buffers

Broadmeadow, S. & Nisbet, T.R. (2004). The effects of riparian forest management on the freshwater environment: A literature review of best management practice. *Hydrology and Earth System Sciences, 8*(3), 286-305.

Review of riparian buffer effectiveness in sediment removal, erosion control, protection of water quality, moderation of shade and water temperature, maintenance of habitat structural diversity and ecological integrity, and improvement of landscape quality. Focus on the width of the buffer and the structure of the vegetation and species choice. Study provides significantly more qualitative info than quantitative.

Carline, R. F. & Walsh, M. C. (2007). Responses to riparian restoration in the Spring Creek Watershed, central Pennsylvania. *Restoration Ecology, 15*(4), 731–742.

In order to test whether narrow buffer zones would be wide enough to improve stream functioning, riparian treatments, consisting of 3- to 4-m buffer strips, stream bank stabilization, and rock-lined stream crossings, were installed in two streams with livestock grazing to reduce sediment loading and stream bank erosion. Cedar Run and Slab Cabin Run, the treatment streams, and Spring Creek, an adjacent reference stream without riparian grazing, were monitored prior to (1991–1992) and 3–5 years after (2001–2003) riparian buffer installation to assess channel morphology, stream substrate composition, suspended sediments, and macroinvertebrate communities. No plantings were made in the buffer strip; all vegetation colonized naturally. Stream bank vegetation increased from 50% or less to 100% in nearly all formerly grazed riparian buffers. The proportion of fine sediments in stream substrates decreased in Cedar Run but not in Slab Cabin Run. After riparian treatments, suspended sediments during base flow and storm flow decreased 47–87% in both streams. Macroinvertebrate diversity did not improve after restoration in either treated stream, however, relative to Spring Creek, macroinvertebrate densities increased in both treated streams.

Hoffmann, C.C., Kjaergaard, C., Uusi-Kämppä, J., Hansen, H.C. B., & Kronvang, B. (2009). Phosphorus retention in riparian buffers: review of their efficiency. *Journal of Environmental Quality, 38*, 1942–1955.

Extensive review of studies examining phosphorus retention in various riparian buffers and wetland areas. Examined how the different flow paths ((i) The diffuse flow path with ground water flow through the riparian aquifer, (ii) the overland flow path across the riparian buffer with water coming from adjacent agricultural fields, (iii) irrigation of the riparian buffer with tile drainage water from agricultural fields where disconnected tile drains irrigate the riparian buffer, and (iv) inundation of the riparian buffer (floodplain) with river water during short or longer periods) in the riparian buffer influence P retention mechanisms. Results given both in kg P/ha/year and in % of P load retained. Variety of results. Seems to be a focus on Denmark.

Mayer, P.M., Reynolds, Jr., S.K., McCutchen, M.D., & Canfield, T.J. (2007). Meta-analysis of nitrogen removal in riparian buffers. *Journal of Environmental Quality, 36*, 1172-1180.

Analyzed data from 89 individual riparian buffers from 45 published studies containing data on riparian buffers and nitrogen concentration in streams and groundwater to identify trends between nitrogen removal effectiveness and buffer width, hydrological flow path, and vegetative cover. Riparian buffers are defined as the zone of vegetation adjacent to streams, rivers, or wetlands. Examined percent effectiveness at removing nitrogen as well as absolute mass of NO_3^- removed. Found that wide buffers (>50 m) more consistently removed significant portions of nitrogen entering a riparian zone than narrow buffers (0–25 m) for both measurements.

McKergow, L. A., et al. (2003). Before and after riparian management: sediment and nutrient exports from a small agricultural catchment, Western Australia. *Journal of Hydrology, 270*(3), 253-272.

From 1991-2000 in a 6km✚2 agricultural catchment near Albany Western Australia, streamflow, nutrient and sediment concentration data were collected downstream of fenced riparian area where, in 1996, a 1.7km stream reach was fenced, planted with eucalyptus species, and managed independently so that livestock were excluded. Suspended sediment concentrations fell significantly after the buffer was built with the median event mean concentration dropping from 147 to 9.9 mg/L. Phosphorus was no longer transferred as sediment but rather filtered reactive P, which increase 60% so that the raw median FRP concentration increased from 0.18 to 0.3 mg/L. Total N changes were unclear with total N concentrations significant at high flows but little change in loads or EMC.

Results demonstrate benefits of riparian management in reducing stream bank erosion, but suggest catchments with sandy low P sorption soils won't be effective at reducing P and N exports.

McKergow, L. A., et al. (2006). Performance of grass and eucalyptus riparian buffers in a pasture catchment, Western Australia, part 2: Water quality. *Hydrological Processes, 20*(11), 2327-2346.

This study compares 10 m wide regenerating grass and Eucalyptus globulus buffer performance.

Surface and subsurface water quality were monitored over a 3-year period. Nutrient and sediment transport were both dominated by subsurface flow (B-horizon), which may limit effectiveness of riparian buffers. The B-horizon subsurface flow pathway carries contaminant loads at least three times greater than surface runoff. Riparian buffer trapping efficiencies were variable on an event basis and annual basis. The grass buffer reduced total phosphorus, filterable reactive phosphorus, total nitrogen and suspended sediment loads from surface runoff by 50 to 60%. The *E. globulus* buffer was less effective, and total load reductions in surface runoff ranged between 10 and 40%. A key difference between the grass and *E. globulus* buffers was the seasonality of sediment and nutrient transport. Surface runoff, and therefore sediment and nutrient transport, occurred throughout the year in the *E. globulus* buffer, but only during the winter in the grass buffer. This study demonstrates that grass and *E. globulus* riparian buffers receiving runoff from pasture under natural rainfall can reduce sediment and nutrient loads from surface runoff.

Meals, D. W. (2001). Water quality response to riparian restoration in an agricultural watershed in Vermont, USA. *Water Science & Technology, 43*(5), 175-182.

Treatment and control watersheds (1,422 ha and 954 ha) in the Missiquoi River drainage of Lake Champlain in Vermont USA (the most agriculture-intensive region of the basin) were monitored under a paired-watershed design since 1994. Both watersheds were mostly dairy operations were some grass feed farms. A calibration period from 1994-1997 allowed researchers to develop baseline inventories and calibration relationships. In 1997, the treatment watershed experienced livestock exclusion from certain stream areas, riparian protection, and riparian restoration projects. Strong statistical calibration between the control and treatment watersheds was achieved, and the first year of post-treatment resulted in a 25% reduction in total P, 46% reduction in E. coli, 52% reduction in fecal coliform, 51% reduction in fecal streptococcus, and a total P export reduction of 42% comparing the treatment to the control, based off the calibration relationships. This early post-treatment data suggests significant reduction in P and bacteria counts because of the treatment.

Osborne, L. L., & Kovacic , D.A. (1993). Riparian vegetated buffer strips in water-quality restoration and stream management. *Freshwater Biology, 29*(2), 243-258.

This paper incorporates a review of known literature on riparian buffers as well as investigated the efficacy and utility of vegetated buffer strips for stream restoration.

- Forested vegetated buffer strips can reduce solar radiation and minimize temperature fluctuations in stream water (Burton and Likens 1973, Karr and Schlosser 1977, Feller 1981). In North American widths between 10-30 m can maintain stream temperature.
- Table 2, p. 246 outlines the effect of buffer strips on sediment loading, suggesting that fairly narrow sheet flows can protect from excessive sediment loads.
- Efficacy of nitrate-N and P removal from surface and subsurface by forested and grass vegetated buffer strips is reviewed on Table 3, p. 247.

The study itself was performed on a 1-km reach of a river in southeastern Illinois and was divided into an agricultural upland zone and a riparian zone composed of row crops, riparian forest and a grass buffer. The effect of riparian treatments on concentrations of nitrate-N, dissolved phosphorus and total phosphorus was examined. Both forested and grass buffers reduced nitrate-N concentrations in shallow groundwater up to 90%, but annually forested buffers were more effective than grass at reducing nitrate-N. Forested buffers were less efficient at retaining total and dissolved phosphorus than grass buffers.

Parajuli, P.B., Mankin, K.R., & Barnes, P.L. (2008). Applicability of targeting vegetative filter strips to abate fecal bacteria and sediment yield using SWAT. *Agricultural Water Management, 95*, 1189-1200.

This study evaluated fecal coliform and sediment removal by vegetated buffer strips using the SWAT model to determine the reduction in load at a watershed scale. It compared modeled removal rates when location of vegetated strips was targeted and chosen at sites deemed to be most effective compared with removal rates when sites were randomly selected for implementation of strips. By selecting a targeted location for the vegetated strips, the removal rates for the watershed as a whole could be improved. When strip implementation was at the 50% of potential sites, fecal coliform removal was estimated at 60% compared with only 42% removal when implementation sites were randomly chosen. Sediment removal was 72% when targeted and 50% when random. By implementing vegetated strips strategically on a watershed scale where they can have the greatest effect, bacteria and sediment removal can be improved compared with randomly installing this BMP.

Polyakov, V., Fares, A., & Ryder, M.H. (2005). Precision riparian buffers for the control of nonpoint source pollutant loading into surface water: A review. *Environmental Review, 13,* 129–144.

Review of the effectiveness of riparian buffers at sediment, phosphorus, and nitrogen retention. Aims to show that "precision" buffers- designed with site-specific characteristics- are best because they manage spatial variability and utilize different protection processes in different areas. Some discussion of cost-benefit analyses at the end of the paper, but no quantitative results in this section.

Salemi, L.F., Groppo, J. D. , Trevisan, R. , de Moraes, J. M., de Paula Lima, W., & Martinelli, L.A.. (2012). Riparian vegetation and water yield: A synthesis. *Journal of Hydrology*, 195-202.

This paper is a review that examines the effect of riparian vegetation on water yield. The paper is broken into two types of studies, nested catchment and paired catchments. Summary of conclusions can be found in Table 1 and Table 1A on pg 199/200; overall riparian vegetation removal results in reduced diurnal fluctuations in ground/streamwater, increasing daily water yield an average of 1.32 mm day-1. Assuming a linear relationship between area treated and water yield, Salemi et al. (2012) found an increase of 62 +/- 35 mm yr-1 for suppressed riparian forest and a decrease of 47 +/- 13 mm yr—1 for regenerated or planted riparian forest. Also, the variation between null to significant water yield increases may be related to the riparian zones size in relation to the whole catchment (Rich and Gottfried 1976).

- Nested catchment
    - » Removal of exotic invasive trees (3.98 ha, or 4% of catchment) of riparian vegetation 37-m on both sides of stream resulted in a 12-m3 day-1 increase per ha, or 13% streamflow increase (Prinsloo and Scott 1999)
    - » Dye and Poulter (1995) found a 12.2-m3 day-1 increase in streamflow after similarly removing exotic invasive trees.

» Dunford and Fletcher (1947) found that removing stream-bank vegetation resulted in a 10-day average gain in water yield of 0.8 mm day-1. The vegetation removal loss was 1.07 ha, or 12% of the watershed

- Paired catchment
  - » Dunford and Fletcher (1947): 1.07 ha (12%) of the catchment had vegetation removed; observed substantial decrease in diurnal fluctuations during the growing season; Johnson and Kovern (1954) found this increase in streamflow to be 3.8%-19%. Average daily gains from 10.3-13.6 m3 for dry days.
  - » Wicht (1941) found in South Africa that between 0.8%-42.% of annual water yield was used by streambank vegetation during dry periods.
  - » Prinsloo and Scott (1999) found a 1.05 mm day-1 increase in streamflow after clearing 16.2% of a catchment in South Africa.
  - » Rowe (1963), in an experiment in California, found a 352-mm increase in water yield during the dry season after 6.1 ha of riparian vegetation removal.
  - » Ingebo (1971) found that suppressed channel-side chaparral cover, which was 15% of the watershed area, lead to a 16 mm yr-1 and 25 mm yr-1 increase in streamflow in two consecutive years.
  - » Rich and Gotfried (1976) found that cutting 0.6% of the total basal area on the catchment did not increase water yield.
  - » Smith (1992) examined the effect of a tree plantation within a riparian zone on water yield in New Zealand. The planted area represented 20% of the catchment and the author found that decreased streamflow ranged from 93-104 mm yr-1 when the pine trees were 8-10 years old and observed for two years.
  - » Scott and Lesh (1996) found that removal of riparian vegetation in South Africa resulted in a 9% increase in streamflow in the first year.

Sweeney, B.W., Bott, T.L. , Jackson, J.K., Kaplan, L.A. , Newbold, J. D., Standley, L. J. , Hession, W. C. , & Horwitz, R.J. (2004). Riparian deforestation, stream narrowing, and loss of stream ecosystem services. *Proceedings of the National Academy of Sciences, 101*, 14132-14137.

