Economic Analysis of an Algal Turf Scrubber and Anaerobic Digester System for Bioremediation of Eutrophic Waters

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Economic Analysis of an Algal Turf Scrubber and Anaerobic Digester System for Bioremediation of Eutrophic Waters

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Abstract

Aquatic environments provide economic value in the form of ecosystem services (ES), such as jobs and food through fisheries, eco-tourism, flood buffering against storms, and nutrient capture (Turner et al., 2000). Globally, marine dead zones have doubled since the 1960s due to excessive nitrogen (N) and phosphorus (P) introduced by synthetic agricultural fertilizers, the burning of fossil fuels, and human and animal waste (Diaz & Rosenberg, 2008; Rabotyagov et al., 2014). There is a range of bioremediation or cleanup methods that can alleviate excess nutrients in waterways. The private firm Hydromentia LLC practices bioremediation through Algal Turf Scrubber (ATS) technology at a variety of sites to capture nutrients, sequester carbon (C), and produce algal biomass from point and non-point source nutrient dense waters. Harvested algal biomass has additional value as crop fertilizer, livestock feed, or as an input to biofuel generation in anaerobic digestion (AD) (Hydromentia, n.d.c). Studies looking at ATS systems' costs and benefits through a comprehensive economic lens are currently lacking and there is opportunity to incorporate more nutrient trading and crediting into analyses (Pizarro et al., 2006; Higgins & Kendall, 2012). The objectives of my study were to evaluate the costs and benefits associated with an ATS-AD system, quantify the ES values an ATS-AD system provides, and provide economic justification to support policy makers in developing additional funding for a wider variety of global ATS-AD projects.

My research included both financial and economic appraisals for the Fall River ATS and ATS-AD site, located in Durham, North Carolina, USA, with a 20-year

timeframe and based on Hydromentia's 2017 Fall River Pilot Report. Hydromentia provided data on financial costs of construction, operations and energy use, and benefits of nutrients captured and dried algal biomass in pounds per year. First, I conducted two comparable financial appraisals, which include the ATS site with and without an AD component used to reduce the energy costs to operate the system. Second, I conducted three economic cost-benefit analyses (CBA), based on different scenarios of household populations, to assess the following questions:

- Do the bioremediation efforts of the ATS-AD system outweigh the costs?
- How much biofuel can be generated from algal biomass from the proposed site?

My research aimed to answer these questions and hypothesized that the economic CBA values are greater than that of both financial CBAs. Including the benefits of avoided eutrophication, nutrient capture, and algal biomass converted to biofuel in an economic CBA provided a more accurate representation than a traditional financial assessment from the business' point of view.

The results of both financial analyses showed negative end values, suggesting the benefits of ATS technology do not outweigh the costs from a traditional business perspective even when AD is incorporated, although the AD component resulted in a smaller negative value. My economic analyses resulted in positive values for all three percentages (75%, 50% and 25%) that ES would be transferred to the Falls Lake Watershed case study population, suggesting that the ES benefits ATS systems provide outweigh the costs of the project. The results support my stated hypotheses and could influence policy makers and governments to further incentivize this bioremediation technology through grants, tax breaks, and subsidies.

Acknowledgments

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Definition of Terms

ASP	Aquatic Species Program
AD	Anaerobic Digester
ATS	Algal Turf Scrubber
ATS-AD	Algal Turf Scrubber Anaerobic Digester
BTU	British Thermal Unit
CBA	Cost-Benefit Analysis
CE	Choice Experiment
CICES	Common International Classification of Ecosystem Services
CSL	Conventional Slurry Lagoon
CV	Contingent Valuation
DALYs	Disability-Adjusted Life Years
DOE	Department of Energy
Dry-g/m^2-day	dry grams per square meters per day
ES	Ecosystem Service(s)
FLW	Falls Lake Watershed
GGE	Gasoline Gallon Equivalent
HABs	Harmful Algal Blooms
HTL	Hydrothermal Liquefaction
kW	kilowatt
kWh	kilowatt-hour

LCA	Life Cycle Assessment
MGD	Million Gallons per Day
MMBTU	Metric Million British Thermal Unit
NRE	Neuse River Estuary
OPEC	Organization of Petroleum Exporting Countries
OWTS/OWT	On-Site Wastewater Treatment Systems (Septic Systems)
PAF	Potentially Affected Fraction
RP	Revealed Preference
SCF	Standard Cubic Foot
SCM	Standard Cubic Meter
SP	Stated Preference
TEV	Total Economic Value
TMDL	Total Maximum Daily Load
USD	U.S. Dollars
VS	Volatile Solids
WTA	Willingness to Accept
WTP	Willingness to Pay

Chapter I

Introduction

The United Nations (2022) expects global human population to increase to 9.8 billion by 2050. Demands for resources, such as food, fuel, and water, will also continue to increase (Gilbert, 2020). By burning fossil fuels, humans have more than doubled the amount of atmospheric nitrogen (N) since the Industrial Revolution (Holtgrieve et al., 2011). The total use of the agricultural fertilizers N and phosphorus (P) has risen about 9-fold since 1970, and their continued use is predicted to increase with the need to produce enough food for projected rising populations (Gilbert, 2020). Figure 1 depicts the worldwide use of N and P fertilizers per country (UNFAO, 2022). Globally, agricultural crops take up less than half of applied N fertilizers (Ritchie, 2021). In Europe, estimates suggest livestock effluent, or waste, contributes about 73% of agricultural water N and P pollution (Leip et al., 2015). Large concentrations of residential septic systems in rural and suburban areas can also contribute to high concentrations of N and P on the watershed scale (Hoghooghi et al., 2016; Humphrey et al., 2020). The effect these fertilizers have on our global waters is impossible to ignore. Excessive N and P from agricultural and septic sources end up in the planet's waterbodies via nutrient runoff; precipitation events carry excess nutrients into groundwater and nearby streams, rivers, and coastal waters. Increased atmospheric N from fossil fuel use offsets the planet's natural N cycle and contributes to an excess of N in our rivers, lakes, and oceans (Holtgrieve et al., 2011).

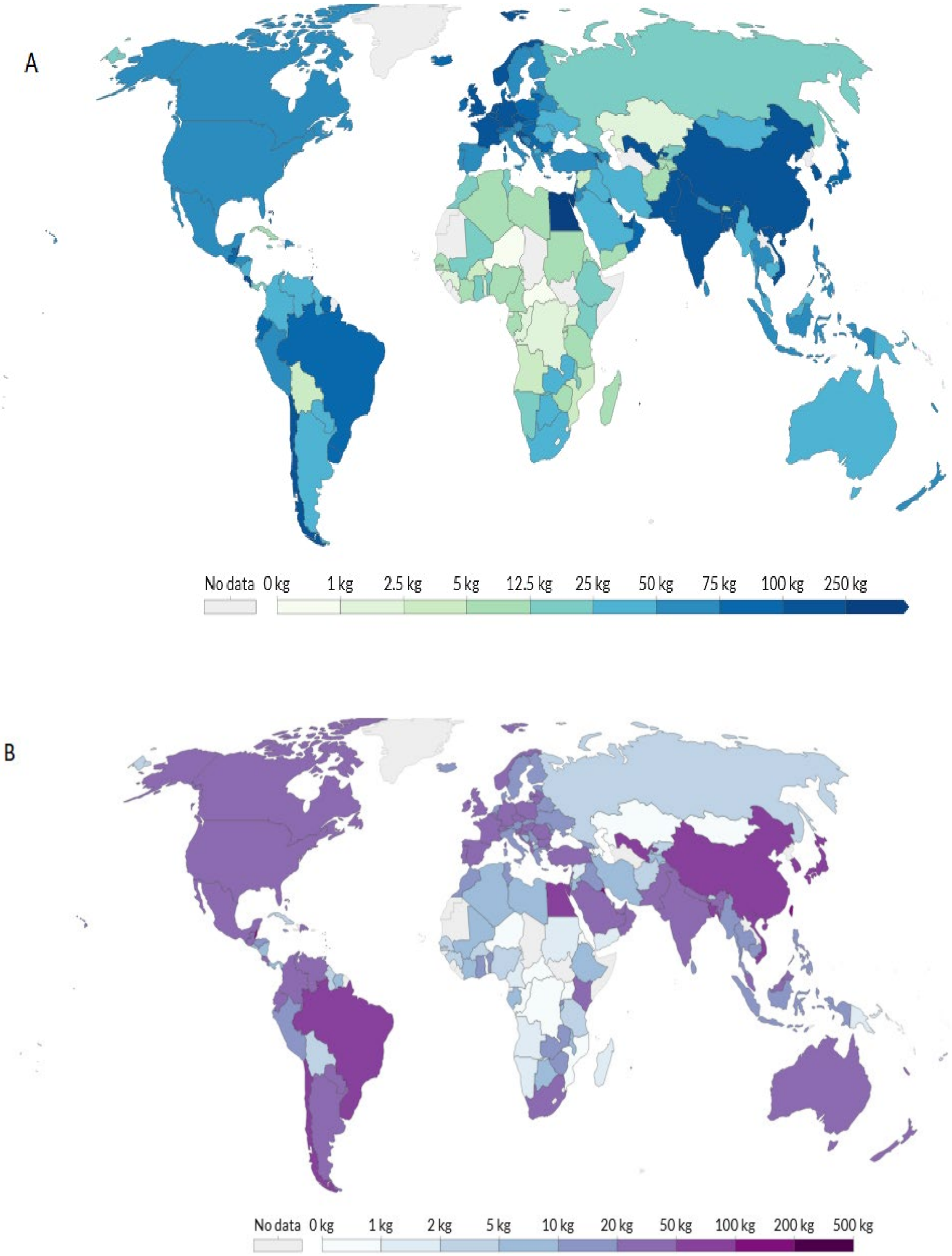


Figure 1. Nitrogen and phosphorus use per country.

Nitrogen (A) and phosphorus (B) use in kilograms of total nutrients per hectare of cropland per country in the year 2019 (UN FAO, 2022).

Globally, extremely eutrophic zones in marine environments have approximately doubled each decade since the 1960s (Diaz & Rosenberg, 2008). Most of the coastal nutrient input has occurred since the 1950s, with hypoxic zones doubling globally since the 1960s, and input continuing to increase (van Beusekom, 2019a). Eutrophication has a positive feedback loop associated with greenhouse gas emissions (Li et al, 2021). The greenhouse gas effect can exacerbate eutrophication by increasing rainfall, which introduces more nutrients into waterbodies, raising temperatures, increasing sediment mineralization, and increasing primary producers (Li et al, 2021). Eutrophication also increases the greenhouse gas effect by producing more algal biomass that turns into harmful algal blooms (HABs), releasing more greenhouse gases, and contributing to climate change (Li et al, 2021). Eutrophication of freshwater ecosystems via anthropogenic sources also reduces overall biodiversity, leading to more homogenous communities that are less resilient to a changing climate, decreases ecosystem efficiency, and are more vulnerable to the dominance of generalist consumers (Cook et al., 2018).

The economic costs associated with eutrophication and dead zones should include the decline in the value of ecosystem services (ES) provided by marine and freshwater environments. Lefcheck et al. (2018) estimated that up to trillions of dollars' worth of ES, such as habitat for commercially valued species, shoreline protection, and C storage, have been lost to nutrient and sediment runoff. Among many harmful effects, eutrophication increases the number of fish kills, reduces the health of coral reefs, reduces species diversity, reduces commercial and recreational fish harvests, creates foul taste and odor in drinking water, increases toxic blooms that are a hazard to many life forms, and reduces the aesthetic value of a body of water (Smith & Schindler, 2009).

Alleviating the negative consequences of eutrophication can lead to economic benefits for society. Keeping excess N and P out of aquatic environments from anthropogenic sources in the first place is the best-case scenario; the effects and benefits of stricter regulation on fertilizer use, incentives for more sustainable agricultural practices, and consumer behavioral shifts towards less nutrient use are too slow to materialize (Kassinger, 2019).

The algal turf scrubber (ATS) system is a bioremediation technology that provides diverse benefits, such as capturing nutrients, sequestering C, and avoiding or reversing the impacts of eutrophication on aquatic environments (Adey et al., 2013). ATS systems consist of long, sloped flow-ways, where eutrophic water is pumped at the top. Algae grows on the surface of the flow-way as nutrients are removed from eutrophic water (Hydromentia, n.d.b). ATS systems are costly to manufacture, requiring the construction of plants using concrete, plastics, and other building materials; costly to operate, requiring energy to pump water over raceways; and costly to maintain, requiring energy to mechanically scrape algae production over time (Higgins & Kendall, 2012). A complete economic analysis would place value on avoided environmental costs, compared to ATS system infrastructural costs, and provide a more accurate depiction of this technology's value in bioremediation.

An ATS system in conjunction with an anaerobic digester (ATS-AD) showed a reduction in the environmental footprint of a dairy wastewater treatment process by reducing eutrophication impacts and providing biogas to generate electricity (Higgins & Kendal, 2012). To date, research is lacking on the economic cost-benefit ratio (CBR) of ATS-AD systems in aquatic environments from a societal viewpoint, including the benefits of avoiding dead zones, restoring ecosystems and their valuable services, and creating marketable byproducts. For example, Adey et al. (2013) estimated algal biogas exceeded the production of biogas made from corn and biodiesel from soybeans per unit area by several times, while certain efficient species of algae produced hundreds of times more oil than corn (Faried et al., 2017) and soybeans (Koyande et al., 2019). Further research on economic CBAs of ATS-AD systems can answer questions of whether the societal and environmental benefits of this technology outweigh its infrastructural costs and whether the existing technology can be made more accessible and affordable to a wider variety of applications in the future.

Research Significance and Objectives

The significance of my research was to expand knowledge of the productivity and profitability of ATS-AD systems to a wider range of uses and project sizes. Currently, ATS systems are large-scale and require significant funding. If more research can show this technology is sustainable and is economically profitable, especially in combination with an AD component, engineering can make the technology more accessible in a wider variety of settings. If costs are reduced and economic gains realized, more probable candidates, such as a local livestock farmer, small municipality, or aquaculture plant, could incorporate ATS-AD systems into their infrastructures.

The goals of my research on ATS-AD systems were to quantify the benefits of nutrients captured and the avoided impacts of eutrophication in an aquatic environment. The calculated environmental benefits were compared to the economic costs of the manufacturing, operation, and maintenance of such systems that use electricity, water pumps, and mechanical and manual labor. Do the bioremediation values exceed the costs in constructing ATS systems? And what is the scope of additional benefits or profits of the technology, such as biogas generation, to cover the operational costs?

Therefore, my objectives were:

- To evaluate a broader scope of impacts and benefits associated with an ATS-AD system
- To quantify/estimate the economic value of ES an ATS-AD system can restore or protect in a specific habitat
- To provide technical and economic justification to support policy makers and governments to develop additional funding and incentives for a wider variety of global ATS-AD projects

Background

Swanson and Johnston (1999) stated that economic development policies in the past have focused on resources such as financial and built capital and labor to produce goods and services because they are seen as more productive and having a higher return compared to natural capital. Presently, human activities are putting more evident pressure on natural resources, such as freshwater, and a radical behavioral shift in how we manage those resources is needed (Sivapalan et al., 2014). The ideology of placing value on natural resources and entire ecosystems is debatable. Some economists say nature only has value due to humans' assignment of it (Harris, 2017). Conversely, ecological economists think nature has inherent value, or value despite its economic value and more for its ethical value (Harris, 2017). This perspective assumes that water resources are important inputs to a range of economic sectors, such as agriculture, industry, tourism, and individual households (UNEP, 2005). To create effective policies, it is crucial to determine the full value of environmental resources and to include those values in private and public decision-making activities (Birol et al., 2006).

Holting et al. defined ecosystem functions as “properties and processes of an ecosystem, such as ecosystem matter and energy cycles, that have a specific function within the ecosystem and are essential for the capacity to provide goods and services” (2019, p. 227). The wellbeing of an ecosystem is associated with ecosystem functions, whereas ES pertain more to the wellbeing of humans (Brockerhoff et al., 2017). Turner et al. (2000) stated that drawing the connection from ecosystem functions to ecosystem uses, goods, and services is necessary for society to place economic value on these functions. An important aspect of ES is their anthropogenic economic value, even if humans never encounter that ecosystem or ES (Turner et al., 2000): “[ES] are the benefits people obtain from ecosystems” (Millennium Ecosystem Assessment, 2003, p. 53).

For example, a coastal housing development may not be physically adjacent to a wetland, and its inhabitants may never visit the wetland; nonetheless, the wetland provides an ES of flood protection, protecting the development from flooding when a large storm, such as a hurricane, raises coastal flood waters.

The Common International Classification of Ecosystem Services (CICES) resources and guidance documents aim to unite researchers, economists, policymakers, and others in valuing ES in economic environmental assessments (Haines-Young & Potschin, 2012). CICES provides a hierarchical structure of provisioning, regulating and maintenance, and cultural sections, which are all further allocated into biotic and abiotic categorizations (Haines-Young & Potschin, 2012). Examples of biotic and abiotic provisioning services include crops grown for energy production or nutrition, or freshwater used as an energy source, respectively (Haines-Young & Potschin, 2012). Examples of biotic and abiotic regulation and maintenance services are vegetation that prevents soil erosion, and a barrier island that protects from storms, respectively (Haines-Young & Potschin, 2012). Examples of biotic and abiotic cultural services include using an outdoor location for recreation or sport, such as rock climbing, and mountain peaks we value as important symbols (Haines-Young & Potschin, 2012). Further down the CICES hierarchical structure are groups, such as cultivated and wild plants; classes, such as cultivated plants for nutrition and materials; and class types, such as cereals and vegetables. (Haines-Young & Potschin, 2012). In the above example of wetlands, flood protection is a biotic regulation and maintenance service by CICES classification (Haines-Young & Potschin, 2012).

More scholars are recognizing that ecosystems provide biogeochemical and biophysical services that influence the climate and day to day weather conditions, and thereby have an impact on humans (Hungate & Hampton, 2012). A biotic regulation and maintenance service that CICES categorizes as a biochemical service is a tree absorbing carbon dioxide over time and mitigating a changing climate (Haines-Young & Potschin, 2012; Hungate & Hampton, 2012). A biophysical and biotic regulation and maintenance service is a tree providing evaporative cooling effects and an absorptive albedo effect in a warm environment (Haines-Young & Potschin, 2012; Hungate & Hampton, 2012). Some C markets already value certain biogeochemical ES, such as those affecting greenhouse gas emissions, through C trading programs (Hungate & Hampton, 2012). However, our capitalistic societies have a long way to go in understanding the full value of benefits to the environment and our species' longevity on this planet. Without applying comprehensive economic value to ES, we will continue to over-extract resources and further degrade habitats (Birol et al., 2006).

Since its patenting in the 1980s, researchers have shown that algal turf scrubber (ATS) systems provide a variety of benefits to society and the environment that represent ES. ATS systems directly remove excessive nutrients like N, P, and C from eutrophic water (Adey et al., 2013; Mulbry et al., 2008b; Mulbry et al., 2010; Pizarro et al., 2006; Sindelar et al., 2015; Torres-Franco et al., 2020) and, thus, improve water quality (Guleri et al., 2020); rebalance decreased oxygen levels in eutrophic waters (Sutherland et al., 2020); create a value-added byproduct of algal biomass that can be used for livestock feed (Catone et al., 2021), crop fertilizer (Kebede-Westhead et al., 2004; Mulbry et al., 2005; Reinecke et al., 2023), omega-3 fatty acids for nutraceuticals (Al-Jabri et al., 2021), and electricity generated from biogas (Adey et al., 2013; Mulbry et al., 2008a; Witarsa et al., 2020). The above ES are directly linked to ATS systems as the technology intends to remediate their associated environmental concerns.

These ES provide nonmarket values as markets do not work to capture their economic value (Raheem et al., 2009). Other researchers have analyzed nonmarket ES to capture their value (Adey et al., 2013; Higgins & Kendall, 2012). ATS-AD systems may provide more nonmarket values through bioremediation that are not contained in current ATS-AD research. ATS-AD systems provide many more services, including but not limited to, controlling the chemical quality of freshwater, cultivating plants in freshwater that are used as an energy source, decomposing and filtering wastes, reducing smells, regulating our global climate, and using the environment for recreation or destress

(Haines-Young & Potschin, 2012). The valuation of ES typically involves economic cost-benefit analysis (CBA).

Cost-Benefit Analysis

The goal of CBAs is to calculate the net present value (NPV), or the difference between the sum of the discounted benefits and the sum of the discounted costs, discounted over time or the length of the project (Talberth et al., 2007). Discounting enables the researcher to reduce bias for the predicted value of future money by assigning a lower weight on a future value than the present value (Pearce et al., 2006). The NPV of a financial analysis will portray the best scenario from the business' point of view, while the NPV of an economic analysis will portray the best scenario from an ecosystem or environmental viewpoint (de Nooij, 2011). When multiple CBAs or projects are compared, the main rule is to choose the project with the highest NPV, as this will ensure the highest net social benefits (Talberth et al., 2007; Boardman et al., 2018). I describe the components of and methods to conduct financial and economic CBAs in the following sections.

