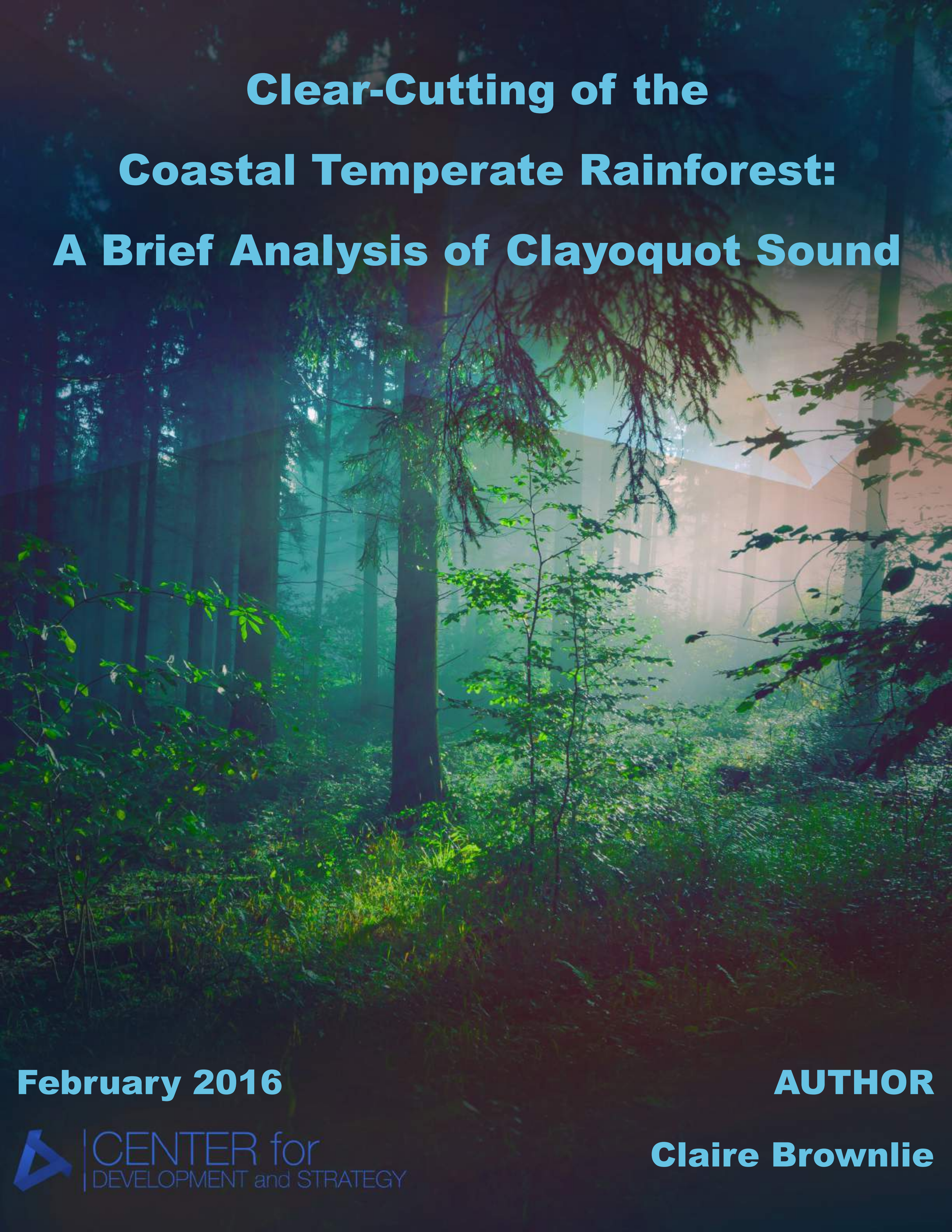

Clear-Cutting of the Coastal Temperate Rainforest: A Brief Analysis of Clayoquot Sound

February 2016

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CENTER for DEVELOPMENT and STRATEGY

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Abstract

Using Clayoquot Sound as a reference, the consequences clear-cut logging has on the coastal temperate rainforest ecosystems was examined. Social and political outcomes from extensive protests in 1993, which opposed the destruction of the natural habitats, were also assessed. Additionally, First Nations' rights were investigated through the Interim Measures Agreement between the Government of British Columbia and the Nuu-chah-nulth community, as well as their co-management of the natural resources in Clayoquot Sound. In 2000 Clayoquot Sound was designated as a UNESCO site. The designation brought the issues in the coastal temperate rainforest to the forefront once again, and allowed for increased non-profit and governmental attention and aid. The economic benefits that can be gained from the region such as its utilization as a carbon sink, ecotourism, and fishing were also evaluated. Lastly, previous research on climate change has estimated the impacts on future biodiversity and ecosystem health of the region.

Introduction

The unique ecosystem of the coastal temperate rainforest will be examined, specifically focussing on the Clayoquot Sound region on Vancouver Island, British Columbia. The British Columbian government's decision to allow for clear-cutting of the region and the outcome of the protests opposing the clear-cutting will be outlined. Specific characteristics of a coastal temperate rainforest and outline its unique biodiversity will be identified. Aside from logging, these old growth forests have many ecological goods and services to offer, and the economic benefits of the alternatives are outlined. First Nations have a vested interest in the area, and have land rights in much of the region, their involvement in the management of the forest will be analyzed. In 2000 the area was designated as a UNESCO biosphere reserve (Reed and Massie, 2013), an examination of the changes to management and biodiversity brought on by the designation will be included. Finally, the current status of the forest and regulations will be evaluated for their usefulness to future sustainability.

Scholarly papers, government documents, and non-profit websites were examined to obtain the data required to evaluate the past and current state of the temperate rainforest in Clayoquot Sound. Due to the older nature of the logging disputes, papers from 1993 onwards were utilized in order to include all relevant information. Websites, reports, and newsletters were taken from non-profit organizations such as The Sierra Club and Friends of Clayoquot Sound as they were directly related to the Clayoquot Sound protection advocacy and implementation. The Government of British Columbia website was used to obtain information about forestry policies and First Nations Treaties. Specifically, the Ministry of Forests, Lands and

Natural Resource Operations was a significant source of information. These sources assisted in achieving the purpose of this research paper: to explore the history of Clayoquot Sound, the implications of government and First Nations intervention, the implications of logging on the biodiversity of the region, and finally any implications for the future.

Background Information

Clayoquot Sound is located on the west coast of British Columbia's Vancouver Island (Lavallee and Suedfeld, 1997). It is a culturally and naturally diverse area, as is evidenced by the biogeoclimatic map shown in Figure 1 (Government of BC, 2003). As the figure demonstrates, the province is home to over 10 unique and distinct biogeoclimatic zones, with temperatures becoming increasingly mild as you move west (Government of BC, 2003). Clayoquot Sound is classified as a coastal temperate rainforest, which is a rare biogeoclimate that covers less than 1% of the world's land base. These regions experience extremely high biodiversity due to the high quality habitat and mild temperatures (Sierra Club, 2009). A unique feature of the BC jurisdiction is that approximately 95% of the province is owned by the BC government, which means that the 90 million hectares of Clayoquot Sound are managed on behalf of the residents (Government of BC, 2003). Since the government is in charge of managing the natural resources of the province, they set the timber annual allowable cut, and are in charge of dispersing this information to the forestry companies, First Nations, communities, and individuals. The government also works closely with First Nations treaty rights to ensure they are given the land they are entitled to (Government of BC, 2003).

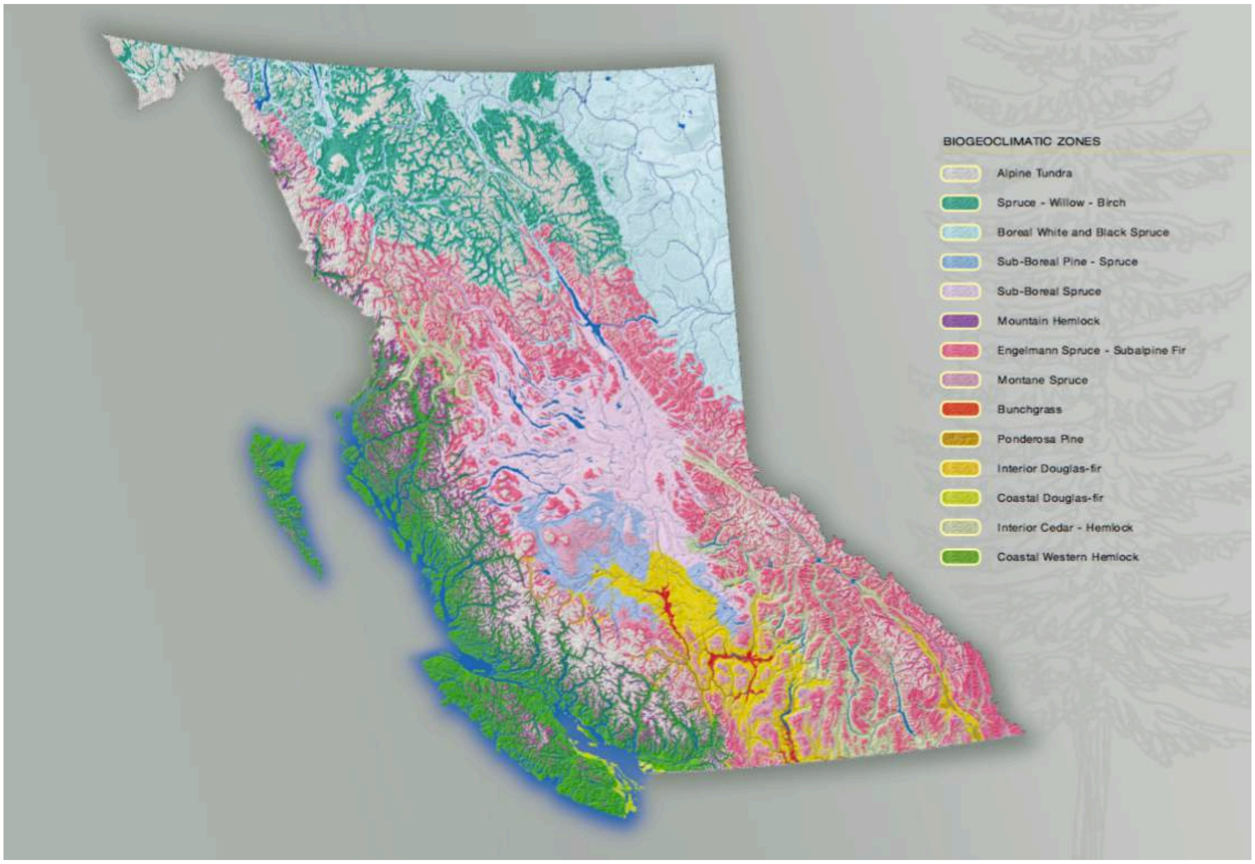

Figure 1: Biogeoclimatic Zones of British Columbia (Source: Government of BC, 2003)

Figure 1 classifies the various biogeoclimatic zones in BC, showing how diverse the region is. It is interesting to note how much of the area is represented by Coastal Western Hemlock, considering this is the main component of coastal temperate rainforest. This region represents much of the remaining coastal temperate rainforest in the world.

The history of Clayoquot Sound has been shaped through many government and First Nation interactions, as well as through input from outside sources. By 1989 most of the region had been allocated for forestry, with MacMillan Bloedel holding the largest claim (Lavallee and Suedfeld, 1997). The Clayoquot Sound Development Steering Committee was formed to prepare a land use plan for the region, however their plan did not receive unanimous support, therefore it did not pass. In April of 1993 the BC government released its land use plan, based on what the Clayoquot Sound Development Steering Committee had previously recommended, for the area (Lavallee and Suedfeld, 1997). The plan designated 33% of the area for protection, 45% for commercial timber use, and 17% for special management areas (Lavallee and Suedfeld, 1997). The special management areas were primarily buffer zones between logging regions and the coastline of the island (Lavallee and Suedfeld, 1997). Through this plan the British Columbian government had also given MacMillan Bloedel permission to clear-cut up to 70% of the 350,000 hectares of Clayoquot Sound (Walter 2007). Clear-cut logging is the process of removing all trees from a portion of forest (NRDC, N.D.). The implementation of this management plan was not supported by the First Nations communities or the general population. In 1993 the Friends of Clayoquot Sound organization set up a protest camp along an active logging road to voice their displeasure for the clear-cutting practices of the logging industry. Over 10,000 protestors participated in the protests that lasted from July to October, making it the largest demonstration of its type in Canadian history (Lavallee and Suedfeld, 1997). In order to keep the protests in line with the initial purposes, a “Peace Camp” was created to house the protesters, as well as to ensure there were rules and standards for continuing a nonviolent protest (Walter, 2007). The protests were nationally recognized, partially due to Clayoquot Sound being a popular tourist destination, as well as its pristine natural beauty (Tindall, 2013). The environmental movement had been blossoming in the years before, also leading to increased attention for the 1993 protests. This movement helped people to understand the importance of ecology and biodiversity, and the health, aesthetics and spiritual aspects of the environment (Tindall, 2013).

Coastal Temperate Rainforest

The coastal temperate rainforest is a unique ecosystem, with approximately half of the global rainforest left worldwide, and half of this occurring in North America. In BC the coastal temperate rainforest occurs on the coast of the mainland and covers most of Vancouver Island, see Figure 1 for a map (Government of BC, 2003). Almost all of the undeveloped coastal temperate rainforest exists in British Columbia (Bunnell, 2008). The forest on Vancouver Island is warmed by passing ocean currents and experiences large volumes of rain due to the prevailing westerly winds (Walter, 2007). Measurable precipitation occurs 200 or more days of the year, and temperatures are relatively mild in the winter and relatively cool in the summer due to the regulating properties of the nearby ocean. The coastal temperate rainforest is made up of three major biogeoclimatic zones: Coastal Western Hemlock, Mountain Hemlock, and Coastal Douglas-Fir (Bunnell, 2008).