North American study of 16 paired forested and deforested streams over one year; purpose was to demonstrate the effects of riparian deforestation. Nutrient uptake (NH_3 and PO_4), pesticides (atrazine, linuron, dursban and methoxychlor), dissolved organic matter, stream metabolism, and fish species and abundance were studied. Forested streams were wider with lower average water velocity. Ammonium uptake in forested reaches was higher than deforested reaches but there was no such effect on phosphorus uptake.

Thompson, R., and Parkinson, S. (2011). Assessing the local effects of riparian restoration on urban streams. *New Zealand Journal of Marine and Freshwater Research, 45,* 625-636.

The effect of riparian restoration on urban stream invertebrate communities was examined in Melbourne, Australia. Three urban streams were sampled at two reaches each, one area of riparian revegetation and one area of low riparian cover. All replantings were at least 15 years old. Results demonstrated that stream width and bank width were less variable in forested reaches in all three streams. Additionally, terrestrial litter input was higher in forested reaches. No difference in biomass of chlorophyll *a* was found between the open and forest reaches. However, invertebrate biomass was higher in the open reaches. Overall, forested reaches had a higher diversity of invertebrate taxa associated with allochthonous resources, while open reaches had more taxa utilizing autochthonous production.

Weigelhofer, G., Fuchsberger, J., Teufl, B. , Welti, N., & Hein, T. (2011). Effects of riparian forest buffers on in-stream nutrient retention in agricultural catchments. *Journal of Environmental Quality, 41*, 373-379.

This study examined to what extent forested riparian buffers can affect in-stream nutrient retention. Four Austrian streams with riparian forest buffers at least 300 m were selected. Each reach was compared to a degraded section of the same stream. Nutrient addition was used to compare short-term nutrient retention. Transient storage exchange coefficients did not vary between forested buffers and degraded reaches. However, the forested buffers had a higher ratio of transient storage area to cross-sectional area of the stream. Additionally, ammonium uptake lengths were shorter in the forested buffers compared to the degraded reaches. Phosphate uptake lengths did not vary between paired sites. Overall riparian forest buffers had a significant impact on hydrologic retention; this general increase in transient storage may support nutrient uptake.

Wenger, S. (1999). *A review of the scientific literature on riparian buffer width, extent, and vegetation*. Report for the Office of Public Service & Outreach, Institute of Ecology, University of Georgia.

Extensive global review of over 140 articles and books on riparian buffer effectiveness with regards to sediment retention, erosion control, phosphorus retention, nitrogen retention, and the retention of other pollutants. Somewhat focused on buffer width, but extent and vegetative cover also examined. Brief exploration of other factors affecting aquatic environments at the end of the paper.

Reforestation/Afforestation

Amazonas, N.T., Martinelli, L. A., de Cássia Piccolo, M., & Rodrigues, R. R. (2011). Nitrogen dynamics during ecosystem development in tropical forest restoration. *Forest Ecology and Management , 262,* 1551–1557.
http://www.lerf.esalq.usp.br/divulgacao/produzidos/artigos/2011femv262n8p1551-1557.pdf

In Sao Paulo state. Large study comparing a protected natural forest to a 21 year old and 52 year old secondary mixed forest. Both secondary forests were the result of assisted revegetation after clear cutting for agriculture. They measured a host of Nitrogen related parameters in the soil and foliage, as well as the surface soil moisture in the three forest types. 15N values, N content, N:P ratio, inorganic N and net mineralization and nitrification rates were all higher, the older the forest. Findings indicate that the recuperation of N cycling has not been achieved yet in the restored forests even after 52 years, but show that they are following a trajectory of development that is characterized by their N cycling intensity becoming similar to a natural mature forest of the same original forest formation. The use of high species diversity with predominance of native trees to restore the studied forests potentially promotes the recuperation of N cycling as restored communities develop.

Buytaert, W., Iñiguez, V., & De Bièvre, B. (2007). The effects of afforestation and cultivation on water yield in the Andean páramo. *Forest Ecology and Management, 251,* 22–30.

The impact of afforestation with *Pinus patula* (representative of pine plantations commonly carried out in this area to improve the economic viability of the páramo grasslands) on the water yield is studied and compared to the more common practice of intensive grazing and potato cultivation in four microcatchments in the Paute river basin in south Ecuador. Two catchments are covered with natural grassland vegetation, one is converted to pine forest, and one is drained, partly intensively grazed, and partly cultivated with potatoes. The results indicate that afforestation with *P. patula* reduces the water yield by about 50%, or an average of 242 mm/ year. The water yield of the cultivated catchment is very similar to that of the natural catchments, but analysis of the flow duration curves suggests a faster response and a loss of base flow.

Farley, K. A., Jobbagy, E. G., & Jackson, R. B. (2005). Effects of afforestation on water yield: a global synthesis with implications for policy. *Global Change Biology, 11*, 1565–1576.

Examined runoff data from afforested regions with a previous land cover of grassland or shrubland (conversion of natural grassland to plantation). Most of the data were from paired catchment studies, in which streamflow from grassland or shrubland catchments was compared with that of nearby afforested catchments. Examined changes in the context of several variables, including original vegetation type, plantation species, plantation age, and mean annual precipitation (MAP). Found that afforestation of grasslands and shrublands will typically result in a loss of one-third to three-quarters of streamflow on average.

Harden, C. P. & Mathews, L. (2000). Rainfall response of degraded soil following reforestation in the Copper Basin, Tennessee, USA. *Environmental Management, 26*(2), 163–174.

To determine the effectiveness of more than 50 years of reforestation efforts, rainfall infiltration, sediment detachment, and soil organic matter were compared for reforested sites, unvegetated sites, and forested reference sites outside the basin. Simulated rain experiments were conducted at 54 sites throughout the Copper Basin, including 11 in Zone 1: Not revegetated- predominantely unvegetated at time of study; 11 in Zone 2: Denuded, then replanted in 1985-1995 (year of study); 11 in Zone 3: Denuded, then replanted in 1971- 1986; 10 in Zone 4: Denuded, then replanted in the 1930s and 40s; and 11 in Zone 5: Forested (since at least 1941) lands outside the basin which were used as reference sites. Found that new "forests" have significantly more erosion, less organic matter, and lower infiltration than forests more than 50 years old.

Ilstedt, U., Malmer, A., Verbeeten, E., & Murdiyarso, D. (2007). The effect of afforestation on water infiltration in the tropics: A systematic review and meta-analysis. *Forest Ecology and Management, 251,* 45–51.

includes quantitative data

Review of four studies, including 14 comparative experiments, dealing with different afforestation treatments (two studies; eight experiments) or agroforestry (two studies; six experiments) on former agricultural land. Forest was defined broadly as any area of trees with more than 10% crown coverage (FAO, 1998). 'Afforestation' denoted plantation on open land that had been free from forest cover as a result of prolonged agricultural use, failed reforestation by active replanting or delayed natural secondary succession. This included pastures, grasslands (non-fallow and fallow) and permanent cultivation. All types of methods for measuring saturated steady state infiltratibility or conductivity in the surface soil (0–10 cm) were included and the type of method was recorded. The overall result of the meta-analysis was that infiltration capacity increased on average approximately three-fold after afforestation or planting trees in agricultural fields (95% confidence interval: 2.4–4.7).

Leopold, A. C., Andrus, R., Finkeldey. A., & Knowles, D. (2001). Attempting restoration of wet tropical forests in Costa Rica. *Forest Ecology and Management, 142,* 243-249.

In 1992 the Tropical Forestry Initiative, a not-for-profit group of seven persons deeply concerned about the depletion of tropical forest resources, purchased 5 ha of land in southwestern Costa Rica in order to attempt a small-scale restoration aimed at replicating native forest species distribution/

diversity/ complexity (as opposed to the monoculture restoration efforts that are common in this country). The land had been cut over in the 1950s and has been used as pasture for about 45 years. Approximately equal amounts of the land were in grass pasture, and in partial scrub growth. Seeds of seven native hardwood species were collected, germinated in a seed-bed, and transferred to soil in plastic bags (10 cm_25 cm). After 3 months in the nursery, the seedlings were about knee-high, and approximately 5000 were planted out in the pasturelands as mixed stands. The saplings were planted into open pasture at spacings of approximately 3 m_3 m; enrichment plantings into partially developed scrub areas were at spacings of approximately 4 m_4 m or more. In the subsequent 5 years the mixed plantations expanded by 3000 or 5000 trees each year, and the range of species planted expanded to include a total of 41 species. Measurements were taken each year of the number of surviving trees, their average height, growth per year, and dbh. In 1997 plots were established to measure changes in species complexity which showed an average of 46% of the ground cover taken over by volunteer scrub and tree species.

Leopold, A. C., & Salazar, J. (2008). Understory species richness during restoration of wet tropical forest in Costa Rica. *Ecological Restoration, 26*, 1.

A follow up to the 2001 restoration effort. Salazar measured and compared the number and diversity of understory plant species found in 1) plantings of mixed tree species (Leopold et al 2001); 2) nearby monoculture tree plantations; 3) secondary unplanted and unmanaged stands; and 4) nearby remnants of primary forest. Only data for mixed tree plantings and primary forest plots are given. Understory species counts range from 88 to 113. Species distribution measurements show a predominance of woody species (54-63%); followed by herbs and vines (13-20%). Families most abundantly represented were Fabaceae (36 species), Melastomataceae (27 species), Rubiaceae and Euphorbiaceae (17 species each). Both species diversity and distribution measurements were similar to primary forest plots (no statistical assessment carried out).

Mapa, R. B. (1995). Effect of reforestation using *Tectona grandis* on infiltration and soil water retention. *Forest Ecology and Management, 77,* 119-125.

***includes quantitative data ***

Steady state water infiltration rates and water retention at various suction pressures, as well as a number of other soil/water capacity parameters (i.e. permanent wilting point) were measured for 3 patches of land within a formerly forested area that was clear cut 50 years ago: one area that has been actively reforested with teak and is now protected; one area that's being cultivated for corn; and one area that has been abandoned and has been taken over by grasses (unsure if native). All three patches have the same type of soil, which is a kind of red clay. The reforested area performed better than either the corn or the grassland across all parameters. The cultivated and grassland soils did not show any significant difference in infiltration or soil water retention.

Rodrigues, R. R., Gandolfi, S., Nave, A. G., Aronson, J., Barreto, T. E., Vidal, C. Y., & Brancalion, P. H. S. (2011). Large-scale ecological restoration of high-diversity tropical forests in SE Brazil. *Forest Ecology and Management, 261,* 1605–1613.

In Sao Paulo state- a study examining the effectiveness of different forest restoration techniques based on different amounts/ types of degradation. A total of 32 ongoing projects, covering 527,982 ha, were evaluated in large sugarcane farms and small mixed farms, and six different restoration techniques have been developed to help upscale the effort. Plantations of native tree species covering the entire area was by far the main restoration method needed both by large sugarcane farms (76.0%) and small mixed farms (92.4%), in view of the low resilience of target sites, reduced forest cover, and high fragmentation, all of which limit the potential for autogenic restoration. Although no hydrologic effects were measured in this study, this is an ongoing project and the authors are aware of the importance of hydrologic factors. Updates to this study/current work/future publications may be found at http://www.lerf.esalq.usp.br/.

Zhou, G.Y., Morris, J.D., Yan, J.H., Yu, Z.Y., & Peng, S.L. (2002). Hydrological impacts of reafforestation with eucalyptus and indigenous species: A case study in southern China. *Forest Ecology and Management, 167*(1–3), 209-222.

** includes quantitative data **

10-year study looking at a eucalyptus plantation catchment, a reforested (started with an invasive "pioneer species") mixed forest catchment, and a bare land catchment in subtropical southern China. Climate, throughfall, streamflow, runoff, soil erosion, moil moisture, water table depth, tree growth, and litter biomass/ understory cover were measured in all 3 catchments (unreplicated). Eucalyptus catchment had a section that contained understory litter because it was protected from fuel foraging, but was mostly bare. Soil erosion from bare land was greater than from the forested catchments and its surface runoff carried a higher proportion of coarse sediments. Soil moisture content was highest in the bare land, but did not show a long term trend in the vegetated catchments. Water table depth averaged 30 cm deeper beneath mixed forest and 80 cm deeper beneath eucalyptus forest when compared with bare land. Runoff was highest in bare land, followed by eucalyptus, with mixed forest showing the lowest runoff.

Forest Recovery (Passive Restoration)

Gonzalez-Iturbe, J.A., Olmsted, I., & Tun-Dzul, F. (2002). Tropical dry forest recovery after long term Henequen (sisal, *Agave fourcroydes* Lem.) plantation. *Forest Ecology and Management, 167,* 67-82.