Financial Versus Economic Cost-Benefit Analyses

In the early 1990s, the World Bank's *World Development Report* advocated for polices that protect the natural environment, while also benefiting nations' economies (Dixon et al., 1994). Shortly after, a financial group, The Global Environment Facility, started funding global projects that would normally not be economically feasible from a more narrow, financial viewpoint (Dixon et al., 1994). Economic cost-benefit analysts have been valuing the environmental costs and benefits of projects rather than valuing direct costs and benefits to the individual, business, or organization (Dixon et al., 1994).

Economic CBAs differ from financial ones in that the short- and long-term environmental health and viability of ES are included only in economic analyses. For example, an economic CBA would consider externalities, such as the impact on human health and species' biodiversity, whereas a strictly financial one might not (de Nooij, 2011). Economic CBAs incorporate both actual and predicted prices, or market and shadow prices, respectively (Guler & Boloş, 2021). CBAs conducted before the beginning of the project (*ex-ante*) determine if the project is worth the distribution of resources; CBAs conducted during the project's timeline (*in medias res*) show if the project should continue; and CBAs conducted after the conclusion of the project (*ex-post*) can show whether assumptions were true or false (Talberth et al., 2007).

Total economic value (TEV) is a concept which represents the additive value of market and non-market values of an environmental resource (N. Raheem, personal communication, June 15, 2023). TEV is further delineated into use and non-use values (van Beusekom, 2019b). Use and non-use values are separated in CBAs and are defined in Table 1 (Birol et al., 2006; van Beusekom, 2019b). Although it is difficult to accurately quantify ES' TEV or losses due to uncertainty, decision makers need to know the best estimation of the consequences associated with ES loss to further protect biodiversity, which provides resiliency to ecosystems (Pearce et al., 2006).

Table 1. Use and non-use values of CBA.

Type	Definition	Examples	Wetland examples
Use values	Values individuals place on the use of environmental resources	Direct uses	Drinking water
		Indirect uses	Flood control
Non-use values	Values places on environmental resources not used by individuals	Existence, bequest, altruism values	
		Value for future generations and overall environmental good	

(Birol et al., 2006; van Beusekom, 2019b).

Conducting Economic CBAs

The textbook, *Cost-Benefit Analysis: Concepts and Practice*, provides detailed information on the theory and practice of CBA (Boardman et al., 2018). Through a basic highway project example, Boardman et al. (2018) proposed a standard format for conducting CBAs:

1. Determine whose benefits and costs are being assessed.
2. Determine the collection of alternative projects.
3. Catalogue, predict, and find value for impacts.
4. Discount or combine benefits and costs to determine present values.
5. Assess uncertainty and include a sensitivity analysis.

Determine whose benefits and costs are being assessed. Any project in society produces positive or negative impacts on a variety of stakeholders (Talberth et al., 2007). Many stakeholders have not been included in CBA, such as individuals living far away from a project but are still impacted by the project's externalities or still place value on the project's benefit (Talberth et al., 2007). To summarize Brent (1996), users can be divided into losers and gainers. A positive change in income represents the benefits of a gainer and the negative change in income represents the costs to losers. Costs are the compensation that would be required to make losers' income positive again. To determine who the stakeholders are in a CBA, it is important to ask who is being made more, or less happy in a specific population (Hanley & Spash, 1993). Determining who is receiving benefits and acquiring costs of a project can also require analyzing how benefits and cost are unequally distributed throughout different sectors of society, liabilities towards impacts on property rights, and clearly stated policy explaining the project's aim (Talberth et al., 2007).

Determine alternative projects. CBA is strengthened by providing multiple scenarios, variety in scale, different options of methodology, and providing different levels of government involvement (Talberth et al., 2007). There may be multiple ways to achieve a project's aim through additional components of the project, the provision of taxes, and additional infrastructure (Talberth et al., 2007).

Catalogue, Predict, and Find Value for Impacts

Impacts in a CBA refer to both positive inputs, or benefits, and negative outputs, or costs. Impacts must be identified, labeled as benefits or costs, and assigned measurement indicators. Measurement indicators refer to the unit of measurement of the impact, such as number of lives saved, or dollar value of gasoline spent. CBA analysts reveal how impacts enhance or degrade some people's situations. For example, are skills improved, incomes increased, or education improved (Boardman et al., 2018)?

Direct and indirect use. It is important to identify stakeholders connected to ecosystem functions and values in a variety of ways (Turner et al., 2000). Turner et al. (2000) proposed a classification of stakeholders in wetland valuation: direct extensive users, direct intensive users, direct exploiters, agricultural producers, water abstractors, human settlements close to wetlands, indirect users, nature conservation and amenity groups, and nonusers. Direct extensive and intensive users directly use ecosystem goods in modest and severe ways, respectively. Direct exploiters overharvest goods in an ecosystem to the point of damaging its health. Human settlements close to wetlands benefit from its ES, but paradoxically create more of a demand to protect water as a valuable natural resource (Turner et al., 2000).

Types of direct use values include extractive and non-extractive values, which include commercial and subsistence fishing and catch-and-release fishing and wildlife boat tours, respectively (Raheem et al., 2009). Indirect users benefit from the ecosystem's indirect services, such as flood control in the wetland example. Other indirect use values include watching nature documentaries (Raheem et al., 2009). Nature conservation and amenity groups advocate for aesthetic and recreational values of the ecosystem. Finally, non-users are stakeholders that recognize the intrinsic value of an ecosystem without being geographically close to one (Turner et al., 2000). Placing a value on the present or future existence of ES, especially for future generations in the latter scenario, are examples of non-use values (Raheem et al., 2009).

Quantifying the monetary value of ES is not always straightforward but it is important for policy makers to understand and attempt to capture their TEV to efficiently allocate and protect limited resources (Birol et al., 2006). Failing to study and estimate existence values because they are impossible to value would designate use values as the only true economic resources (Bishop & Welsh, 1992).

ES valuation methods. There are multiple valuation methods for ES and each method will result in different values. Assessing an individual's or a household's Willingness to Pay (WTP) or Willingness to Accept (WTA) captures the value placed on a good or service, or what it would take to compensate in place of that good or service, respectively (Raheem et al., 2009). The WTP method results in more financially conservative values due to loss aversion and budget constraint and is often the preferred method (Whittington et al., 2017). Using WTA generally results in a higher value, as it captures what compensation would suffice for the loss of an ES.

Two main categories of gathering the public's value on ES include stated preference (SP) and revealed preference (RP). SP includes the use of surveys that ask questions, such as, what would you pay to have access to a particular ES like clean drinking water? SP surveys typically present hypothetical situations. SP can estimate non-use values, such as existence value; however, a disadvantage to SP is the impact the experimental design of surveys can have on participants' responses. RP looks at the public's actual payment behavior on goods or services that are connected to their use of non-market goods, or ES. RP has an advantage in this way, as real valuation data is used; however, RP can only represent use values and not non-use values, such as existence value (Raheem et al., 2009).

Two SP methods are contingent valuation (CV) and choice experiment (CE) methods. CV uses survey questions to elicit participants' values on their WTP or WTA changes in environmental quality. CV survey questions can include asking what the participant or their household would be willing to pay to avoid a reduction in water quality (Peterson, 2003). CV aims to capture the total value of use and non-use resources, for improving river water quality in a CBA, for example (Wattage et al., 2000). CE are surveys that aim to elicit the participants' value of an ES, as well, but by providing a range of scenarios or attributes to the participant. For example, a CE could ask a participant if they preferred an environmental cleanup method valued at $15 per household that leads to a 20% increase in salt marsh restoration with no effect on bird population, or if they preferred a 50% increase in salt march restoration and an increase in bird population but cost $40 per household (Raheem et al., 2009).

Hedonic pricing and travel costs are two RP methods. Hedonic pricing usually pertains to housing values based on the assumption that there is a relationship between the value of homes and their closeness to an environmental good or service (Peterson, 2003). For example, apartment units facing the greenery of Central Park in New York City typically cost more than the same apartment units facing another building or away from the park (N. Raheem, personal communication, June 15, 2023). Travel cost looks at the public's expenditures to get to a place, such as fuel and plane costs, and what opportunities were given up to access an environmental good or service due to travel time (Peterson, 2003).

Discount benefits and costs to determine present values. Discounting applies a lower weight on a future value than in the present (Pearce et al., 2006). There are a variety of ways to apply discount rates and many arguments for and against each method (Pearce et al., 2006). Overall, applying a discount rate reduces bias for the predicted future value of a benefit or cost to justly represent the future predictions of a CBA, as benefits and costs in the future are generally assigned lower values (Pearce et al., 2006). For example, $200 in 1980 is worth under $800 today (CPI Calculator, n.d.). Discounting accounts for the time value of money since resources used in a project are limited and could be used in another activity (Meltzer & Schwartz, 2018). Most people are inclined to consume now rather than in the future (Boardman et al., 2018). Resources in a CBA thus have opportunity costs associated with them and discounting allows analysts to compare 'apples to apples' through calculating costs and benefits at varied points in time and frequencies (Meltzer & Schwartz, 2018). Inflation must be considered with discounting, but discounting is not directly correlated with inflation (Boardman et al., 2018).

A standard method is to apply a constant discount rate, or one that does not change with time (Pearce et al., 2006). Hyperbolic discounting indicates that prices will decline with time (Pearce et al., 2006), that future interest rates are uncertain (Weitzman, 1999), and predicting the future state of the economy is uncertain (Gollier, 2002). Discount rates increase the weight allocated to future impacts, which is important when assessing environmental systems and their values now and in the future with climate change and increasing stressors (Pearce et al., 2006). Boardman et al. (2018) recommend a discount rate of 3.5% for most projects without recurring impacts beyond 50 years. If impacts remain after 50 years, Boardman et al. (2018) recommend time-declining discount rates.

Assess uncertainty and include a sensitivity analysis. A CBA looks into the future and many factors can have an impact on economic values of benefits and costs that are hard to predict (Boardman et al., 2018). Developing different contingencies, or possible outcomes, to a CBA and assigning them probability of occurrence is one way to test for uncertainty. Additionally, a sensitivity analysis is used to explore different scenarios of uncertainty within a CBA. Analysts assign probabilities of occurrence through different sensitivity analyses to account for different states of the world. A partial sensitivity analysis entails varying one assumption of the analysis at a time. A worst- and best-case sensitivity analysis entails altering the combinations of assumptions of the analysis to see how it affects NPV, negatively or positively. A Monte Carlo sensitivity analysis involves using probability distributions for all known quantitative uncertainties, then randomly selecting values for these variables from those probability distributions. Monte Carlo analyses provide sets of values to calculate net benefits. All CBAs should account for uncertainty by using some version of these tests (Boardman et al., 2018).

Economic CBA Examples

Scholars have calculated the overall costs and benefits to society of projects through CBA and results show some projects' gains are not worth the negative environmental impacts that result in overall economic loss. Hodgson and Dixon (1988) appraised the effects of a timber logging project on the nearby marine environment over the course of one year through an economic lens in Bacuit Bay, Palawan, Philippines. The economic appraisal projected results of the logging project ten years into the future. Economic variables included the costs of environmental impacts logging operations had on sedimentation in the marine environment. Results revealed the logging would impose losses to the economy of 40 million U.S. dollars (USD) over ten years due to negative impacts on the area's valuable tourism and fisheries industries. A sensitivity analysis revealed that significant alterations in predicted impacts still resulted in the same conclusion that the costs of the project did not outweigh the benefits. Net economic benefits were portrayed through this analysis, which can be a model for ecologically and economically assessing future projects (Hodgson & Dixon, 1988).

Brown et al. (2009) developed a modeling tool to evaluate the biophysical, socio-economic, and geopolitical impacts of dam construction and removal. The Integrative Dam Assessment Modeling (IDAM) tool incorporated multi-disciplinary perspectives into comparing costs to benefits to provide a more transparent evaluation for the public and decision-makers on environmental impacts of dams (Brown et al., 2009). The IDAM CBA tool can better aid in dam site location and size decisions, and the impacts of dam removal (Brown et al., 2009).

Hammond et al. (2012) conducted both a financial appraisal and an economic CBA for photovoltaic systems in buildings in the UK. The financial appraisal took the point of view of the householder and used direct costs and benefits as the variables for this portion of the analysis. The economic appraisal took the societal perspective. Impact categories included the Disability-Adjusted Life Years (DALYs) and Potentially Affected

Fraction (PAF) to capture impacts on human health and ecosystem quality, respectively (Hammond et al., 2012). The World Health Organization defines one DALY as "the loss of the equivalent of one year of full health" (WHO, n.d.). A PAF is a percentage of species observed above specific concentrations of toxins and quantifies chemicals' toxic effects in nature (Meent & Huijbregts, 2005). The results provided a more integrated approach that revealed more governmental support was necessary to justify the costs of photovoltaic systems for buildings (Hammond et al., 2012).

Economic CBAs valuing ecosystem goods and services to influence policy need to be as comprehensive as possible, implementing integrated modeling to link ecosystem functioning to the value of ecosystems (Turner et al., 2000). In the example of a wetland ecosystem, the traits, composition, and processes of the wetland need to be linked to the economic demand of its goods and services (Turner et al., 2000). Estimating ES values can lead to better policy decisions and enhance the management of our natural resources (van Beusekom, 2019b). Additionally, the CBA should not be interpreted as the final decision when influencing policy (Turner et al., 2000). Policy makers should complement or modify the CBA results, as concerns other than economic efficiency may arise (Turner et al., 2000).

Eutrophication in Aquatic Environments

Photosynthetic organisms in terrestrial or aquatic environments thrive when the elements, C, N, and P, are in a balanced ratio, and when they receive adequate sunlight. When the balance of one of these elements is in excess the organisms' survival is in jeopardy and eutrophication occurs (Le Moal et al., 2019). The Redfield Ratio, developed in 1930 by Alfred Redfield, describes the balance between the nutrients C, N, and P and is a foundational chemical understanding of aquatic environments (Falkowski, 2000). According to Redfield, when the ratio of C to N to P is 106:16:1, photosynthetic organisms thrive (Falkowski, 2000). Generally, P is the limiting factor in freshwater environments and N is the limiting factor in marine environments (Ngatia et al., 2019).

Phytoplankton populations flourish during eutrophication events due to the excess of nutrients (Diaz & Rosenberg, 2008). As these phytoplankton populations boom consumption of these elements leads to more growth, which leads to a higher demand of oxygen and decreased light (Diaz & Rosenberg, 2008). Phytoplankton eventually die and result in dead organic matter sinking to the bottom of the water body (Diaz & Rosenberg, 2008). This upsets the natural balance of the ecosystem through a series of effects. Microbial respiration increases as microbes consume the organic matter of photosynthetic algae, leading to a reduction in benthic oxygen, which is oxygen found in the deeper parts of waterbodies (Diaz & Rosenberg, 2008). Hypoxia, or severely decreased oxygen occurs when dissolved oxygen reaches two or less milliliters of oxygen per liter (Diaz & Rosenberg, 2008). Hypoxic environments lead to abnormal behavior in benthic animals. Organisms like fish and crustaceans that can flee do so, and sessile, or nonmobile organisms die, resulting in dead zones (Diaz & Rosenberg, 2008). Eutrophication can lead to HABs in coastal waters, which lead to fish kills, decreased underwater vegetation, and reductions in shellfish and marine mammal populations (Anderson et al., 2008).

Eutrophication can be caused by natural processes or by human activity. Natural eutrophication can occur in aquatic areas around continental margins of the oceans due to seasonal coastal upwelling, or stratospheric mixing, of nutrients from the benthos to the sea surface (Diaz & Rosenberg, 2008). A lake or other fresh waterbody can become eutrophic without human influence if located in an area containing naturally nutrient-rich soils (Ansari & Gill, 2014). Conversely, anthropogenic eutrophication can be caused by point and nonpoint sources, such as fertilizers from agriculture and lawns, stormwater runoff, the burning of fossil fuels, and human and animal waste (Rabotyagov et al., 2014).

The concentrations of N and P entering salt- and freshwater aquatic environments, respectively, are presently three times higher than preindustrial times (Rabotyagov et al., 2014). Significant increases in nutrient loads into our water sources have occurred since the 1950s (van Beusekom, 2019a). Since 2008, over 400 dead zones have been recognized globally (Diaz & Rosenberg, 2008) and eutrophication is one of the earth's most prominent water quality issues (Khan & Mohammad, 2014). Methods to reduce the influx of excessive nutrients into our waterways include limiting applications of fertilizers and bioremediation techniques.

Bouwman et al. (2011) estimated from 1900 to 1950, excess soil N from agricultural sources increased from 34-51 trillion grams per year, excess soil P increased from 6-9 trillion grams per year, and between 1950 and 2000, soil N and P increased more than 20-fold and sevenfold, respectively (Bouwman et al., 2011). As human populations are predicted to increase, so will the need to grow and produce more food, and with that the generation of more human waste. Figure 2 shows estimated past and future predicted global animal stocks for four half-century time periods (Bouwman et al., 2011). Many countries are predicted to increase the number of livestock they raise.

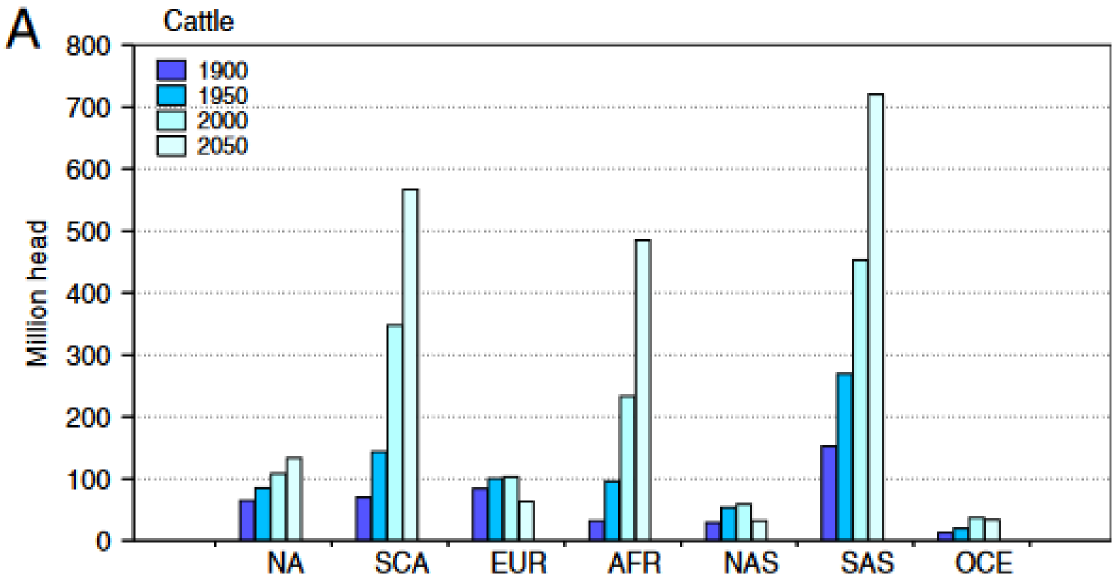


Figure 2. Past and predicted future global cattle stocks.

Estimated past and predicted future global cattle stocks for 1900, 1950, 2000, and 2050 for different regions. Depicted are Africa (AFR), Europe (EUR), North America, including Canada and the United States (NA), North Asia, including Russian Federation, Belarus, Ukraine, Republic of Moldova (NAS), Oceania, including Australia and New Zealand (OCE), South Asia, including the rest of Asia (SAS), and South and Central America (SCA) (Bouwman et al., 2011).

Septic systems, also referred to as on-site sewage treatment and disposal systems (Herren et al., 2021), on-site wastewater treatment systems (OWTS) (Hoghooghi et al., 2016), and on-site wastewater systems (OWS) (Humphrey et al., 2015), can contribute significantly to the source of nonpoint sources of N and P within watersheds (Herren et al., 2021). Testing of watersheds surrounded by OWTS showed higher concentrations of total P and total dissolved P compared to municipal sewer watersheds, revealing that OWTS have an impact on water quality, including downstream aquatic ecosystems (Humphrey et al., 2020). Septic systems surrounding the Indian River Lagoon in east-central Florida, U.S. have been linked to a significant contribution of N enriched wastewater entering the watershed (Lapointe et al., 2015).

Much of the native seagrass coverage of the Indian River Lagoon habitat has been altered due to eutrophication of the waterway from development, agriculture, and sewage over many decades (Lapointe et al., 2020). The seagrasses of this habitat are known as keystone species as they support biodiversity and the many fisheries found in this area (Lapointe et al., 2020). The reduction in seagrass of this habitat is impacting other organisms, such as the Florida Manatee (Trichechus manatus *latirostris*), which have recently experienced an increase in mortality and forced diet changes due to its now limited food sources (Allen et al., 2022). The cleanup of the Indian River Lagoon through limiting nutrients and bioremediation is necessary before species like seagrasses and manatee can rebound (Lapointe et al., 2020).

