ABSTRACT

LINGLEY, BENJAMIN C. Economic Modeling of Drinking Water Costs Associated With Aquatic Biodiversity and Water Quality

Clean surface water has far-reaching implications on human life as well as plant and aquatic species. Decreased water quality can diminish aquatic biodiversity as indicated by benthic macroinvertebrates which in turn affect the entire aquatic food web. Decreases in surface water quality generally result in a loss of desirable game and other aquatic species. There is a strong relationship between forest cover in a watershed and the corresponding water quality. Forests that are uphill in a watershed filter harmful contaminants as water flows through fibrous roots and significantly reduces soil erosion and sediment loading into water bodies as roots help hold soil in place. The measure of water quality in surface waters such as rivers, streams, lakes and reservoirs is often indicated by turbidity, which can be a direct result of sediments in water. This study investigates how surface water quality may affect treatment costs at drinking water treatment plants. Previous studies have indicated a positive relationship between forest cover in a watershed and treatment costs via decreased levels of turbidity and total organic carbon. The necessary data will be collected from various GIS databases including NatureServe, USGS GAP and other NLCD. A survey was developed which aims to gather data about treatment methods and chemical usage. By combining the positive relationship between a forested watershed, water quality, aquatic biodiversity and drinking water treatment costs, stakeholder interest can be maximized to achieve maximum return on conservation efforts.
Economic Modeling of Drinking Water Costs Associated
With Aquatic Biodiversity and Water Quality

by
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Dedication

I would like to dedicate this paper to my friends and family who have supported me throughout the process. To my parents Ron and Lauren, thank you for your unwavering support and words of wisdom and encouragement throughout my academic career. To my sister Kara, brother-in-law Joshua and niece Noelle, thank you for being there to add your support and often needed reprieve.

A special thank you to my wife Christina. Your love and support made this possible. You never questioned me when I wanted to stop working and go back to school and you were always there for me to discuss, ramble and vent to even if you had no idea what I was talking about. Your strength and resolve are an inspiration. I love you.
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I would like to acknowledge those who have helped me in completing my degree. Thank you to my fellow students, friends and colleagues who were there to listen and help me talk through ideas. My knowledge and experience has been greatly enhanced by working with all of you.

Thank you to my committee members, Dr. Kevin Potter and Dr. Marcelo Ardón Sayao. Your constant support and feedback made this possible. I especially appreciate your flexibility and quick response when needed. This would not have been possible without you.

Thank you to Dr. Curt Flather of the Rocky Mountain Research Station for providing funding for me and allowing me to be a part of his ongoing research. I greatly appreciate the opportunity to contribute to this project.

I would especially like to thank Dr. Fred Cubbage. You have been my advisor and mentor since before I enrolled in this program and your continuous support has made all of this possible. I appreciate the time, energy and effort you have taken to guide me through my graduate education. You have had a tremendous impact on my life and for that I am grateful.
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Economic Modeling of Drinking Water Costs Associated with Aquatic Biodiversity and Water Quality

Introduction

Drinking water that is clean, safe and readily available is a precious resource which is the foundation of life and yet it is often taken for granted. Research has shown that vegetative land cover, specifically forests, surrounding lakes, streams and rivers can significantly improve the water quality in reservoirs, rivers and other drinking water sources (Elias et al. 2013; Ernst et al. 2004; Kreye et al. 2014; Matteo et al. 2006; Postel 2005; Postel & Thompson 2005; Wickham et al. 2011). Furthermore, research by Wickham and Flather (2013) has presented a positive relationship between forest cover and aquatic biodiversity, which suggests that higher amounts of forest cover could improve both water quality and aquatic biodiversity.

Figure 1: Overview of land cover relationship to water quality co-benefits
Clean water sources originate from watersheds that share certain landscape-level features. There are many land cover classifications, but the literature primarily focuses on developed, agriculture, bare land, shrub and forests (Freeman 2008; Warziniack 2016; Wickham and Flather 2013). Studies show that agriculture and development pose the highest risk to drinking water sources (Freeman 2008; Warziniack et al. 2016). With development come impervious surfaces such as asphalt, concrete and building roofs. Rainfall cannot penetrate through these layers and creates concentrated runoff which has great erosive power and collects many harmful contaminants that are anthropogenic byproducts. Agricultural land use produces high concentrations of nitrate and phosphorous from fertilization practices (Vitousek et al. 1997). A storm event or excessive irrigation can easily transport chemicals into streams which then feed into rivers, lakes and reservoirs (Warziniack et al. 2016). Excessive nutrient flow into stagnant water bodies such as lakes and reservoirs has been shown to produce detrimental algae blooms that can have devastating effects in an aquatic ecosystem (Postel 2005). High levels of nutrients have proven to be difficult and expensive to treat in drinking water. A watershed that maintains a high level of natural vegetation, such as in forests, shrub and wetlands, usually has a higher relative water quality, in the form of lower nutrients, lower turbidity, and lower total organic Carbon (TOC), than watersheds dominated by agriculture or development (Wickham et al. 2011).
Natural vegetation in a watershed has many positive effects on the health of the watershed and therefore on water quality. The United States Geological Survey (USGS) surveys land at various scales, and at the eight digit hydrologic unit, there are 2,112 watersheds in the lower 48 states (National Gap Analysis Program 2014). These watersheds provide many ecological and human health services (Postel & Thompson 2005). Trees and shrubs provide natural water purification which occurs as water filters through fibrous roots. This filtration removes pollutants and excessive levels of nutrients that may lead to contamination of lakes, rivers and streams throughout the watershed (Postel and Thompson 2005). In addition to filtration, the roots of such vegetation also aid in mitigating runoff and erosion of soil that could cause contamination of water supplies. Erosion of soils throughout the watershed can lead to sediments contaminating water supplies and changing the composition of stream beds or other water bodies (Holmes 1988). Rainfall has strong soil displacement power, and forests and shrub land ease the force of rainfall when it hits the canopy, therefore further reducing erosion.

While natural shrub lands are successful in all of the aforementioned areas, forests are substantially more effective. This is especially the case when comparing the amount of carbon sequestration in forests to all other land cover types because healthy watersheds provide carbon sequestration when they have an abundance of trees and other vegetation, which has positive implications for pollutants and climate change. Forested watersheds are integral in providing habitat for fisheries and other aquatic biodiversity. Shaded spots in streams and rivers are important for the success of many of our freshwater fish species, which play a vital role in food web structure and material cycling (Allan and Castillo 2007).
Effects on Treatment Costs

Legislation such as the Federal Water Pollution Control Act of 1948 and the amendments that
came to be known as the Clean Water Act (1972) helps ensure that clean and potable water is
constantly supplied to the citizens of the United States through various pollution control methods
(CWA 1972). Legislation such as the CWA and the Safe Drinking Water Act (SDWA) of 1974
has led to complacency about public water supply, resulting in rejecting practices that may lead
to cleaner water at the source because it is expected that safe and clean drinking water is
delivered to every home (SDWA 1974). What may help change this perception and garner more
public interest of the role forest cover has in negating the negative effects of poor water quality
on aquatic biodiversity, is an illustration of the negative financial impact of treating
contaminated water.

For example, a watershed with 10% forest cover had 211% higher water treatment costs
compared to a similar watershed that had 60% forest cover (Postel 2005). Furthermore, for
every 10% increase in forest cover, with an upper bound of 60% forest cover, treatment and
chemical costs decreased approximately by 20% (Ernst et al. 2004). Statistics such as these have
resulted in some of the nation’s largest cities, such as New York, Boston, Seattle and Portland,
investing in upland forest conservation to mitigate the treatment costs of their water supply.
New York City has invested $1.5 billion in various efforts to preserve watersheds rather than
building a $6 billion treatment plant with a $300 million per year operating budget (Postel 2005).
The American Water Works Association provides yearly benchmark data on Operation and Management costs for member water treatment plants throughout the country. They show an interquartile range of $364 per million gallons of water treated. We will use these data, as well as data on the quantity and variety of chemicals used to treat the water, to normalize the data across different regions.

**Water Quality**

It is generally agreed that diminished water quality at the intake leads to higher treatment costs (Dearmont 1998). However, correlating aquatic biodiversity and source water quality directly to treatment costs encompasses many factors that make it difficult to assess nationally. Many variables other than water quality have to be weighed when assessing the cost of treatment. Soil type, rainfall, topography and the surface water source (river or lake/reservoir) are a few of the considerations that may influence the treatment cost (Holmes 1988).

In addition to land cover classification, there are numerous other factors that impact surface water quality. For example, when assessing landscape variables in macroinvertebrate tolerance to degradation of streams, topographic complexity was the land form feature which most strongly correlated to impaired waters as indicated by indicator invertebrate indices (Potter et al. 2004). This implies that areas with steeper relief tend to have more erosion of sediments and soils which can have significant effects on water quality. Also, geology and soil type can have varying effects on water quality. Watersheds that have similar relief characteristics can have vastly different water quality issues based on the type of rock or the composition of the soil over which
the rainwater is flowing, which affects the amount of particulates deposited in the water (Potter et al. 2004). Lastly, Potter et al. (2004) noted that watershed shape, which measures the roundness of the watershed, has a significant effect on water quality and can even negate some of the benefits of forested watersheds. The roundness of the watershed can lead to more concentrated runoff with greater erosive power.