Study of native unassisted recovery after abandonment of Henequen plantations on the Yucatan Peninsula, Mexico. Henequen plantations that were abandoned approx. 10 years ago, and subsequently burned to be grazed by cattle, plantations that were abandoned approx. 15 years ago, and where no burning or clear cutting was performed, and areas that had been native forest for at least 50 years were compared on a variety of measures of plant species biodiversity. Found that species diversity, richness, and structural parameters all differed in early and intermediate successional states after abandoned Henequen cultivation.

Hassler, S.K. et al. (2011). Recovery of saturated hydraulic conductivity under secondary succession on former pasture in the humid tropics. *Forest Ecology and Management, 261,* 1634–1642.

Saturated hydraulic conductivity (Ks) was compared in active pasture land, secondary forest of 5–8 years of age (SF5), secondary forest of 12–15 years of age (SF12), and secondary forest of more than 100 years of age (SF100), at two core sample depths (0-6 and 6-12 cm). Recovery of Ks could be detected in the 0–6cm depth after 12 years of secondary succession: P and SF5 held similar Ks values, but differed significantly ($a = 0.05$) from SF12 and SF100 which in turn were indistinguishable. Ks in the 6–12cm depth did not show any differences between the land cover classes; only Ks of the uppermost soil layer was affected by land-use changes.

Stream and Wetland Restoration

Bukaveckas, P.A. (2007). Effects of channel restoration on water velocity, transient storage, and nutrient uptake in a channelized stream. Environmental Science & Technology, *41,* 1570-1576.

Water velocity, transient storage and nutrient uptake were measured in a channelized stream before and after restoration in a 1-km stream segment in Kentucky. Injection experiments were performed over a range of discharge and temperature conditions. Post-restoration data were collected 1-2 years after restoration. Transient storage was mostly unaffected except where backwater areas were created. Water velocity did not change considerably, however the restored channel had lower water velocities. Increased nutrient uptake may have been due to reduced water velocity, even though water velocity was not significantly different due to high among-site variation.

Kaushal, S.S., Groffman, P.M. , Mayer, P.M., Stritz, E. & Gold. A. (2008). Effects of stream restoration on denitrification in an urbanizing watershed. *Ecological Applications, 18*(3), 789-804. http://palmerlab.umd.edu/restoration_course_docs/2008resources/Kaushal_et_al_2008.pdf

To understand the effect of urbanization on stream eutrophication, a restored stream segment in Maryland was compared to an unrestored segment and rates of denitrification were measured. The stream was restored in 1998-1999 and surface water measurements were taken 2003-2005. Mean rates of denitrifcation were greater in the restored reach at 77.4 +/- 12.6 μg N•kg -1 d-1, versus 34.8 +/- 8.0 μg N•kg -1 d-1 in the unrestored segment. Additionally, nitrate-N in groundwater and stream water in the restored reach were significantly lower than the unrestored reach. Overall the efficacy of stream restoration and reconnecting stream channels with floodplains can vary considerably, but generally these activities may increase denitrification rates.

Lepori, F., Palm, D. & Malmqvist, B. (2005). Effects of stream restoration on ecosystem functioning: detritus retentiveness and decomposition. *Journal of Applied Ecology, 43,* 228-238. http://onlinelibrary.wiley.com/doi/10.1111/j.1365-2664.2004.00965.x/full

Compared seven pairs of restored and channelized streams in Sweden. Streams were restored by removing stone walls, replacing boulders, widening channel, decreasing velocity. Coarse organic particulate matter retentiveness and breakdown was estimated, along with taxonomic richness, abundance, biomass and evenness of leaf-eating invertebrates on leaf material. Velocity was the best predictor of biomass loss. Coarse organic particulate matter breakdown rate: in 6/7 pairs of restored and channelized streams, mass loss was lower at restored sites but the difference was not significant ($p=0.197$). Coarse particulate organic matter retained restored sites were on average more than twice as retentive as channelized ones (68% vs 30%), $p<0.001$.

Melesse, A.M., Nangia, V., Wang, X. & McClain, M. (2007). Wetland restoration response analysis using MODIS and groundwater data. *Sensors, 7,* 1916-1933.

Assessment of the effectiveness of a long-term restoration effort in the Kissimmee River Basin. Examined fractional vegetation cover (FVC), latent heat flux (not included in database), and groundwater levels throughout the basin. Measurements taken from 2000 through 2004. The fractional vegetation cover was increased in 2002 and 2004 when compared with 2000 for areas along the Kissimmee River indicating response to the floodplain restoration. Analysis of groundwater level data from eight monitoring wells showed that, the average monthly level of groundwater was increased by 20 cm and 34 cm between 2000 and 2004, and 2000 and 2003, respectively. Taking into account the amount of rainfall, this observation is valid and reasonable.

Pederson, T.C.M., Baattrup-Pedersen, A. & Madsen, T. V. (2006). Effects of stream restoration and management on plant communities in lowland streams. *Freshwater Biology, 51,* 161-179.

Thirty stream reaches in Denmark were selected, each from a different stream; 10 were in natural state, 10 were channelized and 10 were formerly channelized that were restored 3-12 yrs previously. No significant differences were found in size-related parameters width, depth and discharge. Fuzzy classification was used on all physiochemical parameters. Did not find statistical difference in total N, NO3-N, total-P, Ortho-P, total-C or pH. Also examined plant species diversity composition. Did not find that recovery period affected community outcome; did find a large variation in species composition in the group of restored/reference stream reaches. Bonferroni correction made formal significance finding difficult.

Purcell, A. H., Friedrich, C. , & Resh, V.H. (2002). An assessment of a small urban stream restoration project in northern California. *Restoration Ecology, 10,* 685-694.

A 70-m urban stream reach was restored in northern California and compared to both an unrestored section and a section restored 12 years previously. The objective of restoration was to restore sinuosity and riparian vegetation by removing an underground culvert in that stream section. Step pools were created and willows were planted along the banks. The U.S. Environmental Protection Agency's Rapid Assessment Protocols were used for habitat assessments. Benthic macroinvertebrates were surveyed; taxa richness was higher in the restored stream, EPT (proportion of pollution-sensitive individuals present) was also higher in the restored stream. In addition, bank stability was higher in the restored stream, though sediment deposition did not vary between the restored and unrestored stream (detailed results in Table 3 and Table 4). Three years post-restoration recolonization from the upstream site has occurred at the restored site and macroinvertebrate richness is now similar to the creek restored 12 years previously.

Richardson, C. J., Flanagan, N. E., Ho, M., & Pahl, J.W. (2011). Integrated stream and wetland restoration: A watershed approach to improved water quality on the landscape. *Ecological Engineering, 37,* 25–39.

****includes quantitative data* ***

An integrated stream/wetland restoration approach was used to hydrologically reconnect the stream with the adjacent floodplain and allow natural riparian wetland biogeochemistry transformations with the goal of improving stream water quality through increased stream- wetland connection and improved nutrient removal. Four years after restoration low and high bench sites in the riparian floodplain had functioning wetland hydrology, and water quality was improved across a variety of parameters including Fecal Coliform, N, and P concentrations. Downstream water pulses and stream erosion were also reduced.

Roni, P. T., Beechie, J., Bilby, R. E. , Leonetti, F.E. , Polloc, M. M. & Press, G. R. (2002). A review of stream restoration techniques and a hierarchical strategy for prioritizing restoration in Pacific northwest watersheds. *North American Journal of Fisheries Management ,* 22, 1-20.

****includes quantitative data****

This was a review of different site restoration techniques and their effect on improving fish habitat in the Pacific Northwest. The most relevant aspect of this paper was Table 6, listing typical response times for specific restoration actions, longevity or action and their probability of success with respect to improving fish habitat.

Restoration Meta-Analyses

Benayas, J. M. R., Newton, A. C., Diaz, A., & Bullock, J. M. (2009). Enhancement of biodiversity and ecosystem services by ecological restoration: A meta-analysis. *Science, 325,* 28.

A meta-analysis of 89 published scientific assessments of the outcomes of restoration actions undertaken in a variety of ecosystems from all continents except Antarctica. In these studies, ecosystems had been degraded by a wide variety of processes. Restoration actions generally included: Cessation of degrading action only (passive restoration); Extirpation of damaging species (including non-natives); Nutrient removal; Planting of forbs or grasses; Planting of trees; Reinstatement of burning; Reintroduction of herbivores or carnivores; Remodeling of topography; and Soil amendments (to bind or dilute contaminants or restore fertility). Analysis was restricted to those studies that compared restored (Rest), reference (Ref), and degraded (Deg) ecosystems within the same assessment. Results indicate that measures of supporting and regulating ecosystem services and biodiversity across the whole data set were higher in restored than in degraded systems (response ratio > 0) but lower than in reference systems (ratio < 0). Provisioning services showed no effect of restoration, but the sample size for this type of service was low. Our data indicate that supporting services, which provide the basis for provision of other services, were restored more effectively than other service types. Median values of response ratios showed that biodiversity and ecosystem services (all three types combined) in degraded systems were only 51 and 59%, respectively, of those in reference systems. Median response ratios of restored systems were substantially higher than those of degraded systems, with values of 144% for biodiversity and 125% for ecosystem services. However, the restored systems were not fully rehabilitated, as median response ratios for biodiversity and combined ecosystem services were 86 and 80%, respectively, of those in reference systems.

Postel, S. L. & Thompson, Jr., B. H. (2005). Watershed protection: Capturing the benefits of nature's water supply services. *Natural Resources Forum, 29,* 98–108.

Review of watershed protection and restoration projects around the world, including many in Latin America. Examines an array of hydrological benefits that have been seen in the wake of watershed restorations/protections, and relates these benefits to the cost of the projects and of alternative water purification/provision techniques. Focus is largely on financial aspects (as opposed to hydrological data), with detailed discussions of programs in Quito (Ecuador), Costa Rica and New York City.

Protection

Goodale, C.L., & Aber, J.D. (2001). The long-term effects of land-use history on nitrogen cycling in northern hardwood forests. *Ecological Applications, 11*(1), 253-267.

Examined the effect of long-term effects of logging and fire on stream nitrate (plus other forest characteristics) in New Hampshire. Old-growth sites had high nitrification rates and high stream NO3-N concentrations and fluxes as compared to disturbed sites. Present results as a regression of nitrate loss and annual nitrification for all three land use types (burned, logged and old-growth).

Hamilton & King (1983). *Tropical forested watersheds: hydrologic and soils response to major uses or conversions*. Boulder, Colorado: Westview Press.

Apendix B summarized key findings from other studies that examined the effect of deforestation and logging on water yield. Douglass and Swank (1976) found that when 66% of basal area was cut, stormflow during first four years of harvest and regrowth increased 17%, vs. 11% for clearcutting without harvest on a watershed (Hewlett and Helvey 1970). Cutting without timber removal increases peak discharge 22-38% on low-response watersheds (Helvey and Douglass 1971) vs. 7% for high-response watersheds (Hewlett and Helvey 1970). When 66% of basal area was cut over a road system, peak discharge increased 33% during harvest; this increase in peaks declined logarithmically with vegetation regrowth. In general, stormflow volume, peak flow and storm duration are all increased by cutting timber.

Knox , A.K., Dahlgren, R.A., Tate, K.W., Atwill, E.R. (2008). Efficacy of natural wetlands to retain nutrient, sediment, and microbial pollutants. ***Journal of Environmental Quality, 37,*** **1837-1846.**

Research in an agricultural area with two natural wetlands, one that had degraded and incised and a second remaining in a natural state, assessed the performance of each wetland in terms of contaminant removal/retention. Both wetlands were similar in size (0.2 ha.) and both had a low hydraulic retention time (<2 hrs). The channelized, degraded wetland had a median retention of 3% of total nitrogen (TN) and 2% of total phosphorus (TP) compared with 45% retention of TN and 25% of TP by the protected natural wetland. The degraded wetland contributed nitrate, whereas the protected wetland removed 55% of the nitrate. *E. coli* removal in the degraded wetland was 21% compared with removal of 63% in the protected wetland.

Postel, S. L. & Thompson, Jr., B. H. (2005). Watershed protection: Capturing the benefits of nature's water supply services. ***Natural Resources Forum, 29,*** **98–108.**

Review of watershed protection and restoration projects around the world, including many in Latin America. Examines an array of hydrological benefits that have been seen in the wake of watershed restorations/protections, and relates these benefits to the cost of the projects and of alternative water purification/provision techniques. Focus is largely on financial aspects (as opposed to hydrological data), with detailed discussions of programs in Quito (Ecuador), Costa Rica and New York City.

Large-scale Watershed and Land Use Studies

Bernhardt, E.S., Likens, G.E., Buso, D.C., & Driscoll, C.T. (2003). In-stream uptake dampens effects of major forest disturbance on watershed nitrogen export. ***Proceedings of the National Academy of Sciences, 100*****(18), 10304-10308.**

This paper looks at the effect of extensive crown damage from a severe weather event on the change in nutrient export and changes in in-stream nitrate processing. The project was conducted on two watersheds in New York. It found that storm caused increases in watershed export of NO3 for two years post-disturbance, which may be due to greater light from canopy damage, hence increased algal bloom.