Algae-Based Biogas

The Department of Energy (DOE) created the Aquatic Species Program (ASP) to research plants which could potentially be used as alternative fuel sources after the Organization of Petroleum Exporting Countries (OPEC) placed the U.S. in an oil embargo in the 1970s (Kassinger, 2019). At the end of the ASP in the mid-90s, it reported that 500,000 acres of algae production could generate the same amount of energy that eight billion gallons of gasoline generates, or 16,000 gallons per acre. With all the production and processing costs, the cost of algae-based fuel would be $240 per gallon (Kassinger, 2019). Currently, the price of algae-based biofuels still cannot compete with the price of gasoline derived from fossil fuels. A few decades later in 2021, the DOE designated $18.7 million in funding the enhancement of algae farming methods to increase production of algae-based biofuels, as part of a larger funding effort towards improving and producing more biofuels, biopower, and bioproducts (Energy.gov, 2021). In April 2023, the price of regular gasoline was $3.69 per gallon (U.S. DOE, 2023). To date, the price of algae-based biofuels ranges widely depending on the species of algae, the producer, and the scale of the production (Amir, 2022). A production project at Duke University, called the Marine Algae Industrialization Consortium estimated the price of their algae biofuel at $5 per gallon (Duke University, 2022). In the early aughts, Sapphire Energy, an oil company supported by investments from Bill Gates and others, started growing microalgae in ponds with the goal of mixing produced algae-based oil with other fossil-fuel oils (Kassinger, 2019). Sapphire Energy's production refined the growing process efficiently through hydrothermal liquefaction (HTL), a method of extracting oil from algae without drying. Another American company, Solazyme, found ways to produce algae oil from algae that do not need sunlight to reproduce. Hydraulic fracking influenced the rise of American crude oil production in 2013, after algal-based oil companies produced one million gallons of crude algae oil, resulting in the drastic decrease in crude oil prices and the demise of many algae-based oil refineries, such as Sapphire Energy, which might have been able to compete with oil prices at $80 - $90 a barrel, but not $30 (Kassinger, 2019).

The price of algae-based biofuel varies widely for many reasons. Lipid and protein content, and thus energy production, varies among species of algae (Abdo et al., 2016; Templeton & Laurens, 2015). How algae multiply or grow also has an impact on its biomass and biofuel production potential (Sarwer et al., 2022). Similarly, the methods used to extract algal bio-oil to convert it into biofuel also vary in quantity, quality, and thus, cost (Xin et al., 2016).

Algal Turf Scrubber and Anaerobic Digester Systems

The ATS system, invented by Walter Adey in the early 1980s, is a type of bioremediation technology (Adey et al., 2013) consisting of flow-ways or racetracks in which algae is seeded or self-seeded (Adey et al., 2013; Witarsa et al., 2020). Algae proliferate due to the nutrient rich water flowing over the racetracks, thereby removing overloaded nutrients in aquatic environments (Adey et al., 2013; Liu et al., 2016; Mulbry et al., 2008b; Witarsa et al., 2020). Algal biomass is mechanically or manually harvested and can be converted into compost, livestock feed, or biofuel (Leong et al., 2021). Algal growth continues after it is harvested (Adey et al., 2013). The sale or use of algal biomass can add economic value to ATS projects.

ATS systems have been piloted in the bioremediation of livestock manure effluent (Mulbry et al., 2008b) and the cleanup of eutrophic bay waters (Adey et al., 2013). Hydromentia Technologies LLC, a private company operating out of Ocala, FL, U.S., provides pilot and full-scale designs and operations of ATS systems for non-point source pollution, point source pollution, lake restoration, and carbon dioxide recovery (Hydromentia, n.d.b). Sample diagrams of ATS sites are depicted in Figures 3 and 4. Figures 5, 6, and 7 depict Hydromentia’s ATS sites.

In effort to treat nutrient-rich, or eutrophic, water runoff from growing livestock and agricultural operations, some ATS systems are put in place at the point-source, and studies have assessed ATS and ATS-AD systems placed at these point-sources of eutrophic effluent water runoff from livestock farms. A pilot scale study conducted by

Witarsa et al. (2020) estimated that a one-hectare ATS system removed on average up to 8.7 kg of N, 1.2 kg of P, and 52.5 kg of C per hectare per year. The resulting biomass harvested could power a 1.13-kW generator via AD when converted into biofuel, which could provide approximately a third of the energy used to power the five-horsepower pump of the pilot-scale ATS system, consisting of a 122-square meter flow-way (Witarsa et al., 2020). The biogas generated in Witarsa et al.'s (2020) pilot-scale system was calculated from 0.034 kg of algae weighed dry, per square meters, per day, at 20.5% volatile solids (VS), and 107 liters of methane per kg of VS. The generator efficiency was measured at 35% (Witarsa et al., 2020). The potential production of algal biomass into biogas from ATS-AD systems was estimated to surpass the production of ethanol from corn and biodiesel from soybeans by 5.8 and 12 times per unit area, respectively (Adey et al., 2013). Overall, some of the most productive species of microalgae studied from which to produce biodiesel produced over 700 times more oil as corn cultivated per hectare per year (Faried et al., 2017) and over 200 times more biodiesel than soybeans per hectare per year (Koyande et al., 2019).

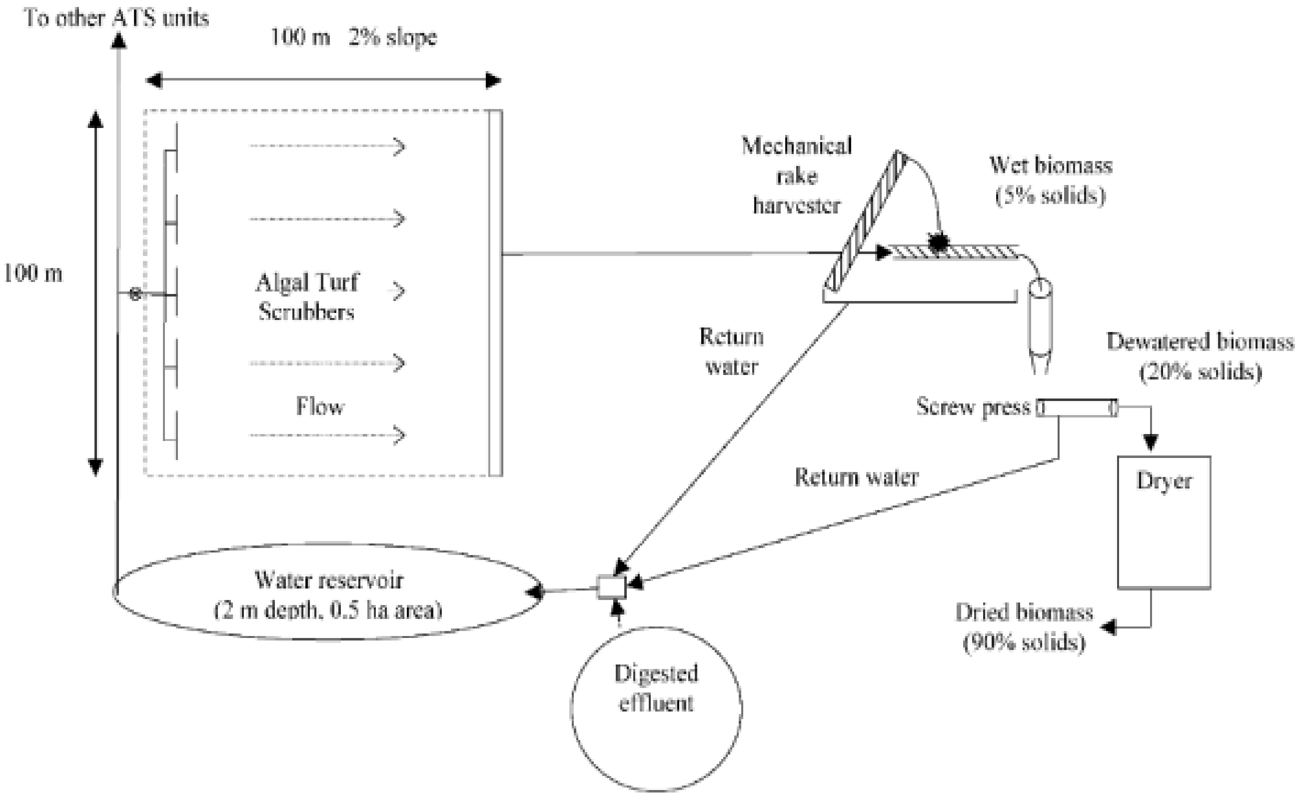


Figure 3. ATS system diagram.

A 1-hectare ATS system diagram with water reservoir, mechanized rake algal harvester, and algal biomass drying components (Pizarro et al., 2006).

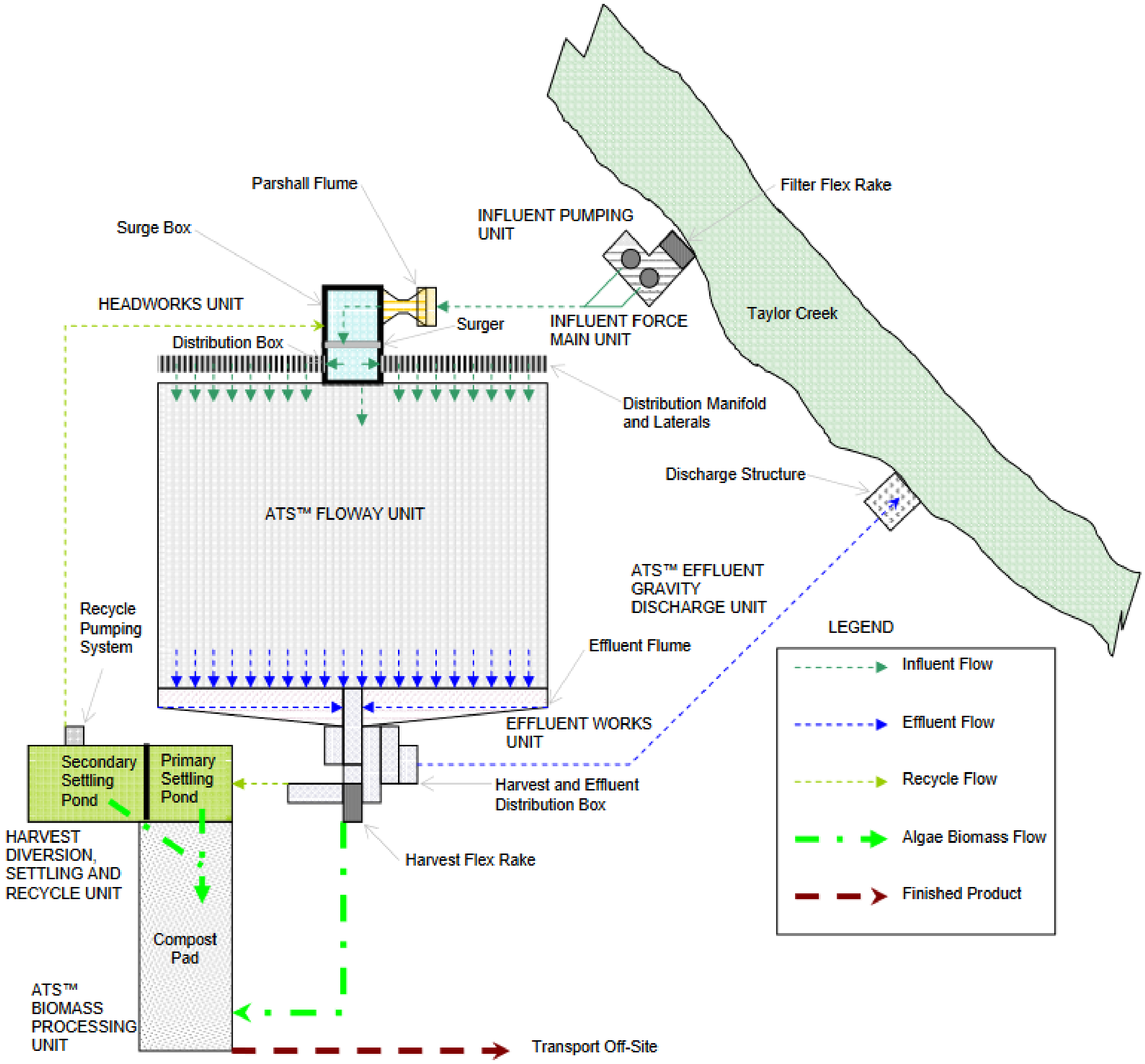


Figure 4. ATS flow-way site.

A 3.6-acre ATS flow-way site depicting the source of eutrophic water, Taylor Creek. Components of the ATS structure are shown, including pumping units, direction of water flow, settling ponds, and compost pads (Hydromentia, n.d.a).

Figure 5. ATS flow-way site from above.

A bird's eye view of an ATS facility, with flow-ways located in the background of the picture and retention ponds in the foreground (Hydromentia, n.d.b).

Figure 6. ATS inflow pipes.

Eutrophic water flowing onto the top of an ATS facility at one of Hydromentia's Florida sites (Hydromentia, n.d.c).

Figure 7. ATS site in Florida.

ATS site with algal biomass looking from the bottom of the flow-way incline (Hydromentia, n.d.d).

ATS systems are expensive to construct and operate. Costs include construction, including the manufacture of materials, such as concrete, steel, plastic liners, and PVC (Higgins & Kendall, 2012), and operations, including energy to power water pumps and to harvest algae. A financial assessment from 1997-2003 on ATS systems treating dairy manure from a point-source on-farm underground sump estimated the operational costs of the technology to be $454 per cow/year with the inclusion of an AD pretreatment of the manure effluent for biogas generation (Pizarro et al., 2006). The cost of the ATS system was $631 per cow/year without the AD pretreatment of manure effluent (Pizarro et al., 2006). The benefits of using a renewable energy source for electricity production are illustrated by this study and portray how AD can lower the costs of operation of ATS systems. Additional benefits exist and should be included to incentivize dairy farmers to partake in ATS-AD systems, such as nutrient trading approaches to manage the health of watersheds, sales from algal biomass byproducts, and state and national incentives and grants (Pizarro et al., 2006).

The results of Higgins and Kendall's (2012) Life Cycle Assessment (LCA) of a simulated ATS-AD system treating dairy manure effluent showed that the ATS system improved eutrophication's negative impacts, as the study assessed life cycle energy, greenhouse gas emissions, eutrophication potential, and cost impacts of the ATS system. The ATS system reduced the C footprint of the dairy wastewater treatment process with low water recirculation rates and algal productivity. However, drying algal biomass to use in AD increased greenhouse gas emissions through energy consumption. Therefore, these results generally align with those of Pizarro et al. (2006). Higgins and Kendall (2012) concluded there was overall cost effectiveness in an ATS-AD system when placed in a dairy wastewater treatment process only if grants and nutrient trading programs were enacted. However, Higgins and Kendall (2012) chose to omit water use and the end-of-life phase of the ATS-AD system.

Preliminary Research on ATS-AD Systems of Livestock Effluent

I conducted preliminary research of a comparative LCA for the use and end of life phases of two dairy manure effluent systems to better understand the inputs, outputs, processes, and carbon footprints of the ATS-AD technology (Trotogott, unpublished). I assessed one conventional slurry lagoon (CSL) operation using slurry to lagoon to field fertilizer and another consisting of a one-hectare ATS-AD system. I based the calculations on data gathered and results obtained by prior studies (Adey et al., 2013; Liu et al., 2016; Mulbry et al., 2008b; Witarsa et al., 2020).

I evaluated impact categories, reference flows, and processes using the web based LCA tool, Earthster. I used impact categories based on the impact assessment method, ReCiPe, a method for life cycle impact assessment developed in 2008 by the Dutch

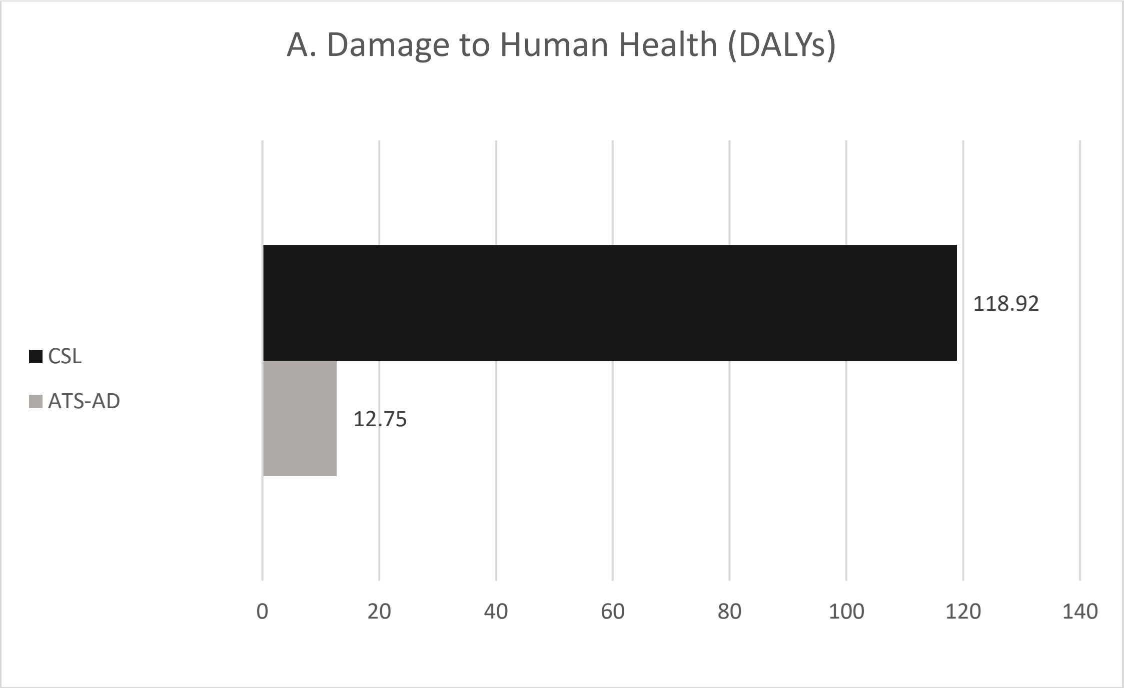
National Institute for Public Health and Environment, Radboud University Nijmegen, Leiden University, and PRé Sustainability, an environmental consultant of the Netherlands. Impact categories included damage to human health measured in DALYs; damage to ecosystems measured in species loss in ecosystems expressed as a fraction within space and time, or species by square (or cubed) meters by year (which is denoted as "species.year"); climate change measured in kilograms of carbon dioxide equivalent; and water use measured in cubic meters (Huijbregts, et al., 2017).

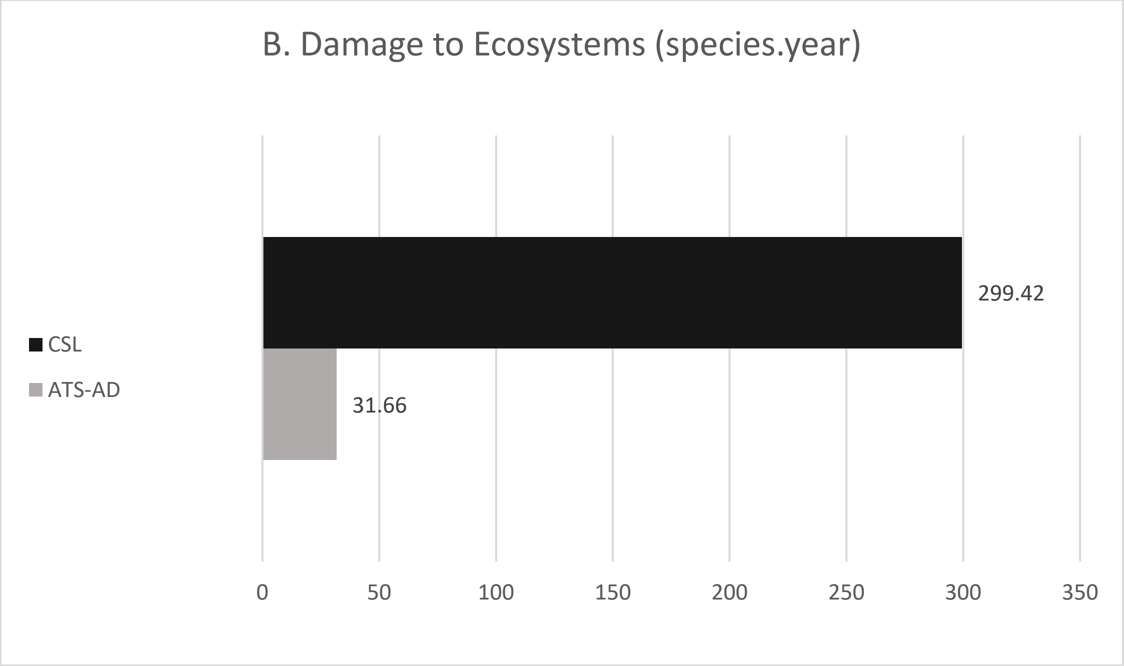
My LCA comparison revealed the ATS-AD system had a smaller carbon footprint than the CSL system in all impact categories of damage to human health, ecosystems, climate change, and water use (Table 2 & Figure 8). This preliminary research suggests that ATS-AD systems reduced greenhouse gas emissions, used less water, and decreased detrimental impacts on human lives and ecosystems through its bioremediation of nutrient-rich water.

Table 2. Impact category results of unpublished LCA.

Impact Category	Unit	ATS-AD	CSL
Damage to Human Health	DALY	12.75	118.92
Damage to Ecosystems	m species.year	31.66	299.42
Climate Change	kg CO2eq	461,440.00	4,290,000.00
Water Use	cubic meters	2,20,000.00	20,810,000.00

(Trotogott, unpublished).