Even in areas where relief and topography do not tend to create significant erosion and sediment loading, such as coastal plains, issues such as saltwater intrusion may prove to have a dramatic effect on drinking water supplies. Salt water intrusion not only affects groundwater but can also cause increased salinity of near-coast waters (SFFP 2012). There are indirect effects of saltwater intrusion as well. As groundwater and surface waters become more salinized, this will lead to large-scale vegetation loss as trees and shrubs that are not adapted for saltwater conditions will die, further compounding the water quality issues (Herbert et al. 2015). This may increase with climate change and projected rising sea levels.

One of the most significant side effects of poor water quality, other than human consumption risk, is the effect decreased water quality has on aquatic ecosystems, specifically aquatic biodiversity. Among the most studied and potentially best indicators of water quality are indices of benthic macroinvertebrate communities. These are small invertebrates that include mussels, crayfish, worms, many different larvae, and snails. One of these indices is the EPT index, which uses three orders of aquatic insects (WSI 2016). The index is named after three commonly found aquatic insects; Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies) (WSI 2016). This index is especially versatile because they are sensitive to human caused and
natural stress. When anthropogenic actions affect their environment, an imbalanced community results from population changes. Effects from physical habitat alteration, point and nonpoint source pollution are displayed by these aquatic insects because they spend the entirety of their lives in the water. EPT insects respond to a vast range of pollutants and are affected by short and long-term conditions that affect water quality (WSI 2016). The fact that they are easy to collect, have limited mobility and are reliably found in certain conditions make them ideal as an indicator of water quality (Washburn & Sanger 2011). If species that are known for being intolerant of polluted conditions are absent and species that are widely considered tolerant of lower quality scenarios are present, it can be concluded that the water is of low quality and can be further analyzed. Furthermore, the species’ variation and abundance can have amplifying effects on the aquatic community and food web. Fish spawning can be particularly affected if low water quality causes a reduction in benthic macroinvertebrates as they are the food supply for spawning fish and their offspring (SFFP 2013).

**Best Management Practices**

Biodiversity has long been a focus of conservation. Species richness and evenness are indicators of a healthy and diverse ecosystem (Hooper et al. 2005; Loreau et al. 2001). Importantly, “biodiversity conservation is proactive; it is not confined to last ditch efforts” (Scott et al. 1993). The Clean Water Act (1972) outlined water quality goals such as Best Management Practices (BMPs), which define methods by which to prevent or reduce nonpoint source pollution. Along with improved general water quality, an expected outcome of BMPs was the preservation of aquatic biodiversity via clean water (Postel & Thompson 2005). One of these BMPs, riparian
buffers or streamside management zones (SMZs) improve water quality by providing shade, producing organic debris, perhaps most importantly, regulating sediment and nutrient flows from upland areas. Matteo et al. (2006) showed that forest BMPs had an especially positive effect on watersheds that are more urbanized. Urban forest buffers reduced sediment and nutrient loading and decreased storm water runoff. Furthermore, watersheds were able to better handle extreme conditions that traditionally had more detrimental effects from events such as storm events, nonpoint source pollution flooding and high winds (Matteo et al. 2006).

However, riparian buffers have been shown to have variable impact in the overall effect on water quality for large scale (watershed) applications. The width of the riparian buffer, topography and soil type of the watershed may influence the effectiveness of the buffer (Momm et al. 2014). The small amount of land cover created by riparian buffers is disproportionate to a large watershed (Potter 2004), and therefore can be overwhelmed by the amount of sediment, nutrients and other contaminants passing through the buffer.

**Turbidity and Total Organic Carbon (TOC)**

Turbidity and Total Organic Carbon (TOC) have been studied to determine the effects they have on water treatment costs. Turbidity measures the sediments in the water by the amount of light that is able to pass through a sample and is measured by Nephelometric Turbidity Unit or NTU. Many factors can affect turbidity but it usually worsens after a rainfall event as a result of erosion and sediment mobilization (Postel & Thompson 2005). Past studies have shown a positive relationship between turbidity and drinking water treatment costs (Freeman et al. 2008,
Warziniack et al. (2016). If there is an increase in forest cover by 1%, turbidity is reduced by 0.5 NTUs or 18% (Freeman et al. 2008). Freeman et al. (2008) reports that for a 1% increase in turbidity there is a 0.07% - 0.30% increase in treatment costs.

Similarly, Warziniack et al. (2016), determined that there is a negative relationship between forest cover and turbidity. If 1% of forested land in a watershed is developed or converted to rangeland, it increases turbidity by 3.9% and 6.3% respectively. Furthermore, it was determined that the intake source has an effect on the level of turbidity (Freeman et al. 2008). River intakes showed higher turbidity than reservoir intakes. It was determined that a 1% increase in turbidity increased treatment costs by 0.19% (Warziniack et al. 2016). If an average watershed had 10% of its forestland developed, there would be a $65,000 per year increase in chemical treatment costs. With a 3% discount rate over 30 years, this would be a present value of $1.3 million (Warziniack et al. 2016).

TOC was not found to have a statistically significant relationship with land cover. However, a 1% increase in TOC did increase treatment costs by 0.46%. This would result in an annual $3,400 increase in chemical cost or a 30-year present value of $66,900 with a 3% discount rate (Warziniack et al. 2016). Freeman et al. (2008) found that with a 1% increase in forest cover, TOC was reduced by 16%.
Study Scope and Methodology

The current research is an extension of *Integrating biodiversity and drinking water protection goals through geographic analysis* by Wickham and Flather (2013). In this study, ‘Hotspot’ watersheds were identified, where at-risk aquatic species of the 90th percentile intersected with sites that had surface drinking water intake sources such as rivers, streams, lakes and reservoirs. Other criteria for watershed identification included the presence of points that occurred in areas with median amounts of protected land and a demonstrated at least a 1% increase in developed land. Wickham and Flather (2013) concluded that these hotspots identify areas where at-risk aquatic species correlate to poor water quality. One issue with analysis on this scale as identified by Harris et al. (2005) is that geographic analysis based solely on hotspots is outside the range of consideration for land managers to make conservation decisions due to the large scale. Wickham & Flather (2013) address the issue of scale by incorporating drinking water data with biodiversity data which allows for ‘geographic specificity,’ by scaling entire watersheds down to smaller parcels where stake-holder buy-in is more likely (Wickham & Flather 2013).

The current research project aims to explore the possibility of mutually beneficial relationships in preserving aquatic biodiversity, water quality, forest cover and drinking water treatment costs. If the data quantifiably demonstrate the joint benefits associated with land use management directed at biodiversity conservation and source-water quality for human consumption, it may lead to increased public interest in preserving aquatic biodiversity.
To develop the necessary data, a few methods will be employed. First, a national survey will be sent to all of the 7,000+ water treatment facilities in the US. This project is being federally funded through the U.S. Forest Service, and a survey of this scale must be submitted for approval to the Office of Management and Budget (OMB). This process can take as long as a year and would significantly limit the data gathering process. By using a NGO to conduct the survey, it may not be necessary to secure OMB approval and it may be possible to send out the survey immediately. The Trust for Public Land (TPL) specifically could help with the survey in an effort to expedite the survey process.

The survey will inquire about the methods and costs of water treatment. The survey questions were created after consulting the American Water Works Association (AWWA) benchmark report (2013) for determining the variables they report, including chemical treatments and costs. The survey was also designed after an on-site visit to the Dempsey E. Benton Water Treatment Plant in Garner, NC in fall 2015 and consulting with the operators on the trends they observe and the methods used at that facility. The questions were also formed using the five benchmark studies on water quality and land cover which are discussed in detail later. Appendix 1 includes a draft of the survey, which has questions that are both qualitative and quantitative in nature. The quantitative questions aim to understand the types and volumes of chemicals used to treat the water. Although there are questions about pricing and costs associated with the chemicals, we anticipate that quantities of chemicals used will help account for regional pricing discrepancies. The survey also seeks to understand how variations in water quality may affect the treatment process. As water from a particular source is generally steady, there is a question about how the process is affected by storm water events. Following a heavy rain, there is an influx of
sediments and surface contaminants which may alter the quantity or even types of chemicals needed. This process can act as an indicator of the effect of decreased forest cover in a watershed. Data on this process can help us understand how an influx of contaminants might cause greater chemical use and in turn higher costs.