Brannan, K.M., Mostaghimi, S., McClellan, P.W. , & Inamdar, S. (2000). Animal waste BMP impacts on sediment and nutrient losses in runoff from the Owl Run Watershed. ***Transactions of the American Society of Civil Engineers, 43*****(5), 1155-1166.**

Multiple BMPs were installed in the Owl Run watershed in Virginia (1,153 ha of dairy farms and some crop production). Pre-BMP conditions (July 1986—July 1989) were compared to post BMP conditions (monitored July 1989-June 1996). From 4 sites, surface runoff was collected. Station A collected from the whole watershed, B from 45 hectares without BMPs implemented, C collected agricultural runoff to demonstrate cropland BMP effectiveness (462 ha), and D collected from 331 ha of 5 dairy operations to demonstrate animal waste BMP effectiveness. All run off samples were generated by storm events. The objective was to measure the effect of a system of multiple BMPs on watershed quality: waste storage facilities, nutrient management based on crop needs, exclusion fencing along streams, water troughs, stream crossings, winter cover crops, field strip cropping, grassed waterways across the whole watershed. Sediment loads and concentration, compared to before BMPs installed in July 1989 were 19 and 35% lower, soluble organic N concentrations were reduced 62%, nitrate N concentration 35%particulate P concentration 78%, and soluble P concentration 39%. The combination of BMPs effectively improved watershed quality.

Faulkner, S., Barrow, W., Keeland, B., Walls, S., & Telesco, D. (2011). Effects of conservation practices on wetland ecosystem services in the Mississippi Alluvial Valley. *Ecological Applications, 21*(3), S31–S48.

Review of the effects of conservation practices on wetlands in the Mississippi Alluvial Valley. There were 68 different conservation practices applied to a combined total of 1.27 million ha in the MAV between 2000 and 2006. These practices fell into two categories: Wetland Conservation Practices and Upland Conservation Practices (i.e. Ag reform). Quantitative results are only given for a select few studies, but a good deal of summary information is available.

Loess Plateau Watershed Rehabilitation Project: Document of the World Bank. Report 25701. April 29, 2003. Rural Development and Natural Resources Sector Unit East Asia and Pacific Region.

This is a technical document that describes the outcomes from the Loess Plateau Watershed Rehabilitation project in China, the goal of which was to increase agricultural production around the watershed and reduce sediment inflow. Results with respect to sedimentation post-management were anecdotal, claiming reduced sediment inflow resulted in irrigated systems that suffered less from large inflows of sediment and that river channels are more stable and maintenance costs lower. Also sediment build-up downstream is lowered. Peak run-off was slowed by retaining rainfall water on terraces, behind dams and in water harvesting pits around planted trees.

Martínez, M. L.**, Pérez-Maqueo, O., Vásquez, G, Castillo-Campos, G., García-Franco, J., Mehltreter, K., Equihua, M., & Landgrave, R. (2009). Effects of land use change on biodiversity and ecosystem services in tropical montane cloud forests of Mexico. *Forest Ecology and Management, 258*, 1856–1863.**

Examined the effects of land use change in the highlands of the central region of the state of Veracruz in Mexico. Analyzed changes in vegetation cover as well as a host of water quality metrics, including total suspended solids, nitrate, and others. Found that water quality was significantly better in uncut cloud forests (compared with coffee plantations and grasslands), and that vegetation was significantly more diverse in cloud forests than in grasslands, but the difference was not significant for coffee plantations. Also did a broad survey of literature calculating ecosystem service values for different land use types in this area and then calculated the changes in these values between 1973 and 2004 in their study area.

Mehaffey, M.H., Nash, M.S. , Wade, T.G. , Ebert, D.W. , Jones, K.B., & Rager , A. (2005). Linking land cover and water quality in New York City's water supply watersheds. *Environmental Monitoring and Assessment, 107*, 29-44.
http://www.springerlink.com/content/x05n82211815x60n/

Thirty-two New York drainage areas were used in a step-wise regression analyses to test landscape and surface-water quality relationships. To look at the land cover and landscape change, four sets of Landsat imagery over 30 years were used. The landscape measurements (stream density, agricultural area, urban area, agriculture or erodible soil and barren earth) were related to surface water total nitrogen concentrations, having a model R2 value of 79.4%. Stream density, percent agriculture and urban development explained more than half of the variability in phosphorous concentration. Percent of agriculture and percent urban development were consistently related to water quality in all models, together explaining between 25-75% of variation. Overall water quality remained high over the 30 years but agricultural land use was the major contributor to nitrogen and phosphorus in streams.

Postel, S. L. & Thompson, Jr., B. H. (2005). Watershed protection: Capturing the benefits of nature's water supply services. *Natural Resources Forum, 29*, 98–108.

Review of watershed restoration projects around the world, including many in Latin America. Examines an array of hydrological benefits that have been seen in the wake of watershed restorations/protections, and relates these benefits to the cost of the projects and of alternative water purification/provision techniques. Focus is largely on financial aspects (as opposed to hydrological data), with detailed discussions of programs in Quito (Ecuador), Costa Rica and New York City.

Shárma, U. C., et al. (2001). Effects of farming system type on *in situ* groundwater recharge and quality in northeast India. *Impact of human activity on groundwater dynamics: Proceedings of a symposium held during the Sixth IAHS Scientific Assembly, Maastricht, Netherlands*. IAHS Press.

Four micro-watershed hillslopes in northeast India were monitored from 1984 to 1991. Four new land use systems were implemented: livestock-based, forestry, agri-pastoral, and agri-horti-silvo-pastoral. Two traditional methods, bun and shifting cultivation, were also present. Researchers measured surface, baseflow, in situ groundwater recharge, and nutrient loads from each system. Groundwater recharge was highest in new systems. Soil loss and total nutrient loads were also less than traditional systems.

US EPA (2009b). Watershed scale efforts reduce bacteria levels. US EPA Office of Water, Section 319 Nonpoint Source Program Success Story. EPA 841-F-09-001JJ.

This Section 319 US EPA success story summarizes the efforts of the Resource Conservation District and community to reduce the levels of *E. coli* in the Lower Nooksack watershed in Whatcom County, Washington. High fecal coliform concentration from the Nooksack River polluted Portage Bay shellfish beds, causing a closure of the beds in 1999. A fecal coliform TMDL by stakeholders brought about the implementation of best management practices in the Lower Nooksack River Basin, which bridges Canada and the US. A tributary of the Nooksack River is Tenmile Creek, a 35 square mile watershed, which showed the highest results for fecal coliform concentration in 2009 at 250 cfu/100 ml. The implementation of 11 miles of hedgerows and riparian buffers along with fixing leaking septic systems in the Tenmile Creek watershed brought the *E. coli* concentration down to 35 cfu/ 100 ml in 2001 and to 10 cfu/ 100 ml in 2003. The Nooksack River Basin landowners have planted more than 100,000 trees since the program started and have replaced failing septic systems. The conservation work in the Lower Nooksack watershed as a whole reduced the *E. coli* concentrations in the shellfish beds to the National Shellfish Sanitation Program (NSSP) standard and allowed for re-opening the harvest.

Model-based Approaches

Brooks, K. N., Gregersen, H. M. , Berghund, E. R. & Tayaa, M. (1982). Economic evaluation of watershed project: An overview methodology and application. *Water Resources Bulletin, 18,* 245-250.

A four-step tool is given to provide an economic overview of projected watershed rehabilitation projects. An example of this model is given through a proposed watershed project in Morocco. Proposed restoration to the 40,000 ha watershed is road construction, channel stabilization, reforestation with gully control, pasture management, and plantation of olive trees. The model assumed the first five years of management did not affect sedimentation rate; similarly vegetation management was not considered effective during this period. The model projects that after 50 years, 15,000 ha of cumulative loss of irrigated land downstream without the project, versus 9,600 ha with the project activities. This paper is primarily useful for the use of a model to predict downstream effects from watershed rehabilitation.

Chandler, D. G. (2006). Reversibility of forest conversion impacts on water budgets in tropical karst terrain. *Forest Ecology and Management, 224*(1-5), 95-103.

A water budget model was developed for three tropical land uses (cropped, heavily grazed, and secondary forest) in the Philippines. Rainfall and runoff were estimated, along with periodic soil moisture and evapotranspiration; these data were used to calculate water budget. Annual precipitation was divided between evapotranspiration and overflow for pasture but for crop and forest precipitation was apportioned more towards to evapotranspiration and bedrock storage. This difference was attributed to depth profile of soil bulk density and drainable porosity. Overall, reforesting compacted landscapes may result in increases in dry season base-flow as forests with bedrock storage may act to recharge aquifers.

Chaubey, I., Chiang, L., Gitau, M. W., & Mohamed, S. (2010). Effectiveness of best management practices in improving water quality in a pasture-dominated watershed. J*ournal of Soil and Water Conservation, 65*(6), 424-437.

171 combinations of BMPs and 250 weather scenarios were simulated using SWAT (a total of 43,000 runs of SWAT) and compared to the baseline SWAT prediction of watershed qualities in 2004 to measure the effectiveness of different BMP combinations on watershed quality. Three types of grazing intensities were used: no grazing, optimum (rotational) grazing, and overgrazing. Three widths of buffer strips were used: 0, 15, and 30 m (placed at the end of the pastured area). Three types of nutrient BMPs were used: controlling poultry litter application rates, litter characteristics, and application timing. After the simulations were run, a 5-way ANOVA model was used to quantify impacts of multiple factors on total N and total P losses using General Linear Models procedure in Statistical Analysis Systems (SAS). Table 6 shows the top 10 individual combinations of BMPs that could be applied to reduce losses of nutrients from pasture areas shown in terms of reduction efficiency. Buffer strips located below the pasture areas decreased total N losses and total P losses significantly. In general for total N, 30 m buffer strips were more effective in reducing total N losses than the 15 m wide strips.

-Tables 3, 4, and 5 show median values from 250 weather scenarios. Table 6 shows the top 10 individual combinations of BMPs that could be applied to watershed to reduce losses of nutrients from the pasture areas. Their effectiveness is indicated by a percentage of reduction efficiency, which I think is the same measurement we are calculating in our impact column.

Parajuli, P.B., Mankin, K.R., & Barnes, P.L. (2008). Applicability of targeting vegetative filter strips to abate fecal bacteria and sediment yield using SWAT. *Agricultural Water Management, 95*, 1189-1200.

This study evaluated fecal coliform and sediment removal by vegetated buffer strips using the SWAT model to determine the reduction in load at a watershed scale. It compared modeled removal rates when location of vegetated strips was targeted and chosen at sites deemed to be most effective compared with removal rates when sites were randomly selected for implementation of strips. By selecting a targeted location for the vegetated strips, the removal rates for the watershed as a whole could be improved. When strip implementation was at the 50% of potential sites, fecal coliform removal was estimated at 60% compared with only 42% removal when implementation sites were randomly chosen. Sediment removal was 72% when targeted and 50% when random. By implementing vegetated strips strategically on a watershed scale where they can have the greatest effect, bacteria and sediment removal can be improved compared with randomly installing this BMP.

Southgate, D. & Macke, R. (1989). The downstream benefits of soil conservation in third world hydroelectric watersheds. *Land Economics, 65*(1), 38-48.

Uses a model-based approach to estimate downstream benefits of reducing erosion in Ecuador's Paute watershed. Estimates downstream benefits of improved management with three scenarios: a) improved management project is fully implemented b) project is implemented four years earlier c) the original schedule for project implementation is followed but sediment load reductions are only half those expected. The results of this study are less interesting (expressed in $/year) than the method to predict effects of sedimentation under differing management regimes.

Wollheim, W.M., et al. (2001). Influence of stream size on ammonium and suspended particulate nitrogen processing. *Limnology and Oceanography, 46*(1), 1-13.
http://www.aslo.org/lo/toc/vol_46/issue_1/0001.pdf

Six streams in Alaska were examined to measure the travel distances of ammonium and suspended particulate organic nitrogen. 15NH4 was added to the stream reaches, which were generally pristine with the exception of fertilization of one reach, which had PO4 added to it. The tracer was added continuously for 3 to 6 weeks for one summer between 1991 and 1997. NH4 and suspended particular organic nitrogen travel length increased with discharge, mainly owing to the changes in depth and velocity. Overall physical gradient contributed more significantly to travel length as compared to biological or chemical changes. Developed two models for how far each will travel.

Wastewater and Stormwater Treatment (Biologically Based)

Alade G.A., & Ojoawo, S.O. (2009). Purification of domestic sewage by water-hyacinth (*Eichhornia crassipes*). *International Journal of Environmental Technology and Management, 10*, 286-294.