A. Damage to Human Health (DALYs)
■ CSL
■ ATS-AD
118.92
12.75
0
20
40
60
80
100
120
140


B. Damage to Ecosystems (species.year)
■ CSL
■ ATS-AD
299.42
31.66
0
50
100
150
200
250
300
350

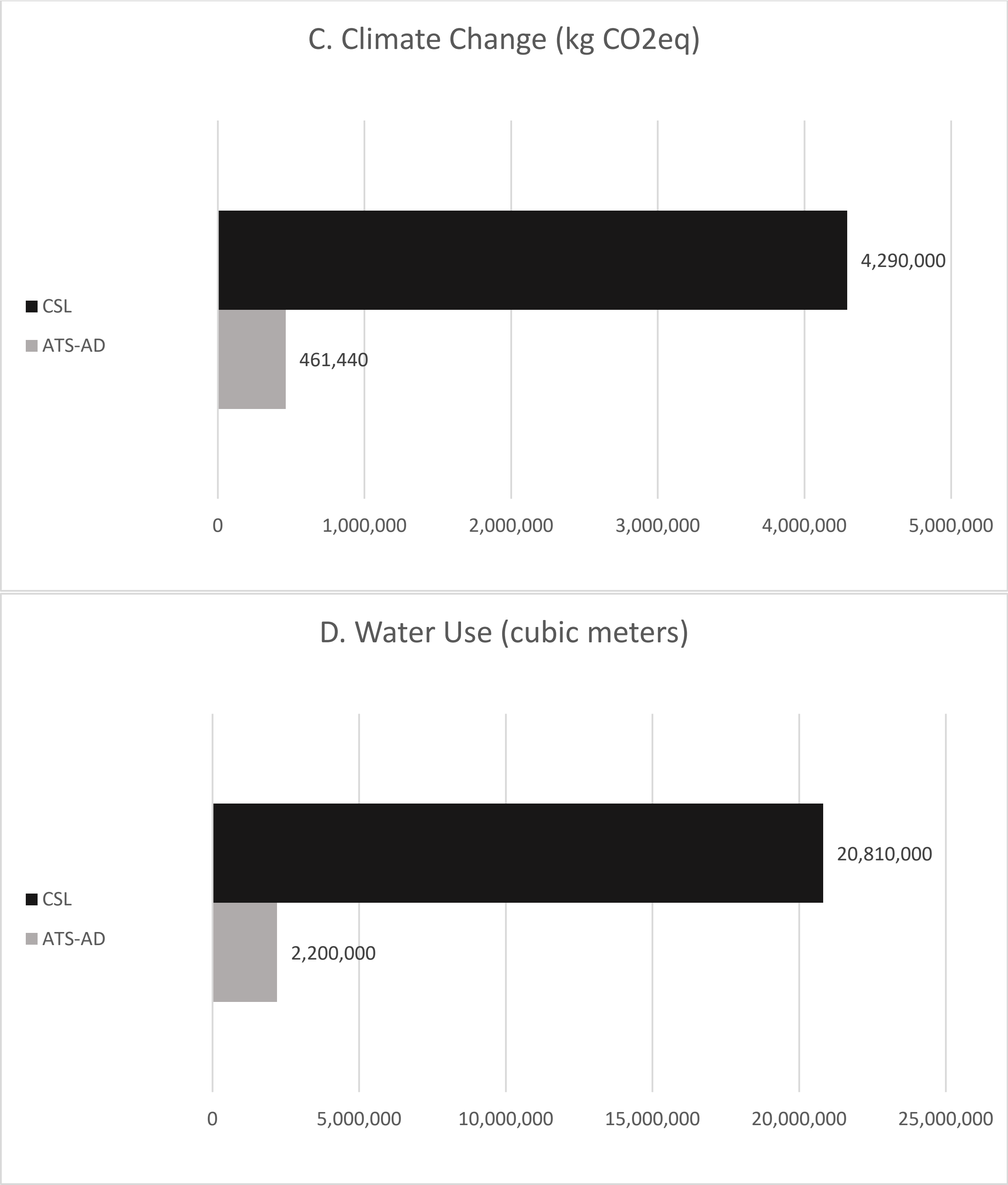


Figure 8. ATS-AD comparison results.

Impact category comparison results of hypothetical CSL livestock farm and a livestock farm with an ATS-AD system treating manure effluent. A. Damages to human health are measured in DALYs, B. Damages to ecosystems are measured in species.year, C. Climate change is depicted in CO2eq, and D. Water use is measured in cubic meters. (Trotogott, unpublished).

In-depth analyses on ATS-AD systems are limited to livestock operations and only concern cost and benefits from the business/financial perspective. Existing research on ATS systems in eutrophic aquatic environments, like Chesapeake Bay, is limited to small-scale systems without using AD (Mulbry et al., 2010). With the rising need for bioremediation in growing eutrophic aquatic habitats, case study assessments are needed to determine whether the ATS-AD system is cost effective from a societal viewpoint. Falls Lake in North Carolina serves as an appropriate case study assessment.

Falls Lake Watershed

Falls Lake is a reservoir constructed in 1981 by the Army Corps of Engineers located on the upper portion of the Neuse River Basin (NRB) (City of Durham, n.d.; Osmond et al., 2015) in the Piedmont region north and northwest of the capital city of Raleigh (Figure 9). The Falls Lake Watershed (FLW) (Figure 10) includes parts of Orange, Person, Granville, Durham, Wake, and Franklin counties (NC Department of Environment and Natural Resources Division of Water Quality, 2009). The reservoir provides drinking water to the residents of Raleigh, Garner, Rolesville, Wake Forest, Knightdale, Wendell, and Zebulon (NC Division of Water Resources, 2021), and provides recreational activities, such as boating, fishing, swimming, and camping (City of Durham, n.d.). The watershed's land cover is comprised of approximately 60% forest, 17% agriculture, 14% developed, 4% grassland/shrub, 3% open water, 2% wetland, and less than 1% barren land (Table 3) (NC Division of Water Resources, 2021). The population demographics, including race, population, number of households, education, and median income, are represented in Tables 4 and 5.

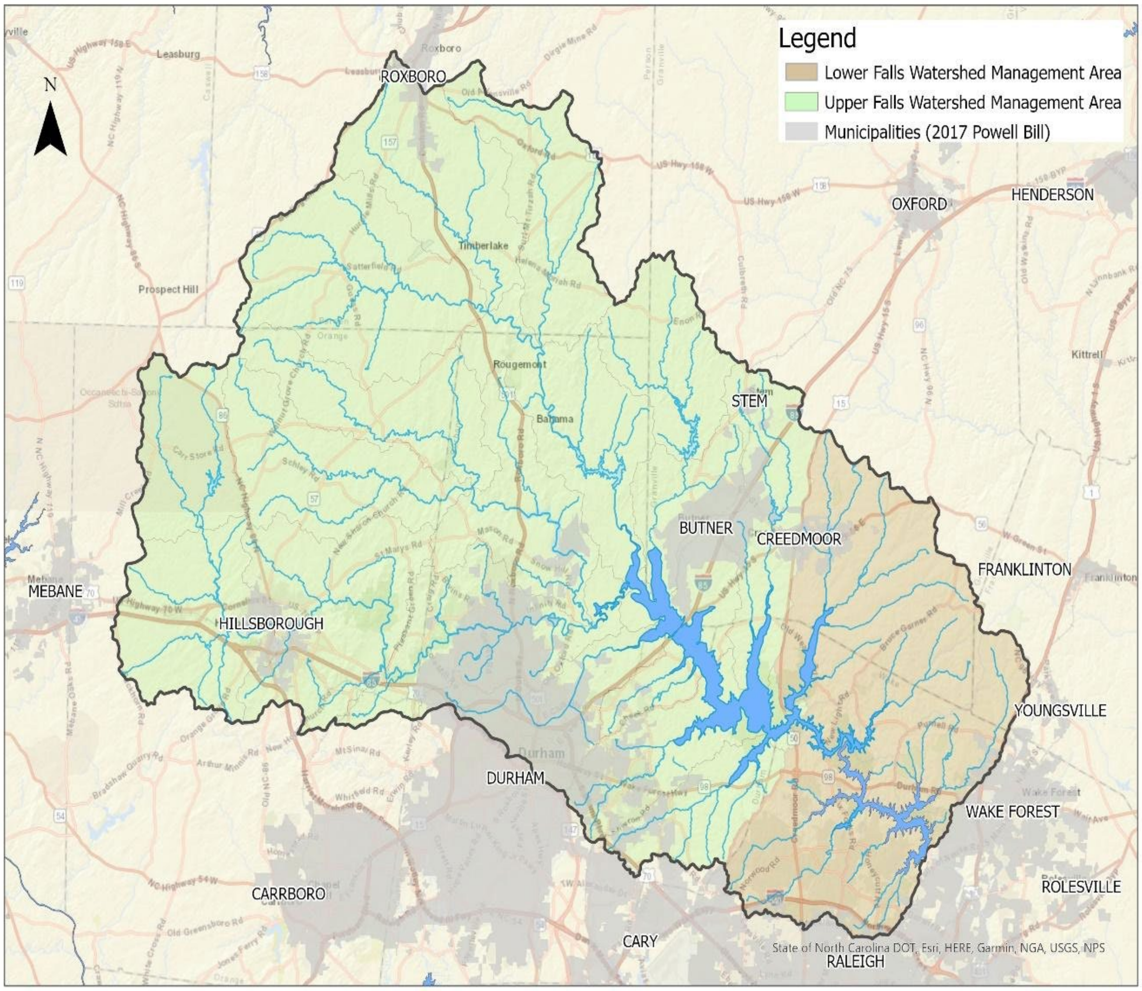


Figure 9. Falls Lake area map.

(North Carolina Environmental Quality, n.d.)

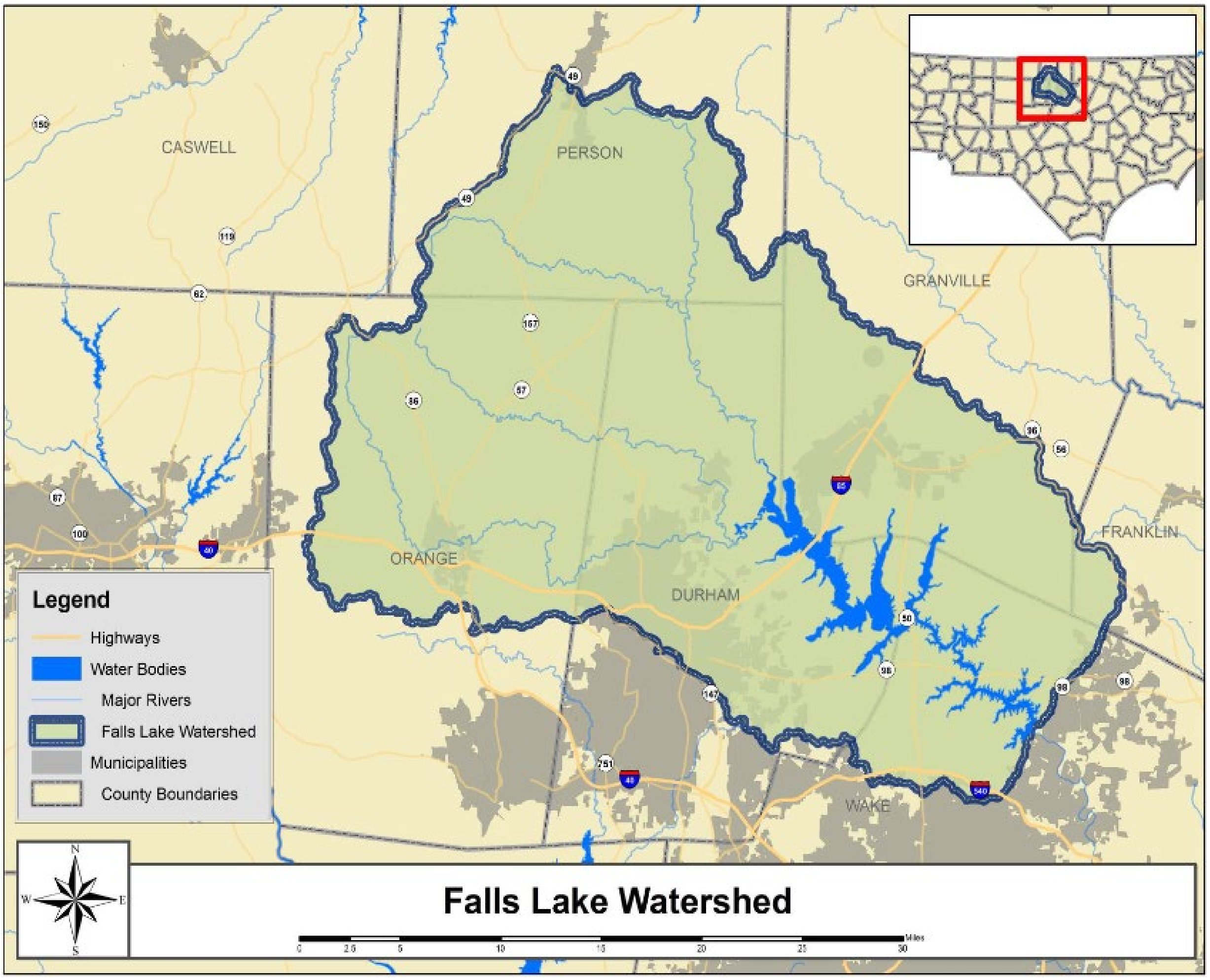


Figure 10. Falls Lake Watershed.

(NC Department of Agriculture and Consumer Services, 2018)

Estimations based on previous state survey data in 2019 approximated that 5.24 million residents of suburban and rural North Carolina used septic systems as a residential wastewater treatment system (Iverson et al., 2021). With an average of two and a half persons per household in the same year, approximately 2.1 million septic systems served North Carolina residents and 50,000 septic systems were within the FLW (Iverson et al., 2021). High concentrations of septic systems in urban, suburban, and rural settings have polluted groundwater leading to eutrophication of fresh, estuarine, and coastal surface waters and watersheds, such as in the Piedmont region of North Carolina (Iverson et al., 2018), the Indian River Lagoon of Florida (Herren et al., 2021), and Cape Cod in Massachusetts (Cape Cod Commission, 2015). Testing of streams surrounded by high concentrations of OWTS leading to Falls Lake showed higher levels of dissolved nitrogen, phosphate, and chloride compared to areas surrounded by a higher concentration of municipal sewer systems (O'Driscoll et al., 2020).

Table 3. Land cover in Falls Lake Watershed.

Aggregated Land Cover Type	2001 (acres)	Percent of watershed	2016 (acres)	Percent of watershed	Change (acres)	Percent change
Forest	297,965	60%	293,337	5900%	-4,628	-2%
Agriculture	82,045	17%	78,086	1600%	-3,959	-5%
Developed	66,984	14%	75,633	1500%	8,648	13%
Grassland/Shrub	18,470	4%	18,017	400%	-453	-2%
Open Water	15,475	3%	16,771	300%	1,296	-8%
Wetland	12,223	2%	11,380	200%	-842	-7%
Barren Land	588	<1%	526	<1%	-63	-11%

(NC Division of Water Resources, 2021).

Table 4. Race demographics of counties within Falls Lake Watershed.

Race and Hispanic Origin	Orange	Person	Granville	Durham	Wake	Franklin
White	69.80%	66.60%	54.70%	64.00%	69.70%	76.30%
Black or African American	25.90%	20.80%	35.30%	31.30%	26.50%	12.00%
American Indian and Alaska Native	1.10%	0.80%	1.00%	1.00%	1.00%	0.60%
Asian	0.90%	8.90%	6.00%	0.90%	0.40%	8.20%
Native Hawaiian and Other Pacific Islander	0.10%	0.10%	0.10%	0.10%	-	0.10%
Two or More Races	2.30%	2.80%	2.90%	2.60%	2.30%	2.80%
Hispanic or Latino	10.30%	10.60%	13.90%	10.30%	5.40%	8.80%
White, not Hispanic or Latino	61.70%	58.30%	43.60%	56.20%	65.50%	68.80%

(U.S. Census Bureau, 2022).

Table 5. Demographics of counties within Falls Lake Watershed.

County	Population[a]	Population[b]	Households	Housing[c]	High school[d]	College[e]	Income[f]
Orange	150,477	374.00	54,783	$399,900	93.70%	61.30%	$79,205
Person	39,386	99.70	15,927	$146,900	87.80%	17.60%	$55,759
Granville	61,903	114.60	24,826	$177,600	86.10%	23.40%	$60,606
Durham	332,680	1,133.70	152,518	$262,400	89.90%	50.70%	$67,000
Wake	1,175,021	1,353.30	493,188	$324,500	93.70%	54.70%	$88,471
Franklin	74,539	139.40	31,695	$174,200	86.50%	22.60%	$62,332

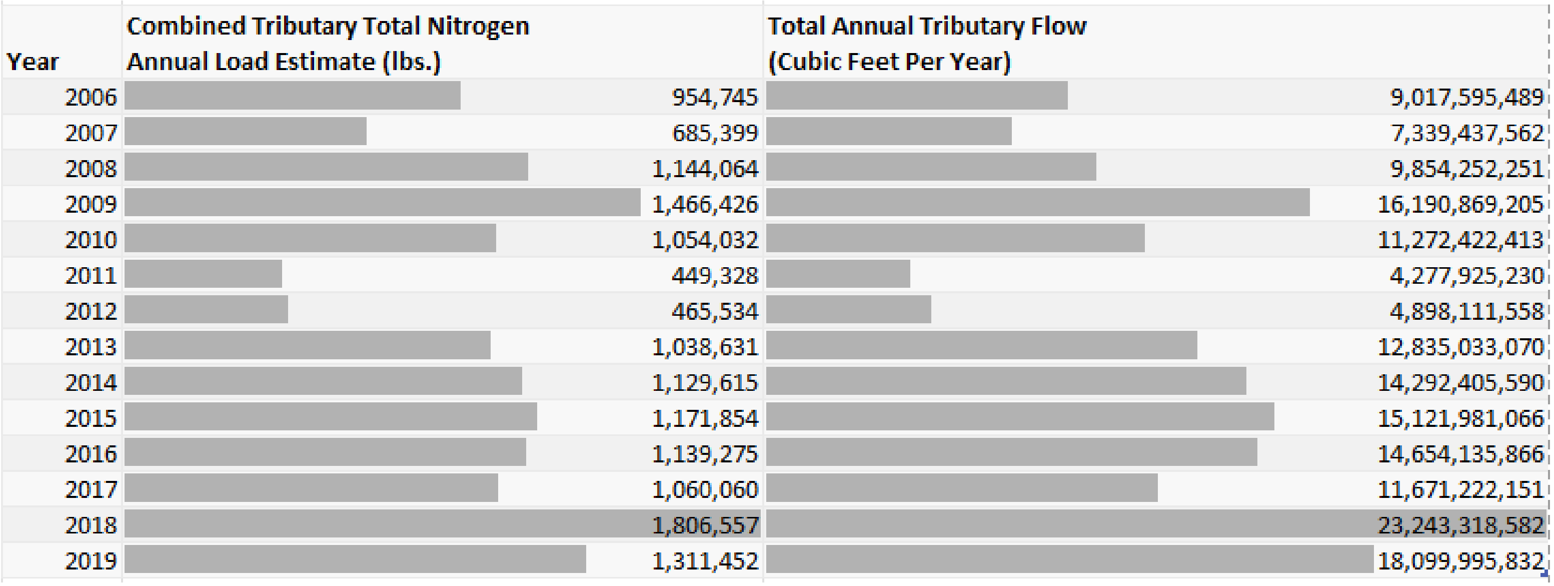
Note.
[a] *July 2022 estimate.*
[b] *per square mile in 2020*
[c] *Median value of owner-occupied housing units, 2017-2021*
[d] *High school graduate or higher, percent of persons age 25 years+, 2017-2021*
[e] *Bachelor's degree or higher, percent of persons age 25+, 2017-2021*
[f] *Median household income (in 2021 dollars), 2017-2021*
(U.S. Census Bureau, 2022).

Surveys of farmer fertilizer decision making within Neuse River, Jordan Lake, and FLW sites indicated farmers did not apply and follow nutrient management plans while fertilizing crops, failed to conduct soil testing guidelines, were hesitant to follow Michigan State University Extension’s recommendation in terms of applied fertilizer amounts, and were more likely to follow fertilizer amount suggestions from fertilizer manufacturers (Osmond et al., 2015; Stuart et al., 2014). Farmers did not trust recommendations made by researchers but instead valued the excessive use of nutrient applications, as fertilizer manufacturers marketed the overuse of fertilizers as insurance (Osmond et al., 2015).

Since 2011, management strategies have been put into place to protect the Falls Lake water supply, fish and wildlife populations, benefits of flood control, and recreational uses from impacts of exceedances in the state’s chlorophyll *a* water quality standard (NC Division of Water Resources, 2021). The most significant sources of N and

P inputs to the FLW are from agriculture and point sources, such as wastewater discharge (NC Division of Water Resources, 2021). Data collected from Falls Lake between 2002 and 2006 showed high levels of chlorophyll *a*, which is a marker for high algal productivity due to high nutrient inputs, revealing the reservoir did not meet water quality standards. N and P loads were measured from the five prominent tributaries leading to Falls Lake from 2006 to 2019 (Figures 11, 12, 13, and 14). These show a 37% increase in N loads over the thirteen-year period from increased water flow or increased rainfall, both of which led to increased nutrient inputs from nonpoint source runoff. The same data show a 17% decrease in P loads. The decrease in P loads despite the increase in rainfall is attributed to increased watershed management activities (NC Division of Water Resources, 2021). The remaining high levels of N are due to the overloaded septic systems in the area (Iverson et al., 2021).