We predict that it will be difficult to show a strong correlation between water source quality, and a statistically significant decrease in water treatment costs, based solely on cost data from the survey. There are many regional and local considerations that may alter price reporting or cost that is not related to incoming water quality. We will attempt to make the data more directly comparable across regions by inquiring about the types and quantities of chemicals used in the treatment process. Once these data are collected they can be compared to national benchmark data that are gathered and published by the American Water Works Association (AWWA 2014).

Other information collected during the survey include gauging the amount of control each plant has over the selection and acquisition of their chemicals, and how sudden changes in water composition from storm water events, soils and topography affect the treatment process.

NatureServe will provide the aquatic biodiversity data. NatureServe maintains a large database of imperiled aquatic species that are searchable in a GIS dataset. The analysis will be conducted on a 12- or 8-digit HUC (watershed level) to examine the density of at-risk aquatic species. Incomplete data from private lands is one of the main challenges with this dataset. Other data sources may be explored such as the impaired waters list from section 303(d) of the Clean Water Act and the USGS GAP biodiversity database tied to 12-digit HUCs.
The USGS GAP is an analysis of conservation lands for mapping biodiversity. The goal of GAP is to answer questions about location and management of biodiversity conservation lands (GAP 2014). It also seeks to understand and map specific plant and animal communities such as who manages lands they are found on and how much of a species’ habitat is on protected lands (GAP 2014). Two datasets that will be especially useful are the Land Cover and Aquatic datasets. These datasets are GIS raster datasets available in Grid or IMG files that can be downloaded and manipulated in GIS software. All of these data will be examined in order to determine if the benefits of conserving watershed land cover provides mutual benefits by decreasing water treatment costs and conserving aquatic biodiversity.

Consulting the list of impaired 303(d) waters will provide a starting point in the analysis of water bodies that are above recommended pollution levels for their designated use. The Clean Water Act’s primary goal is "to restore and maintain the chemical, physical, and biological integrity of the Nation's waters" (33 U.S.C §1251(a)). Section 303(d) requires each state to assess and compile a list of water bodies that do not meet water quality standards (WQS), otherwise known as impaired waters.

In addition to identifying impaired waters, each state must also set a total maximum daily load (TMDL) for each pollutant/water body combination that will bring the water back within acceptable standards. The EPA must approve the state’s TMDL or set a replacement TMDL. The TMDL must account for all pollution sources, both point and nonpoint, and the list from the state must be updated every two years. Section 303(d) listed waters will provide accurate accounts of any water bodies that are not meeting the WQS for their intended uses. This can be
narrowed down to source water intake points from which treatment plants draw their water. These can then be analyzed for land cover characteristics within that particular watershed as Wickham and Flather (2013) did in their study to identify hotspots.

A case study of the B. Everett Jordan Lake in Chatham, Durham, Orange and Wake Counties in central North Carolina also is presented at the end of this paper to demonstrate the types of issues common with land use and water quality problems.

**Comparing Previous Studies, Methods, and Results to Guide Research**

There have been numerous studies which examined the effects of certain factors on the cost of drinking water. Five of these will be used to guide our research based on their direct relevance and common variables that were studied. In addition to the Freeman et al. (2008) and Warziniack et al. (2016) studies, three additional studies will be explored. The first is a 1987 study by D. Lynn Forster on *Soil Erosion and Water Treatment Costs*. The second is a 1988 benchmark paper written by Thomas Holmes titled *The Offsite Impact of Soil Erosion on the Water Treatment Industry*. The third is *Costs of Water Treatment Due to Diminished Water Quality: A Case Study in Texas* by David Dearmont, Bruce McCarl and Deborah Tolman published in 1998.

The methods and results of each study were scrutinized and used to deduce the best approach for determining the variables which have the most significant quality and financial impacts on treating drinking water. Table 1 summarizes these studies in general which will be discussed in detail.
After each paper was studied and outlined, the information that was gathered was used to outline a recommended approach for future research by the USFS and Curt Flather. This recommendation includes suggested methods, areas of focus and the biggest challenges the authors noted as being prohibitive. The recommendation also includes a sample survey to submit to treatment facilities across the country that will obtain the necessary information based on studying these five reports.
Table 1. Summary of Studies

<table>
<thead>
<tr>
<th>Author</th>
<th>Data Sources</th>
<th>Components Studied (Variables)</th>
<th>Methods</th>
<th>Results/Relationships</th>
<th>Cost Effects</th>
<th>Comments</th>
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</thead>
</table>
| Holmes et al. (1988) | 1986 AWWA: O&M expenditures, water production, raw water turbidity & treatments used  
EPA data for typical treatment costs  
Regional wage data from AWWA & Dept. of Energy  
Storage capacity data by ASA from Water Resources Council | - Regional sediment loading rates  
- Raw water storage capacity  
- streamflow | Standard firm model provides approx. sediment related costs.  
Standard firm model linked in recursive fashion with residual discharges via relationship between sediment loading and ambient water quality | 1% increase in turbidity increase expenditures by 0.07%  
1% increase in turbidity will save $0.20 per million gallons treated or $534 annually for average plant or $69,826 for the state of Texas in 1998. | Turbidity damage ranges from $35.33 million - $661.19 annually nationally. | Dearmont, et al. note that total costs of turbidity and chemical contamination would likely be higher if nonchemical costs were considered. |
| Forster et al. (1987) | 12 Communities in western Ohio based on having available information, surface source water and identifiable uphill watersheds.  
Original Data from treatment facilities and observation | - Turbidity  
- volume treated  
- storage capacity  
- watershed soil erosion rates | Multiple regression analysis  
Cost model based on 4 independent variables  
Converted cost function into logarithmic cost function using Cobb-Douglas function (anticipated to be nonlinear). | 1% change in soil erosion would lead to a 0.4% change in treatment costs.  
1% reduction in turbidity will reduce treatment costs by 0.04%  
1% increase in total gallons treated reduces treatment costs by 0.04%  
1% increase in annual precipitation increases treatment costs by 1.74% | 25% reduction in uphill soil erosion would save Ohio $2.7 million in water treatment costs. |  |
| Dearmont et al. (1998) | Selected 12 sample treatment plants based on criteria and database from TWS.  
Treatment cost data: monthly water reports from Texas Dept. of Health, Div. of Water Hygiene.  
Mun. water treatment plants chemical analysis reports | - Turbidity  
- pH  
- volume of water treated  
- contamination variable  
- rainfall | Estimate a model that relates chemical cost per unit of treated water to raw water supply characteristics.  
Cost estimate made using cross-sectional heteroskedastic and time-wise autocorrelation model (Judge 1985). | 1% reduction in turbidity will reduce treatment costs by 0.04%  
1% increase in total gallons treated reduces treatment costs by 0.04%  
1% increase in annual precipitation increases treatment costs by 1.74% | 1% decrease in turbidity will save $0.20 per million gallons treated or $534 annually for average plant or $69,826 for the state of Texas in 1998. |  |
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</table>
| Freeman et al. (2008) | TPL survey to 60 unique treatment plants  
2001 NLCD data  
Water Quality Index derived from EPA water treatment plant model index | - Turbidity  
- TOC  
- Alkalinity  
- Water Quality Index  
- % Land Cover (Ag, urban, forest & non-forest) | Source Water Quality, land cover and chemical treatment cost were analyzed using a general linear model with land cover characteristics as predictor variables.  
Uses method of least squares to fit the models | Increased forest cover significantly related to decreased turbidity  
Increased TOC related to decrease % forest cover  
WQI had positive relationship with forests  
Increased TOC related to higher treatment cost | None Reported                                                                                     | Turbidity alone did not significantly relate to chemical treatment costs. After adding in TOC and Alkalinity for WQI there was significant relationship between low water quality and treatment costs |
| Warziniack et al. (2016) | AWWA survey – focused on costs of chemical treatment (37 respondents)  
2011 NLCD 10-digit HUC from Public Water System Identification (PWSID) | - Turbidity  
- TOC  
- Land Cover | 2 Steps:  
Ecological Production Function: land use affects water quality  
Economic Benefits Function: Relates Water Quality to treatment costs. Measures avoided chemical costs | 1% increase in turbidity = 0.19% increase in costs (Not Significant)  
1% increase in TOC = 0.46% increase in costs (Significant)  
Inc. size of treatment plant = reduced avg. cost of treatment  
Negative relationship of converting from forest and turbidity  
No relationship of converting from forest and TOC | Converting 10% of average watershed from forest to developed would inc. chemical costs by $65,000/yr  
r = .03 the NPV over 30 years = $1.3 million  
Avg. treatment plant annual chemical costs = $742,000. 1% inc. in TOC = $3,400 in inc. costs.  
NPV = $66,900 over 30 years |
Forster et al. (1987) Study

Forster et al. (1987) identified key independent variables which they hypothesized will have a significant effect on water treatment costs. The primary concern is soil erosion from uphill within the watershed. Erosion leads to suspended solids in the raw water that must be removed as part of the treatment process. This is not only for the health of the consumer but also to protect the facilities and not clog the hydraulic machinery. Forster et al. (1987) also predicted that “economies of scale” will affect treatment costs. The larger the treatment facility, the less expensive it is to treat a unit of raw water. Additionally, storage time is factored into treatment costs. It is predicted that as water that is stored in lakes and reservoirs sits idle, the more sediments can settle to the bottom of the lake/reservoir and not have to be removed during the treatment process. There were twelve communities and watersheds in western Ohio included in this study. Each community was chosen based on three criteria: the source water was surface water, officials kept accurate data of variable treatment inputs, and it was possible to clearly identify the upstream watershed and to estimate gross soil erosion rates.