This study compared nutrient and total coliform removal rates from sewage stored over 28 days to sewage treated with floating water-hyacinths. Researchers noted that a waste treatment plant in Nigeria for a small community would cost US $500,000 while a pond with water-hyacinths could be constructed and maintained for a small fraction of this cost. Water-hyacinths treated total coliform from a start point of 40,000 cfu/ml to 460 cfu/ml after 28 days (-98.85%). Storage in the absence of water-hyacinth treated total coliform from a start point of 40,000 cfu/ml to 540 cfu/ml after 28 days (-98.65%). Both nitrate and phosphate concentrations increased over the treatment process. Nitrate in sewage water treated by water-hyacinth increased from a start point of 1.1 mg/L to 1.97 mg/L after 28 days (+77%), and for storage in the absence increased to 2.46 mg/L after 28 days (+122%). Phosphate in sewage water treated by water-hyacinth increased from a start point of 40.02 mg/L to 65.36 mg/L after 28 days (+63%), and for storage in the absence increased to 80.54 mg/L after 28 days (+101%).

Hathaway J.M., Hunt, W.F., Wright J.D., & Jadlocki, S.J. (2009). Field evaluation of indicator bacteria removal by stormwater BMPs in North Carolina. *Proceedings of World Environmental and Water Resources Congress*, 1123-1132.

This study reviewed urban area structural BMPs for bacteria removal in Charlotte and Wilmington, NC. The study included dry detention basins, wet ponds, wetlands and bio-retention areas. The BMPs were installed in a number of different settings, including parking lots, outside office buildings and a school. Most BMPs removed bacteria, with some exceptions. Researchers noted that in cases where the bacteria increased between influent and effluent, often there was a presence of birds or bird droppings and, in one case, a used baby diaper. The most effective BMP was a wetland on a 21 ha watershed, which removed 96% of *E. coli* and 98% of fecal coliform. However a different wetland was ineffective and *E. coli* increased 15% in this wetland, perhaps due to the presence of waterfowl. Inconsistency of treatment effectiveness was also observed in the bioretention structures.

Knox, A.K., Tate, K.W., Dahlgren, R.A., & Atwill, E.R. (2007). Management reduces *E. coli* in irrigated pasture runoff. *California Agriculture, 61*(4), 159.

A 0.5 acre constructed wetland was placed to receive irrigation water runoff travelling from a 12 acre irrigated cattle farm in California. The number of cattle ranged between 56 and 102 head. The residence time of water in the wetland varied between 40 to 120 minutes, and a higher *E. coli* removal rate was observed at the longer residence times than at the short residence time. The timing of grazing and irrigation was also managed to observe its effect, and researchers found that when grazing and irrigation were concurrent there was a higher *E. coli* level in runoff. The wetland inlet *E. coli* level ranged between 420 and 157,800 cfu/100ml with a median of 5400 cfu/100ml and the outlet ranged from 10 to 74,600 cfu/100ml with a median of 1283 cfu/100ml.

Leisenring, P.E., Clary, J., Hobson, P. (2012). International stormwater best management practices database pollutant category summary statistical addendum: TSS, bacteria, nutrients and metals. International Stormwater BMP Database. www.bmpdatabase.org

This paper is a statistical summary of a large number of studies across the U.S. and in New Zealand. Authors reported medians, quantiles and confidence intervals for a number of BMPs used for stormwater management. Statistical analysis was performed on studies found in the International Stormwater BMP Database and included a review of grass strips, bioretention, bioswales, grass-lined surface detention basins, retention ponds, wetlands channels, and wetland basins. It also included some manufactured devices. The study reviews performance of these BMPs for removing indicator bacteria, metals and total suspended sediment. The large amount of data and number of studies that went into compiling these statistics provide insight into the general effectiveness of these BMPs across a wide range of conditions. The website provides useful information about the database and study requirements, and it contains an Excel spreadsheet with results from studies entered up to 2011.

Pennington, S.R., Kaplowitz M.D., & Witter, S.G. (2003). Reexamining best management practices of improving water quality in urban watersheds. *Journal of the American Water Resources Association, 39*(5), 1027-1041.

Using median values reported by the Center for Watershed Protection from the National Pollutant Removal Performance Database for a number of structural best management practices, this study evaluated their potential for use in the Rouge River, Michigan. The best management practices evaluated for effectiveness in pollutant removal in this watershed included wetlands (creation and restoration), grassed swales, dry & wet ponds, filtration practices (surface and sand filters), and infiltration such as porous pavements. Based on current pollutant loads in this watershed, the study concluded that the structural BMPs reviewed were insufficient to achieve the required removal. This study is a good source of storm water treatment practice median removal percentages for a number of urban BMPs, however does not report any real changes implemented in this watershed.

Struck , S.D., Selvakumar, A., & Borst, M. (2008). Prediction of effluent quality from retention ponds and constructed wetlands for managing bacterial stressors in storm-water runoff. *Journal of Irrigation and Drainage Engineering*, 567-578.

This study evaluated bacteria removal in mesocosm tanks (one set up as a retention pond and one as a wetland), each was 1.78 by 0.74 by 0.65 m. in size. First order volumetric decay rate constants (K: hr-1) for each bacteria type (total coliform, E coli, fecal coliform and enterococci) were calculated based on the Kadlec and Knight (1996) K-C* equation for contaminant removal. This decay rate can be used to determine wetland or retention pond sizing needed to remove pollutants if the inlet concentration is known and an outlet target concentration is established. Other factors may influence wetland and pond decay rates, such as temperature, total solids and flow rate. As bacteria may adhere to particles and settle into sediments, re-suspension is possible under turbulent flow conditions. Pilot scale studies, such as these, can be used as a start point, however full scale studies are necessary to verify removal rates.

Regulation/ Policy

Postel, S. L., Thompson, Jr., B. H. (2005). Watershed protection: Capturing the benefits of nature's water supply services. *Natural Resources Forum, 29,* 98–108.

includes qualitative data with good references

Literature review of why and how watershed protection should be implemented. Contains case study of Bogota, Colombia, a high elevation, intact wetland ecosystem covers watershed where vegetation absorbs, filters, releases clean water so only treatment at plant is chlorine to disinfect water. This results in very cheap drinking water but is threatened as wetlands are converted to agriculture to support population growth. A Honduras case study demonstrated negative economic impacts of unprotected land conversion. As a small, sloped watershed is being converted to agriculture/urban development without protection measures, water become expensive as chemicals and sand filters must be used to treat it.

The article describes policy mechanisms to protect watersheds. In Quito, Ecuador, the FONAG trust fund was established to pay landowners in the watershed to protect the supply. 80% of drinking water was from an area used by 27,000 locals for cattle, dairy, and timber. Downstream are users that drink water (municipal water supply), irrigators, flower plantations, and hydroelectric power stations. The FONAG fund received money from government agencies and private organizations while an independent financial manager invested it. However, it is still voluntary and missing quantitative estimates in economic savings. In Costa Rica, a 1996 law gives landowners a dollar amount just over the opportunity cost of converting land in order to maintain services in hydrology, carbon fixation, biodiversity, and tourism/recreation. However, it only contracts with private landowners who have titles, has high transaction costs, and favors medium and large scale private landowners rather than small farmers or indigenous people living on 20% of the natural forest that is outside of protected areas. Maintaining the FONOFIFO fund will require government to increase payments by beneficiaries rather than relying on a national fossil fuel sales tax and World Bank loan. Hydroelectric power producers depend on the watershed to make consistent electricity. So far, one plant has recognized the need for watershed services and pays land users $10/hectare because each lost cubic meter of water results in a loss of one kilowatt of electricity—they're securing 460,000 cubic meters of water per year for energy production. This is about a quarter of what the government reward is to landowners participating in conservation.

PASOLAC Pago por Servicios Ambientales al Nivel Municipal y Microcuencas en Honduras, El Salvador y Nicaragua. 2006.

This is a technical document that describes an ecosystem services project in Nicaragua, which includes forest conservation and natural regeneration in 13 ha. Water fees were introduced and providers received $18.75 ha/yr. After two years there appears to be an increase in water availability from natural springs and creeks. Permanent springs have increased from eight before 2005 to 13 in 2005-2006. Temporary springs have reduced from six to one in 2005 and two in 2006. They suggest 1.23 cubic meters of additional water are available per day.

Social and Economic Impacts

Altieri. M.A. (2002). Agroecology: The science of natural resource management for poor farmers in marginal environments. *Agriculture, Ecosystems and Environment, 1971,* 1-24.

NGOs have lead efforts across the poorer regions of the globe to develop agroecological projects in low income environments, commonly achieving a 50-100% increase in production. Increasing crop yields depends predominantly on knowledge, training, and proper management rather than on expensive capital inputs, as observed for the crops the poor rely on such as rice, maize, cassava, potatoes, barley and beans. Diversifying products by raising fish, growing crops in trees and adding goats and poultry to households are important components to increasing total food production. A primary issue with spreading agroecology to new areas is the need for focused specific applications relevant to local conditions. Success also depends on enhancing personal skills, community training in empowerment and participatory decision making, increased access to markets and developing income generating activities. This study examined agroecology across Africa, Asia and Latin America. Most cases of agroecology reviewed by the authors benefited local communities significantly in terms of increased food security and natural resource conservation. In light of many successful NGO efforts, the article questioned why agroecology is not more widespread and identified strategies for gaining increased adoption. These strategies included capitalizing on previous findings, the sharing of knowledge and techniques across regions, and developing collaboration and support among institutions such as universities, farm organizations and governments. The critical success factors identified included increasing the farmer's ability to manage natural resources in the context of crop production, developing community understanding of ecological principles and biodiversity, adapting the expertise facilitators learn in one region to the local needs in other regions, involving farmers in setting research agenda, and identifying changes in policies to insure the equitable and broad adoption of agroecology.

Ayarza, M., Huger-Sannwald, E., Herrick, E., Reynolds, J.F., Garcia-Barrios, L., Welchez, L.A., et al. (2010). Changing human-ecological relationships and drivers using the Quesungual agroforestry system in western Honduras. *Renewable Agriculture and Food Systems, 25*(3), 219-227.

Since its inception in 1992, thousands of poor farmers in Lempira, Honduras have adopted the Quesungual agroforestry system (QSMAS), resulting in increased food security, improved water conservation and improved soil in hillside farming. QSMAS replaces slash and burn with slash and mulch agriculture, which involves selective tree thinning, direct planting with no tilling, and spot fertilization for annual crops (maize, beans, and sorghum). This practice has helped conserve the deciduous tropical forest in areas where QSMAS has been adopted, conserving 40 different species of trees. The direct benefits to farmers included increased crop yields, reduced costs for labor and agrochemicals, improved soil water availability, and reduced susceptibility to crop failure under drought conditions. The benefits to the communities were increased food security, increased water availability, reduced crop loss from hurricane, reduced susceptibility to crop loss during drought, and increased land value ~30%. This paper outlined the changes since the adoption of QSMAS by assessing human, environmental and social economic factors in terms of the Drylands Development Paradigm. It also outlined issues that must be addressed for the future sustainability. Slash and mulch has been practiced in the Honduras hillsides, Ecuador's Amazon Basin and Costa Rica.

Corbera, E., Kosoy, N., Tuna, M.M. (2006). Equity implications of marketing ecosystem services in protected areas of rural communities: Case studies from Meso-America. *Global Environmental Change, 17,* 365-380.

This paper evaluated the efforts of four different projects to commercialize ecosystem services in Meso-America using a three-tiered equity framework: equity in access, equity in decision-making and equity in outcome. Two projects were related to water management and two to carbon sequestration. I will focus on the two water related projects. The first project in the Las Escobas River Basin (707 ha.) encouraged activities related to sustainable agriculture, low impact eco-tourism, sustainable forest management and hydrographical basin management. The public water provider for over 5000 households in Puerto Barrios began transferring US$17.86/ha/year to insure continuous water flow and a reduction in sediment load. This cost was partially covered by a charge to households receiving water, which was used to pay for the organization and operational costs of protecting the Reserve (Reserva Portectora de Manatiales Cerro San Gil). The NGO involved in negotiating the payment for ecosystem services (PES) in Los Escobas confronted and forbid previous illegal encroachments by peasants such as gravel extraction in the watershed, clandestine logging, and the creation of illegal settlements in the protected reserve. There was no widespread community involvement in the decision making process and lack of mediation with conflicted parties, (e.g., logging), although there was an education initiative to build community awareness of environmental benefits. The study concluded there was an equity issue in this case. The 2nd project was Paso de Los Caballos River Basin (741 ha) in in San Pedro del Norte, Nicaragua where the primary economic activity was cattle ranching and basic grain crop production. The local community related water scarcity problems to timber extraction in a period about 40 years previously as well as water mismanagement, and they created a water committee to establish priority areas and negotiate with landowners. Agreements were made with upstream landowners (39 ha) to adopt reforestation and conservation in these prioritized areas in exchange for $26/ha/year, paid collectively by the other community households in order to increase basin recharge and reduce water pollution for drinking water supplies. This payment did not cover the opportunity cost that could have been gained from the production on crops, which was approximately US$126/ha/year. Landowners agreed to the following activities in return for the compensation: to avoid fires, to develop organic agriculture on a limited subsistence plot, to adopt soil conservation practices, to develop agro-forestry systems and to prevent livestock from invading 39 ha upstream. The project was developed by a community Water Committee involved in strategic selection of protected land, developing social awareness of water restrictions and building commitments to achieving water quotas in each community sector.