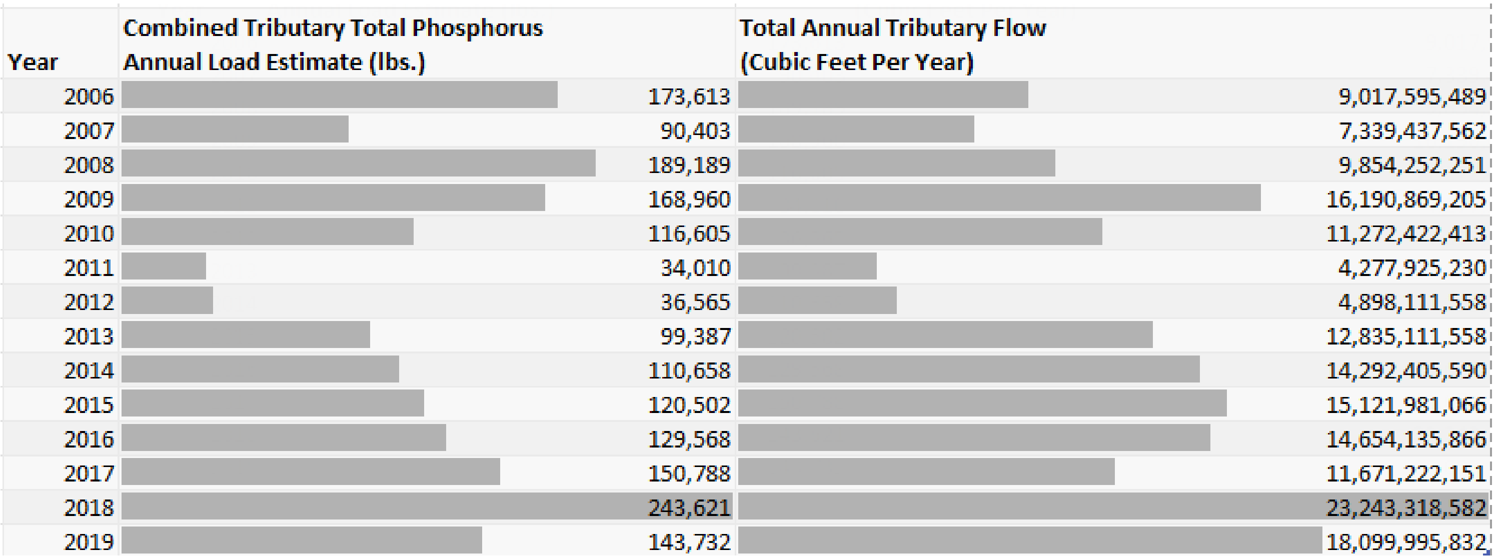
The Neuse River of coastal North Carolina, to which Falls Lake flows, has experienced impacts of eutrophication since the early 1980s, including HABs, fish kills, and hypoxia (Hounshell & Paerl, 2017). An increase of urban and agricultural development over the last few decades has led to an increase in N and P loads into the Neuse despite the enactment of total maximum daily loads (TMDLs) in the 1990s and a reduction in inorganic N (Hounshell & Paerl, 2017). Froelich et al. (2019) looked at N and C only (not including P) nutrient levels in the Neuse River Estuary (NRE) from 2004-2014. An increased prevalence of *Vibrio spp.* -- bacteria present in estuarine waters which play a large ecological role in population dynamics, N fixation, crustacean exoskeleton weakening, and several of which are pathogenic -- were observed and attributed to increased N and C (Froelich et al., 2019).



Year	Combined Tributary Total Nitrogen Annual Load Estimate (lbs.)	Total Annual Tributary Flow (Cubic Feet Per Year)
2006	954,745	9,017,595,489
2007	685,399	7,339,437,562
2008	1,144,064	9,854,252,251
2009	1,466,426	16,190,869,205
2010	1,054,032	11,272,422,413
2011	449,328	4,277,925,230
2012	465,534	4,898,111,558
2013	1,038,631	12,835,033,070
2014	1,129,615	14,292,405,590
2015	1,171,854	15,121,981,066
2016	1,139,275	14,654,135,866
2017	1,060,060	11,671,222,151
2018	1,806,557	23,243,318,582
2019	1,311,452	18,099,995,832

Figure 11. Nitrogen load estimates at Falls Lake.

Annual nitrogen load estimates at Falls Lake based on combined raw tributary nutrient loads (adapted from NC Division of Water Resources, 2021).



Year	Combined Tributary Total Phosphorus Annual Load Estimate (lbs.)	Total Annual Tributary Flow (Cubic Feet Per Year)
2006	173,613	9,017,595,489
2007	90,403	7,339,437,562
2008	189,189	9,854,252,251
2009	168,960	16,190,869,205
2010	116,605	11,272,422,413
2011	34,010	4,277,925,230
2012	36,565	4,898,111,558
2013	99,387	12,835,111,558
2014	110,658	14,292,405,590
2015	120,502	15,121,981,066
2016	129,568	14,654,135,866
2017	150,788	11,671,222,151
2018	243,621	23,243,318,582
2019	143,732	18,099,995,832

Figure 12. Phosphorus load estimates at Falls Lake.

Annual phosphorus load estimates at Falls Lake based on combined raw tributary nutrient loads (adapted from NC Division of Water Resources, 2021).

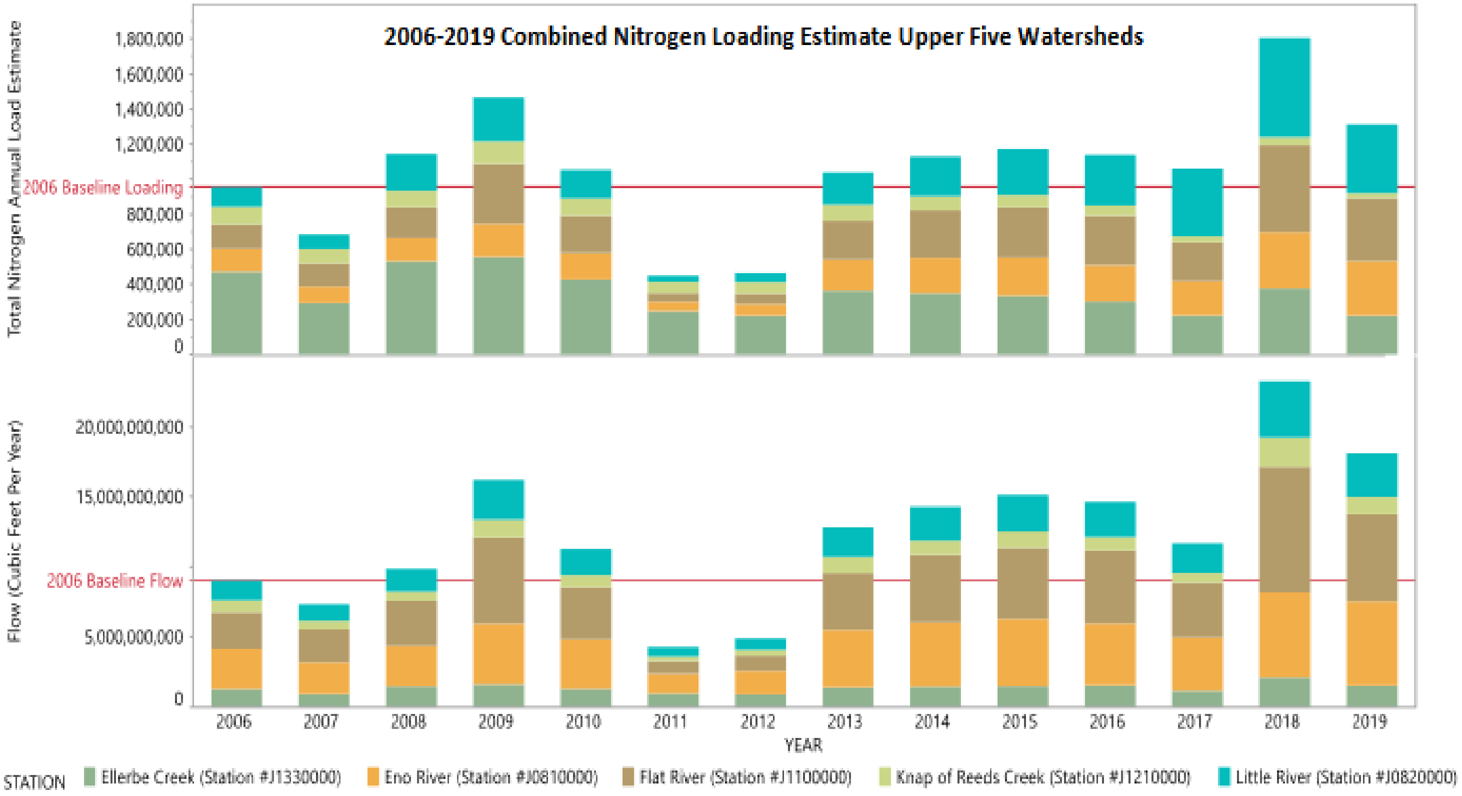


Figure 13. Nitrogen loads based on water flow.

Nitrogen loads based on five main tributaries of Falls Lake with annual water flow (NC Division of Water Resources, 2021).

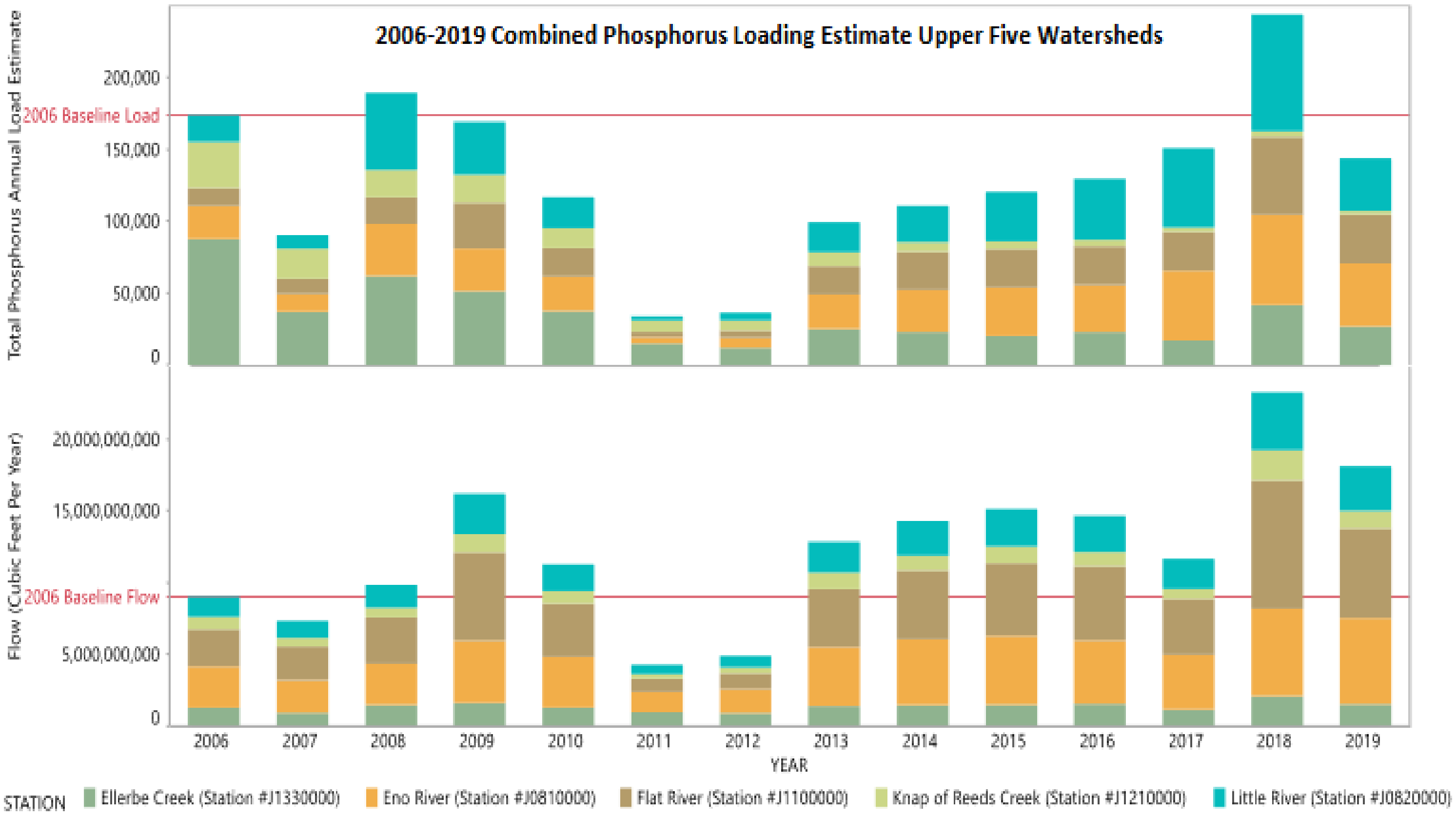


Figure 14. Phosphorus loads based on water flow.

Phosphorus loads based on five main tributaries to Falls Lake with annual water flow (NC Division of Water Resources, 2021).

Falls Lake ATS Pilot Report

Hydromentia Technologies LLC conducted an ATS pilot project within the FLW to measure N and P load reductions, algal biomass productivity, and to provide performance projections for nutrient load reductions and biomass management for larger future sites (Hydromentia, 2017b). The pilot unit was a 500-foot long, one-foot-wide system designed for 20 gallons of inflow per minute (Hydromentia, 2017b) (Figure 15).

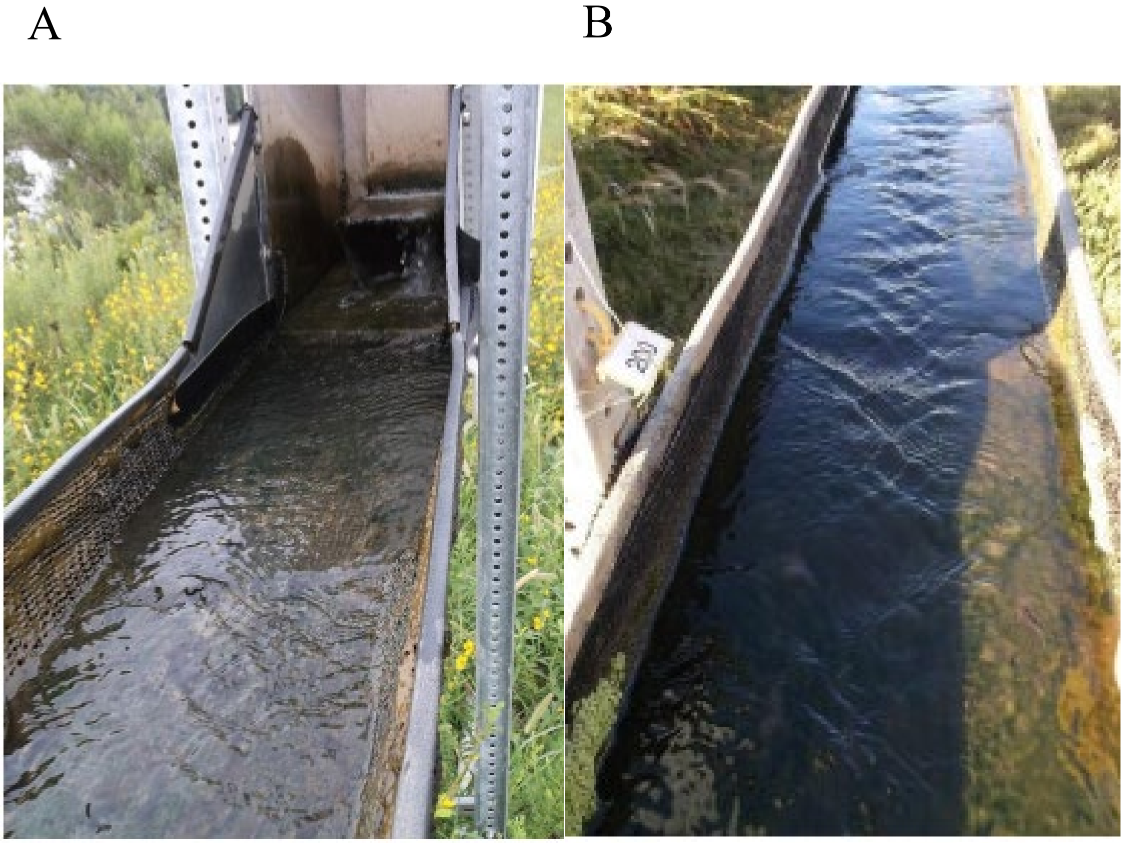


Figure 15. Falls Lake pilot ATS flow-way.

Segments of the flow-way at water entry (A) and 200' down the flow-way (B).

Water quality, algal biomass, and N and P measurements were captured over the course of 448 days (Hydromentia, 2017b). Results of the pilot study are depicted in Table 6. Hydromentia's present worth cost report on the Falls Lake pilot provides projections for similarly located sites, but there is a need for capturing costs and benefits of all operations, including external costs and benefits to the Falls Lake ecosystem.

With this in mind, the FLW provided a case study for both comprehensive financial and economic appraisals of an ATS-AD system.

Table 6. Falls Lake pilot report results.

Variable	Amount	Unit and frequency
Construction	\$6,245,652	per project
Operation	\$279,230	per year
N removed	7,140	pounds per year
P removed	1,260	pounds per year
Harvest algal biomass	6,935	dry grams per m^2 per year

(Hydromentia, 2017b)

Research Questions, Hypotheses and Specific Aims

My central research questions were:

1) How much ATS-AD biogas can be generated from a 25-million gallons per day (MGD) facility pumping eutrophic water from an environment the size of the FLW, and would the electricity generated from the biomass be sufficient to cover the operational costs of the ATS-AD system? These costs include the electricity used for pumping water, the use of raw materials, and the energy used in harvesting the algal biomass.

- To examine this, I tested the hypothesis that an ATS system is more profitable with an AD component – that biofuel generated from algal biomass productivity will offset the costs of running the ATS system.

2) Would incorporating societal benefits, such as avoided eutrophication or bioremediation, harvested algal biomass for livestock feed, crop fertilizer, or biogas generation, in an economic analysis outweigh the overall costs of an ATS-AD project? And if so, by how much would those benefits outweigh the costs compared to a typical business financial analysis?

- I tested the hypothesis that the ATS-AD system is profitable from an economic standpoint – that the avoided environmental impacts of eutrophication and benefits of biogas generation from harvested biomass are worth the costs of construction, land use, maintenance, and operations.

Specific Aims

I completed the following tasks for my research:

1. Determined inputs and outputs for a financial analysis of an ATS-AD system.
2. Determined inputs and outputs for an economic analysis of an ATS-AD system.
3. Collected data on available inputs/costs and known outputs/benefits of an ATS-AD system from Hydromentia.
4. Inquired about collecting potential missing data through Hydromentia or a partner, such as a local conservation organization or city watershed agency.
5. Calculated how much energy an AD system could provide to the specific ATS project.
6. Researched economic values of ES and habitat protection.
7. Researched algal biomass uses and sale prices.
8. Conducted financial analysis with and without an AD system.
9. Conducted a range of economic analyses with avoided environmental costs.

Chapter II

Methods

My general research approach was to provide both financial and economic viewpoints of ATS and ATS-AD systems, while incorporating known ES into an economic CBA. I tested the hypothesis that an ATS system is more profitable with an AD component by conducting a financial analysis based on data provided by Hydromentia (2017a). I tested the hypothesis that a 25-MGD ATS system is profitable from a sustainability standpoint over the 20-year timespan by conducting an economic analysis. I incorporated additional benefits into this analysis that affect the overall health of the surrounding ecosystems and, thus, society at large. I applied the high end of the social project discount rate of 2.8% according to the *Circular A-94* document provided by the Office of Management and Budget (OMB, 2023). I use 2.8% for all CBA scenarios to reflect average annual values over each year of the 20-year project life.

Benefits Transfer

Ideally, a researcher would conduct original valuation research to assess the ES impacts and monetize them. However, due to the time and cost limits of this master's thesis research, I applied benefits transfer in my research methodology to best capture the valuation of ES ATS-AD systems provide. Benefits transfer applies original research of the estimated values of ES from one study location to an unresearched new site with similar characteristics (Richardson et al., 2015). Benefits transfer addresses the demand of valuing more nonmarket ES in more locations in shorter timeframes and on lower budgets (Richardson et al., 2015). Federal agencies and academic researchers use benefits transfer extensively (Richardson et al., 2015). I researched the specific value estimations of ES from areas as geographically similar as possible to my case study site of Falls Lake, North Carolina. I applied values of ES from research conducted in the south-central, south-east, and Midwest United States adjusted to 2023-dollar values using the Consumer Price Index Inflation Calculator (CPI Calculator, n.d.).