Forster et al. (1987) developed a model to relate water treatment costs to the aforementioned variables.

\[ C = f(S,R,T,E) \]

Where C is a community’s variable cost of water treatment, S is the number or gallons treated, R is the retention time of the raw water, T is a variable that represents the required turbidity improvement during the treatment process, and E is the soil erosion rate upstream. It is noted that each of the 12 communities studied have similar size, topography, and soils, so the sediment
delivery ratios are all approximately the same. It was therefore possible to use gross erosion rates instead of sediment yield estimates.

The study took place over 25 months at the conclusion of which daily averages were computed for each variable. The data were input into the model using a logarithmic cost function:

\[ c = a_0 s^{a_1} t^{a_2} e^{a_3} e^{a_4} \]

Where \( a_0, a_1, a_2, a_3 \) and \( a_4 \) are regression coefficients. The results of the regression analysis are summarized in Table 2.

Table 2. Regression Analysis Data Examined by Forster (1987)

<table>
<thead>
<tr>
<th>Variable</th>
<th>Description</th>
<th>Geometric Mean</th>
<th>Minimum</th>
<th>Maximum</th>
</tr>
</thead>
<tbody>
<tr>
<td>( s )</td>
<td>Volume of water treated (thousand gallons/day)</td>
<td>933.0</td>
<td>275.0</td>
<td>6,166.0</td>
</tr>
<tr>
<td>( r )</td>
<td>Retention time (days)</td>
<td>331.0</td>
<td>2.5</td>
<td>3.8x10^5</td>
</tr>
<tr>
<td>( T )</td>
<td>Turbidity improvement (turbidity units)</td>
<td>3.8</td>
<td>0.3</td>
<td>38.9</td>
</tr>
<tr>
<td>( e )</td>
<td>Soil erosion upstream</td>
<td>2.0</td>
<td>0.9</td>
<td>4.5</td>
</tr>
</tbody>
</table>

Source: Soil Erosion and Water Treatment Costs Forster et al. (1987)

Each variable was assessed based on percentage change in cost associated with a 1% change in the corresponding variable, meaning water treatment costs would change by 0.4% for a 1% change in soil erosion rates upstream. Extrapolated, this would lead to a $2.7 million savings in treatment costs in Ohio if conservation efforts were successful in reducing soil erosion by 25% (Forster et al. 1987)
Holmes (1988) Study

Holmes (1998) used two economic modeling methods to estimate the influence of water quality on treatment costs. The first is the standard firm model which provides an initial estimate of sediment related costs. These estimates were then used to gauge how reasonable the results were from the second method, the hedonic cost function model. The hedonic cost function “embodies attributes of the production process not typically considered by neoclassical models” (Holmes 1988). Lastly, in order to estimate the linkage between sediment loadings to waterways and the subsequent economic costs, Holmes uses a “black-box” model. This model assumes that “environmental variables interact in the production of ambient environmental quality…and the marginal effect of sediment loading on water quality is nonlinear.”

Both models use operating and maintenance costs as the dependent variable. For the hedonic cost function, in addition to treatment plant costs the dependent variable also includes costs associated with water acquisition and distribution. The primary source for data for the hedonic cost function is survey data that were collected by the American Water Works Association (AWWA). This dataset has survey responses from 430 of the 600 largest utilities in the U.S. and has specific data on operating and maintenance expenditures, water production, raw water turbidity and specific treatment costs for individual facilities. Holmes (1988) also accounted for salary data for water utility employees from AWWA and used local Department of Energy data in order to determine a general price per kilowatt hour for the facilities in question. The hedonic cost function is:

\[ C_j = C_o Y_j^{a_1} Q_j^{a_2} (\pi R_{ij}^{a_3}) \]
where $C_j$ is expenditure of firm $j$; $C_0$ is the intercept; $Y_j$ is the water production of firm $j$; $Q_j$ is influent water quality of firm $j$ and $R_{ij}$ is the price of input $i$ to firm $j$.

After determining which facilities had all the required data, Holmes was left with 132 facilities for the hedonic cost equation method and 230 for the typical treatment cost function. Holmes quantified the relationship between sediment discharges in waterways, which he coined environmental linkage. Environmental linkage represents the correlation between sediment loading in waterways and the associated economic costs. For environmental linkage, the data for sediment loading came from Resources for the Future while storage capacity and streamflow data were from the Water Resource Council’s Aggregated Sub-Areas (ASA). As with the previous two methods, water quality data are from the AWWA. In all, there were 66 data points available to estimate environmental linkage. Environmental linkage is represented as:

$$Q_k = A_0 S_k^{a_1} F_k^{a_2} R_k^{a_3}$$

Where $Q_k$ is the ambient water quality in region $k$; $A_0$ is an intercept term; $S_k$ is sediment loading in region $k$; $F_k$ is streamflow in region $k$ and $R_k$ is the storage capacity of reservoirs in region $k$.

The results showed that a 1% increase in turbidity resulted in a 0.07% increase in operating and maintenance costs. Furthermore, Holmes estimated that efforts to mitigate turbidity would cost between $4.40 per million gallons and $82.34 per million gallons. These numbers were extrapolated to a national scale, with treatments of suspended sediments in surface water costing
between $35 million and $661 million annually. Holmes provided a table that shows a side by side comparison of regions with the most sediment discharge and the greatest benefits caused by a 10% decrease in sediment loading. This table shows minimal overlap between regions indicating that soil erosion criteria may not show areas with the greatest offsite benefits.

Table 3. Results from Holmes (1988) Study of Water Quality Treatment Costs

<table>
<thead>
<tr>
<th>Rank</th>
<th>Incremental Benefit</th>
<th>Sediment Load</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>$1,000</td>
<td>ASA</td>
</tr>
<tr>
<td>1.</td>
<td>1458.5</td>
<td>1806</td>
</tr>
<tr>
<td>2.</td>
<td>335.8</td>
<td>1804</td>
</tr>
<tr>
<td>3.</td>
<td>238.8</td>
<td>408</td>
</tr>
<tr>
<td>4.</td>
<td>195.6</td>
<td>1009</td>
</tr>
<tr>
<td>5.</td>
<td>132.3</td>
<td>1005</td>
</tr>
<tr>
<td>6.</td>
<td>132.2</td>
<td>1102</td>
</tr>
<tr>
<td>7.</td>
<td>130.3</td>
<td>701</td>
</tr>
<tr>
<td>8.</td>
<td>126.4</td>
<td>406</td>
</tr>
<tr>
<td>9.</td>
<td>124.9</td>
<td>1103</td>
</tr>
<tr>
<td>10.</td>
<td>113.3</td>
<td>1007</td>
</tr>
</tbody>
</table>

Holmes developed a cost function approach to “obtain econometric estimates of water treatment costs induced by suspended sediment mitigation measures.” Turbidity related cost was estimated using engineering cost data. “A hedonic cost function was then specified and estimated using operating cost data. The hedonic model fit the data well and the estimates produced appeared reasonable.”

**Dearmont et al. (1998) Study**

David Dearmont, Bruce McCarl and Deborah Tolman (Dearmont et al. 1998) conducted a study titled *Costs of Water Treatment Due to Diminished Water Quality: A Case Study in Texas*. In this study they focused on chemical costs as a function of raw surface water quality. Their
results are discussed in the context of the presence of raw water contamination and the effects of turbidity on chemical costs.

Dearmont et al. (1998) state, “Following other studies, we use sediment as a primary indicator of water quality… (It) accounts for 68% of total suspended solids in waterways.” Dearmont examined water treatment costs associated with sediment and chemical contaminants in an area of Texas. Of the 142 surface water treatment facilities, 10 were selected to geographically represent the varied conditions in Texas. The study area included four main river systems, and two sites from each were selected. One system had three selections and some of the cities chosen had more than one treatment facility. Ultimately, 12 sites were chosen for study with an average treatment of 222 million gallons of water per month. Dearmont et al. (1998) gathered treatment data from the Texas Department of Health, Division of Water Hygiene. Specifically, data are from monthly water reports which are filed by treatment plant operators. The reports include quantity treated and types and amount of chemicals used. The reports also include data on turbidity, pH and alkalinity levels for both raw and treated water. Chemical cost data were obtained by contacting sales representatives of chemical companies. The chemical costs per million gallons of water treated ranged from $20.21 to $286.14 with an average of $88.38. Turbidity had a range from 5.58 - 89.16 NTUs with an average of 23.05 NTUs.