Current, D. & Scherr , S.J. (1995). Farmer costs and benefits from agroforestry and farm forestry projects in Central America and the Caribbean: implications for policy. *Agroforestry Systems, 30*(1), 87-103.

This is a comparative study of 21 donor, NGO, and national government agency-sponsored projects promoting agroforestry and farm forestry in Central America, Haiti and the Dominican Republic evaluating effective policies and financial benefits. Consultants from each country examined projects within their country. Guidelines were developed to ensure uniform data collection and workshops were organized, areas were visited in person to obtain information from both farmers and project personnel, cost and benefit studies of individual agroforestry systems in each country were performed for a total of 56 agroforestry technologies: woodlots, *taungua* (landowners grow trees and ag crops together during establishment phase of trees planted. Once established shade crops, cropping is stopped. Different from more common practice of allowing landless farmers access to public lands to raise crops during establishment phase of government tree plantation), managed woody fallows, perennial intercrops, homegardens, alley-cropping, windbreaks, trees on contours, and trees in pastures. Table 3 summarizes each agroforestry system's CBA results.

Total value of the agroforestry projects were $150 million US dollars and included 50,000 participants total. Of the agroforestry systems, 75% had positive NPV at 20% discount rate. In two thirds of the cases, NPV and return to labor were superior to the alternative land uses they were compared to. Payback period for woodlots was 5-20 years but for other practices were 1-6 years. Even if the labor input was high, systems were profitable if the output prices and productivity of the system were high. Labor input varied from 0.5 man-days ha -~ yr -1 for one boundary planting of 33 trees to 211 man days ha -~ yr -~ in one inter- cropping system. "Labor input for the agricultural intercropping systems was highly variable, ranging between 24 and 211 man-days ha -1 yr-~; there is thus obviously considerable scope for farmers to modify spacing,management and intensity of agricultural crop production. *Eucalyptus camaldulensis, Gliricidia sepium* and *Leucaena leucocephala* were the most commonly used species, while *Tectona grandis, Acacia mangium* and Cassia spp. were mentioned in at least four projects.

Therefore, findings confirmed the ability of on farm tree-planting to provide rural communities with tree products needed for household use and sale and created employment opportunities and generated income while protecting and improving soil quality and improving other environmental conditions.

Farmers were interviewed, and profitability was not always a predictor of adoption of agroforestry practices; there were many reasons for adoption. Initially, most farmers interviewed adopted systems because they were interested in tree products for household use and in other cases for improving the soil and crop/soil protection. Beyond on-farm, private benefits, other social benefits included windbreaks becoming biological corridors in Costa Rica, the provision of wood products from the farm rather than from the natural forest in Honduras, and substituting trees from agroforestry practices as posts instead of harvesting mangrove forests, which are important for off-shore fisheries in Guatemala. Also, in Costa Rica, reforestation efforts created employment opportunities and generated forestry based on community development in a community that had lost 50% of its population to migration. Therefore, "policy makers should more aggressively use agroforestry as a tool for rural development and poverty alleviation, as well as environmental goals." Most projects had some sort of planting incentive or subsidy. The most successful incentives were those providing minimal material inputs and technical assistance to produce and plant seedlings (eliminate out of pocket expenditures for families). The most sustainable programs were shown in extension registers, where farmers started with few trees and expanded plantings later once they were trained to see the benefits. Once farmers see concrete results, incentives can be gradually eliminated.

Markets also played an important role in motivating farmers to plant trees. In Costa Rica and the Pacific coast of Guatemala, tree-planting was motivated by markets and by planting incentives: once household needs met, famers desired to market tree products to generate income. In Guatemala, the market for tobacco drying posts motivated tree plantings on the Pacific coast and didn't occur where markets didn't exist. In Costa Rica, farmers didn't perform required woodlot tree thinnings where there were no markets and in turn lost productivity and planting benefits. Also inhibiting project participation and success were harvesting laws and regulations that protect forests Farmers feared possible expropriation of land and no longer being able to harvest trees without permits. To overcome this disincentive, most projects had developed contracts and written agreements granting the farmer explicit tree harvesting rights.

Three mechanisms worked well for implementing agroforestry projects: Some projects trained and used local community members as liaisons between project extension agents and the community, providing for continuation of the project long-term. Some projects trained farmers through community and family nurseries, which reduced the need for continued technical assistance and made communities self-sufficient. This ultimately saved money by eliminating costs of government assistance in the future. Some projects took off because farmers saw successful results of tree-planting on other farms, and were especially motivated to being agroforestry when they were able to discuss the results with the farmer who had implemented tree-planted in the past on their own farm.

In summary, projects have shown farmers will plant trees for household use and beyond that point, market incentives become important to keep motivating and maintaining agroforestry practices, such systems produce private and social benefits, and extension activities to promote tree planting are needed. There are inexpensive approaches to promote agroforestry, and public resources should be invested in research and monitoring of programs, training support for commnity-based extension (including making preferred tree species available to farmers), minimal subsidy provision, and removal of regulatory constraints. NGOs, public agencies, and farmer organizations can work with the private sector to expand markets that will further promote tree-planting and promote interagency coordination.

Dumanski, J., Peiretti, R., Benites, J.R., McGarry, D., & Pieri, C. (2006). The paradigm of conservation agriculture. *Proceedings of World Association of Soil and Water Conservation,* 58-64.
http://www.unapcaem.org/admin/exb/ADImage/ConservationAgri/ParaOfCA.pdf

This article outlines Conservation Agriculture (CA). Zero tillage is the cornerstone of CA, which emphasizes soil as living body. Zero tillage strategy is practiced on 95 million hectares worldwide, 47% of which are in South America. 39% of this is practiced in the US and Canada, 9% in Australia, and about 3.9% in the rest of the world (Europe, Africa, Asia). CA promotes minimal disturbance of soil by tillage, balanced applications of chemical inputs, careful water and residue management, all of which reduces land and water pollution and soil erosion, improves water quality, water use efficiency, and other ecosystem services unrelated to the watershed (like carbon sequestration). At the same time, it promotes food sufficiency, less human labor input, and production related to market opportunities. Article explains how zero tillage works and why it's beneficial to watershed services. CA principles: maintain soil cover at all times, balance applications and precision placement of fertilizers or pesticides, promote legume fallows/composting/organic amendments, and practice agroforestry to enhance on-farm biodiversity and alternate income sources. CA supports ecosystem services: mitigates land degradation, climate change, improves air quality, enhances biodiversity and agricultural diversity, and *improves water quality.*

Grieg-Gran, M., Porras, I., & Wunder, S. (2005). How can market mechanisms for forest environmental services help the poor? : Preliminary lessons from Latin America. *World Development, 33*(9), 1511-1527.

This paper defines market mechanisms as "initiatives that involve the sale of environmental services to change incentives of forest managers and to generate resources to finance conservation efforts." Researchers are interested in the development benefits and livelihood impacts of these mechanisms. Researchers found PES programs benefited locals' incomes, provided more land tenure security, and strengthened communities while some private carbon sequestration ventures provided employment opportunities.

In Pimampiro, the PES system contributed 30% of the household expenditure on food, medicine, and schooling. However, the opportunity cost of foregone land uses is unknown. In the Ecuador PROFAFOR PES program, internal rates of return for the five communities ranged from 12-27% over 30 years (a favorable rate) while net present value per family ranged from US$46.6 - $2,481. Currently, opportunity costs are lower than what the main alternative land use would generate (livestock), but this is subject to change in the future. In Huetar Norte PES program in Costa Rica, 60% of the plantation establishment costs are covered so that the main financial benefit will be from timber sales. This enables diversification of income at the farm level through the incorporation of forestry, but participants lose eligibility for subsidized housing and bank credit as the opportunity cost. Across almost all cases (all but Pimampiro, Ecuador), effects on social capital were positive with the strengthening of community organization and networking. Indirect impacts were relevant to projects by private ventures where market transactions for ecosystem services are not intended to provide local community members with payments. More employment opportunities were created by the Peugeot project (PSA Peugeot-Citroen hoped to improve the environmental image of the automotive manufacturing industry and to learn more about the emerging forest carbon market). in Brazil than the previous cattle ranching land use. Also, pressure from the public resulted in the Peugeot project setting up environmental education programs and distribution of seedlings of native species to 83 farmers for agroforestry systems by 2003. The Plantar project by Plantar, a reforestation company prevented the loss of 1,270 jobs in forestry and in charcoal and steel production in an area where there are few alternatives for jobs. The goal is to use carbon credits as a subsidy to maintain the use of charcoal in the pig-iron industry and to promote reforestation with eucalyptus (jobs created are in forestry and industrial operations) .

Ultimately, more research is needed and opportunity costs must be quantified over a longer period of time to see the true benefits of PES programs to local communities.

Kosoy, N., Martinez-Tuna, M.M., Muradian, R., & Marinez-Alier, J. (2006). Payments for environmental services in watersheds: Insights from a comparative study of three cases in Central America. *Ecological Economics*, 1-29.

This report reviewed stakeholder perception, opportunity costs, and the local economic implications of payment for ecosystem services for three case studies: Jesus de Otoro (Honduras), San Pedro del Norte (Nicaragua) and Heredia (Costa Rica). The authors considered three variables to estimate the opportunity cost associated with maintaining forest cover: 1) the net profits from farm activities that could be produced, 2) the price the provider was willing to accept for payment for ecosystem services (PES), and 3) the fair rental value of the property. Jesus de Otoro, Honduras with a population of 5200 people consumes water from the Cumes River watershed (3180 ha), which is covered by 70% forest. In response to serious water problems in the early 1990's, a grass roots organization was established to manage water and sanitation services. Expansion of coffee cultivation in the upstream reaches in 1996 increased water pollution and lead to conflicts between upstream landowners and downstream water users. In 2001, the Program for Sustainable Agriculture in Hillsides of Central America developed a PES scheme to compensate upstream land owners for ecosystem services and gained landowner agreement to a set of actions: no burning; establishment of ditches, terraces and fences; establishment of agro-forestry systems, organic agriculture and forest protection/ reforestation. The PES amounted to only 3.6% of the landholders' opportunity cost, and these landholders did not feel they were fairly compensated for the environmental services provided. The water users in Jesus de Otoro consume water directly without pretreatment. Users feel water service is reliable and water quality has improved over the past 2 years. Most users are largely unaware of the PES scheme; however after it is explained, they feel the payment is fair. Heredia, Costa Rica is in the Virilla River watershed (11340 ha), with an urban area with 28,600 households, was the location of the second case study. About 34% of the Virilla watershed is covered by forest. A PES scheme developed in 2002 was created to protect forest and avoid deterioration of water resources. In this PES scheme, a local beverage company and household users paid an additional amount (~6% on top of their normal water fee) for ecosystem services. The activities promoted by PES were prevention and control of fires, elimination of hunting and extraction of wood, forest conservation, and no conversion of forests to cattle ranching or agriculture. Land acquisition in the upstream areas was also paid for by the PES. Surveyed water users in Heredia felt water quality and service had not changed. These water users had a higher willingness to pay than the fee added to their water service. The third location was Paso de Los Caballos River Basin (741 ha) in in San Pedro del Norte, Nicaragua. This PES project was described in the previous study (Corbera et al. 2006) and this report repeated much of the same information. There was an emphasis in this report on the cost to landowners, who received only about 20% of the opportunity cost of farming the land. Surveyed water users in San Pedro reported bad water service, and 71% of those interviewed disagreed with the amount paid for ecosystems services.

Marquardt, K., Milestad, R., & Porro, R. (2012). Farmers' perspectives on vital soil-related ecosystem services in intensive swidden farming systems in the Peruvian Amazon. *Human Ecology*, 1-13.