To transfer values of ES to this research, I estimated the number of households within the FLW. I estimated the number of households based on a population of 90,000 persons within the watershed from the 2021 Falls Lake Nutrient Status Report (NC Division of Water Resources, 2021). I used the same population growth rate of 2% as the 2021 Falls Lake Nutrient Status Report for two years to account for the year of this research, 2023. Iverson et al. (2019) estimated the average household in North Carolina in 2019 to be 2.52 persons. Therefore, I calculated the number of households for this research by increasing the 90,000 population of the watershed by 1,800 individuals for the year 2022, increasing the population by 1,836 individuals for the year 2023 (2% increase for two years), dividing by 2.52 persons in each household, and arrived at 37,157 households within the watershed.

Financial Cost-Benefit Analysis

I conducted two financial analyses in spreadsheet model form, one portraying the overall business financials of an ATS system without an AD, and another with an AD incorporated.

The CEO of Hydromentia, Mark Zivojnovich, provided cost and benefit data taken over the 20-year timespan of the Falls Lake Pilot Present Worth Cost Analysis of a

25-MGD, 500-foot ATS system, which included capital costs, operations and maintenance costs, algal biomass productivity, nutrients captured, the price of nutrients captured in dollars per pound, and costs of disposing algal biomass to the landfill (Hydromentia, 2017c). Table 7 details all financial cost and benefit variables and their units of measurement for the simple financial ATS appraisal and the financial ATS-AD appraisal. I converted all values to 2023-dollars using the Consumer Price Index Inflation Calculator (CPI Calculator, n.d.).

Cost variables included:

- design
- construction
- engineering, surveying, permitting, and administration
- operations and maintenance of the facility, including labor associated with harvesting the algal biomass
- algal biomass disposal to landfill, and
- energy use to operate the water pumps (Hydromentia, 2017b).

Land costs were not included in this analysis as Hydromentia's Falls Lake report states the ATS facility is located on NC state government owned land (Hydromentia, 2017a). End of life, or demolition, costs were not included in this analysis (Hydromentia, 2017a). Water use was not included in this analysis as water use permits are not issued on any Hydromentia sites (M. J. Zivojnovich, personal communication, September 23, 2022). Eutrophic water flows into the ATS system over the flow-ways and enters a receiving body within ten minutes (M. J. Zivojnovich, personal communication, September 23, 2022). Evaporation and impervious surfaces collecting rainfall will alter the amount of water flowing out from the system, but water use was not a system cost in this research (M. J. Zivojnovich, personal communication, September 23, 2022).

Table 7. Financial CBA of ATS and ATS-AD systems.

ATS System	
Benefits	Units of Measurement
Nutrients captured	USD/pounds/year
Costs	Units of Measurement
Construction	USD/year
Operations and maintenance	USD/year
Algal biomass productivity	Pounds dry weight/square meter/year
ATS-AD System	
Benefits	Units of Measurement
Algal biofuel generation	UDS/year
Nutrients captured	USD/pounds/year
Costs	Units of Measurement
Construction	USD/year
Operations and maintenance	USD/year

Economic Cost-Benefit Analysis

I conducted an economic analysis in spreadsheet model form, which encompassed all previously captured costs and benefits of the second financial analysis (including the AD component). I included additional added impacts from environmental externalities. I determined the ES variables the ATS-AD system provides using the CICES spreadsheet resource (Haines-Young & Potschin, 2012) (Table 8). I used these codes when conducting both a literature review for my thesis and to find economic valuation through benefits transfer on specific ES that ATS-AD systems provide. Table 9 details all final economic cost and benefit variables and their units of measurement.

Table 8. CICES codes and descriptions for economic CBA research.

Code	Section	Simple descriptor
1.1.2.3	Provisioning (Biotic)	Plants that are cultivated in fresh or salt water that we can use as an energy source
2.1.1.1	Regulation & Maintenance (Biotic)	Decomposing wastes
2.1.1.2	Regulation & Maintenance (Biotic)	Filtering wastes
2.1.2.1	Regulation & Maintenance (Biotic)	Reducing smells*
2.2.3.2	Regulation & Maintenance (Biotic)	Controlling disease*
2.2.5.1	Regulation & Maintenance (Biotic)	Controlling the chemical quality of freshwater*
2.2.5.2	Regulation & Maintenance (Biotic)	Controlling the chemical quality of salt water*
2.2.6.1	Regulation & Maintenance (Biotic)	Regulating our global climate*
3.1.1.1	Cultural (Biotic)	Using the environment for sport and recreation; using nature to help stay fit
3.1.1.2	Cultural (Biotic)	Watching plants and animals where they live; using nature to destress*
4.2.1.1	Provisioning (Abiotic)	Drinking water from sources at the ground surface
5.1.1.1.	Regulation & Maintenance (Abiotic)	Diluting wastes
5.1.1.3.	Regulation & Maintenance (Abiotic)	Natural processing of wastes

**All ES that provide value but were not included in the economic CBA due to lack of current findings or applicability (Haines-Young & Potschin, 2012).*

I did not use all CICES codes that apply to ATS-AD systems in the economic CBA due to lack of existing research, or lack of research conducted in a similar geographic region as Falls Lake, NC. I found and collected research on the sources provided in Table 10 for a literature review of the applicable ES and their economic valuation. I only used ES valuation research I could find or that was conducted in a similar geographic location to Falls Lake, NC; therefore, my research is likely to understate the value of ES that ATS and ATS-AD systems provide.

Table 9. Final economic CBA variables.

Benefits	Unit of measurement
Nutrients captured	USD/pound of N & P/year
Drinking water	USD/household/year
Recreation	USD/household/year
Cultural heritage	USD/household/year
Air quality	USD/household/year
Freshwater provision	USD/household/year
Algal biogas	USD/year
Costs	Unit of measurement
Construction	USD/year
Operations & maintenance	USD/year

Table 10. Literature review of valued ES research.

Ecosystem service	Citation	Location	2023 USD	Economic valuation method
Nutrient removal (N)	Hernández-Sancho et al. (2010)	Spain	$145.64 million/year	Avoided costs/Shadow pricing
Nutrient Removal (N)	Campbell et al. (2020)	Maryland, U.S.	$18.34/kg N	Hedonic pricing
Nutrient removal (P)	Hernández-Sancho et al. (2010)	Spain	$74.23 million/year	Avoided costs/Shadow pricing
Nutrient removal (P)	Molinos-Senante et al. (2011)	Europe	$62.44/kg P removed	Avoided costs/Shadow pricing
Nutrient removal (P)	Verberg (2016)	Wisconsin, U.S.	$30.41/kg P	Hedonic pricing
Nutrient removal	Chaikae et al., (2017)	Suwanee, FL, U.S.	<$2/household/year	WTP
Drinking water/Water quality	Schinck et al. (2020)	Quebec, Canada	$235/household/year	CV survey
	L'Ecuyer-Sauvageau et al. (2019)	Quebec, Canada	$353/household/year	CE questionnaire
	Ureta et al. (2022)	South Carolina, U.S.	$0 - $3.07/resident/month	WTP
Recreation	Nelson et al. (2015)	Utah, U.S.	$13.63/month/user; $8.31/month/nonuser	CV survey
	Zhang & Sohngen (2018)	Ohio/Lake Erie, U.S.	$9.63 – $12.04 more per trip for 1 less mile of boating through HABs en route to fishing site	CE
Recreation	McDougall et al. (2020)	Lochs Lomond & Leven, Scotland	$17.43/household/year (Lomond); $12.19/household/year (Leven)	WTP
Freshwater Provisioning	Moeltner et al. (2023)	Florida, U.S.	$17.37/household/year	CE

Ecosystem service	Citation	Location	2023 USD	Economic valuation method
Algal Based Biogas	Ranganathan & Savithri (2019)		$5.08 gasoline gallon equivalent (GGE)	Wastewater-based algal biofuels; cash flow analysis to calculate minimum selling price (MSP)
	Xin et al. (2016)		$2.81/gallon	Wastewater-based microalgae cultivation, solar drying, pyrolysis of biomass
	Zhu et al. (2016)	China	$19.72/gallon	Artificial wastewater biofuel
	Hydromentia (2017)	Florida, U.S.	$4.20/MMBTU	U.S. Energy Information Administration
Mediation of Smell/Odor Reduction	Han et al. (2022)	South Korea	$25.60/household/year	CV
	Tyndall (2009)	Iowa, U.S.	$0.25/pig/year	WTP
Regulating Climate/Carbon Sequestration	Chaikaew et al., (2017)	Suwannee, Florida, U.S.	< $2.50/household/year	WTP
	Shrestha & Alavalapati (2004)	Lake Okeechobee, Florida, U.S.	$48.44 - $113.99/household/year over 5 years; $92.98/household/year at moderate improvement level	WTP

Ecosystem Services

All results of studies were described in 2023 USD, unless otherwise noted.

Nutrient removal. ATS-AD systems provide the ES of nutrient removal, decomposing, or filtering wastes, or controlling the chemical quality of freshwater (Haines-Young & Potschin, 2012). Through the uptake of N and P, algae bioremediates fresh water of the wastes or toxic nutrients that come from anthropogenic sources (Haines-Young & Potschin, 2012). HABs have a negative effect on economies by degrading fisheries, tourism, and public health (Hoagland et al., 2002; Jin et al., 2020; Plaas & Paerl, 2021). Researchers have gathered data on the economic value of nutrient removal through various methods.

Smith and Crowder (2011) quantified part of the value of improved ecosystem function in the NRE, into which Falls Lake flows. Through a bioeconomic model, Smith and Crowder (2011) valued a 30% reduction in N loading at $3.44 million by modeling the response of N reduction and its effect on primary production. With excessive N loading comes hypoxic conditions, which affects the prey of the NC commercial blue crab fishery, and thus, the blue crab fishery rents (Smith & Crowder, 2011). Jenkins et al. (2010) valued N mitigation at $1,737 per hectare per year in the Mississippi Alluvial Valley by valuing the prevention of nitrate from entering local waterways via agricultural systems, and the removal of nitrate through the denitrification process. Jenkins et al. (2010) used benefits transfer from studies that used the N credit transfer model, the U.S. Agricultural Sector Mathematical Programming, and the simulation model Environmental Policy Integrated Climate, for agricultural N values. These values aimed to justify restoring forested wetlands (Jenkins et al., 2010). Campbell et al. (2020) valued seven ES in Maryland, U.S. via hedonic pricing. The researchers valued the price of N removal at $18.34 per kilogram to better inform decision makers on conservation land designation on the values ES provide (Campbell et al., 2020).

Molinos-Senante et al. (2011) estimated the economic valuation of P removal through wastewater recovery projects via shadow pricing while including environmental benefits. Their findings revealed a price of $62.44 per kilogram P removed. Molinos-Senante et al. (2011) showed that P removal was viable not only from a sustainability standpoint but from an economic one. Verberg (2016) valued the price of P through hedonic pricing of restoration treatments at $30.41 per kilogram of P. Sena et al. (2020) used Verberg's (2016) estimation on cleaning up P pollution and found that restoration treatment for nutrient removal was more expensive once pollution had entered the environment than participants' WTP to keep the pollution out of the environment in the first place.

Hernández-Sancho et al. (2010) estimated shadow prices for nutrients removed during wastewater treatment processes to justify investment policies in water resource management. These researchers valued the environmental benefit N and P pollution treatment at $145.64 million and $74.23 million per year, respectively (Hernández-Sancho et al., 2010).

Chaikaew et al. (2017) assessed the ES of the Suwannee River Basin of Florida, including nutrient control, or water quality. Although nutrient control proved most valuable of all the ES, the average WTP value was comparatively low at less than $2.50 per household per year, suggesting that residents in this area might have held other amenities than the environmental provisioning, regulating, and supporting services, at higher value (Chaikaew et al., 2017).

I used Campbell et al.'s (2020) value of $18.34/kg N and Verberg's (2016) value of $30.41/kg P to represent the prices of nutrient removal in my CBA analyses due to the study areas being the closest geographical matches for NC. I converted price per kilogram to pound, since Hydromentia measured N and P removal in pounds in their pilot report, and I resulted with values of $47.45 and $84.06 per pound of N and P, respectively.

Drinking water/water quality. In the case of ATS systems, algae are an example of a natural ecosystem providing the benefit of clean drinking water (Haines-Young & Potschin, 2012). Through nutrient removal, ATS-AD systems provide clean surface water for drinking, or "potable water in [a] public supply system" (Haines-Young & Potschin, 2012, CICES V5.1), such as the Falls Lake drinking water reservoir. The earth's natural and healthy ecosystems, such as forests and wetlands, filter pollutants from precipitation that collects underground, in lakes, and reservoirs and becomes drinking water for humans and other organisms (Venkataraman, 2023). However, natural ecosystems cannot keep up with the excessive anthropogenic nutrient input on their own.

Schinck et al. (2020) assessed the economic value of detection and treatment tools that filter out cyanotoxins in drinking water facilities in Quebec, Canada and found that residents, on average, were willing to pay $235 per household per year for facilities to acquire diagnostic treatment tools to avoid public water bans. These results are promising since expert opinion placed implementation of such tools at $110 per household per year at the time of the study. Since Quebec often enacts water bans due to cyanotoxins, the WTP results from this study show promise in technological implementation (Schinck et al., 2020).

Elsin et al. (2010) used benefits and function transfer approaches to quantify the economic benefits of water quality improvements for drinking water protection in the NRB in NC. Function transfer is a more sophisticated benefits transfer approach that takes the WTP values from one study area and applies it to another using coefficient values of both sites' physical features, socio-economic, and demographic characteristics (Pearce et al., 2006). Elsin et al. (2010) resulted with a mean NPV between $2.7 million to $16.6 million, valued at the time of their study, with a 30% improvement in water quality over a 30-year period, including a more conservative mean value with the function-transfer approach.

Mamun et al. (2023) used spatially explicit data on roughly 674,000 property transactions and lake water quality of 1,632 U.S. lakes to provide extensive U.S.-wide estimates for the benefits of water quality. By extrapolating this data nationally, the researchers arrived at the conclusion that property values appreciated $9.22 billion with a 10% improvement in water quality in the U.S. These results are important to guide lake property ownership policies (Mamun et al., 2023). L'Ecuyer-Sauvageau et al. (2019) conducted a CE WTP questionnaire to understand the preferences of water users in Quebec, Canada. Through this study, residents showed they value water quality for recreational activities, ecological health, and aesthetics, in that order of importance, with an average WTP value of $353/household per year towards resolving water quality issues (L'Ecuyer-Sauvageau et al., 2019).

Ureta et al. (2022) analyzed payments for ecosystem services (PES) by surveying 1,560 residents in South Carolina on their WTP for ES improvement. Ureta et al. (2022) found that residents were willing to pay a mean value of \$0-\$3.07 per month per resident for water quality improvement. All residents surveyed were willing to pay a mean value of \$0-\$6.54 per month per resident for wildlife habitat improvement. Only residents located in the upstate region were willing to pay a mean value of \$0.32 per month per resident for water availability. Ureta et al. (2022) estimated ES benefits to this community from \$4.7-\$6.4 million per month, indicating PES could substantially develop conservation programs financially.

Recreation. ES also provide cultural value through physical and experiential outdoor interactions with living ecosystems, or through recreational activities, such as fishing, hunting, diving, and hiking (Haines-Young & Potschin, 2012). Water-based recreation also includes indirect water contact through activities such as walking or running around lakes or along rivers, viewing water from a distance, and other activities pertaining to water bodies, or blue spaces, that can provide stress relief (McDougall et al., 2020). ATS-AD systems provide enhanced recreational activities as they bioremediate nutrient-rich freshwaters and avoid HABs in fresh waterbodies, such as Falls Lake.

Nelson et al. (2015) developed a mail CV survey that captured the WTP of households in Utah, U.S. to use Utah's waters recreationally. Recreation users were willing to pay up to \$208 per year to prevent water pollution on recreational water. These data can support the benefits of nutrient regulation in the state (Nelson et al., 2015). McDougall et al. (2020) used CV to survey the public's preference for protecting lake water quality in recreational views at Loch Lomond and Loch Leven in Scotland and found the average WTP was $17.43 per household per year at Loch Lomond and $12.19 at Loch Leven, indicating that the Scottish public was willing to pay for lake water quality. This research can aid in cost-effective and optimal water management policies (McDougall et al., 2020).

Zhang and Sohngen (2018) surveyed 767 recreational angler residents using Lake Erie, in Ohio through CE and found the first monetary quantification of the impacts HABs have on U.S. recreational anglers. Anglers from the study were willing to pay $9.63 to $12.04 more per fishing trip for one less mile of navigating through HABs to a site. Visitors to the Apalachicola River region of Florida were surveyed by Shrestha et al. (2007) to find out what value was placed on visiting natural sites, such as forests, parks, and preserves. The authors found that visitors were willing to pay on average $108.03 per visit per day for nature-based recreation. These results can help natural resource management better protect this region's unique ecosystems (Shrestha et al., 2007).

Freshwater provision. While CICES does not include an ES code for freshwater provision, Castro et al. (2016) collected WTP data on both freshwater provision and water regulation and defined them as provisioning and regulating services, respectively. Holtgrieve et al. (2011) defined water regulation as the "regulation of physical, chemical, [and] biological conditions" (CICES V5.1, 2011) of water; provisioning services as "nutritional, non-nutritional material, and energetic outputs from living systems as well as abiotic outputs" (p. 10); and regulation and maintenance services as "the ways in which organisms can mediate or moderate the ambient environment that affects human health, safety or comfort, together with abiotic equivalents" (p. 10). Freshwater provision could mean providing water for more than drinking. This ES was the most economically important to Oklahoma residents, according to Castro et al. (2016), reporting an average WTP of $5.42 per household annually to preserve. Through SP and CE methods, Moeltner et al. (2023) measured the value of avoiding airborne toxins caused by red tide in Florida, U.S. Results suggest that residents valued prediction of air quality and algae bloom location. Residents were willing to pay $17.37 per household per year, or $14.5 million per year, which can represent the WTP for costs associated with forecasting HABs (Moeltner et al., 2023).

Algae-based biogas. ATS-AD systems provide the ES of algal biomass used for energy generation (Haines-Young & Potschin, 2012). Algal biomass productivity flourishes from the uptake of N and P and is harvested regularly with the maintenance of ATS-AD systems. The price range of algal-based biomass is wide due to different algal biomass production methods, the use of different algae species, and the various processes of creating biogas from biomass. Researchers have gathered data on the economic value of algal-based biogas through various methods.

Ranganathan and Savithri (2019) used simulation to calculate the price of wastewater-based algal biofuels using HTL processing and resulted with the minimum selling price of $5.08 per gasoline gallon equivalent (GGE). Xin et al. (2016) conducted an analysis of wastewater-based algal biofuel production via solar drying of biomass and pyrolysis, which creates biogas through heating at temperatures between 400 and 1000 degrees Celsius without oxygen present. The authors arrived at the selling price of $2.81

per gallon. Zhu et al. (2016) cultivated wastewater microalgae in a pilot-scale system using photobioreactors in southern China. The goal of the research was to assess net energy ratios of oil production and the cost of bio-oil, which resulted in \$5.21 per liter, or \$19.72 per gallon, in this pilot-scale set up (Zhu et al., 2016). Dong et al. (2016) priced algal biogas at \$12.49 GGE, and Davis et al. (2016) at a range from \$5.48-\$5.66.

I calculated the benefit variable of algal biogas from Hydromentia's (2017b) report and sources. Hydromentia found that 674 tons of compost, or growing material, per year could be harvested from the Falls Lake pilot ATS system in a 25-MGD system. According to Table 11 (adapted from Hydromentia's report), the optimal mean annual algal production of a 10-acre facility was 19 grams, measured dry, per square meter per day (dry-g/m^2-day) (Hydromentia, 2017b). One acre is equivalent to 4,046.86 m^2, therefore, 10 acres is 40,468.6 m^2. Annual dry production of algal biomass was 280,649,741 g/year (Table 12). The cost to dispose of the algal biomass in this form at the landfill was \$43.58 per ton. At the time of the report, several value-added products were determined for the algal biomass produced from the ATS system (Figure 16). Hydromentia priced methane gas at \$4.20 per Metric Million British Thermal Units (MMBTU) based on an average taken from the U.S. Energy Information Administration (M Zivojnovich, personal communication, August 22, 2023). Table 12 shows the steps I took to calculate an annual benefit value of \$35,978 for electricity production based on facility size and dry algal biomass production weight (M. Zivojnovich, personal communication, August 22, 2023). The costs of machinery, operations, and maintenance of an AD were not included in this calculation.