The Coastal Western Hemlock zone covers about 87% of the temperate coastal rainforest on Vancouver Island and occurs at elevations below 900m sea level. Flora and fauna flourish in the zone and some of the largest and oldest trees grow there (Bunnell, 2008). Western hemlock is the most abundant tree in the zone with Western red cedar and Douglas-fir also occurring. Wetter and higher elevation sites grow the Amabilis fir and yellow cedar. 1000 to 4400 mm of precipitation falls in the zone, and the average temperature ranges from 5.2-10.5°C (Bunnell, 2008). An abundance of bryophytes and lichens grow in the area, many endemic, or native, to the region. Mountain Hemlock represents 12% of the land and the remaining one percent is represented by Coastal Douglas-fir, in the southern rain shadow (Bunnell, 2008).

Biodiversity is extremely high in the coastal temperate rainforest. Approximately 175 forest-dwelling, terrestrial vertebrate species breed within the area, which reflects the complex stand structure (Bunnell, 2008). Lichens, bryophytes, and vascular plants prosper in regions where there are dead trees and rotting wood, as is common on the forest floor of the region. More species use these structures, cavities and downed wood, than in any other forest type in the forest, emphasizing the diversity of the region (Bunnell, 2008). The area is naturally very diverse and is important not only for economic reasons, but also in preserving the culture of the First Nations who traditionally lived in the area.

First Nations' History on Vancouver Island

The West Coast of Vancouver Island is home to 14 Nuu-chah-nulth First Nations. The Ahousaht, Hesquiaht, and Tla-o-qui-aht are the three First Nations that have traditional territories in the region of Clayoquot Sound (Mabee and Hoberg, 2006). The Nuu-chah-nulth First Nations have interacted and maintained relationships with non-Native people since as early as the 1770s. These relationships began with the fur trade and European interests that were focused on profits through sea otter pelts (Goetze, 2005). Nuu-chah-nulth were some of the first communities to take part in the fur trade on the Pacific Coast due to their strong bargaining abilities and confidence in negotiating with the European settlers (Goetze, 2005). When the fur trade ended in the 1850s the Nuu-chah-nulth helped the Europeans create settlements on the land. This proved to be less beneficial to the First Nations as the settlers looked to develop more permanent economies on the land, and did not require the trade assistance of the communities (Goetze, 2005). In 1849, after Vancouver Island was established, the governor purchased 14 segments of land from First Nations living along the south and north east coasts. The First Nations were free to continue their routines such as hunting, fishing, and trapping in the area; although their subsistence lifestyle did not continue for long as the settlers quickly bought the land for their own gains (Goetze, 2005).

Comprehensive treaties were required to be negotiated in order to identify their traditional territories and rights (Goetze, 2005). As logging concerns arose in the 1970s the Nuu-chah-nulth Nations brought up the conflicts surrounding irresponsible resource use by the government. Damage to streams due to the logging debris, among various other issues, was addressed through negotiations with the government and logging companies (Goetze, 2005). Throughout the 1980s and 1990s the Nuu-chah-nulth Nations had many land claims surrounding their traditional territories. By 1994 they entered a treaty process with the BC Government (Goetze, 2005). Their aim with the treaty formation was to recognize and protect their Aboriginal rights to resources, but also ensure decision making would continue within a cooperative framework (Goetze, 2005).

The Nuu-chah-nulth Tribal Council is currently negotiating a treaty with the Government of British Columbia. They are in Stage 4, which is outlined by the BC Treaty website as the agreement that will form the basis for the eventual treaty (BC Treaty, 2009). Throughout this process both parties must include their essential points of agreement and develop plans for implementation of the treaty (BC Treaty, 2009). The framework must be agreed upon by both parties before it can move on to Stage 5, which is the negotiations that will finalize the treaty (BC Treaty, 2009).

Government Management of Clayoquot Sound

The government management of old-growth temperate rainforests in Clayoquot Sound was catered to multinational forestry companies. Originally, the resources were harvested in pursuit of short-term profit through clear-cutting, the most efficient mode of extraction (Goetze, 2005). The province was aiming to encourage economic growth and job creation. Due to the recession in the 1980s the government saw logging as a large economic benefit and relaxed their sustainability guidelines around the amount of timber logged (Goetze, 2005). The increase in logging resulted in a decrease in consultation with the First Nations communities, with the forestry companies taking charge in what was deemed to be sustainable for the forests and the surrounding ecosystems (Goetze, 2005). While the government did recognize that clear-cutting was an unsustainable solution to forest management, they were primarily focused on the economic bottom line (Goetze, 2005). After the 1993 Land Use decision, 900 square kilometres, or 34% of Clayoquot Sound, were designated for protection by the government (BC Ministry of Forests, N.D.). The land was designated to ensure the protection of the environment, local communities, and the economy (BC Ministry of Forests, N.D.). The reserve forms a link from the mountains of interior BC to the coast line of Vancouver Island, providing less chance for fragmentation of key species and habitat areas (BC Ministry of Forests, N.D.). Of the 900 square kilometres, 700 of them are temperate rainforest, providing a key ecosystem for approximately 29 rare plant species, significant old growth forest, salmon spawning habitat, and rare marine ecosystems (BC Ministry of Forests, N.D.). It was critically important that the government set aside this area for protection, as all other stands of temperate rainforest ecosystems in the world are under some sort of threat from human destruction. Aside from the 34% of Clayoquot Sound that was completely protected, the government also placed 21% more of Clayoquot Sound under “special management” (BC Ministry of Forests, N.D.). This special management allows for some logging, but no clear-cutting, and still emphasizes the importance of protecting wildlife, along with recreational and aesthetic values (BC Ministry of Forests, N.D.). After the Land Use Decision was enacted only 40% of Clayoquot Sound was open to integrated resource management (BC Ministry of Forests, N.D.). Integrated resource management allows for logging and other resource extraction such as mining and fishing (BC Ministry of Forests, N.D.). In order for a forestry company to receive approval from the Government of BC to undergo any logging in Clayoquot Sound they must meet standards for forest management planning, road building, and harvesting limits (BC Ministry of Forests, N.D.). Conventional clear-cutting has been replaced by variable retention harvesting – which is a system that ensures key elements of the forest are left intact, allowing for the forest to healthily regenerate (BC Ministry of Forests, N.D.). The Scientific Panel also recommended that ecological assessments should be conducted for undeveloped watersheds before any additional resource extraction was undertaken (BC Ministry of Forests, N.D.).

Discussion

The coastal temperate rainforest is an essential asset to the province of British Columbia, and should be protected accordingly. A 2009 report by the Sierra Club of BC outlines the importance of old-growth forests in BC's greater ecosystems. In order to avoid species extinction in the coastal temperate rainforest, a minimum of 30% of old-growth forests need to be conserved, while 70% of natural levels should be conserved to ensure low risks to species loss (Sierra Club, 2009). Much of the ecosystems on Vancouver Island are below this threshold, with many species extinct or close to extinction. A potential benefit of the coastal temperate rainforest is its proximity to the ocean, which may buffer it from some effects of climate change (Sierra Club, 2009).

Coastal temperate rainforest is an excellent carbon sink, with an estimated carbon storage potential of 1,000 tonnes per hectare (Sierra Club, 2009). Taking advantage of this carbon storage opportunity will be important in achieving carbon reduction goals faced by Canada in the future. On Vancouver Island alone, approximately 1 million hectares of old-growth forest have already been lost, amounting to a loss of approximately 100 million tonnes of carbon reservoir (Sierra Club, 2009). Emissions from logging also contribute close to 370 million tonnes of carbon to the atmosphere (Sierra Club, 2009). This loss of a carbon reservoir coupled with the emissions from logging lead to an unbalanced emissions profile. The coastal temperate rainforest is set to become an important resource in a potentially carbon focused market of the future.

These important ecosystems are rapidly declining, with approximately 50% of all coastal temperate rainforest on Vancouver Island at high risk for species loss. It is essential to create protection and conservation areas on the island, especially since 13% of the land on Vancouver Island has already been converted from old-growth forest (Sierra Club, 2009).

Climate Change Impact on the Coastal Temperate Rainforest

Climate change has to the potential to impact almost every ecosystem on the planet, and coastal ecosystems are often the first to feel the effects. Coastal temperate rainforest is no different. Shanley et al. (2015) conducted a study of the Intergovernmental Panel on Climate Change (IPCC) models predicting the possible scenarios due to anthropogenic climate change. They found the results for indicators on the coastal temperate rainforest in Alaska and British Columbia are all projected to increase. Specifically, the report found, through analysis of IPCC models and representative concentration pathways:

Table 1: Potential Increases Due to Climate Change (Source: Shanley et al., 2015)

Year	Temperature (°C)	Precipitation – Rain (mm)	Precipitation – Snow (mm)
1961-1990	3.2	3130	1200
2080	4.9 – 6.9	3320 - 3690	720 - 500

This table outlines the potential for a large increase in temperature and precipitation in the form of rain, but a decrease in precipitation in the form of snow. There are many outcomes for the region based on these increases. These results will cause a cascade effect on the ecosystem, with a plethora of new and extreme weather events. Some issues will include an increase in floods, reduced snowpack, effects on river flow, shifts in suitable wildlife habitat, and many additional impacts (Shanley et al., 2015). The people of the region typically rely heavily on the ecosystems goods and services, such as fishing, forestry, and ecotourism (Shanley et al., 2015). If these extreme events start occurring with more frequency the First Nations and other community members will lose their livelihood.

Climate change could potentially affect the ecosystem goods and services that the population depend upon for their economic benefits. Particularly, the fishery habitat could be altered, hydropower opportunities may become less dependable, and ecotourism activities could decline (Shanley et al., 2015). It is not expected that climate change will have an affect on forestry, further to the restrictions already put in place in the region (Shanley et al., 2015). Climate change also has the potential to effect the biodiversity of the region, so measures must be taken to protect the endemic and rich variety of species living in the coastal temperate rainforest.

Biodiversity Conservation Strategies

As outlined by the Government of British Columbia in the 1970s and 1980s, the logging company MacMillan Bloedel was granted logging rights of the region. Forestry in the region is difficult to pursue due to steep slopes, wet soil, and large equipment. After strong opposition to the clear-cutting of the region, MacMillan Bloedel agreed to stop clear-cutting in 1998 (Bunnell, 2008). In order to create a more sustainable forestry management plan, the company divided the forest into three different harvest zones, based on intensity of harvest. The first zone is the timber zone, and is classified as the primary source of economic value, and most of the productive harvest was found in the area (Bunnell, 2008). The provision of late-seral features is meant to allow for species to survive that would not otherwise if clear-cutting was in place (Bunnell, 2008). Secondly, the habitat zone has higher retention levels than the timber zone, and only 70% of the forest is available for harvest. The goal of the habitat zone is to conserve organisms that make up the biological diversity of the area (Bunnell, 2008). Finally, the old growth zone is mostly protected as to maintain late-seral forest conditions (Bunnell, 2008).

Variable retention was implemented in all zones to retain appropriate habitat structures to maintain biodiversity. Objectives of structure retention include:

increasing species richness in managed stands through connection across the landscape to provide refuge and survival for species after harvesting of timber; creating opportunities to meet market demand of harvesting trees that will not be detrimental to forest health, vigour, genetic composition, or timber quality; meeting social expectations of stewardship and visual aesthetics; and, meeting site-specific needs for regeneration and habitat (Bunnell, 2008).

The three categories of retention are shown on Figure 2. Retention is managed differently based on the amount of trees that need to be kept in place to achieve biodiversity. Small groups of trees are retained together on the same cut block for

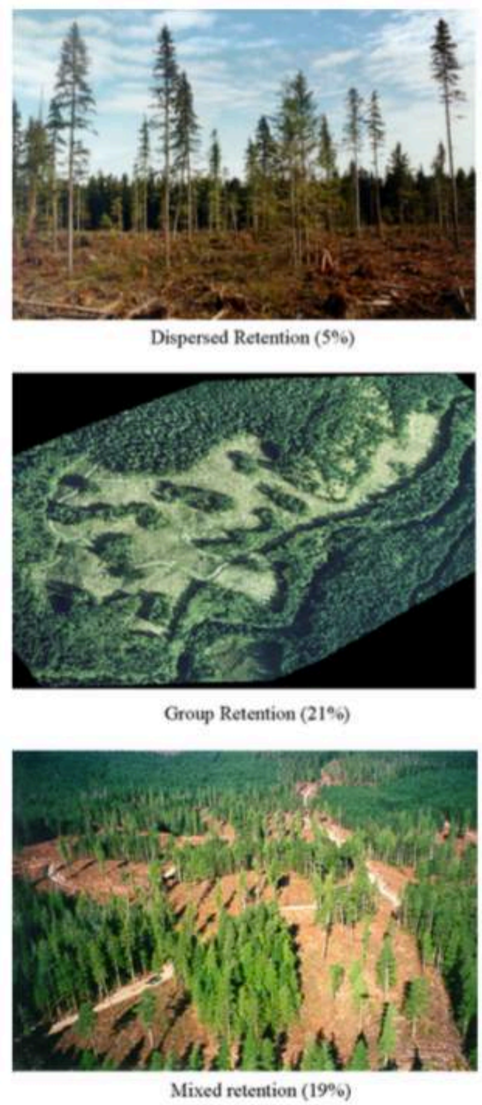

Figure 2: Retention Practices (Source: Bunnell, 2008)

group and mixed retention, while substantial amounts of trees are cut without leaving any in groups for dispersed retention (Bunnell, 2008). These methods are important to reduce potential habitat fragmentation that occurs with logging roads and powerlines. For species to have the best chance of survival, large areas of habitat must be preserved.