Dearmont et al. (1998) constructed a model which related chemical cost per unit of treated water to raw water supply characteristics. They represented per unit chemical costs as a function of: gallons treated, turbidity, pH, rainfall and a proxy variable for chemical contamination.
Cost/1,000 gallons = $b_0 + b_1(\text{total gallons}) + b_2(\text{turbidity} \times \text{pH}) +$

\[ b_3(\text{turbidity} \times \text{pH})^2 + b_4(\text{turbidity} \times \text{pH})^3 + b_5(\text{contamination dummy}) + \]

\[ b_6(\text{average annual rainfall}) \]

This is not a formal model of a cost equation or cost function from production economics. These aforementioned models would require costs for all inputs for items such as surface water, labor and energy. This is due “to the unavailability of prices; the difficulty of relating the use of some of these items to water volume; and the multicollinearity induced by fixed relative prices during the short timeframe of the study with relatively constant levels of input usage.”

The model from Dearmont et al. (1998) included total volume of water treated to account for scale. They also used a polynomial form due to the information available about sedimentation process, because a low-turbidity raw water supply requires less coagulant than a more turbid one, so as turbidity increases less coagulant is needed. Once turbidity reaches a critical level, the amount of coagulant rises. Turbidity and pH are combined because of the chemical relationship between coagulants and pH adjusters. Coagulants are generally acidic and lower pH. If pH falls below 7.5, lime or another base must be added to raise pH. A raw water supply with high pH reduces the likelihood of this, thus the relationship between turbidity and pH.

Rainfall effects on raw water quality were accounted for via runoff and sedimentation data. Also, a contamination dummy variable was included to account for the Texas Water Commission report of potential chemical contamination in groundwater. The variable is meant to capture the
“change in the intercept of the regression line representing additional treatment cost due to potential contamination of the surface water supply.”

Dearmont et al. (1998) made their cost estimate using a cross-sectional heteroskedastic and time-wise autocorrelation model. This model can correct for “autocorrelation of differing degrees in each cross section and for heteroscedasticity.” By doing this, it provides an estimate that is efficient and unbiased for the model across time and cross sections.

| Table 4  Estimation Results for Chemical Cost of Treatment per Thousand Gallons |
|-----------------|-----------------|-----------------|
| Variable        | Estimated Coefficient | t Ratio        |
| Constant        | -0.1314          | -6.5053         |
| Total gallons   | -1.6950 × 10^{-8} | -4.1604         |
| Turbidity × pH  | 1.3496 × 10^{-4}  | 4.3989          |
| (Turbidity × pH)^2 | -1.5130 × 10^{-7} | -2.6375         |
| (Turbidity × pH)^3 | 5.3013 × 10^{-11}| 1.9374          |
| Contamination dummy | 0.0947      | 7.7713          |
| Average annual rainfall | 5.6024 × 10^{-3} | 8.3164          |

One gallon equals 0.003785 m³, or 3.785 L.

The results showed that chemical treatment costs increase at a decreasing rate as the level of turbidity increases. A 1% decrease in turbidity reduces treatment costs by 0.27%. They also showed there is an elasticity associated with total gallons treated. A 1% increase in total gallons treated lowered treatment costs by 0.04%. A 1% increase in rainfall causes a 1.74% increase in treatment costs. The results further indicated that when groundwater contamination is present, treatment costs increase by $94.75 per million gallons. Lastly, a 1% reduction in turbidity (23.05 NTUs to 22.82 NTUs) reduces chemical costs by $0.20 per million gallons treated or $534 annually per plant for the average gallons treated. If this number is expanded to all of the 142
cities in Texas that use and treat surface water, it would mean a $69,826 per year in chemical costs based on a total of 350 billion gallons.

**Freeman (2008) Study**

Freeman (2008) conducted a two phase analysis; the first part sought to determine if there was a positive relationship between percent forest cover and water quality. The second analysis examined whether treatment costs and water quality have a negative relationship.

The data came from multiple sources. In total 60 water treatment plants were surveyed. Twenty plants were surveyed in 2004 and 40 more in 2006. The 60 plants were chosen based on a number of criteria. Plants that collected water from extremely large bodies of water, such as the Great Lakes, were excluded. The treatment plants also met the following criteria:

- At least 90% of raw water coming from surface water
- Having drinking water source areas less than 500 square miles
- Treatment plant treating at least 1 million and no more than 100 million gallons per day

In 2006, in addition to the listed criteria, facilities were limited to a study area defined by the USFS Northeastern Area, which includes 20 states in New England, the Mid-Atlantic and part of the upper Midwest. States that did not have critical land cover data in GIS format as of 2001 were eliminated. Freeman (2008) also excluded plants in states that had not produced or were unwilling to create hydrologically accurate source area delineations. The source area delineations were obtained through the EPA after signing a confidentiality agreement. Fifteen of the 20 USFS Northeastern Area states had data that fit these criteria. Using 2001 National Land Cover
Database (NLCD) information, boundaries and statistics were generated from the source area delineations. The land cover data were simplified into 6 from the original 21 categories as follows: forest, non-forest vegetation, wetland, urban, agriculture, and landform feature. Furthermore, the classifications were determined for 100 and 300 ft. buffers around water bodies within each source area.

A water quality index (WQI) was developed to express the relationship between water quality parameters provided by the individual treatment plants. These parameters were total organic carbon (TOC), turbidity, alkalinity, conductivity, temperature and pH. Freeman (2008) used TOC as the best indicator of general water quality. The WQI equation incorporates turbidity and alkalinity into a TOC-based indicator. Specifically, WQI was calculated by using mean annual TOC and adding median annual alkalinity and turbidity multiplied by coefficients for alkalinity and turbidity. The alkalinity coefficient of 0.003 was from the Disinfectant/Disinfection Byproduct Rule used by the EPA in developing minimum TOC removal criteria. The turbidity coefficient was from Freeman’s experience of turbidity’s effect on coagulant doses and sludge production:

\[
WQI = \frac{1}{(\text{median } TOC + 0.003 \times (\text{median alkalinity}) + 0.01 \times (\text{median turbidity}))}
\]

A general linear model that uses the method of least squares to fit the models was used to examine the relationship between source water quality, land cover, and chemical treatment costs. Only the 40 plants from 2006 had available data for this model.
Freeman found that land cover had the most significant impact on turbidity, accounting for 66% of the variation in turbidity. Increased percentages of agriculture and urban land cover led to higher turbidity while increased forest land cover was significantly related to decreased turbidity as well as TOC and the WQI. TOC and the WQI had strong correlations to treatment costs as increases in each led to increased treatment costs. Higher percentages of urban land cover also significantly correlated to higher costs. One inconsistency with Freeman’s findings was with non-forest land cover. Increased non-forest land cover correlated to decreased treatment costs. While turbidity did not independently link to treatment cost, the WQI, which factors in turbidity along with TOC and alkalinity, did show a significant positive relationship.

**Warziniack et al. (2016)**

Warziniack et al. (2016) used a two-step process in analyzing the effect of land use on water treatment cost, an ecological production function and an economics benefit function. The ecological production function factored in watershed stressors such as land development, roads and agriculture. In addition to the stressors, the ecological production function focused on turbidity and TOC to measure water quality as it pertains to treating drinking water. Similar to Freeman et al. (2008), Warziniack et al. simplified the land cover data into six land uses: forest, rangeland, developed area, agricultural area, water and barren land. A variable was included to account for the source of the water as well. Surface water intakes are generally either rivers or reservoirs. Reservoirs allow for sediments to settle and potentially cause fewer issues associated with sediment loading.
\[ \log(Q_i) = \beta_0 + \beta \text{LANDUSE}_i + \gamma \log(\text{STRESSORS}_i) + \delta_i \text{RIVER}_i + \epsilon_i \]

\(Q_i\) was the overall water quality determined by many factors. \(\text{STRESSORS}_i\) represented land development, roads, and agriculture which occur within the watershed. The variable \(\text{LANDUSE}_i\) was one of the six land use types (forest, rangeland, developed area, agricultural area, water and barren land) with forested being the reference condition. \(\text{RIVER}_i\) was the indicator for the source of the water. This was a binary system and was set to 1 if the intake was on a river or 0 for reservoir.