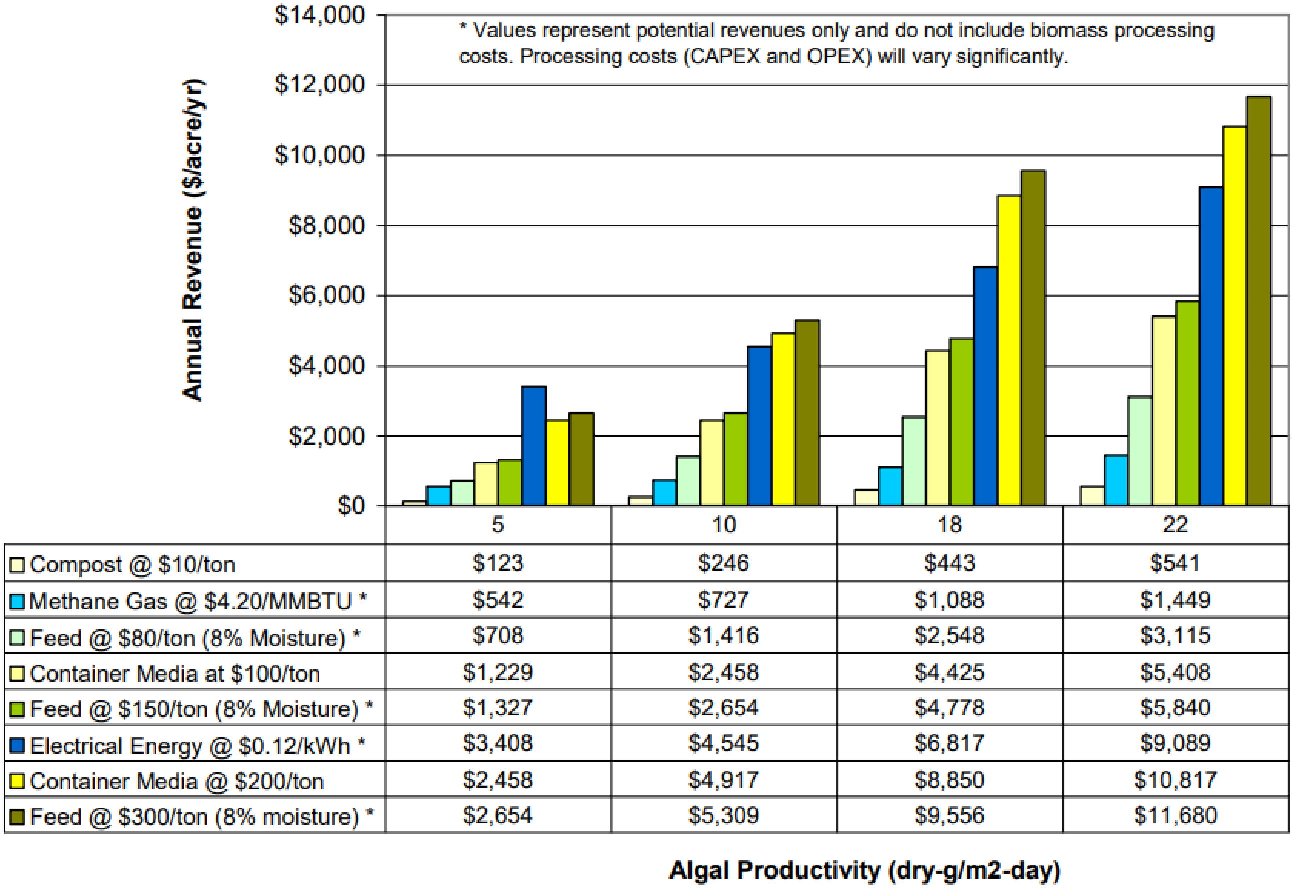
Table 11. Algal biomass and compost production at Falls Lake ATS facilities.

Scenario	Data Source	Facility size (acres)	Mean Annual Algal Production (dry-g/m2-day)	Compost Production Model	Finished Compost (tons per facility-yr)
High Rainfall Years	CY2016 Actual Pilot Performance Data Dec 29, 2015 - Dec 20, 2016	4	19	Optimal	270
		4	19	Conservative	231
High Rainfall Years	CY2016 Actual Pilot Performance Data Dec 29, 2015 - Dec 20, 2016	10	19	Optimal	674
		10	19	Conservative	578

(Hydromentia, 2017b).

Mediation of smell/odor reduction. Smell reduction for inhabitants is valued for its harmful or stressful effects and the costs associated with it (Haines-Young & Potschin, 2012). Examples of odor reduction in CBA are the costs associated with shelter belts or decomposers that remove rotting material (Haines-Young & Potschin, 2012).

Han et al. (2022) estimated South Korean households' WTP for government funding to address the issue of livestock odor through a CV method. The average household was willing to pay $25.60 annually for government subsidies to improve bad odor conditions from livestock production, which can be used to support better livestock management practices to alleviate conflicts between producers and residents (Han et al., 2022). Tyndall (2009) surveyed Iowa hog producers' WTP and demand for shelterbelts, or forested areas surrounding farms, which can alleviate malodor from livestock production. 75% of producers surveyed expressed a mean WTP of $0.25 per pig annually, while 13% of those surveyed were willing to pay more. Younger pig producers



* Values represent potential revenues only and do not include biomass processing costs. Processing costs (CAPEX and OPEX) will vary significantly.

	5	10	18	22
■ Compost @ $10/ton	$123	$246	$443	$541
■ Methane Gas @ $4.20/MMBTU *	$542	$727	$1,088	$1,449
■ Feed @ $80/ton (8% Moisture) *	$708	$1,416	$2,548	$3,115
■ Container Media at $100/ton	$1,229	$2,458	$4,425	$5,408
■ Feed @ $150/ton (8% Moisture) *	$1,327	$2,654	$4,778	$5,840
■ Electrical Energy @ $0.12/kWh *	$3,408	$4,545	$6,817	$9,089
■ Container Media @ $200/ton	$2,458	$4,917	$8,850	$10,817
■ Feed @ $300/ton (8% moisture) *	$2,654	$5,309	$9,556	$11,680

Figure 16. Proposed annual revenues of harvested algal biomass.

(Hydromentia, 2017b).

Table 12. Algal biogas price calculation.

Units	Description	Value
Algal Production Rate (dry-g/m^2/day)	Based on a 25-MGD 10-acre facility	19
Algal Production Rate (dry-g/day)	Multiply by 40,468.60 to get dry g/day	768,903.40
Algal Production Rate (dry-g/year)	Multiple by 365 to get annual dry production in g/year	280,649,741
Facility Annual Dry Production (kg)	Divide by 1,000 to get annual dry production in kg	280,650
Facility Total VS (kg)	Assume 50% (VS) to get dry weight VS in kg	140,325
Facility VS Destroyed (kg)	Assume 50% of available VS destroyed in digester	70,162
Facility Gas Yield (m^3)	Expected gas yield of 1.0 m^3/ kg. VS destroyed to get gas yield in cubic meters (m^3)	70,162
Gas Value (BTU)	Gas Value 65% methane from biofuels yields 600 BTU/standard cubic feet (SCF). 23,312 BTU/standard cubic meter (SCM). Multiply gas production in SCM by 23,312 to get BTU production.	1,635,626,691
Gas Value (MMBTU)	Divide BTU by 1,000,000 to get MMBTU production per facility	1,636
Gross Energy Value (kWh/year)	Multiply MMBTU by 292.70 to get kWh/year	478,748
Net Energy Value (kWh/yearr)	Methane to Electricity Conversion Efficiency 25%	359,061
Electrical Energy Value ($)	Assume electrical value of $0.10/kWh	$35,906

Note. Typical digester gas, with a methane concentration of 65%, contains about 600 BTU of energy per cubic foot. 1,012 BTU/standard cubic foot methane (Chiton et al., 1973, p. 9). 10-acre facility; 4,046.86 square meters per acre; 110,000 BTU per gallon biofuel (butanol); 3,500 assumed gallon of biofuel per acre; 385,000,000 total BTU production per acre; 385 total MMBTU per acre. (M. Zivojnovich, personal communication, 2023, August 22).

expressed the knowledge and assurance that shelterbelts worked and showed higher WTP for their costs and maintenance (Tyndall, 2009).

Regulating climate/carbon sequestration. Climate regulating ES reduce greenhouse gas concentrations and lower or avoid costs associated with climate change (Haines-Young & Potschin, 2012). Lakes and forests are examples of C sinks, or sequestration habitats, that help reduce the overall greenhouse effect (Tranvik et al., 2009; Liski, et al., 2006).

Chaikaew et al. (2017) assessed residents' WTP at less than \$2.50 per household annually for the climate regulation, or C sequestration, ES in the Suwannee River Basin of Florida, U.S. Chaikaew et al. (2017) suggested the very low result, compared to similar studies, may indicate that residents have other priorities competing for their money other than environmental protection.

Shrestha and Alavalapati (2004) estimated the public's valuation of silvopasture, which integrates forestry with livestock operations and has potential in C sequestration benefits. Using a SP methodology, Shrestha and Alavalapati (2004) found households were willing to pay between \$48.44 and \$113.99 annually for five years for the ES of C sequestration. These estimations provide information to policy makers to implement and subsidize silvopasture in the Lake Okeechobee watershed (Shrestha & Alavalapati, 2004).

I did not include mediation of smell and regulating climate change in my CBAs due to not finding existing research in similar enough regions to NC.

Ecosystem Services in the Kiamichi River Watershed

The Kiamichi River watershed flows through Choctaw, Pushmataha, and LeFlore counties of OK. The demographics of those counties are shown in Tables 13 and 14. The land use types of the watershed are represented in Table 15.

Table 13. Race demographics of Kiamichi River Watershed.

Race and Hispanic Origin	Choctaw	Pushmataha	LeFlore
White	61.3%	72.2%	75.5%
Black or African American	9.9%	1.3%	2.2%
American Indian and Alaska Native	19.4%	19%	14.8%
Asian	0.7%	0.6%	0.9%
Native Hawaiian and Other Pacific Islander	-	-	0.2%
Two or More Races	8.7%	6.9%	6.4%
Hispanic or Latino	5.8%	4.7%	7.9%
White, not Hispanic or Latino	57.9%	69.2%	69.5%

(U.S. Census Bureau, 2022).

Table 14. Demographics of Kiamichi River Watershed

County	Population[a]	Population[b]	Households	Housing[c]	High school[d]	College[e]	Income[f]
Choctaw	14,358	18.4	5,756	$102,100	83.70%	14.20%	$38,854
Pushmataha	10,769	7.7	4,189	$89,100	85.50%	15.10%	$40,721
LeFlore	48,907	30.8	17,623	$96,700	84.2%	15.30%	$43,049

Note.

[a] July 2022 estimate

[b] per square mile in 2020

[c] Median value of owner-occupied housing units, 2017-2021

[d] High school graduate or higher, percent of persons age 25+, 2017-2021

[e] Bachelor's degree or higher, percent of persons age 25+, 2017-2021

[f] Median household income (in 2021 dollars), 2017-2021

(U.S. Census Bureau, 2022).

Table 15. Land use type of Kiamichi River Watershed.

Land Cover Type	Percentage
Water/Wetland	4%
Urban/Barren	3.30%
Forest	63.60%
Shrub/Grassland/Pasture	29%
Cropland	0.10%

Note. Based on 2006 data (Castro et al., 2014).

Many of the WTP values I used to calculate benefits in my economic analysis were from Castro et al.'s (2016) research on the ES of the Kiamichi River watershed in Oklahoma. I used Castro et al.'s (2016) study as a foundation for my research in Falls Lake, NC due to the similarity in geographic location. Both the Kiamichi River watershed and the hypothetical ATS-AD system of Falls Lake provide several of the same ES classified by Holtgrieve et al.'s (2011) CICES classification system.

Castro et al. (2016) calculated average WTP values for ES based on the WTP survey results from a variety of stakeholders, including business tourists, visiting tourists, watershed residents, experts, and city residents (Table 16). I used the average WTP values for the economic CBA of the ATS-AD system for the following ES: water quality, recreation, cultural heritage, air quality, and freshwater provision. Based on Castro et al.'s (2016) survey dates, I converted values from June 2013 to April 2023 dollars using the Consumer Price Index Inflation Calculator (CPI Inflation Calculator, n.d.).

Table 16. Valued ES in Kiamichi River Watershed.

Citation	Ecosystem service	Valuation at time of original study	Valuation in April 2023	Economic valuation method
Castro et al. (2016)	Drinking water/Water quality	$9.59/household/year	$12.46	WTP
	Recreation	$4.99/household/year	$6.48	WTP
	Cultural Heritage	$2.03/household/year	$2.64	WTP
	Air Quality	$3.49/household/year	$4.53	WTP
	Freshwater Provisioning	$4.17/household/year	$5.42	WTP

(Castro et al., 2016).

Percentage of Households for Benefits Transfer in Falls Lake Watershed

In order to better design a benefits transfer approach, I consulted with Lou Nadeau, Ph.D., a senior economist at and vice president of Eastern Research Group, Inc, a consultation and research company in the environment and sustainability services sector. As a result of conversations with Dr. Nadeau and my thesis director, I decided to provide three economic CBA scenarios based on a high, medium, and low range for the percentage of FLW households that would experience the benefits indicated by the ES values represented in the literary review. Comparisons of different populations of households, one in the FLW of NC and another in the Kiamichi Watershed of OK, cannot be guaranteed 100% transferable. Populations of households do not benefit equally due to differences in geographic location, class, income, and other demographics. By applying a range of percentages, high/75%, medium/50%, and low/25%, at the household level, I portrayed at which percentage the benefits outweigh the costs (L. Nadeau, personal communication, August 14, 2023). The upper bound, or 75% of the population of FLW represented 27,868 households. The middle and lower bounds of the population at 50 and

25% represented 18,579 and 9,289, respectively. The low percentage of 25% was acceptable since OK and NC are similar but not identical, containing slightly different demographics and ecosystems. If the benefits outweigh the costs at the low percentage of 25%, the analysis can support a policy argument for using the ATS-AD technology (L. Nadeau, personal communication, August 14, 2023). Nutrients captured and biogas generation values remained the same despite the varying scenarios since they are calculated by price per pound and are not dependent on number of households.

Chapter III

Results

The first financial CBA without an AD component resulted in a negative NPV of $12,896,923 (Table 17). The benefits did not outweigh the costs in this scenario. In the first year of the project, 2023, there were no benefits of N and P captured because algal productivity and performance did not stabilize until after the first year of system operation (Hydromentia, 2017b). In 2023, there were no costs of algal disposition to the landfill and operations and maintenance for the same reason. Capital costs, mainly construction of flow-way and influent pump installation, were only for the first year (Hydromentia, 2017b).

Financial Cost-Benefit Analysis of ATS-AD System

The second financial CBA with the AD component resulted in a negative NPV of $4,911,406 (Table 18). The benefits still did not outweigh the costs in this scenario, although the NPV was closer to breaking even due to the algal biogas generation offsetting some of the costs. Like the first financial CBA scenario, the benefits of captured nutrients and algal biogas generation did not begin until the project's second year, 2024. The costs of operations and maintenance began the second year, and capital costs were only for the first year of the project, 2023. There were no costs of disposing algal biomass to the landfill in this scenario since algal biomass was used for energy generation.

Table 17. Financial CBA of ATS without AD.

	2023	2024	2025	2026	2027	2028	2029	2030	2031	2032	2033	2034
Year	1	2	3	4	5	6	7	8	9	10	11	12
Benefits												
N captured in lbs * $/lb of N	$0	$338,793	$338,793	$338,793	$338,793	$338,793	$338,793	$338,793	$338,793	$338,793	$338,793	$338,793
P captured in lbs * $/lb of P	$0	$105,916	$105,916	$105,916	$105,916	$105,916	$105,916	$105,916	$105,916	$105,916	$105,916	$105,916
Total Benefits												
Costs												
Capital cost	$7,740,229	$0	$0	$0	$0	$0	$0	$0	$0	$0	$0	$0
Algal disposition net Cost (landfill)	$0	$374,550	$374,550	$374,550	$374,550	$374,550	$374,550	$374,550	$374,550	$374,550	$374,550	$374,550
Operations and maintenance	$0	$346,049	$346,049	$346,049	$346,049	$346,049	$346,049	$346,049	$346,049	$346,049	$346,049	$346,049
Total Costs												
Benefits - Costs												

	2035	2036	2037	2038	2039	2040	2041	2042	2043	Total	NPV
Year	13	14	15	16	17	18	19	20	21		
Benefits											
N captured in lbs * $/lb of N	$338,793	$338,793	$338,793	$338,793	$338,793	$338,793	$338,793	$338,793	$338,793	$6,775,860	$5,134,863
P captured in lbs * $/lb of P	$105,916	$105,916	$105,916	$105,916	$105,916	$105,916	$105,916	$105,916	$105,916	$2,118,312	$1,605,293
Total Benefits										$8,894,172	$8,651,918
Costs											
Capital cost	$0	$0	$0	$0	$0	$0	$0	$0	$0	$7,740,229	$7,529,406
Algal disposition net cost (landfill)	$374,550	$374,550	$374,550	$374,550	$374,550	$374,550	$374,550	$374,550	$374,550	$7,490,991	$5,522,180
Operations and maintenance	$346,049	$346,049	$346,049	$346,049	$346,049	$346,049	$346,049	$346,049	$346,049	$6,920,988	$5,101,988
Total Costs										$22,152,209	$21,548,841
Benefits - Costs										-$13,258,037	-$12,896,923

Table 18. Financial CBA of ATS with AD.

	2023	2024	2025	2026	2027	2028	2029	2030	2031	2032	2033
Year	1	2	3	4	5	6	7	8	9	10	11
Benefits											
N captured in lbs * $/lb of N	$0	$338,793	$338,793	$338,793	$338,793	$338,793	$338,793	$338,793	$338,793	$338,793	$338,793
P captured in lbs * $/lb of P	$0	$105,916	$105,916	$105,916	$105,916	$105,916	$105,916	$105,916	$105,916	$105,916	$105,916
Algae based biogas	$0	$35,906	$35,906	$35,906	$35,906	$35,906	$35,906	$35,906	$35,906	$35,906	$35,906
Total Benefits											
Costs											
Capital cost	$ 7,740,229	$0	$0	$0	$0	$0	$0	$0	$0	$0	$0
Operations and maintenance	$0	$346,049	$346,049	$346,049	$346,049	$346,049	$346,049	$346,049	$346,049	$346,049	$346,049
Total Costs											
Benefits - Costs											

	2034	2035	2036	2037	2038	2039	2040	2041	2042	2043	Total	NPV
Year	12	13	14	15	16	17	18	19	20	21		
Benefits												
N captured in lbs * $/lb of N	$338,793	$338,793	$338,793	$338,793	$338,793	$338,793	$338,793	$338,793	$338,793	$338,793	$6,775,860	$4,995,002
P captured in lbs * $/lb of P	$105,916	$105,916	$105,916	$105,916	$105,916	$105,916	$105,916	$105,916	$105,916	$105,916	$2,118,312	$1,561,569
Algae based biogas	$35,906	$35,906	$35,906	$35,906	$35,906	$35,906	$35,906	$35,906	$35,906	$35,906	$718,120	$529,381
Total Benefits											$9,612,292	$9,350,479
Costs												
Capital cost	$0	$0	$0	$0	$0	$0	$0	$0	$0	$0	$7,740,229	$7,529,406
Operations and maintenance	$346,049	$346,049	$346,049	$346,049	$346,049	$346,049	$346,049	$346,049	$346,049	$346,049	$6,920,988	$5,101,988
Total Costs											$14,661,217	$14,261,885
Benefits - Costs											-$5,048,925	-$4,911,406

Economic Cost- Benefit Analysis of ATS-AD System

I depicted three economic CBA scenarios to provide a range in valuing ES from Castro et al. (2016) in relation to the number of households within the FLW. The NPV results are portrayed in Table 19 based on the high, medium, and low scenarios. Tables 20, 21, and 22 portray the high, medium, and low scenarios of my economic CBAs, respectively. All three scenarios showed the benefits of the ATS-AD system outweigh the costs with NPVs of $12,183,341 at 75% of households, $6,485,401 at 50% of households, and $786,691 at 25% of households (Tables 19, 20, 21, & 22). An analysis of 15% of the population revealed a negative NPV of $1,492,180 (Table 23).

Table 19. CBA NPVs scenarios based on percentages of households.

Total Households in Falls Lake Watershed	% of Population	Number of Households	CBA NPV
37,157	100	37,157	$17,881,594
	75	27,868	$12,183,341
	50	18,579	$6,485,401
	25	9,289	$786,691
	15	5,574	-$1,492,180
	10	3,716	-$2,631,922
	5	1,858	-$3,771,664

The Economic CBA turned the NPV negative between 15 and 25% of the population. This means that if roughly a minimum of 25% of households in the FLW expressed the values from Castro et al.'s research, the project would benefit from the ATS-AD system.

Table 20. Economic ATS-AD CBA at 75% of FLW households.