Economic Valuation of the Coastal Temperate Rainforest

Clayoquot Sound and The Great Bear Rainforest are both coastal temperate rainforest. The Great Bear Rainforest is located on the west coast of inland British Columbia. Both are positioned to provide economic benefits for the province, with and without logging. The book "*Great Bear Markets: The Interface of Finance, Forestry and Conservation in BC's Great Bear Rainforest"* by Andrew Norden and James Tansey outlines the economic gains that the Great Bear Rainforest can produce. Since Clayoquot Sound is the same ecosystem, and under the same provincial governance, comparisons and suggestions can be made between the two. Reduction of logging in the area can create carbon offsets through the carbon that is being stored in the remaining trees (Norden, 2011). Potential ways to earn revenue in the area include: ecotourism, hunting, logging, fishing licenses, and carbon offsetting. Carbon offsetting has the largest and most immediate potential to be economically viable based on the large stands of forests. The initial obstacle is the development of a carbon market with formal compliance. In BC, this began with the formation of the Pacific Carbon Trust (Norden, 2011). In 2013 Pacific Carbon Trust, responsible for carbon offsets for the provincial public-sector organizations, announced that the organization would be transitioned to fall under the Climate Action Secretariat through the Ministry of Environment (Pacific Carbon Trust, 2014). The potential offset inventory of the Great Bear Rainforest, which covers 6.4 million hectares of land (Price et al., 2009), is 1 million tonnes per year for the first 30 years. A typical carbon transaction for a single buyer is usually 30,000 to 100,000 tonnes (Norden, 2011). This means that the revenue coming from carbon offsetting alone could be valued at $4 million a year. This will be split 50/50 between the First Nations in the area and the provincial government (Norden, 2011). In Clayoquot Sound, First Nations have similar land rights, and the government would have to reach an agreement with them to start carbon offsetting projects. Clayoquot Sound covers 350,000 hectares, and is substantially smaller than the Great Bear Rainforest, but there is still economic potential for the First Nations and government to profit from a carbon offsetting project (Mabee and Hoberg, 2006).

Ecotourism is a main source of income for communities on northern parts of Vancouver Island (Dodds, 2012). Tourism growth began in the mid 1980s when activities such as whale watching became popular. By the year 2007 approximately 35,000 people visit the area per year. These tourists bring in an economic value of close to $50 million a year for the region (Dodds, 2012). While there is the economic incentive to have tourists in the area, the influx of people also causes issues within the community. A survey was conducted regarding the benefits and effects of tourism in the region, and half of the respondents suggested that tourism was a good livelihood but did not enhance the community, and their sense of place was in jeopardy (Dodds, 2012). There are also potential issues surrounding infrastructure, such as sewage, in the summer months when the population is more than doubled. The small communities directly dump their sewage in the ocean and allow the strong currents to disperse it (Dodds, 2012). In the summer when there is the large influx of people the currents are not always fast or strong enough to dispose of the excess sewage. Another issue with ecotourism is the potential for human and wildlife interactions (Dodds, 2012). There is the potential of feeding from tourists as well as close encounters, especially with bears. When tourists feed the wildlife this creates a situation where wildlife becomes dependent on the food, and are unable to survive on their own (Norden, 2011). Bear interactions typically cause more harm to the bears as they become familiarized with humans and are more likely to become pests. Unfortunately, this typically ends in their destruction by wildlife officials (Norden, 2011).

Fishing licenses are important on the West Coast due to the large salmon population of the area. There is also an abundance of aquaculture farms because of the suitable ocean habitat and ease of access. The Department of Fisheries and Oceans (DFO) are responsible for determining the rate at which fish are caught, which is a vital calculation to ensuring the sustainability of the salmon and other fish in the region (Norden, 2011). Commercial salmon farming, or aquaculture, was valued at $215 million in 2005 with the recreational fishing industry worth nearly $230 million (Norden, 2011). It is clear that salmon fishing is vital to BC as a whole, which provides further incentive to preserve the integrity of the ecosystem. Aquaculture is growing as the demand for seafood has grown, therefore it has the potential to be a key industry in British Columbia’s economy.

Forestry has the opportunity to contribute to a green economy if improved forest management practices are implemented. Selective logging and longer rotation periods will contribute to more jobs than clear-cutting would, and introduction of value-added products will also help move toward a sustainable forestry sector (Sierra Club, 2009). If the practices, such as variable retention, suggested above were to be implemented, Clayoquot Sound could once again become a viable economy for Vancouver Island and the surrounding areas. Clear-cutting has already been phased out due to government policies and agreements with logging companies such as MacMillan Bloedel, but clearing practices such as those suggested by Bunnell (2008) need to be implemented to ensure continued sustainability of the forests. Working towards a carbon economy, such as the model suggested by Norden (2011) is essential for the economic wellbeing of the region. If Clayoquot Sound and the neighbouring communities are able to achieve accreditation for their carbon sinks, this could be a profitable endeavour. Finally, ecotourism must continue to grow in a sustainable way. As Dodds (2012) outlined in her surveys, ecotourism must be completed, but the local communities must not feel as if their traditional ways are in jeopardy. The industry cannot grow at an unsustainable pace that does not allow for infrastructure to keep pace. Issues such as sewage and lodging for tourists in the summer months must be acknowledged and addressed if the industry is to continue to grow.

UNESCO

United Nations Education, Scientific, and Cultural Organization (UNESCO) biosphere reserves were first created in 1976 to create a better understanding on how to conserve biodiversity and improve human-environment interactions (Reed and Massie, 2013). Reserves are created due to the sustainability desire of local communities (Reed, 2007). All biosphere reserves are designated to “demonstrate three functions: environmental protection; logistical provisioning for scientific research; and sustainable resource use” (Reed, 2007). These reserves contain three zones: “a core that must be protected by legislation; a buffer where research and recreational uses compatible with ecological protection are allowed; and a transition zone where sustainable resource use is practiced” (Reed, 2007). Clayoquot Sound was designated as a reserve in 2000 to help promote the conservation of biological and cultural diversity; advance sustainable development; and provide support for research, learning, and public education (Tindall, 2013; Reed and Massie, 2013). A biosphere reserve is not a protected area, but is known for being a living laboratory or a learning site. Periodic reviews are conducted for the Clayoquot Sound Biosphere Reserve (Reed and Massie, 2013). Clayoquot Sound receives much media and stakeholder attention due to the large amount of work done by non-profits in the area to raise awareness and encourage involvement in preservation of the region (Reed, 2007). These non-profits, along with government sources and First Nations, have a high level of involvement in managing Clayoquot Sound (Reed, 2007).

The idea that a biosphere reserve is a learning site encourages inclusion of local people in the process of creating a sustainable site (Reed and Massie, 2013). First Nations inclusion in the decision making process is vital for Clayoquot Sound, with the government and First Nations co-managing the area. In general, Clayoquot sound has a strong status of environmental management through work with private, public, and governmental needs to address resource management (Reed, 2007). In her study, Reed found that First Nations of the region were directly involved with resource management of Clayoquot Sound, specifically within a co-management role (2007).

First Nations' Involvement in Conservation Strategies

As the logging disputes arose in the mid-1980s First Nations and NGOs showed their displeasure through non-violent protests. One of the government solutions to the disputes was to set up an Interim Measures Agreement (IMA) between BC and the First Nations, which allowed for the co-management of the region's resources (Mabee and Hoberg, 2006).

The idea of co-management was proposed, so both the First Nations and the BC Government would have a say in how the resources were used (Mabee and Hoberg, 2006). The ultimate goal of co-managing an area is to achieve a state of equality between the two groups (Mabee and Hoberg, 2006). These types of initiatives have been prevalent in BC due to the historic treaties, as outlined above. BC has been a strong example of how First Nations' and Government's ideologies can be aligned.

Co-management cannot be seen solely as an environmental issue, for First Nations it is also a socio-political issue (Goetze, 2005). When the agreement toward co-management was created this gave the First Nations of the area more governance over "their territories and resources, protection of cultural heritage sites, and pursuit of traditional harvesting activities" (Goetze, 2005). Through the negotiation of resource use this requires the government to recognize First Nations' governance structure, where the Chiefs are responsible for land management and distribution (Goetze, 2005). First Nations Chiefs are responsible for dividing their land equitably, and in a manner that will ensure the sustainability of their tribes (Goetze, 2005). The Interim Measures Agreement (IMA) which resulted in co-management acted as a way to address the socio-cultural, political, and legal issues and land rights issues faced by the First Nations communities (Goetze, 2005). The co-management process has been successful for the Nuu-chah-nulth First Nations as it has “advanc[ed] their aspirations concerning political and structural equity, or “systematic change,” and the protection and practice of indigenous rights” (Goetze, 2005). British Columbia has the chance to share their successes surrounding co-management with the rest of Canada as well as neighbouring countries with a vested interest in First Nations negotiations. Their successes should be shared as such, and failures should be outlined as a process for improvement.

Interim Measures Agreement for Clayoquot Sound

The Interim Measures Agreement (IMA) was a result of the protests in 1993 and negotiated over a period of several months, and was updated in 2008 to extend the co-management practices (Goetze, 2005). The government was initially unwilling to negotiate regarding their involvement with land rights issues with the Nuu-chah-nulth Nation (Goetze, 2005). The persistence in negotiating allowed for the IMA to recognize many key political claims of the Nuu-chah-nulth that are closely related to resource management, their governance structure, Chief authority, and the relationship between the Nuu-chah-nulth governance and the Government of BC (Goetze, 2005). The agreement gave the Nuu-chah-nulth much control of the natural resources of their home (Magnusson and Shaw, 2003). The IMA agreements around resource management led to the formation of the Central Region Board and the Scientific Panel, which included equal representation from First Nations and BC Government (BC Government, 1994; Magnusson and Shaw, 2003). The Board provided cooperative management of the terrestrial and marine resources, except for ocean fisheries, in Clayoquot Sound (BC Government, 1994). The unique feature of the board was that a double majority was required for any voting matters. This meant that a majority from the government representatives and the Nuu-chah-nulth representatives was required (Goetze, 2005). The IMA give Nuu-chah-nulth the position of “co-managers tangible, determinative authority to make decisions about resource use in Clayoquot Sound” (Goetze, 2005). Finally, the IMA provides a sense of positive interaction between Nuu-chah-nulth, the BC Government, and local communities (Goetze, 2005).

Conclusions

The history of Clayoquot Sound has been varied and shaped through social, political, and environmental tensions. In the 1970s and 1980s when the BC Government was looking to expand their economic activity there was little regard for the environmental goods and services provided from the coastal temperate rainforest.

The biodiversity and range of natural habitat of the coastal temperate rainforest is vast, and becoming increasingly rare. Steps must be made to ensure that the area is protected from further damage that arises through clear-cutting of an area. The protests in 1993 helped bring awareness to the issues facing the area, and non-profit, community, and governmental involvement and interest continued after the UNESCO biosphere reserve designation in 2000. This also allotted the area as a protected zone, showing British Columbia’s dedication to protecting their unique ecosystem and its biodiversity. Alternative options have been implemented to continue to obtain economic benefits from Clayoquot Sound. Utilizing the old growth forest is especially beneficial as a carbon offsetting tool as the dense and large old trees are excellent carbon sinks. Ecotourism is also very important to the region, as it boasts a mild climate, diverse scenery, and many environmental activities that are attractive to tourists. While the region is very small, it benefits from thousands of tourists a year bringing in millions of dollars to the local communities. Fishing and hunting are also popular activities, although not as economically profitable as carbon offsetting or ecotourism. First Nations' involvement in the natural resource management process has been continuous, with the Nuu-chah-nulth community being the most involved due to their close connections to Clayoquot Sound through traditional territory and subsistence activity. A significant achievement for the Government of BC and the First Nations was the agreement that allowed for a co-management of the land in and surrounding Clayoquot Sound. This was significant due to the required levels of cooperation, allowing both parties to have an equal say in the management of the ecological goods and services of the area. Initiatives similar to this should be implemented throughout Canada, with the BC Government providing relevant information for the other Provincial governments.

Climate change has the potential to significantly impact the biodiversity of Clayoquot Sound. Climate models project a large increase in mean annual temperature, and increase of approximately 100 additional millimetres of rain per year, and a decrease in annual precipitation in the form of snow. The compilation of the effects of climate change could be potentially very damaging for the coastal temperate rainforest; therefore, this area must continue to be protected by First Nations, non-profits, and the Provincial and Federal Governments. The case study of Clayoquot Sound shows that environmental awareness about a region or specific issue is key in helping work towards preservation. When people are able to come together in peaceful protest many new agreements and regulations can be made to solve the problem. It is vital to include First Nations in the recommendation proves for ecological goods and services as they have lived off the land for much longer than we have, and are able to sustainably manage the resources. Overall, British Columbia is doing a commendable job in protecting the coastal temperate rainforest but needs to ensure future commitments are set and followed in order to preserve the forest. As the last remaining intact coastal temperate rainforest it is vital that the area is kept pristine and used as a strong example of successful environmental policies and sustainable standards.

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Effects of Conventional and Organic Agricultural Techniques on Soil Ecology

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Abstract

This paper explores the differing effects that conventional and organic agriculture have on soil ecosystems. The findings are primarily based on a review of published literature found in journal articles and government reports. Conventional agriculture is found to generally have higher yields than organic agriculture. However, the monetary benefits of high yielding conventional agriculture systems use monoculture cropping, tilling, pesticides, and fertilizers that have a significant negative effect on the natural processes soil ecosystems. As a result, these natural process need to be replaced by the continual and increased use of external inputs such as pesticides and fertilizers which present issues from a sustainability perspective. Furthermore, conventional agriculture is a main culprit of soil loss. This paper explains intricate relationships between soil organisms and argues for an increase in the use of organic agriculture strategies to maintain healthy soil in the long term. The paper focuses on the relationship between soil ecosystems and the following: tilling, pesticides, biodiversity, fertilizers. The effects of these relationships are explored in terms of yield, resource use, and sustainability. The conclusion is that when all costs are considered, organic agriculture is at least competitive with conventional agriculture systems.

Keywords: *Soil Ecology, Sustainable Agriculture, Biodiversity, Monoculture*

Introduction

Soil is an important natural resource on Earth. Soil performs an essential function in terrestrial ecosystems and forms the base of a food chain, where primary producers grow and access nutrients and water. Healthy soil contains an immense diversity and abundance of organisms that perform ecological services. As such, it is essential that soil is cared for, maintained, and respected, specifically in agriculture. Unfortunately, the dominating conventional, monoculture agricultural systems use strategies and technology which can negatively impact the life in soil. Therefore, understanding how soil ecosystems work with plants is important for human decision making in moving towards more sustainable and ecologically sound agriculture. Organic agriculture offers an alternative to conventional monoculture agriculture systems. The main differences between organic and conventional agriculture are the use of tillage methods, pest control methods, and fertilizers, and the consequent effects on biodiversity, yields, sustainability and resource use. This paper explores the fundamental processes of soil ecosystems in each of these differences. Through an extensive literature review, this paper argues that conventional agriculture has significant negative effects on soil ecology, and organic agriculture is at least as competitive with conventional agriculture when externalized costs are considered.