Dilemma to conserve Amazon forest while allowing locals to secure livelihood. Swidden (slash/burn) agriculture depends on continued ecosystem service provision to enable agricultural conditions. Looked at soil ecosystem services need for agriculture from farmers' perspectives. 6 farm communities in Peru discussed land uses, swidden systems that work, and failed swidden systems. Noted changes in their production systems and described any eco services that existed in terms of soil quality, crop production quantity and quality, burning practice, forest regeneration, and farming skill. Fallow management and crop diversity were central elements to farmers' strategies for managing ecosystem services. Farmers considered a "good" fallow as a field that produced ample organic matter that would be converted into humus, which is important for a "good" field because it regulates soil moisture and nutrient cycles to the plant. They observed that crops in a "good" field matured faster while the field remained productive for multiple years. Most of the land in the study area had been converted to agriculture so all ecosystem services needed to maintain productive crops must be produced on the same land because there is little reserve or unutilized land. As conservation conflicts with agriculture, it is important for agricultural land to produce ecosystem services in order to sustain the population and conserve Amazonia, and farmers are aware of the benefits of fallow management and crop diversity.

Nair, P.K.R., Tonucci, R.G., Garcia, R., Nair, V.D. (2011). Silvopastoral and carbon sequestration with special reference to Brazilian savanna (*Cerrado*). In Nair, P.K.R. (Ed.), *Carbon sequestration potential of agroforestry systems: Opportunities and challenges* (pp. 145-163). New York: Springer Science.

The *Cerrado*, a Brazilian savanna covering 200 million ha., has been undergoing conversion to beef cattle pasture since 1960, with an estimated current pasture area of 35 - 50 million ha. For the past 20 years, there have been efforts to convert pastureland to eucalyptus tree plantations as a silvopastoral systems. Silvopasture coverage is estimated at 14,000 ha total. Silvopastoral conversion from pastureland begins with rice and soybean cultivation for 2 years followed by tree planting (eucalyptus) in rows with spacing between rows for cattle grazing. Tree cultivation improves soil moisture, thereby creating better forage quality in the dry season. In the third year, grass (Brachiaraia brizantha) is sown and sixty days later, cattle are introduced. Fertilization of grass with N and K at the Fazenda Riacho improved grass production and resulted in higher animal live weight gain in direct proportion to the amount of fertilizer applied. Forage legumes are an alternative to fertilization. One rancher observed a 40% higher weight gain of cattle in silvopastoral compared with those on grass pastures. This article sited other studies and the differences in cattle weight gain under high and low tree density silvopastoral systems in Brazil. An additional benefit to the improved cattle weight gain is the carbon sequestration potential of land under agroforest. The carbon storage potential of silvopastoral systems depends on tree density, soil type, tree species, and other factors.

Pagiola, S., Arcenas, A., & Platais, G. (2005). Can payments for environmental services help reduce poverty?: An exploration of the issues and the evidence to date from Latin America. *World Development, 33*(2), 237-253.

Although the Payments for Ecosystem Services approach (PES) was created as a way to improve natural resource management efficiency, some argue that PES can also positively impact poverty by awarding poor farmers with money for good land stewardship. The extent of the impact depends on how many of the PES participants are actually poor, the amount that is paid to them, and their ability to participate in the program.

In some cases, the positive impact is assumed to occur automatically while in other PES systems, the program targets poor land users specifically. The Western Altiplano Natural Resources Management Project in Guatemala targets poor rural farm households as beneficiaries. In Guatemala, the 77 most hydrologically sensitive (on a slope of 8% or more and between agriculture and forest) have a poverty rate of 70% and include a third of the country's poor. Therefore, it is likely the poor are benefiting from this PES program monetary payments. Also, the National Environmental Management Project in El Salvador explicitly links poverty reduction goals with market-based natural resource management, targeting small farmers to benefit from this strategy. However, the PES payment itself may not be a good measure of financial benefit to participants. Rather, measuring the payment net of the opportunity cost of adopting the PES promoted land use would be more indicative. However, an important aspect to the benefit to recipients of PES payments is having a stable source of income, even if it is a very small percent of what they make.

The establishment of PES programs can also socially and culturally impact the lives of the poor because it requires strengthening or creating institutions, which draws stakeholders together and can foster community relationships. The PES program in Pimampiro, Ecuador helped make an institutional capacity in the Nueva America community that gave them power to influence the municipality decision to enforce environmental regulations. In this way, social capital is a benefit from PES programs.

Pagiola, S., Rios, A.R., Arcenas, A. (2008). Can the poor participate in payments for environmental services? : Lessons from the silvopastoral project in Nicaragua. *Environment and Development Economics, 13,* 299-325.

This paper examined the ability of the poor and extremely poor to participate in payments for ecosystem services (PES) in the Matiguas-Rio Blanco area of Nicaragua during an effort to convert pastureland to silvopastoral production. The question researchers addressed was whether the poor households who were eligible to participate in a PES program were able to do so, as they were required to make land use changes and take actions prior to receiving PES compensation. PES programs are attractive to landowners because payment is predictable and extends for the life of the contract, and payment does not fluctuate with weather extremes and market conditions, as does agricultural income. At the project start, land-use in Matiguas-Rio Blanco was about 63% pastureland , with about 50% of the pastureland degraded and 25% treeless. Landowners meeting basic requirements were offered participation in silvopastoral operations which required land use changes. There were 28 different land uses identified in this project, each with an environmental service score so that more payment was made for more difficult changes with greater benefits to conserving biodiversity and carbon sequestration. The first year of payment for land use changes was 2004, and changes involved 17% of the total area. Changes ranged from sowing improved grass seed to high density tree planting. After the first year, the area of degraded pasture was reduced from 869 ha to 402 ha., natural pasture with high density trees increased from 382 ha. to 471 ha., and improved pasture with high density trees increased from 167 to 279 ha. An additional 110 km of fencing was installed. The area used for crop production fell from 232 ha. to 161 ha. Poor and extremely poor households were heavily involved in land-use changes, contributing to 51% of the decline in degraded pasture and 70% of the decline in crop production. Poor household participation was not limited to the cheaper simpler practices despite a lower ability to obtain credit; however there was evidence that extremely poor households were less able to participate.

Paris, T. (2002). Crop–animal systems in Asia: Socio-economic benefits and impacts on rural livelihoods. *Agricultural Systems, 71*(1), 147-168.

** includes qualitative data**

Contains case studies of agricultural systems integrated with crop and animal enterprises. To increase crop and animal productivity, incomes, and maintain ecological balance, several options have been developed through farm studies and international research organizations. This review focuses on socioeconomic impacts of the new systems and strategies on poor farmers in South Asia. Contains relevant cost-benefit analysis information pertaining to silvopastoral and agroforestry systems with implied watershed improvements or qualitatively stated improvements.

Pascual, U., Muradian, R., Rodriquez, L.C., & Duraiappah, A. (2010). Exploring the links between equity and efficiency in payments for environmental services: A conceptual approach. *Ecological Economics, 69,* 1237-1244.

This article assesses the potential for the reduction of poverty from the provision of environmental services if a Coasean policy approach is adopted by the project, arguing that the efficient outcomes of a Coasean approach can change or reinforce existing power structures and sustain unequal distributions. The issue of "fairness" in the distribution of payment for ecosystem services (PES) has been addressed in different ways by different projects. In Mexico's nationwide PES scheme for carbon sequestration, funds are distributed to communities rather than individuals and collective decisions are made for allocating payments toward the common good by the betterment of infrastructure, education, health, etc. Costa Rica's forest conservation PES scheme run by FONAFIFO pays all individual property owners an equal amount of money per hectare implemented. The Jesus de Otoro project in Honduras pays on the basis of expected service provision, determining the amount based on the type of forest and number of practices adopted by a farmer. Ecuador's PROFAFOR project pays landholders based on the estimated quantity of carbon sequestered on individual lands. The article points out that income from PES may not result in beneficiaries being better off, noting that changing land-use from agriculture to forest may result in a decrease in the local food supply for example. The authors also mention the potential social issues associated with PES schemes, such as the threat to pastoral tribes' identity, way of life and livelihood by excluding livestock from large land parcels. They also point out that PES schemes can be effective means of income redistribution, as occurred in Mexico for water related services by very poor participating forest communities. Their conclusion is that efficiency and equity must both be considered in the development of PES schemes, and that better balance can be achieved by equal stakeholder bargaining power in the definition of fairness of the distribution.

Postel, S. L., Thompson, Jr., B. H. (2005). Watershed protection: Capturing the benefits of nature's water supply services. *Natural Resources Forum, 29,* 98–108.

includes qualitative data with good references

Literature review of why and how watershed protection should be implemented. Contained case study of Bogota, Colombia, a high elevation, intact wetland ecosystem covers watershed where vegetation absorbs, filters, releases clean water so only treatment at plant is chlorine to disinfect water. This results in very cheap drinking water but is threatened as wetlands are converted to agriculture to support population growth. A Honduras case study demonstrated negative economic impacts of unprotected land conversion. As a small, sloped watershed is being converted to agriculture/urban development without protection measures, water become expensive as chemicals and sand filters must be used to treat it.

The article described policy mechanisms to protect watersheds. In Quito, Ecuador, the FONAG trust fund was established to pay landowners in the watershed to protect the supply. 80% of drinking water was from an area used by 27,000 locals for cattle, dairy, and timber. Downstream are users that drink water (municipal water supply), irrigators, flower plantations, and hydroelectric power stations. The FONAG fund received money from government agencies and private organizations while an independent financial manager invested it. However, it is still voluntary and missing quantitative estimates in economic savings. In Costa Rica, a 1996 law gives landowners a dollar amount just over the opportunity cost of converting land in order to maintain services in hydrology, carbon fixation, biodiversity, and tourism/recreation. However, it only contracts with private landowners who have titles, has high transaction costs, and favors medium and large scale private landowners rather than small farmers or indigenous people living on 20% of the natural forest that is outside of protected areas. Maintaining the FONOFIFO fund will require government to increase payments by beneficiaries rather than relying on a national fossil fuel sales tax and World Bank loan. Hydroelectric power producers depend on the watershed to make consistent electricity. So far, one plant has recognized the need for watershed services and pays land users $10/hectare because each lost cubic meter of water results in a loss of one kilowatt of electricity—they're securing 460,000 cubic meters of water per year for energy production. This is about a quarter of what the government reward is to landowners participating in conservation.

Pretty, J. N., Morison, II, J., & Hine , R.E. (2003). Reducing food poverty by increasing agricultural sustainability in developing countries. *Agriculture, Ecosystems & Environment, 95*(1), 217-234.

In a survey from 1999-2000, 208 projects in 52 developing countries in which 8.98 million farmers adopted farming BMPs using environmentally sensitive practices and technology on 28.92 million hectares, which represents 3% of the 960 million hectares of arable and permanent crops in Africa, Asia, and Latin America. Improvements in food production occurred in intensification of a single component of the farm system, addition of a new productive element to the farm systems, better use of water and land to increase crop intensity, and improvements in per hectare yields of staples through introduction of new regenerative elements and locally appropriate crop and animal varieties. Improved food production occurred through one or more of four mechanisms: (i) intensification of a single component of farm system; (ii) addition of a new productive element to a farm system; (iii) better use of water and land, so increasing cropping intensity; (iv) improvements in per hectare yields of staples through introduction of new regenerative elements into farm systems and new locally appropriate crop varieties and animal breeds.

89 projects with yield data showed an average per project increase in food/ha produced of 93%. Weighted average increases across these projects were 37% per farm and 48% per hectare. In 80 projects with small farms of cereal plants, 4.42 million farms on 3.58 million hectares increased household food production by 1.71 tons/year. Practices leading to these increases were increased water use efficiency, improvements to soil health and fertility, pest control with minimal or zero pesticide use.

Pretty, J.N., Noble, A.D., Bossio, D., Dixon, J., Hine, R.E., Penning de Vries, F.W.T. & Morison, II, (2006). Resource-conserving agriculture increases yields in developing countries. *Environmental Science &d Technology, 40*(4), 1114-1119.

Scientists surveyed 286 small scale agricultural projects in 57 poor countries where various resource-conserving techniques were implemented (interventions) and have resulted in increased crop production. Sustainable agriculture practices include integrated pest management, integrated nutrient management, conservation tillage, agroforestry, aquaculture, water harvesting, and livestock integration. They sampled 268 projects once over a four-year period, and 68 projects twice over a four-year period and classified these projects into 8 FAO/World Bank categories Average crop yield in these projects were 79%. All crops showed water efficiency gains, especially in rainfed crops. Increased output on small, poor farms that have used sustainable agriculture techniques over just a four-year period imply considerable potential for avoiding environmental and economic costs. Although there are no specific dollar amounts of costs averted, watershed services impacted are erosion control, water yield, water quality, flood control, and benefits extend to other environmental and social issues, such as climate change and biodiversity loss, and global hunger due to population growth.

Rosset, P.M., et al. (2011). The Campesino-to-Campesino agroecology movement of ANAP in Cuba: Social process methodology in the construction of sustainable peasant agriculture and food sovereignty. *Journal of Peasant Studies, 38*(1), 161-191.