The results of these variables inserted into a multivariate regression model are found in Table 5:

<table>
<thead>
<tr>
<th>Dependent variable:</th>
<th>Log(median turbidity)</th>
<th>Log(median TOC)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>(1)</td>
<td>(2)</td>
</tr>
<tr>
<td>Developed area (%)</td>
<td>0.038***</td>
<td>0.039***</td>
</tr>
<tr>
<td></td>
<td>(0.012)</td>
<td>(0.013)</td>
</tr>
<tr>
<td>Agriculture area (%)</td>
<td>-0.093**</td>
<td>-0.082*</td>
</tr>
<tr>
<td></td>
<td>(0.041)</td>
<td>(0.043)</td>
</tr>
<tr>
<td>Rangeland (%)</td>
<td>0.061**</td>
<td>0.049**</td>
</tr>
<tr>
<td></td>
<td>(0.015)</td>
<td>(0.020)</td>
</tr>
<tr>
<td>Barren area (%)</td>
<td>-0.40**</td>
<td>-0.38**</td>
</tr>
<tr>
<td></td>
<td>(0.18)</td>
<td>(0.18)</td>
</tr>
<tr>
<td>Water area (%)</td>
<td>0.007</td>
<td>0.002</td>
</tr>
<tr>
<td></td>
<td>(0.68)</td>
<td>(0.076)</td>
</tr>
<tr>
<td>Source (River = 1; Reservoir = 0)</td>
<td>0.74*</td>
<td>0.69**</td>
</tr>
<tr>
<td></td>
<td>(0.32)</td>
<td>(0.33)</td>
</tr>
<tr>
<td>Animal units density (AU/km2)</td>
<td>0.031</td>
<td>0.039</td>
</tr>
<tr>
<td></td>
<td>(0.028)</td>
<td>(0.030)</td>
</tr>
<tr>
<td>Constant</td>
<td>-0.60</td>
<td>-0.47</td>
</tr>
<tr>
<td></td>
<td>(0.49)</td>
<td>(0.52)</td>
</tr>
<tr>
<td>Obs.</td>
<td>37</td>
<td>36</td>
</tr>
<tr>
<td>R-squared</td>
<td>0.56</td>
<td>0.46</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Regression characteristic:</th>
<th>All observations</th>
<th>1 outlier eliminated (100 NTU)</th>
<th>Only conventional treatment</th>
<th>All observations</th>
<th>Only conventional treatment</th>
</tr>
</thead>
<tbody>
<tr>
<td>All observations</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1 outlier eliminated (100 NTU)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Only conventional treatment</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Notes: *** p-value<0.01; ** p-value<0.05; * p-value<0.1.; Std. errors are in parenthesis.
The economic benefits function examined how treatment cost is affected by numerous variables surrounding the treatment process itself. These included four different types of treatment technologies: unfiltered/disinfection only, direct, conventional and advanced. There were also variables for the size of the watershed and total volume produced in millions of gallons treated per day. \( (\text{COST}_i) \) represented average chemical treatment cost measured in dollars per million gallons treated. \( \text{SIZE}_i \), was the variable to normalize the variously sized treatment plants in the study area, measured in million gallons treated per day. The size of the watershed was represented by \( \text{DRAINAGE} \). CONVENTIONAL, DIRECT, and ADVANCED were dummy variables which indicated the method used by a particular plant. “The random variable \( \nu_i \) is assumed to be normally distributed with mean zero and constant variance” (Warziniack et al. 2016).

\[
\log(\text{COST}_i) = \beta_0 + \beta_Q \log(Q_i) + \beta_{pop} \log(\text{SIZE}_i) + \beta_{dr} \log(\text{DRAINAGE}_i) \\
+ \beta_c \text{CONVENTIONAL}_i + \beta_d \text{DIRECT}_i + \beta_a \text{ADVANCED}_i + \nu_i
\]

The results of the economic benefits function are as follows:
The data used in this study came from multiple sources. For data from treatment facilities, a two-phase survey was sent out to AWWA member facilities. This was conducted as part of a joint AWWA Technical and Education council and U.S. Endowment of Forestry and Communities study to examine the value of watershed protection as it relates to cost reductions in treatment of drinking water. The study was concerned with water sources in forested ecoregions in Eastern Temperate Forest and Northwestern Forested Mountains. Based on a ZIP code database of applicable facilities, a long form survey was sent out via email.

This survey focused on information pertaining to characterization of treatment process and all chemicals typically used. The survey asked for the quantity of each chemical that is used during...
the survey period (unspecified), the cost per unit (gallon or pound) and total cost of specific chemicals for survey year. Warziniack et al. (2016) normalized the data to a cost per gallon standard by dividing by the gallons produced for that year. For incoming water quality, the survey requested data for a number of variables: minimum, median and maximum turbidity and TOC during the most recent calendar year for which data were available. Lastly, this first survey asked for intake geographic location and the source of intake (river or reservoir).

A second shorter version of the survey was sent out which only focused on chemical costs and raw water quality, mainly omitting the specifics about types and amounts of chemicals used. In total there were 37 treatment facilities that provided adequate data. Twenty-six of the facilities used conventional treatment, seven used direct treatment and two each of advanced treatment and no treatment, only disinfection. As part of the response, each facility reported the Public Water System Identification (PWSID) number associated with it. The PWSID was used to match the treatment facility to the appropriate watershed based on the EPA’s Safe Drinking Water Information System (SDWIS). Land cover data were determined at the 10-digit HUC watershed using 2011 USGS NCLD.

The results showed that rivers had higher and more variable turbidity. TOC levels showed no difference in water from reservoirs or rivers. There is a strong negative relationship between forest cover and turbidity. A 1% decrease in forest land caused a 3.9% increase in turbidity when converted to developed land and a 6.3% increase when converted to rangeland. Despite insufficient data to draw conclusions, agriculture and barren land decreased turbidity. There was no relationship between TOC and land cover, possibly explained by the lack of agriculture,
where there tends to be a relationship to TOC. A 1% increase in turbidity and TOC increased cost by 0.19% and 0.46%. There was a turbidity outlier that more than likely influenced the results and made for a stronger correlation.

Warziniack et al. (2016) estimated that if 10% of the average watershed is converted from forest to developed status, chemical treatment costs would increase by approximately $65,000 per year. A discount rate of 3% over 30 years would be $1.3 million. A 1% increase in TOC increased costs by 0.46%. With an average annual chemical cost of $742,000, this would increase annual cost by $3,400, with a present value of $66,900 over 30 years with a discount rate of 3%. Turbidity had a very weak relationship with chemical costs that became insignificant when the high turbidity outlier was removed. These results are summarized in Table 7.

Table 7 - Increase in treatment costs from development in the watershed

| Percentage change in turbidity from a 10% change in forest cover | 46% |
| Percentage change in treatment costs from a 46% in turbidity    | 8.7%|
| Average annual chemical treatment cost in sample              | $742,000 |
| Annual increase due to turbidity                               | $65,000  |
| Increase per million gallons treated (based on 19.3 MGD)       | $9.16    |
| Present value over 30 years (3% discount rate)                | $1,300,000 |

Potential Challenges

There are many challenges when studying the relationship of independent variables affecting drinking water treatment costs. One of the biggest challenges is examining data from treatment facilities. Freeman et al. (2008) attempted to get chemical cost data from chemical suppliers in
order to determine the quantities sold, but when they surveyed suppliers that had other plants under their purview, some were unable to present exact cost data because costs were based on the collective enterprise and not isolated for specific plants. Furthermore, different treatment plants vary in treatment methods. Some rely more heavily on chemicals while others rely more on mechanical methods. It will be necessary to account for different treatment methods so that the data are not skewed. One potential approach would be segmenting the study into two (or more) different data sets depending on the treatment method.

Freeman et al. (2008) noted the great variation in sequence of treatment and types of chemicals used by individual plants. Additionally, some previous studies have focused solely on chemical costs, which may ignore a large component of the treatment process. Warziniack et al. (2016) noted that some facilities use filtration prior to disinfection while others do not need to filter. Facilities that do not filter may use more chemicals than facilities that do, which can also be affected by the degree to which the water is filtered prior to chemical treatment or disinfection. Variations in raw water sampling methods can also affect the data. Whether the data for raw water are collected as fixed frequency samples, event based (storms, draught, etc.) samples or random samples fluctuates based on the facility. Despite all methods being acceptable, comparing results from differing methods will increase variability in the results (Freeman 2008).

It has been shown that the effectiveness of forest cover in improving water quality plateaus at around 60% (Ernst 2004). Many watersheds that will be surveyed are likely to occur in areas with well over 60% forest cover, particularly in the Southeast, which can skew the data and may need to be acknowledged by a regional breakdown of the results. In the Warziniack et al. (2016)
study, the average watershed had at least 60% forest cover while 21 out of 37 samples had
greater than 60% forest cover. The raw water source can have a decisive impact on the
treatment process as well. Nationally, there are numerous intake sources including, rivers, lakes
and reservoirs. Lakes and reservoirs are more likely to have less sediment in the raw water taken
in, as many reservoirs are designed to allow sediment to settle to the bottom prior to intake. This
has the potential for significant impact on the turbidity of the raw water when compared to
intakes along rivers, in which the water constantly moves and therefore does not allow the
sediment to settle.