Background Analysis

Over thousands of years, humans have learned how to domesticate the plants which provide both the necessities for our diet and the products for our lifestyles. This domestication brought about the agricultural revolution, which was a large step towards what some might term as humanity's domination of nature and the subsequent development of our first civilizations. As time passed, economies became increasingly mechanized with technological advances made during the industrial revolution. The ways in which plants are were grown became an issue of efficiency, output, and productivity. This shift, along with new ways of increasing plant growth through fertilizer use and pest control, led to the green revolution. Throughout the world, production costs were reduced while yields increased, primarily through the planting and cultivating of vast tracts of land with homogenous crops, organized in rows for easier harvesting, maintenance, and control. This is now the established method of planting called conventional farming or monoculture. The prefix “mono” refers to “one”, “only”, or “single”. The word “culture” in the context of farming or planting refers to the cultivation of a piece of land.

Because of the wide variety of soil types, agricultural strategies, fertilizers, pesticides, and research methods used in agricultural studies, there is a diversity of research on this subject. Many studies are limited to researching the effects of specific pesticides on specific plants and soil types. Some studies present conflicting evidence of the benefits and costs of organic and conventional agriculture in comparison to other studies. Literature reviews that have drawn conclusions based on the analysis of significant amounts of research are especially valuable. The references used in this paper are diverse and numerous in order to present a broad comparison between conventional/monoculture agriculture and organic agriculture.

Very little is known about communities of soil microorganisms (Dance, 2008). This is because of the extreme diversity and abundance of soil microorganisms. A 2007 study aimed to resolve the differences between existing estimates of the number of bacteria species per gram of soil that had ranged from 2,000 to 8.3 million. The study concluded that the average number of unique species in a gram of soil was closer to 52,000 through studying ribosomal RNA sequences (Roesch, 2007). This study only considered bacteria. Other types of organisms are abundant as well. A study found that one cup of undisturbed soil contained 200 billion bacteria, 20 million protozoa, 100,000 metres of fungi, 100,000 nematodes, and 50,000 arthropods (Moravec et al, 2014). Science does not have a thorough understanding of soil ecology due to the numbers of species and organism interactions. Only a small percentage of soil organisms is known, documented, and studied. The interactions of soil organisms that have been studied have been shown to be complex (Ingham, 2015). Conventional agriculture encompasses the idea that humans can bypass these complex natural systems and develop superiorly efficient systems through the use of tilling, pesticides, monocultures, and fertilizers.

Tilling

Tilling has a variety of purposes. It is used to loosen and aerate the soil, kill weeds, dry out soil after wet winter seasons, and mix organic matter into the soil. There are various ways to measure the effect of plow tillage on soil. Cone penetration testing is a method used in determining soil stratigraphy which is the variation in soil composition with depth. It is done by pushing a cone through soil at a constant pressure, thereby measuring soil penetration. Studies by Alvarez and Steinbach (2009) showed that soils that are not tilled after harvest or before planting (no till) can have 50% higher penetration than plow tillage. They also tested bulk density, an indicator of soil compaction, and found that non tilled soils had less compaction compared to plow tillage. Soil penetration, compaction, and density are important determinants in soil ecology. Grant (1993) explains that compaction increases soil resistance to root extension. Soil compaction also reduces overall porosity, limiting oxygen movement to root surfaces which is required for carbon respiration and nutrient uptake. The increased compaction and density of soil resulting from plowing hinders the growth of roots and affects the ability of microorganisms to provide nutrients to plants because of limited oxygen.

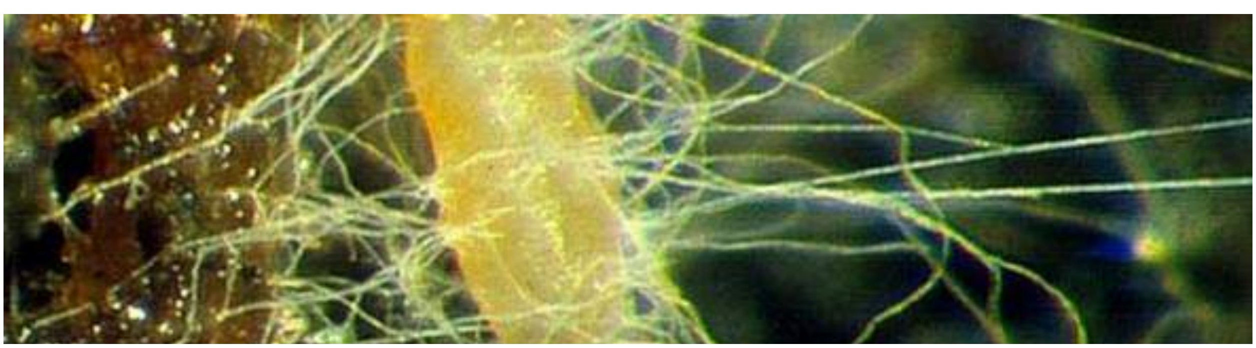
Tillage also affects aggregate stability in soils. Aggregate stability measures the ability of soil to resist disintegrating when disruptive forces such as plow tillage and water/wind erosion come in contact. When aggregate stability is high, it means organic matter content, biological activity, and nutrient cycling in soil are at healthy levels (Andrews & Wander, 2008). When comparing no till to plow tilled soils, Alvarez and Steinbach (2009) found that aggregate stability is higher in non-tilled soils and 70% higher in certain soil types. This indicates that plowing negatively affects levels of organic matter content, biological activity, and nutrient cycling and is disruptive to the soil ecosystem.

Soil compaction also affects how water interacts in soil ecosystems. Lipiec and Hatano (2003) explain that saturated hydraulic conductivity, which measures the ease with which water infiltrates pore spaces or fractures, is drastically reduced in compacted soil. In fact, the water infiltration in no till can be twice as high compared to low tillage (Alvarez & Steinbach, 2009). The reduction of saturated hydraulic conductivity in soil also results in increased water runoff and erosion. To exacerbate the issue, tractors, which are used to plough the soil and distribute pest control and fertilizer, further compact soil into ruts from the tires. Studies show that these traffic ruts are the main source of soil erosion and runoff even though they only make up 5% of the surface of a field (Fleige & Horn, 2000). Soil erosion and water runoff could be drastically reduced by no till methods. This would decrease sedimentation and the levels of fertilizers causing eutrophication in aquatic ecosystems and replenish water tables and aquifers.

Tilling greatly influences the biology in soils (Gomiero et al, 2011). Earthworms have been regarded as a useful indicator of soil fertility but have largely been displaced in farmland soils because tilling exposes them to predation from birds (Paoletti, 1999). No till, or minimum till soils tend to reduce the loss of earthworm biomass. Studies have confirmed this by highlighting a significant difference in the abundance of earthworms found in non tilled soils than under conventional tilling (Capperton et al, 1994). After healthy soil is tilled, worms become exposed to the outside air which attracts birds as shown in figure 1.

Figure 1: Tilling soils. Source: Mogalander, 2012

A specific soil layer that is affected by tillage is the rhizosphere. This is the region where plant roots associate with various fungi and bacteria. In the rhizosphere, the mycorrhizae process occurs. Mycorrhizae is the symbiotic relationship between plants roots and fungi where a plant provides carbohydrates derived from photosynthesis to arbuscular mycorrhizal fungi. In return,



the arbuscular mycorrhizal fungi, which can extend beyond the plant's roots, have access to more surface area and provide the plant roots with various minerals. This is shown in figure 2 (Aitken, 2015). The arbuscular mycorrhizal fungi have higher absorption capacity for water and can convert phosphorus minerals into a form that is useable to plants. This mycorrhizae symbiotic relationship is essential for a plant's health, specifically in the uptake of nutrients (Balzergue et al, 2011), (Kabir, 2005). Tilling has been shown to affect the colonization of arbuscular mycorrhizal around the rhizosphere which affects the plant's uptake of water and minerals (McGonigle et al, 1999). Under no till systems, arbuscular mycorrhizal survive at higher rates. Furthermore, when fields are not tilled, arbuscular mycorrhizal can follow old root channels and encounter other arbuscular mycorrhizal to form networks that create more effective nutrient channels (Kabir, 2005). Kabir (2005) notes that conserving arbuscular mycorrhizal is essential for maximizing benefits to crops and is also important for maintaining aggregate soil stability.

Organic farming tends to use no till methods for soil management because of the reasons explained above. But these reasons have not always been known to science. The National Oceanic and Atmospheric Administration (NOAA) (2015) explains that the dustbowl of the 1930s in the American Great Plains was due not only to a multi year drought, but also to land management practices which aimed to maximize agricultural production though intensive tillage methods. The US Department of Agriculture reports that no-till farming is growing in the United States at around 1.5% per year (Plumer, 2013). This is due to new technology in seeding techniques, and government subsidies for soil conservation. However, only 10% of American Agriculture uses no-till techniques. The worldwide percentage of no till agriculture is even smaller, with Europe, Asia, and Africa making up only 15% of the world's no till acreage as seen in figure 3.

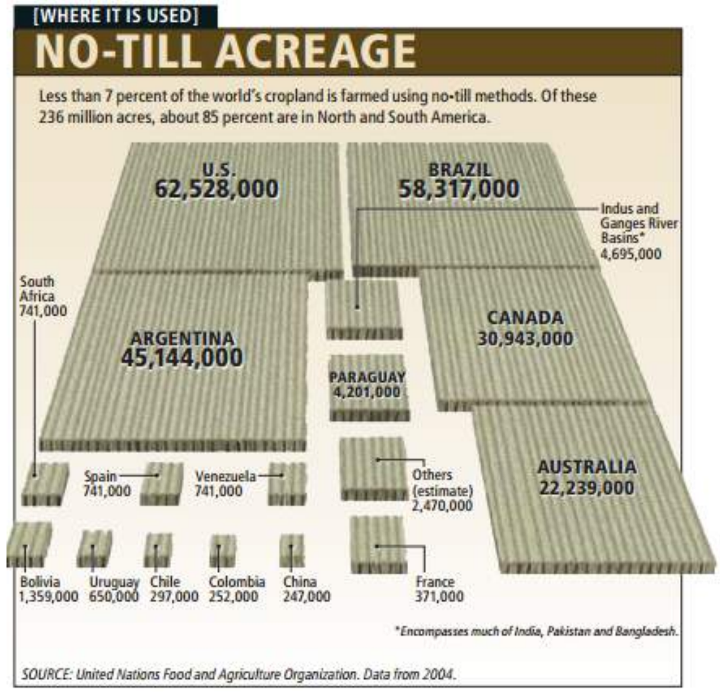


Figure 3: No-till farming around the world. Source: Plumer, B. (2013),

Pesticides

The United Nations Food and Agriculture Organization (UNFAO) defines pesticides as follows:

> "Pesticides refer to insecticides, fungicides, herbicides, disinfectants and any substance or mixture of substances intended for preventing, destroying or controlling any pest, including vectors of human or animal disease, unwanted species of plants or animals causing harm during or otherwise interfering with the production, processing, storage, transport or marketing of food, agricultural commodities, wood and wood products or animal feedstuffs, or substances which may be administered to animals for the control of insects, arachnids or other pests in or on their bodies. The term includes substances

intended for use as a plant growth regulator, defoliant, desiccant or agent for thinning fruit or preventing the premature fall of fruit, and substances applied to crops either before or after harvest to protect the commodity from deterioration during storage and transport." (UNFAO, 2013)"

This definition is indicative of an anthropocentric view of nature within the agricultural industry. The very existence of an abundance of pests is indicative of an unbalanced ecosystem. When these pests interfere with the production, processing, storage, transport or marketing of food, it is seen as harmful. But when human agriculture activity affects the production, processes, and the health of soil itself, there are not necessarily immediate monetary losses. The losses are only recognizable in the long run. The result is dirt, which is soil that has become devoid of life. Pesticides are a crude and simple, yet effective way to eliminate pests. However, many beneficial soil organisms are unintentionally killed at the same time.

Numerous studies confirm the effect that pesticides have on beneficial soil organisms (Bunemann et al, 2006) (Friedrich, 2005) (Gomeiro et al, 2011) (Makawi, 1979) (Merrington et al, 2002) (Smith et al, 2000). However, there are hundreds of thousands of different pesticides and not all of them have been studied. The Pesticide Action Network (2014) has a list of more than 375,000 current and historical chemical pesticides registered in the United States alone. As a result, upon studying literature reviews of the effects of pesticides, there is a lack of comprehensive data showing the overall effects of pesticides and their long term effects on soil health (Bunemann et al, 2006). Furthermore, some of the research that has been done on pesticides is kept confidential by chemical companies. Regardless, studies exist that show effects of specific pesticides on specific organisms and even the broader effects of pesticides. For example, one study showed that pesticides in general will sharply depress the counts of azotobacter and clostridia microorganisms (Makawi et al, 1979) which are especially important for providing nitrogen to plants. The abundance of pesticide types makes researching the broad topic difficult, although literature reviews have been completed in an attempt to assess categories of pesticides including herbicides, insecticides, and fungicides.

Herbicides, which are used to control weed growth, had little to no effects on soil biology, although some herbicides have been found to be toxic to earthworms and also affect the enzyme activity in microorganisms (Bunemann et al, 2006). Studies show that no till farming requires the use of more herbicides because weeds are not disturbed since the soil is not turned over (Teasdale et al, 2007). However, other studies indicate that integrated pest management strategies that do not use any chemicals have been demonstrated successfully. It is more of a lack of awareness and understanding that leads farmers to believe that no till farming requires more herbicides (Friedrich, 2005).

Insecticides have a much greater effect on the soil biology than herbicides. Organophosphate insecticides such as chlorpyrifos have been shown to have a significant impact on bacterial populations. The application of this insecticide reduced bacterial populations by 53.4% after 15 days and then a further 70.6% after 30 days. It took 120 days for the populations to return to normal rates comparable to the control (Pandy & Singh, 2004).