This paper outlines how Cubans were able to boost food production without the expensive inputs of conventional agriculture by substituting ecological inputs and making more diverse farming systems (transitioned to agroecology).The paper provides a history of the movement and how it took off in Cuba, definitions of agroecology, explanations for why the social movement is effective, and evidence of increased production by the peasant sector and resilience to climate change through agroecology implemented by the movement. This occurred more so because of the social process methodology and grassroots movement that formed (Campesino-a-Campesino (CAC) and the National Association of Small Farmers (ANAP and is part of La Via Campesina international agrarian movement) are key participants) rather than just because alternatives to conventional agriculture were made available.

Essentially, international organization La Via Campesina views two models of farming: peasant agriculture and agribusiness, where reproducing that ag business model on one's land through purchased chemicals, commercial seeds, and heavy machinery, results in exclusion and environmental destruction. The costliness of agribusiness has driven development for alternative inputs that are separate from ever-varying petroleum prices that drive the cost of conventional inputs. Campesino-a-Campesino Agroecology Movement in Cuba (MACAC) is an important example of sustainable peasant agriculture and farmer-to-farmer extension methodology so this report was conducted to evaluate the Cuban experience, identify new steps, and make strategies available to LA Via Campesina in other countries working with agroecology.

Researchers traveled throughout Cuba twice in 2008 and 92009 They visten dozens of farms, held workshops, and met with leaders of ANAP at different levels. They reviewed all internal files and documents of MACAC.

In the 1990s lost trade relations resulted in a collapse in food production because of the loss of imported fertilizer , pesticides, tractors, and petroleum. The country reoriented its agriculture to depend less on imported inputs and ended up outperforming other Latin American countries' agriculture with a 4.2% annual growth rate per capita food production from 1996-2005.

The most successful methodology for promoting farmer innovation and sharing and learning is the farmer-to-farmer method in the MACAC movement. Beginning in the 1970s, this form of communication spread from Guatemala and eventually to Cuba by the 1990s. Promoters of agroecology are recruited from farmers who are recognized by their peers for successful innovations of the agro-ecological practices on their own farms. There is no compensation for being a leader except for the satisfaction of being a good role model. Otherwise, people believe farmers would not believe in their technologies and just promote it to get a salary. There are facilitators who match and arrange visits of farmers who need solutions to promoters who have them, organize workshops to help keep things running. If a particular facilitator is not seen has effective, they can be ied by the farmers.

Statistics presented in the paper include the following:

- In 2008, farms were classified by promoters, facilitators, and coordinators according to the degree of agroecological advance and integration. 1 is low integration, 3 is high, and it is based on 31 criteria.
- Families receiving highest scores gained the respect of the community and cooperative and honor.
- 12 years after the CAC came to Cuba, between 2008 and 2009, results were impressive: Number of families grew from 200 in 1999 to 110,000 in ten years. In 2009, there were less than 350,000 families in the peasant sector, so this is a third of families joining in just ten years. 12,000 farmer-promoters, 3,000 facilitators, 170 coordinators.
- Non MACAC members have also adopted agroecological practices from observing neighboring participants that 64% of all peasant farms use organic soil amendments and 82% use ecological pest management methods.
- Figure 4 shows invoiced sales for 2008 and shows that on 33 farms in Sancti Spiritus, the higher the level of agroecological integration (category 3), the greater the value of production. Category 1 shows 600 pesos/hectare/year and 2,300 pesos/worker/year. Category 2 shows 900 pesos/ha/year and 3,700 pesos/worker/year. Category 3 shows 2,400 pesos/ha/year and 6,700 pesos/worker/year
- In 2002, CAC became a movement in Cuba and input substitution gave way to agroecological integration. In 2006-2007, agroecology was advanced under normal conditions, in 2008 Cuban agriculture was hit by 3 hurricanes, but peasant agriculture was resilient and production only dropped 13%. In 2009, production by the peasant sector exceeded expectations in the National Production Plan.

- Figure 5 shows total production from the peasant sector in Cuba. Total production in 2002 was about 75 , 2006 about 175, 2007 about 205, 2008 175 (hurricanes), and projected to be at least 275 in 2009.This coincides with the growth of the agroecology movement.

Another way to see the relationship between peasants, food production, and agroecology is to look at production data and agrochemical use:

- Vegetable production fell 65% from 1988-1994, but by 2007 was 145% over 1988 levels. This corresponds with 72% fewer agricultural chemicals in 2007 than in 1988.
- Bean crops reduced 77% in 1994 but 351% over 1988 levels by 2007 with a 55% decreased use of agrochemicals.
- Roots and tuber crops down 42% in 1994 but at 145% 1988 levels by 2007 with an 85% decreased use in agrochemicals.
- Sugarcane NOT a peasant crop. Contrasting data: Yields fell 25% below 1988 levels and fell another 3% by 2007, which is the same time period during which production of peasant crops leaped. Did not change use of agrochemicals.
- Climate: In 2008, 3 hurricanes struck Cuba. In Holguin and Las Tunas provinces 40 days after Hurricane Ike, researchers observed large areas of industrial monoculture with only about 5% of crops still standing. Agroecological farms with multi-storied agroforestry farming systems had only knocked down the taller 50% of crop plants while the lower story were growing rapidly because of newly exposed sunlight. Peasant families had saved the trees that had blown down and stood them back up the first morning after the storm
- Social structure: Researchers also observed that transitioning to agroecological farming may have impacts on power structures. Peasant interviews revealed that in conventional farming systems, men were in charge because all money wen to them and all inputs. As farm diversified, researchers observed and were told that the row crops are managed by men while animals, vermiculture, and medicinal plants could be in the woman's domain and allowed her potential income. Adolescents and children were observed to be associated with animals while fruit trees and preserves were managed by grandparents. However, only 12 % of facilitators and 8% of promoters in the CAC movement are women so more effort needs to be put into making women activists and training them so they can be promoted.
- Factors that slowed developing the CAC process and spreading agroecology were when peasant promoters didn't retain humility and were viewed the same as an outside extension agent or technician. Also, peasant promoters that had to deal with bureaucracy like paperwork for reporting slowed the process even though monitoring is so important.
- Factors that made successful the CAC movement were respect for local culture and customs in EACH locality The most successful cases involved and built on the skills of and respect for local leaders, used local structures like the cooperative assembly, and involved potential local allies, whether that be schools and teachers to physicians.
- MACAC important to creating new grassroots leadership within ANAP as peasants who become promoters train, help others, and gain respect from peers. Then they get elected to leadership positions in their cooperatives and can rise to national positions—bottom-up of new leader generation.

- Story of MACAC in Cuba shows more integrated agroecosystems are more productive but cannot spread without this social process of peer-to-peer knowledge sharing and peasant self-organization. Limiting factor is not technical but rather social and methodological .
- In order to scale up agroecology, need CAC methodology and peasant self-organization

Ruiz-De-Ona-Plaza, C., Soto-Pinto, L., Paladino, S., Morales, F., & Esquivel, E. (2011). Constructing public policy in a participatory manner: From local carbon sequestration project to network governance in Chiapas, Mexico. In Nair, P.K.R. (Ed.), *Carbon sequestration potential of agroforestry systems: Opportunities and challenges* (pp. 247-). New York: Springer Science.

This article reviewed the Scolel Te project associated with carbon sequestration through agroforestry and forestry in Chiapas, Mexico starting in 1996. The article reports the Scolel Te has developed into a structured system for carbon transactions that is now being applied to other countries under the Plan Vivo System. Authors recommend network governance based on solutions achieved through the consensus of multiple stakeholders, rather than an exclusively market-based transaction approach, as stakeholder collaboration can better consider social and ethical matters in conserving ecosystems. At the time of publishing, a total of 677 producers from 62 communities in Chiapas had participated in the project, engaging in activities amounting to 2000 ha. of carbon sequestration, 2660 ha. of avoided emissions, and over 7500 ha of land in conservation or restoration. The participants were small landholders from 5 different Maya language groups who participated either on individually owned land or on collectively owned community lands. Working groups were involved in decision making with representation from farmers in these groups. The agroforestry activities implemented included improving fallow areas, planting trees in coffee production, conservation, restoration and the use of living fences in pastures. Water quality impacts were not reported. Although participants were paid on a per hectare basis for activities, the economic impact to households has not been substantial. Although the data suggests 1% to 25% of household income is from carbon payments, higher income could be gained from maize production on this same land. No changes in livelihood strategies have resulted from the carbon payments nor has a substantial boost in household income; however the report identifies sustainably managed sale of timber in the future as a potential income producing activity.

Southgate, D., & Wunder, S. (2007). *Paying for watershed services in Latin America: A review of current initiatives.* Working Paper 07-07. Sustainable Agriculture and Natural Resource Management Collaborative Research Support Program and Office of International Research, Education and Development, Virginia Polytechnic Institute and State University.

This report briefly summarizes payments for environmental services (PES) projects related to watershed conservation in several countries in Latin America. They noted that strong skepticism of natural resource management and PES in Bolivia is slowing adoption of these practices, based on suspicions that the disguised objective is to privatize public resources. Venezuela lags in adoption of PES because it has a large, closed economy, and the Bolivian Highlands because of the nature of their strong indigenous culture. Venezuela and the Andes of Peru have potential for such projects, but service buyers have been unwilling to pay for these services. Ecuador and Columbia have the most examples of successful implementation of PES strategies. Three case studies of successful PES implementation were summarized in the report: Pimampiro (Ecuador), Fondo para la conservacion de agua (FONAG) in Quito (Ecuador), and payments for the National Program for Hydrologic-Environmental Services in Mexico. The PES in Pimampiro is based on voluntary participation by both buyers and sellers of environmental services. PES are conditional payments based on sustaining land use practice, either protection of pristine forests or unassisted restoration, with periodic review of the land to insure compliance. These payments are supported by charging an addition 20% fee on top of normal water fee to the 1350 households receiving water. As a result of PES, timber extraction has nearly ceased in Pimampiro (while continuing in neighboring areas) and 14% of previous agricultural lands have been allowed to revert to natural vegetation. Payments to households of $US0.50 to US$1.00 per ha. are far below the opportunity cost, however landowners with limited land-clearing capacity gain economically by accepting the payment. No mention was made of economic or social shifts resulting from the second case study of FONAG. The third case study of hydrologic-environmental service in Mexico observes a strategy to target places with high natural forest cover for protection, paying US$40/ha/year for cloud forests and $30/ha/year for other tree-covered land. These charges are covered by fees paid by federal water users and support impoverished people who inhabit the forested land. Between 72-83% of the payments were made to areas characterized as having high or very high poverty levels, and 31% of beneficiaries were below the poverty line. By enrolling in the program, impoverished households increased family income by up to 10%. Payments for community owned forests are often used to improve infrastructure in the village. Hydrologic improvements resulting from the hydrologic-environmental service have not yet been identified or quantified.

Wunder, S. (2005). *Payments for environmental services: Some nuts and bolts*. Center for International Forestry Research: Jakarta, Indonesia.

This book examines payment for ecosystem services (PES) as one approach within the overall portfolio of conservation approaches and also outlines practical considerations in the choice of location and design of PES schemes. Wunder defines the scope within which PES is likely to succeed based on a literature review and his direct observations of projects (the book is rife with beautiful images). He asserts that PES users will only pay for schemes that can clearly demonstrate additionality, and that PES is best suited to moderate threat scenarios, often in marginal lands. In the Brazilian Amazon, governments have declared large areas as protected lands and expressed interest in PES rewards for this conservation; however low deforestation rates in these remote areas make them unlikely candidates for PES, as the threat of forest destruction is low and far off in the future. At the other end of the spectrum is Mato Grosso (Brazil) where rapidly expanded soy farming and ranching is causing deforestation. Although the threat to the environment is high, the economic forces of the opportunity costs of profitable farming and ranching activities in Mato Grosso make a PES scheme unlikely to succeed in this area. He suggests that a moderate threat scenario such as that observed in Acre (Brazil) where road projects are creating linkages for the potential expansion of timber and beef production, but where people are creating a grass-roots conservation movement, Governo de Floresta, is a more likely candidate for a successful PES scheme. Acre represents the intermediate threat scenario, where opportunity costs are rising (but not onerous) and the environmental threat is foreseeable, as the most likely location for a successful PES scheme. The author also provides insight on other questions about PES: Who to pay? How to pay? What are the issues in payments to poor? and How to design a PES scheme? He provided examples of the effect of PES on sellers noting the following examples of cases where people have benefited: Costa Rica where PES was 10% of the families' income in a quarter of the families, Virilla (Costa Rica) where PES averaged 16% of the family cash income, the Oca Peninsula (Costa Rica) where PES payments lifted more than half the recipients living in poverty to above the poverty line, and Pimampiro (Ecuador) where payments amounted to 30% of recipient's spending on food, medicine and schooling.