**Case Study: Everett Jordan Lake**

In September of 2007, The North Carolina Department of Natural Resources (DENR) submitted
its compliance plan to the EPA for violations of water quality standards in the Jordan Reservoir
and throughout the Jordan watershed. The reservoir had been found to be in direct violation of
the Clean Water Act. It is under the CWA that the EPA is enforcing its claim that the water of
Jordan Lake is impaired, based on the high levels of storm water pollution or nutrient loading in
the lake. Similar violations of the section 303(d), namely around the Chesapeake Bay area, set a
precedent for enforcing TMDLs and regulations (Clean Water Act 1972).

Excessive amounts of nutrients have led to intense algae blooms that adversely affect the aquatic
ecosystem necessary for a healthy lake. The increasing Blue Green Algae, or Cyanobacteria,
concentration at B. Everett Jordan Reservoir in central North Carolina is causing disruption to
the aquatic ecosystem as well as dangerously polluting the water. The increased algae bloom
taints the water supply by releasing toxins and promoting other harmful bacteria and chemicals
in the water (Paerl and Otten 2013). These blooms are fed by high levels of nitrogen and phosphorous. By lowering the nitrogen and phosphorous levels in the lake, the natural ecological cycle can control the algae once it is back to normal levels.

The high nitrogen and phosphorous also lead to increased chlorophyll-a concentrations. The effects of the algae blooms fed by chlorophyll-a include low levels of dissolved oxygen in the water, blockage of the sun from penetrating the water, and fish kills. Another side-effect can be increased pH. There have been recorded levels of a pH of 9, which is enough to burn mammalian skin and which can have a huge impact in wildlife as well as human recreational use (Paerl and Otten 2013).

The letter of concern from the EPA office in Atlanta, GA, cited North Carolina law 15A NCAC 02B .0211, setting water quality standards for freshwater lakes and reservoirs, as not being met in the Jordan reservoir. The EPA recommended setting Total Maximum Daily Load for various sources, both new and old, on pollution sources within the watershed. The North Carolina DENR responded with ‘B. Everett Jordan Reservoir, North Carolina Phase I Total Maximum Daily Load’ and ‘Jordan Rules.’ This agency policies that define the compliance efforts that have been required by the EPA to address the nutrient loading nitrogen and phosphorous in the reservoir (DENR 2007).

Specifically, this policy targets both point source pollution and nonpoint source pollution. Point source pollution comes from 14 identified facilities that deliver over 99% of the total permitted flow from point sources. Due to this, the TMDL does not allow for any additional point source
facilities, stating that, “All of the available loading was allocated to the existing facilities” (DENR 2007). For nonpoint pollution, the TMDL identifies key areas that will be regulated and/or monitored with the new program. Agricultural operations would need to meet nitrogen and phosphorous output goals as a collective system.

Storm water management is an integral part of this policy as well. The TMDL report targets new development as well as retroactively lowering allowed storm water output from existing developments. These must all meet the National Pollutant Discharge Elimination System (NPDES) Phase II requirements. This requirement was to have been met by December 2007, but has not yet happened.

Another important provision is that all local governments must enforce standards for riparian buffers as outlined as part of the Jordan Rules. These 50-feet wide buffers are thought to serve multiple uses in first slowing the nutrient loading and in turn, helping to reverse the effects of the high phosphorous and nitrogen inputs into the Jordan reservoir. Other measures that were outlined in the TMDL ‘Nonpoint Source Strategy’ are to be taken to help minimize nonpoint pollution of the watershed include nutrient management training for those who apply fertilizers (funded by a tax on fertilizers), a trading program between nonpoint and point sources, and development of programs to reduce nitrogen and phosphorous loading from on-site wastewater (DENR 2007).

In response to the policy set forth by DENR, as forced to do so by the EPA, there was a considerable backlash on multiple fronts to the implementation due primarily to two factors. First
was that total cooperation and implementation would cost up to $1 billion (projected amount when report filed in 2007) (Reeder 2014). Another was opposition from the potential limits these rules would have placed on development and the concern over restrictions in the middle of the economic crisis beginning in 2008.

There have been many local, regional and statewide organizations that have been successful in delaying the implementation of the TMDL. The EPA has been involved in the early stages since 1997 in the analysis of the Jordan Reservoir and in outlining the potential policy changes that would be necessary. The 2007 timeline does not fully represent the extensive effort that the EPA has taken to attempt to get regulation and rules in place. Once the EPA was successful in setting the appropriate regulations and TMDL, there have been numerous votes in the North Carolina legislature to delay the implementation of these new standards.

This culminated in the passing of Senate Bill 515 or the ‘Jordan Lake Water Quality Act’ in the summer of 2013. Groups who have led the opposition to the TMDL policy include local governments, private citizens, development/construction companies and housing groups. In a document released by the Triad Real Estate and Building Industry Coalition (TREBIC), that summarizes the open public comment period, many of the groups are homebuilders (Apartment Association of NC, Builders Associations, NC Home Builders Association, and NC Association of Realtors), civil engineering agencies, and town/city governments totaling over 60 listed groups in the public comments summary. A policy on this scale has major financial impacts on groups such as these. Initial budgeting shows that upwards of $1 billion will need to be spent on measures of compliance with TMDLs (Reeder 2014).
Although some of these groups may agree with the need to clean up the pollution and prevent further degradation to the reservoir, they oppose the regulations purely on financial grounds. Many organizations find, “The existing development rule is unprecedented, burdensome, infeasible…” (TREBIC 2007). This can be understood as the preliminary budget projections given by Tom Reeder, the Director of Division of Water Resources, project that the city of Greensboro would be responsible for nearly $100 million in total (Reeder 2014).

Additionally, the activities of many of the organizations associated with developing land for either housing or other municipal projects will be greatly restricted by these new regulations. Furthermore, if construction and development of new areas is allowed, there is fear that a significantly large amount of money will need to be put into the projects to comply with the new rules. This is a financial burden that is one that adds to the nearly $1 billion that was estimated.

Groups that have come out in support of this policy are namely environmental groups (Sierra Club, Haw River Assembly, NC Conservation Network, NC Wildlife Federation, etc.), some citizen groups such as Chatham Citizens for Effective Communities, and some towns and local governments. Many of the comments pertaining to the support of the policy originate in protecting the ecosystem for the valuable habitat it provides to wildlife as well. Noticeably, the groups that comment in support for the rules omit the discussion of the potentially devastating hazards of continued eutrophication of the lake as it pertains to a drinking water source for over 300,000 people at over 100 million gallons of water per day (DENR 2007).
The NC Department of Transportation (NCDOT) has its own section in the comments summary document. The NCDOT does not take sides on the policy but rather expresses concern over some of the implications and support for other areas. One area of particular concern for the NCDOT is the burden placed on it to be largely financially responsible for nutrient loads. It claims that the analysis indicates that “NCDOT is responsible for 2% of the nutrient loading into the lake” yet they are charged with an “estimated $600 million cost of compliance” (TREBIC 2007). On top of this, “the Jordan rulemaking process did not include an accurate technical analysis of DOT nutrient loads, nor did the process allow NCDOT participation in such an analysis.” The claim therefore is that the number applied to NCDOT is arbitrary.

The NCDOT supports the rules and actually suggests that they be taken a step further when it comes to existing riparian buffers. There is a provision in the rules that allows for storm water treatment BMPs. It is suggested that the rules prohibit the construction of storm water treatment BMPs in zones 1 and 2 (the entire buffer) in order to avoid further destruction of the buffer as well as avoid unaccounted-for costs of maintaining these in the event of a flood, a cost that is implied would fall on the NCDOT.

As discussed above, the implementation of the TMDL policy has been delayed numerous times by the North Carolina state legislature. The reason for this has been justified by stating the need to finding cheaper, yet still effective alternatives to those extensive and costly solutions put forth. The financial burden to not only stop, but then reverse the nutrient load in the Jordan reservoir has been shown to be the motivating factor behind the delay.
One of the results of Senate Bill 515 was an experimental fix to explore the possibility of addressing the pollution and subsequent algae bloom in the lake by fixing it at the lake level and not the surrounding watershed. The proposed, implemented and failed alternative involved a device made by Medora Corp. known as a SolarBee. The SolarBee is a solar powered aerator and mixing device that churns the water to help mix the top and bottom levels of the water and minimize algae blooms by reducing chlorophyll-a. The thought behind this alternative is that it costs a fraction of the estimated budget to implement the Jordan rules (approximately $1.4 million allocated by state legislature).

In 2016 the SolarBee experiment ended with some controversy. A report was completed and submitted by a 40 year veteran of the state Division of Water Quality and the chairman of the Environmental Management Commission’s Water Quality Committee but was quickly redacted and a new version submitted two days later. The initial report claimed that the SolarBee was ineffective. The employee was demoted and a new report filed to the state legislature. The original report stated that the devices had not achieved the intended results and that their continued use would be of no value. The original report suggested, “A comprehensive, adaptive, and science-based approach to reducing nutrient inputs to the watershed remains the most viable option for recovering these waterbodies from impairment” (DEQ 2016). Furthermore,

“Technologies that do not result in actual nutrient reductions may simply shift nutrient problems or impairments to downstream waters... Finally, efficacy of in situ and lakeside technologies are generally predicated on first reducing upstream nutrient inputs to the maximum extent practicable. Nutrient management approaches that rely heavily on these technologies instead of reducing nutrient inputs to the watershed would appear to disregard the fundamental premise of the Clean
Water Act, which seeks reduction of pollutants to waters of the United States. The U.S EPA notes in their guidance ‘...it is clear that nearly all (in-lake) restoration procedures will be quickly overwhelmed by continued high incomes of silt, organic matter and nutrients. Protection and watershed management are therefore paramount to restoration’ (U.S. Environmental Protection Agency, 1990).