Fungicides are found to have an even more significant impact than herbicides or insecticides. While fungicides are meant to stop fungal disease, they also affect beneficial soil bacteria. Fungicides that contain copper have been shown to accumulate in soils and significantly reduce and stress microbial biomass. These copper residues accumulate in soils because they cannot dissipate from biodegradation, this has long term negative effects on soil (Merrington et al, 2002). Other studies show how beneficial fungi, such as arbuscular mycorrhizal are significantly affected by fungicides. Smith et al. (2000) found an 80% decrease in arbuscular mycorrhizal root colonization from benomyl fungicide application. With less arbuscular mycorrhizal fungi, the bacterial biomass, abundance of fungal-feeding, and predatory nematodes were reduced by 20, 12, and 33% respectively.

The absence of normal bacteria levels for significant periods of time (almost 4 months) coupled with the absolute losses of beneficial fungi, means plants are not receiving the ecological service that these bacteria and fungi provide. One of these services is protecting plants from diseases and pests. In *Symphony of the Soil,* Ingham (2013) explains that plants excrete exudates into the soil through their roots to promote the growth of bacteria and fungi. These exudates are made of simple sugars, proteins, and carbohydrates that attract bacteria and fungi that the plant needs. When the bacteria and fungi are killed by pesticides, they cannot defend the plant from disease and pests. When plants are more susceptible to disease and pests because of the absence of bacteria and fungi defenses, disease and pests are able to spread more quickly. The response might be to add more pesticides. This can become a vicious positive feedback loop as the application of pesticides can lead to the need for more and more pesticide application.

Another ecological service that is forgone without bacteria and fungi is providing plants with nutrients (Smith et al, 2000), and this absence of nutrients then results in the need to apply artificial fertilizers. This will be further explored in later sections.

The impact of pesticides on soil health as well as other ecosystems is becoming more apparent. In addition, the use of pesticides is an extremely inefficient process as only a small fraction of the pesticide reaches the intended target while the rest is effectively wasted (Aselage et al, 2009). This extra pesticide reaches untargeted environments such as ground and surface waters and distant terrestrial ecosystems (Aselage et al, 2009). Farmers who are aware of these issues are searching for alternatives. A promising alternative is integrated pest management (IPM) which uses natural predators, pest resistant plants, and other strategies to maintain yields and healthy soils without using pesticides. A major IPM strategy is intercropping, interplanting, and polycultures, which increase the biodiversity of agriculture ecosystems. Regardless, pesticide production and imports are expected to continue growing based on historical trends shown in figure 4 (Tillman et al, 2002).

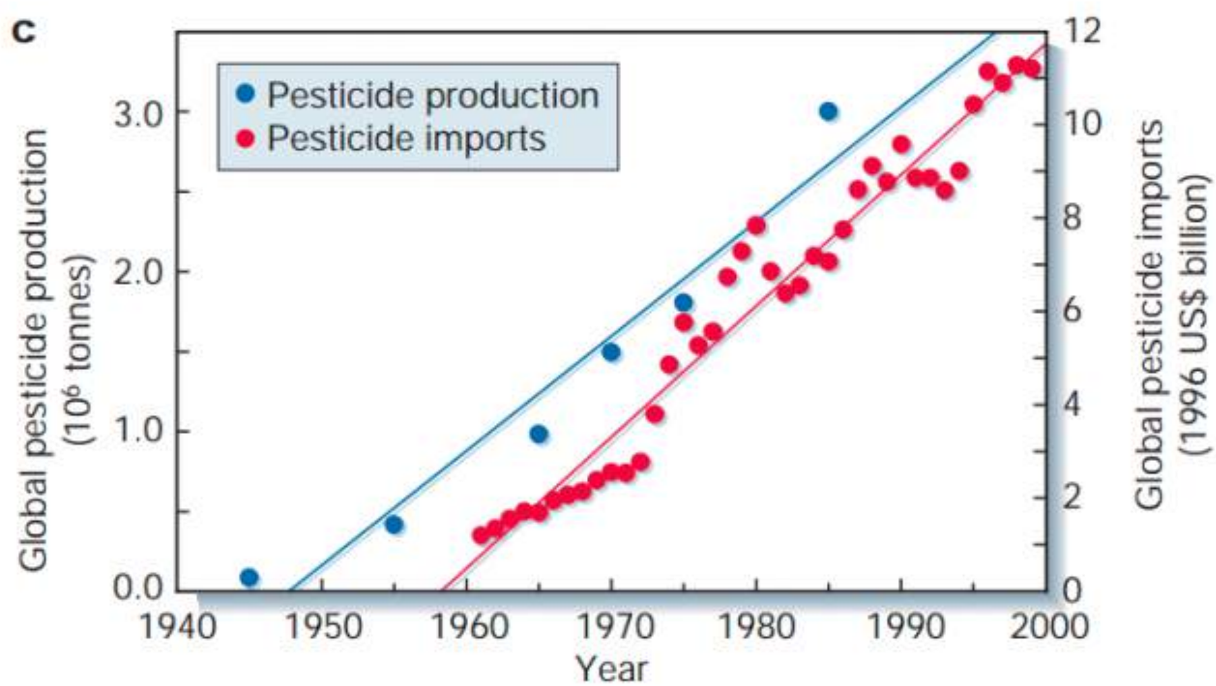


Figure 4: Total global pesticide production and global pesticide imports 1940s-2000 Source: Tillman et al, 2002

Biodiversity and Monoculture

Part of the reason pesticides are needed and used in conventional monoculture methods is because there are lower levels of biodiversity in these systems. Monocultures are more vulnerable to pests because homogeneous and simplified ecosystems are weak and less resilient. The higher levels of microbial activity and biodiversity associated with organic agriculture enhance nutrient balance in plants which increases the plants ability to respond to pests (Phelan et al, 1996). Furthermore, pests cannot cause as much damage in organic systems because there are more predators that can keep the level of pests in balance. These characteristics of monoculture systems make them inherently vulnerable to pests. This point is argued by Naeem et al. (1994) who found that reduced biodiversity alters the performance of ecosystems. This particular study measured five factors including community respiration, decomposition, nutrient retention, plant productivity, and water retention. All of these factors increased with higher levels of biodiversity.

The application of pesticides, as explained previously, will harm the life in the soil. With less biodiversity, plant productivity decreases. This is counterintuitive to the ultimate goal of agriculture which is to produce plants in the most cost effective and sustainable way. Conversely, without using pesticides, organic agriculture, while still using row cropping to some extent, usually increases species richness with 30% higher levels of species compared to conventional agriculture (Bengsston et al, 2005). The study showed increase richness and heterogeneity of birds, insects, plants and soil organisms.

Not only is species diversity increased in organic agriculture, but organisms are 50% more abundant as well. The abundance of birds, insects, plants and soil organisms all increased while pests did not (Bengsston et al, 2005). However, this finding was limited to organic agriculture on smaller sized plot and field scales while effects were less significant for larger agricultural areas. The study noted that species diversity and abundance were dependant on intensively managed agricultural landscapes. In other words, large farming plots that are comprised of hundreds of hectares of land would have less diversity and abundance. Bengsston et al. (2005) suggest that measures to preserve and increase organism biodiversity and abundance should be farm and landscape specific. This type of strategy is contradictory to the standard accepted strategies of conventional agriculture which favour universal models that are convenient for replication and mass scalability.

Fertilizer

The use of fertilizers in agriculture has increased productivity significantly since they were introduced in the 1930s in the developed world and in the 1960s in the developing world. The use of fertilizers in developed countries is show in figure 5 (Roser, 2008). Synthetic fertilizers were part of the technology that contributed to the Green Revolution. The dramatic increase in worldwide agricultural productivity significantly reduced starvation rates in the developing world (Lal, 2009). Synthetic fertilizers have been and continue to be beneficial in that regard but they also have negative effects on soil ecosystems.

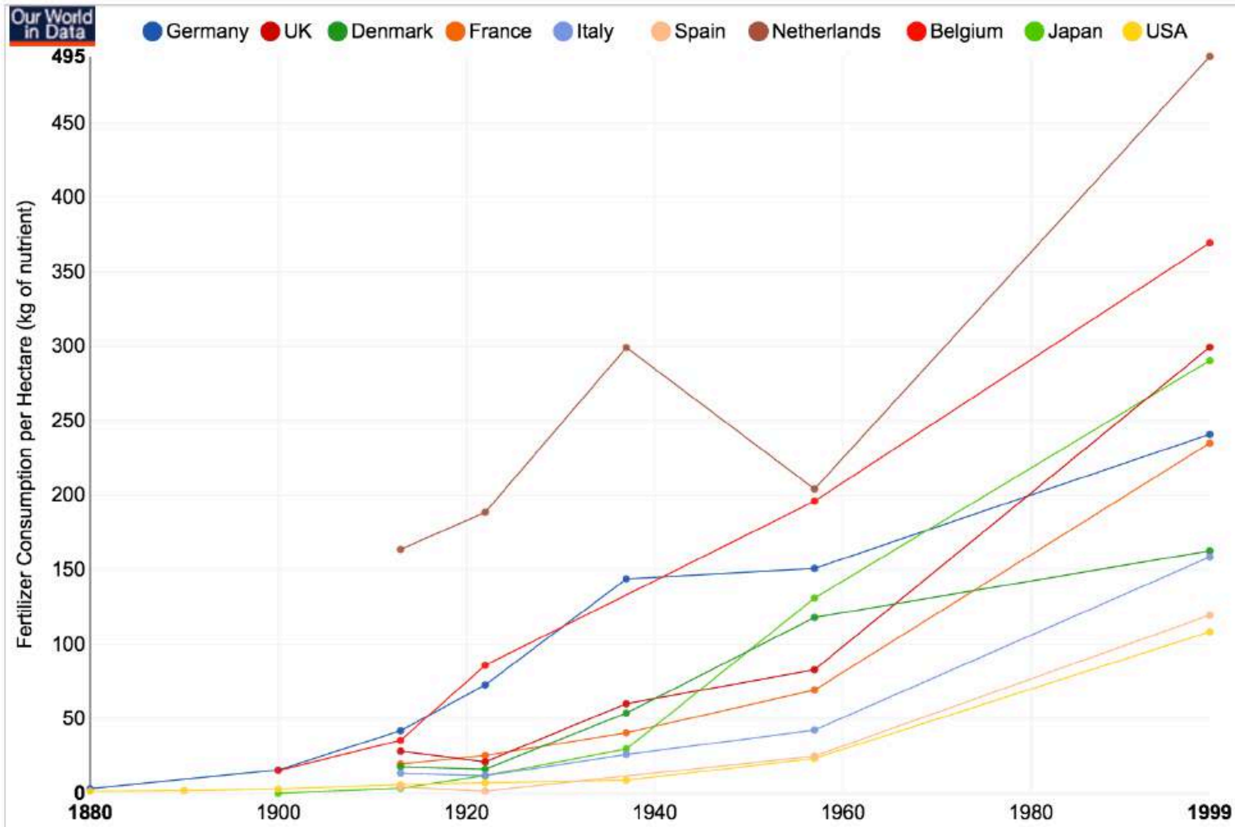


Figure 5: Fertilizer consumption per hectare in developed countries over the long run. Source: Roser, 2008

All living things require nutrients. In any type of agriculture, carbon, nitrogen, potassium, and phosphorus are critical nutrients for growing and fruiting. There are also a number of important trace elements such as sulphur, magnesium, calcium, iron, zinc, and copper. The relevant difference in conventional and organic agriculture is how plants access these nutrients and minerals. In conventional agriculture, synthetic chemical fertilizers are applied and in organic agriculture, organic matter and compost are applied. Understanding how nutrient cycle through ecosystems is important for determining how plants take up nutrients from soil ecosystems.

Soil is the biggest terrestrial reservoir of carbon, another essential element needed by plants and soil ecosystems (Murck et al. 2013). Carbon is taken into plants through photosynthesis where carbon dioxide in the atmosphere is converted to sugars and starches, becoming part of the plant's carbon based organic matter. This organic matter is returned to the soil when the plant dies, or drops its leaves, and feeds decomposing organisms and bacteria. Carbon can remain in the soil for thousands of years and is also emitted back into the atmosphere in significant amounts through soil respiration of carbon dioxide (Gaudinski, JB et al, 1995).

Nitrogen is a basic building block of life and promotes growth in plants. It is abundant in the atmosphere but plants cannot use it in a gaseous state. Specialized bacteria have developed symbiotic relationships with certain plants to combine nitrogen gas from the atmosphere with hydrogen ions to form ammonium or $NH4^+$. This is a form of nitrogen that plants can use. The specialized bacteria that provide this ecological service live freely in the soil and in the root nodules of leguminous plants and some other plants. This natural cycle has been used in an industrial form by humans as the Haber-Bosch process, developed in 1909 (Murck et al, 2013). This process converts atmospheric nitrogen into ammonia, by reacting nitrogen (N_2) to ammonia (NH_3) by a reaction with hydrogen (H_2) using a catalyst at high temperatures. The industrialization of this process has doubled the rate of nitrogen input into the terrestrial nitrogen cycle, which has caused losses in other soil nutrients, increased the acidification of soils and aquatic systems, and caused biodiversity loss especially in plants that have adapted to the efficient use of nitrogen (Vitousek et al, 1997).

The phosphorus cycle is another major part of soil ecosystems. The phosphorus cycle is relatively slow compared to other cycles because it is usually trapped in rock sources and then eroded over time into soils. Plants take up phosphorus from the soil and it is transformed into organic matter. Animals will eat plants and the phosphorus can become part of their system or be excreted back into the environment. Over time, phosphorus minerals will be washed into oceans and will accumulate into sedimentary layers. Over longer periods of geologic time, these rocks will become exposed to terrestrial environments and once again repeat the process. In organic agriculture, phosphorus is added to soil in the form of manure while in conventional agriculture, phosphorus in added through fertilizers derived from mined phosphorus.