	2023	2024	2025	2026	2027	2028	2029	2030	2031	2032	2033	2034
Year	1	2	3	4	5	6	7	8	9	10	11	12
Benefits												
N captured in lbs * $/lb of N	$0	$338,793	$338,793	$338,793	$338,793	$338,793	$338,793	$338,793	$338,793	$338,793	$338,793	$338,793
P captured in lbs * $/lb of P	$0	$105,916	$105,916	$105,916	$105,916	$105,916	$105,916	$105,916	$105,916	$105,916	$105,916	$105,916
Drinking water	$0	$347,232	$347,232	$347,232	$347,232	$347,232	$347,232	$347,232	$347,232	$347,232	$347,232	$347,232
Recreation	$0	$180,583	$180,583	$180,583	$180,583	$180,583	$180,583	$180,583	$180,583	$180,583	$180,583	$180,583
Cultural heritage	$0	$73,571	$73,571	$73,571	$73,571	$73,571	$73,571	$73,571	$73,571	$73,571	$73,571	$73,571
Air quality	$0	$126,241	$126,241	$126,241	$126,241	$126,241	$126,241	$126,241	$126,241	$126,241	$126,241	$126,241
Freshwater provision	$0	$151,043	$151,043	$151,043	$151,043	$151,043	$151,043	$151,043	$151,043	$151,043	$151,043	$151,043
Algae based biogas	$0	$35,906	$35,906	$35,906	$35,906	$35,906	$35,906	$35,906	$35,906	$35,906	$35,906	$35,906
Total Benefits												
Costs												
Capital cost	$7,740,229	$0	$0	$0	$0	$0	$0	$0	$0	$0	$0	$0
Operations and maintenance	$0	$346,049	$346,049	$346,049	$346,049	$346,049	$346,049	$346,049	$346,049	$346,049	$346,049	$346,049
Total Costs												
Benefits - Costs												

	2035	2036	2037	2038	2039	2040	2041	2042	Total	NPV
Year	14	15	16	17	18	19	20	21		
Benefits										
N captured in lbs * $/lb of N	$338,793	$338,793	$338,793	$338,793	$338,793	$338,793	$338,793	$338,793	$6,775,860	$4,995,002
P captured in lbs * $/lb of P	$105,916	$105,916	$105,916	$105,916	$105,916	$105,916	$105,916	$105,916	$2,118,312	$1,561,569
Drinking water	$347,232	$347,232	$347,232	$347,232	$347,232	$347,232	$347,232	$347,232	$6,944,640	$5,119,423
Recreation	$180,583	$180,583	$180,583	$180,583	$180,583	$180,583	$180,583	$180,583	$3,611,660	$2,662,430
Cultural heritage	$73,571	$73,571	$73,571	$73,571	$73,571	$73,571	$73,571	$73,571	$1,471,420	$1,084,696
Air quality	$126,241	$126,241	$126,241	$126,241	$126,241	$126,241	$126,241	$126,241	$2,524,820	$1,861,237
Freshwater provision	$151,043	$151,043	$151,043	$151,043	$151,043	$151,043	$151,043	$151,043	$3,020,860	$2,22,906
Algae based biogas	$35,906	$35,906	$35,906	$35,906	$35,906	$35,906	$35,906	$35,906	$718,120	$529,381
Total Benefits									$27,185,692	$26,445,226
Costs										
Capital cost	$0	$0	$0	$0	$0	$0	$0	$0	$7,740,229	$7,529,406
Operations and maintenance	$346,049	$346,049	$346,049	$346,049	$346,049	$346,049	$346,049	$346,049	$6,920,988	$5,101,988
Total Costs									$14,661,217	$14,261,885
Benefits - Costs									$12,524,475	$12,183,341

Table 21. Economic ATS-AD CBA at 50% of FLW households.

	2023	2024	2025	2026	2027	2028	2029	2030	2031	2032	2033	2034
Year	1	2	3	4	5	6	7	8	9	10	11	12
Benefits												
N captured in lbs * $/lb of N	$0	$338,793	$338,793	$338,793	$338,793	$338,793	$338,793	$338,793	$338,793	$338,793	$338,793	$338,793
P captured in lbs * $/lb of P	$0	$105,916	$105,916	$105,916	$105,916	$105,916	$105,916	$105,916	$105,916	$105,916	$105,916	$105,916
Drinking water	$0	$231,494	$231,494	$231,494	$231,494	$231,494	$231,494	$231,494	$231,494	$231,494	$231,494	$231,494
Recreation	$0	$120,392	$120,392	$120,392	$120,392	$120,392	$120,392	$120,392	$120,392	$120,392	$120,392	$120,392
Cultural heritage	$0	$49,049	$49,049	$49,049	$49,049	$49,049	$49,049	$49,049	$49,049	$49,049	$49,049	$49,049
Air quality	$0	$84,163	$84,163	$84,163	$84,163	$84,163	$84,163	$84,163	$84,163	$84,163	$84,163	$84,163
Freshwater provision	$0	$100,698	$100,698	$100,698	$100,698	$100,698	$100,698	$100,698	$100,698	$100,698	$100,698	$100,698
Algae based biogas	$0	$35,906	$35,906	$35,906	$35,906	$35,906	$35,906	$35,906	$35,906	$35,906	$35,906	$35,906
Total Benefits												
Costs												
Capital cost	$7,740,229	$0	$0	$0	$0	$0	$0	$0	$0	$0	$0	$0
Operations and maintenance	$0	$346,049	$346,049	$346,049	$346,049	$346,049	$346,049	$346,049	$346,049	$346,049	$346,049	$346,049
Total Costs												
Benefits - Costs												

	2035	2036	2037	2038	2039	2040	2041	2042	2043	Total	NPV
Year	13	14	15	16	17	18	19	20	21		
Benefits											
N captured in lbs * $/lb of N	$338,793	$338,793	$338,793	$338,793	$338,793	$338,793	$338,793	$338,793	$338,793	$6,775,860	$4,995,002
P captured in lbs * $/lb of P	$105,916	$105,916	$105,916	$105,916	$105,916	$105,916	$105,916	$105,916	$105,916	$2,118,312	$1,561,569
Drinking water	$231,494	$231,494	$231,494	$231,494	$231,494	$231,494	$231,494	$231,494	$231,494	$4,629,887	$3,413,042
Recreation	$120,392	$120,392	$120,392	$120,392	$120,392	$120,392	$120,392	$120,392	$120,392	$2,407,838	$1,775,001
Cultural heritage	$49,049	$49,049	$49,049	$49,049	$49,049	$49,049	$49,049	$49,049	$49,049	$980,971	$723,149
Air quality	$84,163	$84,163	$84,163	$84,163	$84,163	$84,163	$84,163	$84,163	$84,163	$1,683,257	$1,240,857
Freshwater provision	$100,698	$100,698	$100,698	$100,698	$100,698	$100,698	$100,698	$100,698	$100,698	$2,013,964	$1,484,646
Algae based biogas	$35,906	$35,906	$35,906	$35,906	$35,906	$35,906	$35,906	$35,906	$35,906	$718,120	$529,381
Total Benefits										$21,328,209	$20,747,285
Costs											
Capital cost	$0	$0	$0	$0	$0	$0	$0	$0	$0	$7,740,229	$7,529,406
Operations and maintenance	$346,049	$346,049	$346,049	$346,049	$346,049	$346,049	$346,049	$346,049	$346,049	$6,920,988	$5,101,988
Total Costs										$14,661,217	$14,261,885
Benefits - Costs										$6,666,992	$6,485401

Table 22. Economic ATS-AD CBA at 25% of FLW households.

	2023	2024	2025	2026	2027	2028	2029	2030	2031	2032	2033	2034
Year	1	2	3	4	5	6	7	8	9	10	11	12
Benefits												
N captured in lbs * $/lb of N	$0	$338,793	$338,793	$338,793	$338,793	$338,793	$338,793	$338,793	$338,793	$338,793	$338,793	$338,793
P captured in lbs * $/lb of P	$0	$105,916	$105,916	$105,916	$105,916	$105,916	$105,916	$105,916	$105,916	$105,916	$105,916	$105,916
Drinking water	$0	$115,741	$115,741	$115,741	$115,741	$115,741	$115,741	$115,741	$115,741	$115,741	$115,741	$115,741
Recreation	$0	$60,193	$60,193	$60,193	$60,193	$60,193	$60,193	$60,193	$60,193	$60,193	$60,193	$60,193
Cultural heritage	$0	$24,523	$24,523	$24,523	$24,523	$24,523	$24,523	$24,523	$24,523	$24,523	$24,523	$24,523
Air quality	$0	$42,079	$42,079	$42,079	$42,079	$42,079	$42,079	$42,079	$42,079	$42,079	$42,079	$42,079
Freshwater provision	$0	$50,346	$50,346	$50,346	$50,346	$50,346	$50,346	$50,346	$50,346	$50,346	$50,346	$50,346
Algae based biogas	$0	$35,906	$35,906	$35,906	$35,906	$35,906	$35,906	$35,906	$35,906	$35,906	$35,906	$35,906
Total Benefits												
Costs												
Capital cost	$7,740,229	$0	$0	$0	$0	$0	$0	$0	$0	$0	$0	$0
Operations and maintenance	$0	$346,049	$346,049	$346,049	$346,049	$346,049	$346,049	$346,049	$346,049	$346,049	$346,049	$346,049
Total Costs												
Benefits - Costs												

	2035	2036	2037	2038	2039	2040	2041	2042	2043	Total	NPV
Year	13	14	15	16	17	18	19	20	21		
Benefits											
N captured in lbs * $/lb of N	$338,793	$338,793	$338,793	$338,793	$338,793	$338,793	$338,793	$338,793	$338,793	$6,775,860	$4,995,002
P captured in lbs * $/lb of P	$105,916	$105,916	$105,916	$105,916	$105,916	$105,916	$105,916	$105,916	$105,916	$2,118,312	$1,561,569
Drinking water	$115,741	$115,741	$115,741	$115,741	$115,741	$115,741	$115,741	$115,741	$115,741	$2,314,819	$1,706,429
Recreation	$60,193	$60,193	$60,193	$60,193	$60,193	$60,193	$60,193	$60,193	$60,193	$1,203,854	$887,453
Cultural heritage	$24,523	$24,523	$24,523	$24,523	$24,523	$24,523	$24,523	$24,523	$24,523	$490,459	$361,555
Air quality	$42,079	$42,079	$42,079	$42,079	$42,079	$42,079	$42,079	$42,079	$42,079	$841,583	$620,395
Freshwater provision	$50,346	$50,346	$50,346	$50,346	$50,346	$50,346	$50,346	$50,346	$50,346	$1,006,928	$742,283
Algae based biogas	$35,906	$35,906	$35,906	$35,906	$35,906	$35,906	$35,906	$35,906	$35,906	$718,120	$529,381
Total Benefits										$15,469,935	$15,048,575
Costs											
Capital cost	$0	$0	$0	$0	$0	$0	$0	$0	$0	$7,740,229	$7,529,406
Operations and maintenance	$346,049	$346,049	$346,049	$346,049	$346,049	$346,049	$346,049	$346,049	$346,049	$6,920,988	$5,101,988
Total Costs										$14,661,217	$14,261,885
Benefits - Costs										$808,718	$786,691

Table 23. Economic ATS-AD CBA at 15% of FLW households.

	2023	2024	2025	2026	2027	2028	2029	2030	2031	2032	2033	2034
Year	1	2	3	4	5	6	7	8	9	10	11	12
Benefits												
N captured in lbs * $/lb of N	$0	$338,793	$338,793	$338,793	$338,793	$338,793	$338,793	$338,793	$338,793	$338,793	$338,793	$338,793
P captured in lbs * $/lb of P	$0	$105,916	$105,916	$105,916	$105,916	$105,916	$105,916	$105,916	$105,916	$105,916	$105,916	$105,916
Drinking water	$0	$46,301	$46,301	$46,301	$46,301	$46,301	$46,301	$46,301	$46,301	$46,301	$46,301	$46,301
Recreation	$0	$24,080	$24,080	$24,080	$24,080	$24,080	$24,080	$24,080	$24,080	$24,080	$24,080	$24,080
Cultural heritage	$0	$9,810	$9,810	$9,810	$9,810	$9,810	$9,810	$9,810	$9,810	$9,810	$9,810	$9,810
Air quality	$0	$16,833	$16,833	$16,833	$16,833	$16,833	$16,833	$16,833	$16,833	$16,833	$16,833	$16,833
Freshwater provision	$0	$20,141	$20,141	$20,141	$20,141	$20,141	$20,141	$20,141	$20,141	$20,141	$20,141	$20,141
Algae based biogas	$0	$35,906	$35,906	$35,906	$35,906	$35,906	$35,906	$35,906	$35,906	$35,906	$35,906	$35,906
Total Benefits												
Costs												
Capital cost	$7,740,229	$0	$0	$0	$0	$0	$0	$0	$0	$0	$0	$0
Operations and maintenance	$0	$346,049	$346,049	$346,049	$346,049	$346,049	$346,049	$346,049	$346,049	$346,049	$346,049	$346,049
Total Costs												
Benefits - Costs												

	2035	2036	2037	2038	2039	2040	2041	2042	2043	Total	NPV
Year	13	14	15	16	17	18	19	20	21		
Benefits											
N captured in lbs * $/lb of N	$338,793	$338,793	$338,793	$338,793	$338,793	$338,793	$338,793	$338,793	$338,793	$6,775,860	$4,995,002
P captured in lbs * $/lb of P	$105,916	$105,916	$105,916	$105,916	$105,916	$105,916	$105,916	$105,916	$105,916	$2,118,312	$1,561,569
Drinking water	$46,301	$46,301	$46,301	$46,301	$46,301	$46,301	$46,301	$46,301	$46,301	$1,389,041	$1,023,968
Recreation	$24,080	$24,080	$24,080	$24,080	$24,080	$24,080	$24,080	$24,080	$24,080	$722,390	$532,529
Cultural heritage	$9,810	$9,810	$9,810	$9,810	$9,810	$9,810	$9,810	$9,810	$9,810	$294,307	$216,956
Air quality	$16,833	$16,833	$16,833	$16,833	$16,833	$16,833	$16,833	$16,833	$16,833	$505,004	$372,277
Freshwater provision	$20,141	$20,141	$20,141	$20,141	$20,141	$20,141	$20,141	$20,141	$20,141	$604,222	$445,418
Algae based biogas	$35,906	$35,906	$35,906	$35,906	$35,906	$35,906	$35,906	$35,906	$35,906	$718,120	$529,381
Total Benefits										$13,127,256	$12,769,705
Costs											
Capital cost	$0	$0	$0	$0	$0	$0	$0	$0	$0	$7,740,229	$7,529,406
Operations and maintenance	$346,049	$346,049	$346,049	$346,049	$346,049	$346,049	$346,049	$346,049	$346,049	$6,920,988	$5,101,988
Total Costs										$14,661,217	$14,261,885
Benefits - Costs										-$1,533,961	-$1,492,180

Chapter IV

Discussion

The objective of my research was to demonstrate to other researchers, agriculturalists, municipalities, engineers, and policy makers the effectiveness of ATS-AD systems in environmental cleanup, to inspire reducing the costs of construction through design, to increase incentives through grants and tax breaks, and to provide a template to evaluate the total costs and benefits of this technology. I expected to find that an AD component to an ATS system makes the operation more cost effective overall compared to without an AD component. I also expected to find that incorporating more ES and long-term benefits to society at large in an economic CBA portrays ATS-AD systems are worth the initial cost of construction and operation. My estimates were quite conservative as I provided reasonable assumptions with my range of economic CBA scenarios and the ATS-AD system was cost-effective at a percentage of the FLW household number lower than 25% (L. Nadeau, personal communication, 2023, August 14). The project was a positive and viable construct towards a policy agreement since the NPV was positive at 33% (L. Nadeua, personal communication, 2023, August 14).

The high capital, operations, and maintenance costs of ATS projects still indicate a need for improving the technology and lowering the costs of construction, maintenance, and operations. The high costs involved also support that this technology serves a specific purpose and is not a one size fits all approach to bioremediation and eutrophication clean up.

The low, medium, and high range of values I calculated using Castro et al.'s (2016) data was in place of a sensitivity analysis or contingencies. Given the scope of this thesis research, an ideal version would use a complete sensitivity analysis. To achieve this, input and output values would come from the same source, which is rare to achieve, even at the federal level in the U.S. Approximations, like this thesis research using benefits transfer are common (N. Raheem, personal communication, 2023, October 10).

My calculations of benefits and costs for the ES of ATS-AD systems could present limitations or assumptions to this research. I relied on peer reviewed studies to quantify the benefit and cost variables. My research was limited by the available data and research conducted on valuing the ES ATS-AD systems provide. I was not able to find applicable economic values for odor reduction, disease control, and controlling the chemical quality of freshwater.

Further research limitations may exist in calculating cost and benefit variables based on data provided by Hydromentia LLC. The Falls Lake Algal Turf Scrubber Pilot Program Final Report (2017a) provided data for nutrients captured, algal biomass production, capital cost, and operations and maintenance variables. The Final Report was based on a pilot flow-way which ran for a monitoring period of 448 days, during which there were four system water flow interruptions that lasted from 26 to 59 hours, totaling 170 hours (Hydromentia, 2017a). Flow interruptions were due to flooding of the ATS site and impacted algal growth during and after the interruptions (Hydromentia, 2017a). Other research limitations include that each ATS system and its measured results on algal productivity, nutrients captured, energy costs, construction costs, and land costs depend on the site location, its climate, precipitation levels, sunlight levels, nutrient load, water flow, water temperature, and air temperature. A site in southeast Oklahoma may have different result values than a site in central North Carolina, for example. The Falls Lake Pilot report provides a sound starting point for a 20-year projection of this location. These limitations should be recognized for overall conclusions drawn on my research. Costs associated with AD machinery, operations, and maintenance were not included in this study.

Research continues to expand on the added benefits algae may be able to provide to our polluted water systems and could be considered in further ATS-AD research. Pharmaceuticals and personal care products are known micro-pollutants in our waterbodies that can cause negative effects on organisms (Hena et al., 2021). Hena et al.'s (2021) review of microalgae as a bioremediation use showed its promise in sustainably transforming and adsorbing polluted water. Zeller et al.'s (2013) review took algae-based N and P bioremediation of livestock farm effluent waters even further and showed the value-added bioplastic products harvested biomass can provide. Specific microalgae species' proteins are being transformed into a variety of materials and applications, such as edible and single-use plastic products, agricultural plastic products, and vegetative planting containers (Zeller et al., 2013).

ATS-AD systems are a form of green-gray infrastructure, which is defined as mixing conservation and restoration of natural systems with conventional, built approaches (Conservation International, n.d.). An example of grey infrastructure alone in freshwater management is a wastewater treatment plant (Conservation International, n.d.). An example of green infrastructure are wetlands that uptake nutrients (Conservation International, n.d.). A more in-depth and ideal study would compare the ATS-AD system with other best management practices (BMPs), which avoid, or cleanup freshwater eutrophication caused by excessive nutrients from point and non-point sources. Examples of structural and nonstructural BMPs include green infrastructure, such as, wetland restoration projects (Steinman et al., 2018), deliberate agricultural management to limit animal waste and crop fertilizers into water bodies (Utah State University Extension, n.d.), and riparian, or vegetative, buffers (Cao et al., 2018). However, measuring the effectiveness of BMPs for eutrophication in quantitative form is not always practiced or available (Steinman et al., 2018) and make a comparative study with ATS-AD systems difficult.

With a changing climate, it is often recommended to implement integrated and hybrid approaches to water resource management and protection (Nicholls, 2011). Residents and city planners of Pleasant Bay Watershed of Cape Cod, Massachusetts have managed N pollution through a hybrid approach, leading to a collective reduction (Pleasant Bay Alliance, 2018). Examples of integrated, watershed wide approaches include implementing traditional and non-traditional watershed management technologies, including but not limited to sewer maintenance, reducing the use of residential and golf course fertilizers, constructing on-stie denitrifying systems, building more permeable reactive barriers, propagating shellfish, and implementing more sustainable golf course fertigation, or fertilizing through irrigation (Pleasant Bay Alliance, 2018). On-site denitrifying systems filter N from wastewater and reduce TMDLs (US EPA, 2007). Permeable reactive barriers are underground permeable walls that clean up contaminated groundwater after the groundwater flows through it (US EPA, 2021). More sustainable golf course fertigation captures N in groundwater via irrigation wells, which irrigates and fertilizes the golf course green (Pleasant Bay Alliance, 2018). These examples of hybrid and integrated approaches to water resource management illustrate different sustainable solutions, as one size does not fit all water resource issues and, with a changing climate, technology and policy need to be adaptive (Mohammad & Ang, 2021).

Conclusions

All the bioremediation technology and water resource management practices in the world will still not resolve our eutrophication issues without proper regulation. There will always be an excess of nutrients from anthropogenic sources without regulation and improved wastewater treatment (Nwankwegu et al., 2019). A national plan or strategy to limit excess nutrients from entering our coastal waterways does not exist (Board & National research Council, 2000). The 1972 Clean Water Act does not limit most agricultural fertilizers (Kling, 2019). State governments can and do enact policy to limit the excessive use of nutrients from entering waterways: Wisconsin banned using fertilizers on frozen ground and Florida reduced P loading in the Everglades with more precise agricultural fertilizer application (Kling, 2019).

Without more regulation on nutrient use and cleanup, there will always be an excess of N and P in our waterways from anthropogenic sources. Implementation of gray-green technology, such as the ATS system, along with other traditional and non-traditional BMPs structured to fit each unique geographic location of eutrophic concern and adaptable to a changing climate, will help alleviate the excessive nutrients and their negative impacts on our environment.

It is my hope that bioremediation technologies, such as the ATS-AD system, now and in the future can be recognized not only for their immediate eutrophication clean up or avoidance but also their external value in ES restoration and protection. Implementing and incorporating such technologies into more industries will be costly to construct and manage. Implementing policy towards regulating nutrient input into our ecosystems will be costly. But if more individuals, governments, and businesses recognize the value of restoring and protecting the services our ecosystems provide us, we will hopefully be more equipped, and act more urgently, to address the continually exacerbating symptoms of a changing climate.

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