The entirety of this section was removed and the final report suggested further study of the devices through 2018. The original report is in line with the approach that this study is focusing on and would indicate that the co-benefits of aquatic biodiversity and drinking water benefits is exceptionally exemplified in this instance.

Many of the opponents to the TMDL backed the SolarBee program with the hopes that it will improve the water quality enough so that the costly implementation will not be necessary. A policy of this nature has explicitly stated objectives that need to be obtained in order to determine if the policy has been successful. The TMDL report clearly identifies current (baseline) numbers for nitrogen and phosphorous in different parts of the lake. The report summarizes the necessary reductions from each area of the lake by using the following equation and the baseline numbers:

\[ \text{TMDL} = \sum \text{WLA} + \sum \text{LA} + \text{MOS} \]

Where WLA is wasteload allocations for point sources, LA is load allocations for nonpoint sources and MOS is margin of safety. The following chart (DENR 2007) summarizes the required change in nutrient levels to meet compliance:
As long as the TMDL are met, the objectives of the EPA to rectify the status of an impaired water have been met. Meeting these objectives should mean increased health of the lake which can be seen by decreased algae blooms, a lack of unpleasant odor and a number of improved quantitative data on oxygen levels, fish stocks and water clarity.

The Cary/Apex Water Treatment Plant serves the towns of Cary and Apex, NC and draws its entire municipal water supply from Jordan Lake. The Jordan Lake pollution issues have exhibited a direct effect on the municipal water supply and the citizens of these towns. There has been a 26.8% and 37.4% increase in the average cost of consumption (for 10,000 gallons per month) in Cary and Apex respectively from 2006 to 2016 (NC League of Municipalities 2016). Likewise, town budget data for the Cary/Apex Water Treatment Plant has shown a 42% increase in chemical costs from 2009 to 2016 (Cary Budget Reports). Tables 2 & 3 show the change in cost data for these towns and treatment plant.
The Jordan Lake case study exemplifies the impacts of nonpoint source pollution within a watershed on water quality. Many of the pollutants of Jordan Lake come from agricultural, development and other human-caused pollutants. While there are some point source polluters, these are minimal and relatively easy to regulate. The original proposed rules placed emphasis on TMDLs and other BMPs such as forested buffers to combat pollutants throughout the watershed. Many experts have assessed the Jordan Lake case and have concluded that whole watershed approaches are necessary to stop and reverse the detrimental effects on Jordan Lake. Jordan Lake demonstrates many of the negative characteristics that poor water quality can have on aquatic ecosystems and biodiversity as well has economic impacts such as clean-up costs and treatment costs to the consumer. The intense alga blooms have decimated many aquatic plant
and food species which have resulted in decreased populations of certain fish species. The algae blooms have also caused lower oxygen levels in the water which has caused a widespread issue throughout the food web.

One of the main goals of this research is to lay the foundation for how poor water quality and the resulting effect on aquatic biodiversity has implications for various stakeholders. One of the most integral groups of stakeholders is the general public and consumers of municipal drinking water. The towns of Cary and Apex have shown significant increases in both chemical costs to treat the water as well as the average water bill in the area serviced by Jordan Lake. This strong financial relationship may be the needed link to increase public awareness and interest in measures to preserve source water quality via watershed conservation methods. The New York watershed and Jordan Lake case started on approximately the same timetable in the mid 1990’s. New York has taken an aggressive conservation approach which has resulted in hundreds of millions of dollars saved from reduced treatment and operation costs as well as one of the cleanest surface drinking water supplies in the country. Conversely, Jordan Lake has been marred in political strife and controversy, resulting in a lack of progress in almost twenty years.

**Conclusions**

There is a definitive positive relationship between forest cover throughout a watershed and the water quality of water bodies within that watershed. There is also a demonstrated positive relationship between water quality and aquatic biodiversity. The five studies examined in this paper have shown varying levels of correspondence between forest cover, water quality and treatment costs based on turbidity and total organic carbon in the incoming source water.
The survey has been designed with an effort to focus on gathering data on treatment methods, chemical use and cost as well as information on irregular effects on the treatment process. It is expected that chemical and energy cost data will have significant variation across different regions due to differences in landscape features. Although gathering data from as many sources across the country could be ideal, it may not be feasible to gather representative data from all across the country. Given that the effectiveness of forest cover in a watershed plateaus around 60%, targeting the survey to treatment plants that occur in areas that fall above 60% forest cover, 50-59% forest cover, 40-49% forest cover, etc. as determined by GIS analysis. This should help give data that can be assessed based on differing tiers of forest cover. This will help prevent too many data points that have above 60% forest cover and data that does not adequately represent the effects on treatment costs given different levels of forest cover.

As was mentioned in discussing the survey, accounting for regional differences will be integral to the success of this study. The landscape features of the Southeast are different from those of the Southwest, Midwest, Pacific Northwest, etc. Much of the southeast is known for having large amounts of forested land and sparse populations, other than a few cities. The northeast is more heavily populated but has forest cover as well. The southwest and midwest are characterized by far less forest land and desert, shrub land a grass land. These inherent differences in land cover will significantly alter the treatment methods and chemicals used. These regions also have vastly different water characteristics. For example, the southeast generally has an abundance of rainfall and water while the southwest is much more arid and the water supply is often stressed by drought. Because of these variations, it is recommended to
conduct the study on water treatment cost regionally. This will allow for the most direct comparison of the data. The data can then be combined and analyzed on a national scale to understand the national trends and implications. Conducting a national analysis from the outset will more than likely result in skewed results or results that are not statistically significant.

In addition to current land cover, change in land cover over time should be analyzed. Through GIS data, it will be important to identify areas in which survey respondents occur where land cover change has taken place over a set period of time. The changes that are most likely to have an effect on the data are from forested or agriculture to developed or converting agriculture lands to forest. Areas that have experienced change from forest to developed area might see significant changes in treatment costs and could be another method of analysis. Further study could be done to gather historic water utility bills and chemical cost data (when available) that corresponds to the NLCD sets.

Another important assessment that should be done in addition to overall changes in land cover is forest fragmentation. Logging activities and other forms of development can lead to tracts of land that have a significant (over 60%) level of forest cover but the amount of continuous forest has decreased significantly (Riitters et al. 2002). These fragmented forests can be far less effective at filtering water as it moves through a watershed and may even lead to increased amounts of sediment and nutrients in storm water events (Riitters 2002). This can be analyzed vis GIS and the effects of such fragmentation can be studied further.
Although monetary measurement is easier to understand and may be more symbolic to stakeholders, chemical quantities will provide the most accurate representation across a region or even nationally. Quantities of the most commonly used chemicals should have significant overlap across the study area and be the most accurate way to assess variations in the treatment process in various conditions. It is recommended that the primary analysis of water treatment cost be done in chemical quantities and not dollar amount. Also, operation and management costs should not be considered due the high variability in treatment methods, energy costs and labor rates.

In completing this study, if there is a strong positive relationship between forest cover, water quality and drinking water treatment costs, this relationship needs to exhibit benefits to the general public and consumers of municipal water supplies. If this benefit can demonstrated, stakeholder buy-in will influence policies and practices that can have a perpetual positive impact on watershed health and water quality.
References


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Appendix 1

Proposed Water Quality Treatment Survey Questions

1. Please describe briefly (2-3 sentences) your drinking water treatment methods. (Please attach flow chart if possible).

2. What is the average annual daily volume of finished water produced?

3. Is your source water drawn from a river, reservoir or lake, or both? If both, what proportion of source water comes from each, and are the different sources blended before treatment?

4. Are the chemicals used/treatment method determined by plant manager, regional manager or at the state level?

5. How do you monitor incoming water quality?

6. What are the primary pollutants you need to treat?

7. What are the average annual chemical quantities for the most commonly used and expensive chemicals in treatment?

8. What are your average annual costs for those chemicals to produce finished water? How do these costs change with changes in source water quality?

9. What are your total annual chemical costs to treat water?
10. What are your average annual operation and maintenance costs for residual management (please do not include capital costs)? How do these costs change with changes in source water quality?

11. How do storm water events affect the treatment process? Are more chemicals used, and if so, which?

12. Has there been a change in chemical use/treatment costs over the past 10 years? Please explain and indicate the degree to which it has changed.

13. Where do the treatment chemicals come from? Is pricing of these controlled or variable?