The potassium cycle is similar to the phosphorus cycle and is the final nutrient cycle discussed in this report. Potassium is an essential nutrient for plant functions such as carbohydrate metabolism, enzyme activation, and protein synthesis. Potassium is also important in the growth of the edible parts of plants such as grains and tubers (University of British Columbia, (2015). Potassium is a highly reactive alkali metal and is usually bonded with sodium to form salt deposits commonly known as potash. Similar to phosphorus, potash is mined from the earth. Potash deposits are formed when inland seas evaporate and salt layers remain. Parts of Saskatchewan were covered by an inland which dried around 400 million years ago to form one of the world's largest potash deposits (Ladurantaye, 2010). Both potassium and phosphorus based fertilizers are essentially non-renewable resources as their formation is based on long geological time scales.

In conventional agriculture, chemical fertilizers are needed because the soil has been depleted of the biota that are necessary for transporting nutrients and minerals to plants in an accessible form. These biota, including bacteria, fungi, and nematodes, are harmed by the tilling and pesticide use that is so common in conventional agriculture systems. Without these biota, important ecological services that are essential to healthy plants are not available. Nitrogen, potassium, and phosphorus agricultural fertilizers have been developed in a form that plants can readily access and which bypass the ecological services provided by the soil biology. This presumably makes the various biology in the soil ecosystem less important. However, as explained in the previous section, discussing the rhizosphere, plants need the soil biota to provide other trace minerals and nutrients that are not as readily available in synthetic chemical form. The trace minerals and nutrients exist in the soil already but are only made available to plants through their relationship with the beneficial fungi and bacteria (Ingham, 2015).

Fertilizers do effectively increase plant growth. However, compost application is a viable substitute for fertilizer when there is access to large quantities. Large quantities of compost are a limiting factor and may be out of reach to many agricultural ventures trying to cut back on fertilizer use. Nitrogen, is found to be the limiting nutrient when compost is used in place of nitrogen, phosphorus, and potassium fertilizers (Evanylo et al, 2008). However, the compost provides benefits that fertilizers do not. Compost tends to improve the physical properties of soil such a bulk density, porosity, and water holding capacity. The compost amended soils reduce mineral and sediment runoff. Evanylo et al. (2008) also showed that compost increased the rate of carbon sequestered in the soil. It is important to note that the various benefits of using compost accrue over time and are not realized until compost has been applied for a number of years. This could be a major issue for farmers who are struggling with marginal returns and need to maximize yields to make a steady income.

Earthworms, an indicator of soil health, are also affected by fertilizer use. Organic matter and compost are food sources for worms and are essential to their survival. Earthworms respond better to organic manure and compost than to chemical fertilizers (Paoletti, MG, 1999). Because worms eat dead organic matter and compost, the application of compost in agricultural settings will promote earthworm biomass. When worms are present in soil, their waste creates highly fertile worm castings. The ecological service that worms perform is becoming increasingly recognized for the soil improvement without the risk of pathogens or other polluting substances. Furthermore, worms have been found to eliminate harmful chemical substances and heavy metals which also improve the the quality of polluted soils (Iozon et al, 2003). Worm castings are also shown to increase yields and fruiting quality of certain plants (Panicker et al, 2009). These studies are limited to specific plants and their fruits. More research is needed to make a broader statement about yields and fruiting quality relating specifically to worm castings. Regardless, the source of nutrients and the success of plant growth is largely associated with biology in the soil. The application of compost and organic matter increases the activity and biomass of decomposers while fertilizers tend to have a negative effect on their abundance.

Another biological effect that fertilizers have is on the arbuscular mycorrhizal. The use of phosphorus fertilizer makes crops less reliant on arbuscular mycorrhizal. Numerous studies indicate that the colonization and spore numbers of arbuscular mycorrhizal is invariably reduced in soils with phosphorus fertilizer applications (Gosling et al, 2006). One study showed that even moderate application of phosphorus fertilizer of can reduce arbuscular mycorrhizal colonization and spore numbers by 50% (Martensson & Carlgren, 1994). This finding indicates that fertilizers that are meant to increase plant growth actually may diminish the natural ability of soils to provide nutrients to plants. The resiliency of soil ecosystems is thereby decreased and this can create positive feedback loops. As explained in the tilling section, arbuscular mycorrhizal increase a plants ability to absorb phosphorus. By adding more inorganic phosphorus to the soil, the plants ability to absorb it is decreased because of the diminished mass of arbuscular mycorrhizal. Arbuscular mycorrhizal not only help plants uptake phosphorus, but create transportation networks for many other nutrients as well. The presence of arbuscular mycorrhizal is therefore essential for healthy soil ecosystems.

In addition to harming the biology of the soil, fertilizers have been found to affect the very chemistry of soil. Many experiments have been conducted which show the use of nitrogen fertilizers leads to the losses of nutrient cations (positively charged ions) and increases soil acidification (Vitousek, et al, 1997). A study at the University of Hawai'i (2015) indicates that this happens because nitrate fertilizers are anions (negatively charged ions) and will move through the soil with water and attract cation nutrients and trace minerals such as calcium. The soils are then depleted of calcium and other nutrients and trace minerals. When calcium in particular is absent from soils, there is increased leaching of toxic inorganic aluminum. Soils become more toxic, which decreases the ability for plants to absorb nitrates (Durka et al, 1994). This creates a positive feedback loop in the soil system with fertilizer use. When fertilizers are meant to increase the fertility of the soil and boost productivity and output, they will invariably cause the soil to reduce a plant's capacity to absorb nutrients in the long term. If a farmer determines that his crops are not growing well because they are not taking up nitrates, the first solution he might have is to add more nitrogen fertilizer. This solution is sought out because nitrogen fertilizer, in particular, is a relatively inexpensive commodity. It is one of the least expensive and most effective ways of increasing yields (Vitousek, et al, 1997). However, without the farmer's knowing, this will further diminish the soils capacity for nutrient uptake and increases acidification as well as quantities of toxic compounds. The overall efficiency of fertilizer has thus been found to have diminishing returns as shown in figure 6 (Tillman et al, 2002).

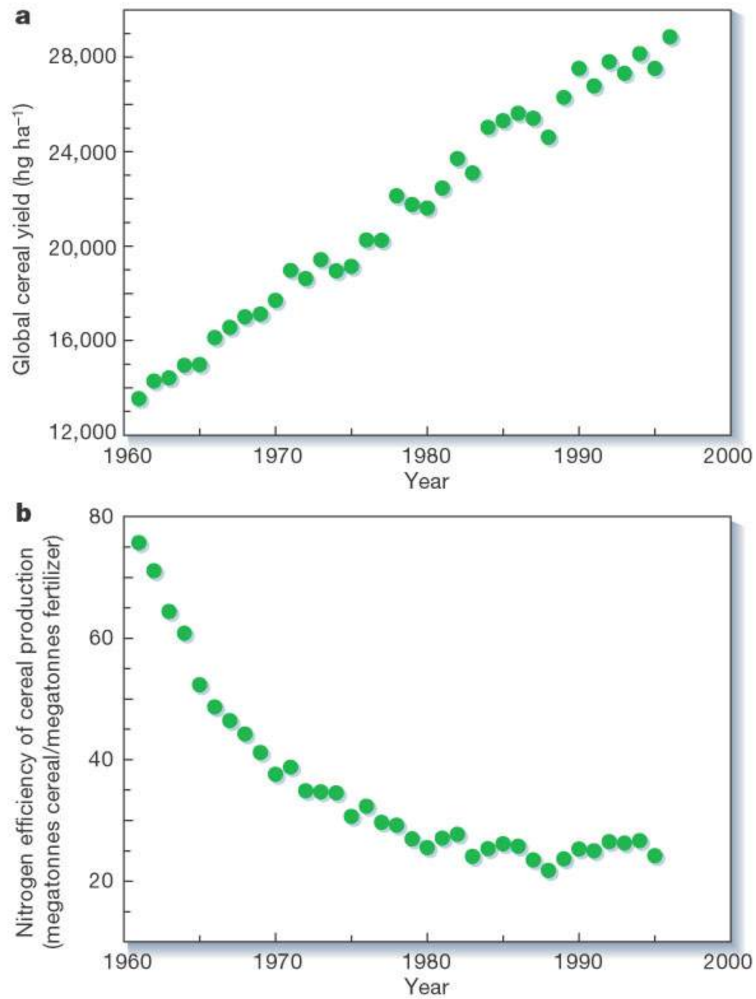


Figure 6: a, Trends in average global cereal yields; b, trends in the nitrogen-fertilization efficiency of crop production (annual global cereal production divided by annual global application of nitrogen fertilizer) Source: Tillman et al, 2002

Increasing the application of compost, and leaving more organic matter on soils instead of tilling soil and leaving it exposed to wind and rain, will benefit the very chemistry of the soil over time (Clark et al, 1998). The ratio of carbon to nitrogen is a factor that correlates a with soils capacity to retain nitrogen. Greater levels of carbon in the soil allow for higher levels of nitrogen retention (Vitousek, et al, 1997). As mentioned in the carbon cycle section, carbon in soil comes from decomposing organic matter.

Regardless of the issues that fertilizers pose, they are important for maintaining or even increasing yields, which is necessary for the food security of the world's growing population. Figure 7 shows the correlation between fertilizers and yields (Dyson, 1996). It is difficult to foresee fertilizers being omitted from agriculture from this equation entirely. However, it has been demonstrated that fertilizer use can decrease when used in conjunction with compost. A partial substitution of nitrogen, potassium, and phosphorus with compost and green manure (undecomposed plant matter) was as good as a 100% recommended use of NPK fertilizer without any compost and green manure for wheat and rice cropping systems in India (Yadav et al, 2000).

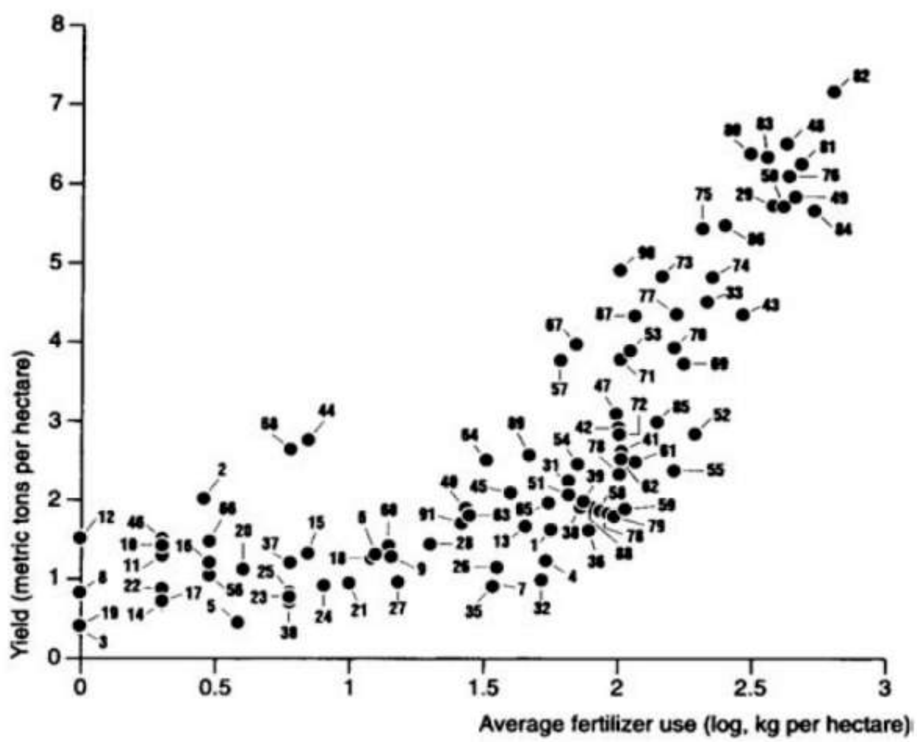


Figure 7: Fertilizer use and cereal yields in sample countries, 1989-1992. Source: Dyson, 1996

Fertilizer use can also be decreased with technological and knowledge based solutions. A study in Hawaii of sugar cane plantations compared two techniques for applying nitrogen fertilizer (Matson et al, 1996). The more technological and knowledge based approach involved applying dissolved nitrogen fertilizer through irrigation systems in small amounts, and with frequent applications based on timed requirements of the growing sugar cane. The other approach involved a more conventional and simple method that involved applying larger amounts of nitrogen less frequently. The technological and knowledge based approach using irrigation systems used 2/3 as much fertilizer and the losses of nitrates were ten times less than the more fertilizer intensive system. In addition, there were increased yields and it was more profitable as well.

These two significant findings show how more sustainable agriculture management methods do exist and can be incorporated into conventional farming practices. As such, creating a dichotomy or mutually exclusive perspective between organic and conventional agricultural methods can be unnecessarily polarizing. This polarization between organic and conventional agriculture can affect the perception that farmers, regulators, and other stakeholders have about adopting practical, technological, and scientific changes in their activities.

Yields, resources, and sustainability

The importance of yields cannot be overstated. At the same time, the world is finite and increasing yields through the continued reliance on external chemical fertilizer, pesticide, and water inputs poses growing challenges and issues from a sustainability perspective. Soil is not necessarily an easily renewable resource as it takes a very long time for soil to accumulate (Murck et al, 2013). Most of the inputs that are required for conventional farming are derived from non sustainable sources. Pesticides are derived from fossil fuels and chemical sources, while the concept of peak phosphorus and peak potassium, which is similar to peak oil, poses the question of when the finite resources of phosphorus will be depleted. These issues need to be considered when evaluating the sustainability of agricultural yields in the long run.

Yields from conventional/monoculture agriculture are generally higher than organic agriculture. A quantitative synthesis of studies conducted by Badgley et al (2007) show that yields in organic agriculture were around 8.7% lower than conventional agriculture with a 95% confidence interval based on 138 comparisons. Meanwhile, Kremin & Miles (2012) conclude there is still uncertainty about the comparison between differing yields from organic and conventional agriculture because of a general lack of available studies. This is still an area where more research is needed. Regardless, the large scale tendencies of conventional farming operations result in lowering direct costs of production through achieving economies of scale. However, like other industries, conventional farming has brought about significant costs that have been externalized to society and natural ecosystems. It can take 500 to 10,000 years for one centimetre of topsoil to accumulate depending on climate, biome, and rate of mineral accumulation (Murck, 2013). Soil then, is technically a renewable resource, but not in human lifetimes. To manage any renewable resource, the exploitation cannot exceed regeneration. The Food and Agriculture Organization of the United Nations has estimated that 24 billion tons of fertile soil are lost each year across the world. This represents $490 billion in lost ecological services in one year. As soil is lost, humanity’s capacity to grow food decreases. This is a sobering fact when considering the future demands for food as the world’s population grows. Tilman et al. (2002) expect food demand to double by 2050 based on the increasing consumption of calorie and meat intensive diets.

Society and future generations will bear the cost of lost soil resources. But when farmers are working with small profit margins, yields are one of the major factors in determining operational strategy. As such, farmers will be hesitant to adjust their strategies from conventional agriculture to organic agriculture. However, by using net present value analysis, it has been shown that cropping systems in certain areas which are transitioning from conventional to organic are competitive with conventional systems when there is a price premium for organic produce. On organic farms, yields are shown to be smaller for corn, generally but not significantly smaller for wheat and alfalfa, and no significant difference in soybeans. In addition to these marginal decreases in yields, larger investments in production for farm labor, equipment and machinery were necessary on the organic farming systems in this study. The cost savings came from less expenditure on purchased fertilizer and pesticide, and fuel inputs, as well as the cost savings from not tilling the soil (Archer et al, 2007). These studies show that the competitiveness and profitability of organic agriculture systems was dependant on a price premium. The study did not take into account the long term resilience of conventional or organic systems as it was conducted over four years, but shows that, in general, savings from fertilizer, pesticide, and fuels inputs, coupled with revenues from price premiums on organic produce make up for lower yields. This is an important finding which shows the viability of organic farms from an economic cost perspective.

There is an abundance of energy in our present day economies. But as the world economy becomes more energy stressed, energy efficiency will become a more important determinant in agricultural systems. Energy efficiency in agriculture can be analyzed to highlight differences in sustainability. It can also be used to forecast future competitive advantages. As energy becomes less available in the long run, farms that are more energy efficient will have a competitive advantage. Agricultural energy efficiency is calculated as a ratio between yields and total energy inputs. Although yields tend to be lower in organic agriculture systems, energy efficiency is generally higher when indirect costs are taken into account (Dalgaard et al, 2001), (Wood et al, 2006).

Considering time scales is important for analyzing organic and conventional agriculture with respect to yields because of the relatively recent rise and use of pesticides and fertilizers. A long term trial has lasted for 150 years in the Rothamsted Experimental Station in the UK which has looked at the yields for organic wheat for an organic plot using compost and a conventional plot using chemical NPK fertilizers. Nitrogen levels are shown to have increased by 120% in the organic plot over 150 years compared to the conventional plot with an increase of around 20%. Soil organic matter levels were also much higher in the organic plot. The application of compost resulted in increased nitrogen and soil organic matter which are main contributors to higher yields in the organic plot. Yields were 3.45 tons per hectare in the organic plot, compared to 3.4 tons per hectare on the conventional plot (Gomiero et al, 2011). While the yield increase is marginal, the need for external inputs of the organic plot is significantly lower. At the same time higher levels of soil organic matter indicate healthier soils.

The findings from studies at Rothamsted Experimental Station have important implications for the sustainability of agriculture as well as yields. External inputs of pesticides and fertilizers may increase short term gains but when long term sustainability is the goal, soil health is arguably more important than yields. Organic agriculture avoids the use of external inputs of fertilizers and pesticides and is even more effective at improving soil water content and increasing water use efficiency over time (Gomiero et al, 2011).

As the world experiences climate change and becomes more water strained, humanity needs to ensure that soil ecosystems are resilient enough to support agriculture. Successful and resilient agriculture will have the characteristics of yield stability and resource use efficiency facing the adversity of climate change and water and energy shortages. Lotter et al indicate that organic cropping systems under drought conditions had higher yields than conventional systems. Under severe drought conditions organic yields can be 70-90% higher than conventional systems. The mechanism allowing for this is healthy soil that has higher water holding capacity. Furthermore, organically managed systems in the long term were found to have lower yield variability as well as higher cropping system variability (Lotter et al, 2003). When considering the broader scope of organic agriculture, Kremin & Miles (2012) and Gomiero et al (2011) suggest that the benefits of maintaining and enhancing soil ecosystems increases resilience and sustainability of human agricultural systems. Furthermore, the lower productivity that tends to be characteristic of organic agriculture is balanced out by the environmental benefits and reduction of negative externalities.

Conclusion

This paper has demonstrated that soil is a highly complex ecosystem of interactions between minerals, nutrients, bacteria fungi, living plant roots, larger decomposing organisms, decaying organic matter, and also humans and their strategies for achieving high rates of productivity. Conventional/monoculture agricultural strategies that aim to simplify ecosystems in order to achieve high rates of productivity have generally been found to have higher yields but also significant negative effects on soil ecosystems. The use of tilling, pesticides, fertilizers, and homogenous planting systems in conventional agriculture tend to increase yields and profitability but the associated negative effects are too often ignored, unaccounted for, and externalized on society and other ecosystems. When agriculture is analyzed from a holistic approach that considers a wide range of variables beyond just profits and yields, the argument for implementing organic agriculture systems is robust, since conventional agriculture systems are unsustainable in the long run. The findings described in this paper suggest that the the benefits of maintaining and enhancing soil ecosystems through organic agriculture increases the resilience and sustainability of human agricultural systems. Furthermore, the lower productivity that tends to be characteristic of organic agriculture is balanced out by the environmental benefits and reduction of negative externalities. Organic agriculture will become an increasingly important component in providing for humanity’s growing food needs in a sustainable manner especially considering the issues of climate change and soil loss.

Franklin Roosevelt said, “The nation that destroys its soil destroys itself.” Understanding soil ecosystems and expanding the research area and its findings are important for sustainable agricultural decision making and humanity’s future.

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Toxicological Effects of Nanomaterials on Aqueous and Terrestrial Ecosystems

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Toxicological Effects of Nanomaterials on Aqueous and Terrestrial Ecosystems

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Abstract

This paper provides a holistic overview of the effects of engineered and natural nanomaterials (ENMs and NNMs) in the environment. Through increasing production and technology, NMs are contaminating the air, soil, and water with large toxicological implications. The proposed pathways of nanomaterials in the environment suggest that they accumulate in terrestrial and aquatic environment due to emissions, surface runoff and effluent from wastewater treatment facilities. Nanomaterial toxicological studies have also shown large bactericidal and mutagenic effects due to the metal ion properties, stabilizing agents and the size and shape of the NMs. (Suresh et al., 2013). These adverse effects that were seen on bacteria and microbes were also observed in plants and fish. Plants were seen to have inhibited growth, and even death when coming into contact with specific nanoparticles. (Burke et al., 2015) However, some conflicting data exist where plants have shown improvement in growth under specific exposure. (Lahiani et al., 2008) Fish and aquatic organisms were also studied in order to determine the health impacts on these species as well as the potential for biomagnification. Organ failure was observed in zebrafish and carp due to acute exposure to metal NPs (Griffit et al., 2011). Other harmful effects have also been observed in aquatic ecosystems, with many engineered nanoparticles settling in the sediment of freshwater and marine environments causing risk to benthic species (Handy et al., 2011). The impact on human health was also reviewed, revealing that the exposure to nanoparticles has lead to nausea, vomiting and even death. Human organs can also be severely affected by inhaling, ingesting or coming into contact with large amounts of nanomaterials. (Tang et al., 2015).

Key Words: Nanomaterials, Nanotechnology, Nanotoxicology, Aqueous Ecosystems, Terrestrial Ecosystems, Biomagnification, Engineered Nanoparticles

1.0 Introduction

Nanotechnology and nanomaterial (NM) containing products are part of a multi billion-dollar industry, with a wide variety of uses ranging from electronics to biomedical applications. Due to growing industry, it is inevitable that engineered and natural NMs will be released into the atmosphere, affecting both the water and soil. This could be from intentional or unintentional release, which means that it is vital to determine the potential impacts this growing industry can have on environmental, human, animal and plant health.

Nanoparticles are naturally occurring in the environment, however through technological breakthroughs the abundance of new and engineered nanoparticles not found naturally are becoming prevalent in the atmosphere. Due to lack of knowledge there is no regulation on the emissions and many scientists are concerned that this could pose a threat to as a new class of environmental hazards. Nanomaterials are “structures with at least one dimension less than 100nm” which, due to their small size and large surface area NMs are widely used in various products, increasing the routes in which they can come in contact with organisms and plants. (Van Aken, 2015) (Hou. et al., 2013) Release of nanomaterials in the environment through emissions, industrial and commercial products can have a large effect causing them to end up in wastewater treatment plants and furthermore to surface water. (Hou. et al., 2013) The nanoparticles that are not filtered in wastewater treatment plants are most likely to “accumulate in benthic sediments” which can possibly cause problems for many species in lakes and rivers. (Batley. et al., 2011) Aquatic organisms are “particularly susceptible to pollutants due to their large, fragile, respiratory epithelium”. (MacCormack. et al., 2008) Changes in pH, water temperature, oxygen levels can also increase the risks associated with nanomaterials in aqueous environments and must be considered when addressing risk. Plants are also susceptible to high exposure of nanomaterials through soil contamination or accidental release. Nanomaterials have been shown to enter living organisms and "exert toxic effects at the cellular level, including membrane disruption, protein inactivation, DNA damage, and disruption of energy transfer and release of toxic substances". (Van Aken, 2015) Due to the importance of plants and bacteria in the food chain it is crucial to understand the impact that this large industry could have in the future of environmental and human health. This paper will focus on the toxicological effects of waste engineered nanomaterials in terrestrial and aquatic environments and their implications on human health and environmental safety.

2.0 Background

2.1 Definition of Nanomaterials

Nanoparticles are naturally occurring in both aquatic and terrestrial environments in form of colloids, "mineral precipitates (Al, Fe, MgO, OH)" and dissolved organic matter". (Batley et al., 2011) The International Organization for Standardization (ISO) has classified three groups of nanomaterials. First, nanoparticles, which include all "three dimensions between 1 and 100 nm", are the most commonly referred nanomaterials. However, dimensionality plays a large role in determination of NM's showing that nanoplates (two dimensions between 1 and 100nm) and nanofibers (one dimension between 1 and 100nm) are also classified as NM's by ISO. (Batley et al., 2011) NNM's (natural nanomaterials) have always existed in the environment with little known toxicological effects. However, these definitions based only on size may not be sufficient in addressing toxicological risk, due to the fact that the nanomaterial version of a material exhibits different properties than it's non-nano counterpart. (Boverhof et al., 2015) With increasing nanotechnology and new products, engineered nanomaterials (ENM's) are now emitted into the air, water and soil. These ENM's can be categorized in "seven main classes: carbonaceous nanomaterials (carbon nanotubes (CNT's));

semiconductors (quantum dots); metal oxides (zinc oxide); nanopolymers (dendrimers); nanoclays; emulsions (acrylic latex); and metals (silver, gold).” (Buckley et al., 2011)

2.2 Properties of Engineered Nanomaterials and Microbial Toxicity

Suresh et al., (2013) have written a comprehensive review of the properties of nanomaterials and their relationship to microbial toxicity. Due to the fact that “microbial consortia underlie environmental processes” it is crucial to study the toxic effects of nanoparticles and the impacts this can have on “trophic balances”. Nanomaterials are used in medicine because of their bactericidal and fungicidal properties, the most well known being Ag and CuO nanoparticles. (Suresh et al., 2013) However, when these engineered NM’s are released into the environment such properties might have serious long-term impacts on both terrestrial and aquatic species. Some studies have tried to correlate the parent metal toxicity with that of its nano-form, however the size, shape, coating and the way the nanoparticle is synthesized determine that its toxicity is different than its parent metal. Furthermore, manufacturing processes add detergents, additives and other chemicals, which are not fully removed from the final product, increasing the toxicity. (Suresh et al., 2013) Metal nanoparticles are studied because they dissolve into ions in aquatic environments, which is “often a primary step and common cause for nanoparticle toxicity”. (Suresh et al., 2013) The most widely studied and used ion for its microbial toxicity is silver, yet there are many studies showing that the “correlation between nanoparticle toxicity and that of its dissolved ion” is seen with other materials as well. (Suresh et al., 2013) Secondly, the coating of a nanomaterial determines the toxicity and effect it will have on bacteria. Usually, engineered nanoparticles are “surrounded by a shell or cap to act as a stabilizing, biocompatibility and or reacting agent”. (Suresh et al., 2013) This shell can affect the charge of the nanoparticle and its interaction with the environment creating a completely different toxic effect than the nanoparticles parent material. Baumann et al., studied the acute effects on

Daphnia Magna of four different types of coatings on iron oxide nanoparticles which stabilized the NPs yet had toxic effects on the organisms, including death. Lastly, the size and shape of a nanoparticle can have a large effect on bacteria "as the particle size decreases, the ratio of surface area to mass increases" resulting in changes to the "physical-chemical properties" of the nanoparticle. (Suresh et al., 2013) This creates novel applications due to "surface atom reactivity, electronic and optical properties" which can influence binding characteristics in bacteria and increased reactivity. (Suresh et al., 2013) A general trend of increased toxicity with a decrease in size has been observed due to increased reactivity of smaller particles. (Suresh et al., 2013) Below is a table highlighting the known microbial nanotoxicity studies.

Table 1. Known effects of nanoparticles on bacterial strains. (Source: Suresh et al., 2013)

Nanomaterial	Size (nm)	Surface coating	Dosage[a] $mg\ L^{-1}$	Bacterial strains	Mode of action[b]	Ref.
Ag	10	Myrasmistin	2.5–5	*B. subtilis, S. aureus, L. mesenteroides*	Bactericidal	74
Ag	6.7 or 7.2	Mercaptopropionic acid or polylysine	7	*E. coli*	Growth inhibition and inactivation	104
Ag	25	-	4.8 ± 2.7	*S. mutans*	Cell membrane damage	63
Ag	1	-	10	*P. putida*	Bactericidal	105
Ag	9–21	-	>1	Nitrifying bacteria	ROS	106
Ag	20 80	Phosphate	0.25–1 ppm	*N. europaea*	Decreased ammonia monooxygenase activity and outer membrane destabilization	46
Ag-oleate	4 ± 1	Protein	5–7.5	*E. coli, B. subtilis, S. oneidensis*	Non-inhibitory	3
Ag-colloidal Ag-biogenic	9 ± 2 4 ± 1.5	- Protein	2–12	*E. coli, B. subtilis, S. oneidensis*	Cell membrane damage	3
Ag–TiO_2	10–80	-	-	*B. subtilis, P. putida*	Bactericidal	80
Ag_2S	2–20	Protein	50–150	*E. coli, B. subtilis, S. oneidensis*	Non-inhibitory	41
Ag@MESs	100	MES	-	*E. coli, B. anthracis*	Growth inhibition	107
Ag–MgO	ND	-	4	*E. coli, B. subtilis*	Cell-membrane damage	48
Al_2O_3	179	-	0.1–1	*E. coli*	Minor growth inhibition	108
Al_2O_3, SiO_2, TiO_2, ZnO	1–100	-	10–200	*E. coli, B. subtilis, P. fluorescens*	Bactericidal	38
CeO_2	3–50	-	50–150	*E. coli, B. subtilis*	ROS	4
CeO_2	3–50	-	50–150	*S. oneidensis*	Non-inhibitory	4
CeO_2	10 25 50 60 5000	-	2.4–29.6	*P. subcapitata*	Cell wall and membrane disruption	109
SiO_2, TiO_2, ZnO	10–1000	-	205–480	*E. coli*	ROS, cellular internalization, membrane disorganization	15
CuO	10	-	70–300	*P. putida*	Bactericidal	105
CuO, spiked multiarms	500–1000	Uncoated	500	*E. coli, S. typhi, S. aureus, B. subtilis*	Bactericidal activity	70
CdSe–CdS	2–10	-	-	*P. aeruginosa*	Non-toxic	57
CdSe	8	Citrate	50	*P. aeruginosa*	Membrane damage, impaired growth and ROS	40

Nanomaterial	Size (nm)	Surface coating	Dosage[a] mg L^{-1}	Bacterial strains	Mode of action[b]	Ref.
CdTe	3.6	Alkanethiols	21.2 mM 11.6 mM 17.4 mM 11.1 mM	*E. coli, S. aureus, P. aeruginosa, B. subtilis*	Toxic, charge transfer	77
CdTe nanowires	40–60	Mercaptosuccinic acid	100 nM	*E. coli*	Oxidative damage	110
Fe_3O_4@TiO_2	>100	-	2.57	*S. pyogenes, S. saprophyticus*	Photokilling	111
FePt	9	-	2.5 per plate	*E. coli, S. typhimurium*	Mutagenicity, DNA damage	112
Iron doped ZnO	0.3–43	-	-	*E. coli, B. subtilis*	Bactericidal	51
Si	50	AAPTS and TES	0.8	*P. aeruginosa*	NO release	66
Si	100 200	AAPTS and TES	1.5	*P. aeruginosa*	NO release	67
TiO_2/Fe_3O_4	22–33	Oleic acid	-	*S. iniae, E. tarda*	Photokilling	113
TiO_2	Varying	-	-	*C. metalliruans, E. coli*	ROS	71
TiO_2	-	-	0.2 mM	*E. coli*	Bacterial inactivation	114
TiO_2	15–20	-	0.05–0.5	Soil bacterial communities	Substrate induced respiration and DNA damage	83
ZnO	20–30	-				
ZnO, CuO	1.9–79		-	*V. fischeri*	Bactericidal	37
ZnO	260 ± 40 6.8 ± 2	Uncoated DMF	0.1–10	*E. coli, S. aureus*	Cell division arrest and oxidative stress	34
ZnO	20–40	2-Mercaptoethanol	4	*E. coli*	Bactericidal	64
ZnO	13	Diethylene glycol	3.4 mM 1 mM	*E. coli* *S. aureus*	Bactericidal	115
ZnO	27–71	PEG and PVP	-	*E. coli*	Non-toxic and increased stability of the particles	62
ZnO rods	250	Hexamethylene tetramine	7.5 mM 1 mM	*E. coli, B. atrophaeus*	ROS mediated membrane damage	75

[a] Concentration is mg L^{-1} or as noted. [b] ROS: reactive oxygen species, ND: not determined; - unknown/commercial.

As previously mentioned; the coating, size, dosage and type of nanomaterial have different effects on different bacterial strains. Some surface coatings such as PEG and PVP are used as stabilizing agents for ZnO nanoparticles and display no known toxic effects on *E. coli* whereas an MES stabilized Ag^+ ion displays inhibition in *E. coli.* PVP was also found to be the most promising stabilizer for medical applications due to its polymer coating, which reduces the release of iron ions through high colloidal stability. (Baumann et al., 2014) Most of the studied nanoparticles however, present possibilities of being bactericidal, mutagenic, cause membrane damage, act as inhibitors and destabilizers. Very few of the studied nanoparticles have no side effects in bacteria.

2.3 Potential Mechanism of Biological Uptake and Toxicity

Klein et al. have described some possible mechanisms of nanomaterial toxicity to bacteria, as shown below in **Figure 1.**

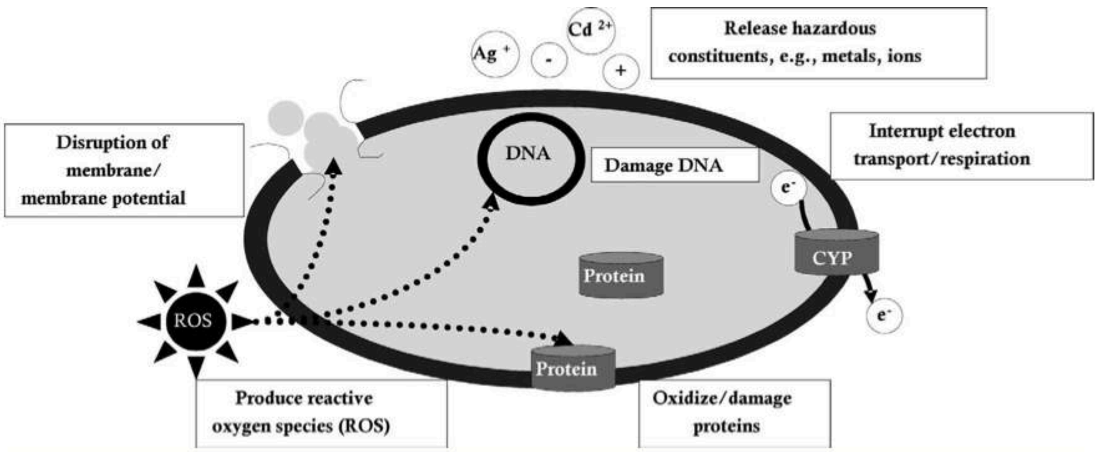

Figure 1. A proposed schematic diagram by Klein et al., of the possible mechanisms of nanomaterial toxicity to bacteria. (Image retrieved from Klein et al., 2008)

Nanoparticles are shown to enter the cell by diffusion through the membranes as well as through endocytosis and adhesion. (Klein et al., 2008) Quantum dots and carbon nanotubes (CNTs) are designed to interact with "proteins, nucleic acids or cell membranes for drug delivery purposes" which makes the unintentional interactions with the environment potentially hazardous. (Klein et al., 2008) As mentioned previously, CNTs are also some of the most mass produced nanoparticles for many applications, which increases the risk of environmental exposure through product use and industry leaks. The diagram depicts possible mechanisms, which include "disruption of membranes or membrane potential, oxidation of proteins, genotoxicity, interruption of energy transduction, formation of reactive oxygen species and release of toxic constituents." (Klein et al., 2008) These effects are all seen in the data provided in **Table 1.** of the known effects of nanomaterials on different bacterial strains.

3.0 Discussion and Analysis

3.1 Pathways of Nanomaterials in the Environment

NMs find their way into the environment intentionally and unintentionally, through consumer products such as medicinal devices, paints, electronics, cosmetics, car catalysts, plastics, ceramics and more. Other pathways could include industrial runoff, spills and waste. Most research is focused on ENMs that have a high potential for industrial spills or waste such as carbon nanotubes and metal oxides due to their mass production and possibly toxic effect on the environment. (Lowry et al., 2010) Some large risks associated with the manufacturing of these NMs are the unintentional leakage that can occur during the transportation of the NP's to secondary or tertiary sites.

Kalavrouziotis et al., 2008) have studied the impact of the platinum group elements (Pt, Pd, Rh) emitted into the atmosphere through automobile catalyst converters. The study suggests that even though catalytic converters minimize the pollution emitted by the car exhaust fumes, the platinum group elements are being emitted in forms of particulate matter and accumulating in soil, plants and air. (Kalavrouziotis et al., 2009) Due to the nature of these emissions, the particulate matter is being transported over long distances and have "increased significantly during the last ten to fifteen years, especially along the road side of high ways". (Kalavrouziotis et al., 2009) Intentional releases of nanomaterials also occur when remediating contaminated soils, and the "use of iron NPs to remediate groundwater". (Klaine et al., 2008) Other pathways of intended release are through consumer goods, such as sunscreen, health care products, fabrics and paints, which enter the environment "proportional to their use". (Klaine et al., 2008) In order to effectively filter these NM's requires "a new class of nanostructured sorbents" which is not widely available due to unregulated emissions.

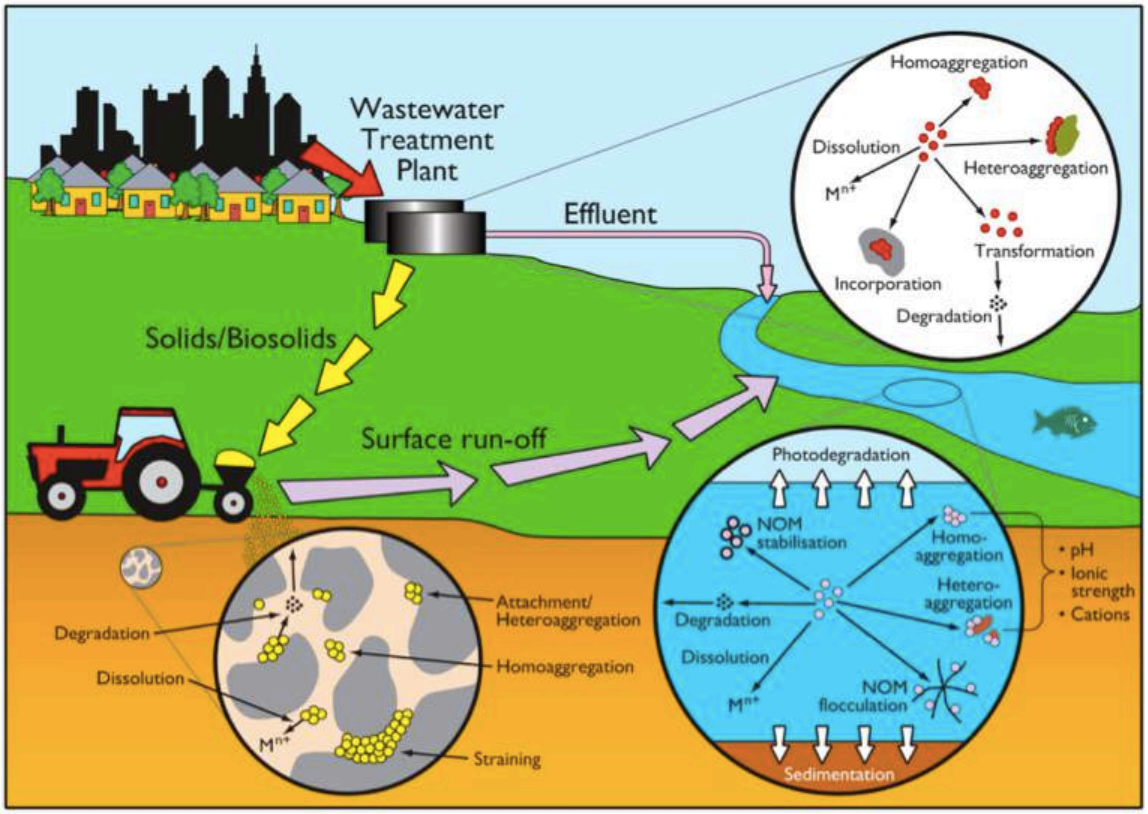

Figure 2. Proposed schematic diagram of pathways of NPs into the environment. (Image retrieved from Batley et al., 2011)

Despite knowledge of the pathways into the environment, trying to analyze the life cycle of the nanoparticles and the exact mechanisms of entry is quite challenging. Most studies have focused on *in situ* experiments, which can be vague and unlikely to show the correct processes. (Nam et al., 2014) However, "researchers have turned to microcosm and mesocosm systems as miniaturized ecosystems to simplify experimental conditions", which helps to understand "the uptake and bioaccumulation" of NPs. (Nam et al., 2014)

3.2 Uptake of Nanoparticles in Aquatic Ecosystems

Since nanoparticles are designed to "persist as particulate matter in aqueous media", they are able to pass through biological membranes due to their size. (Velzeboer et al., 2008) Nanoparticles in aqueous solutions form colloidal suspensions, which can potentially interact with aggregates, other colloidal suspensions and agglomerates. (Velzeboer et al., 2008) This occurs in marine ecosystems because they are generally more alkaline, have higher ionic strength and already have a "wide variety of colloids and natural organic matter". (Klaine et al., 2008) Coastal zones are very likely to have a high concentration of colloids, organic matter and nanoparticles, being closer to discharges or spills from plants or industries than deep ocean water.

In freshwater, the nanoparticle aggregates have a high chance of sinking slowly to the bottom and accumulating in the sediment; this can have a negative effect on the benthic species. In marine ecosystems, it is possible that "nanoparticles will accumulate at the interface between cold and warm currents" which is not likely with freshwater. (Klaine et al., 2008) Another possibility for the mechanism of NPs in marine ecosystems could be a recycling through biota. This can increase the risk of the species, which feed within these cold and warm zones such as tuna. Lastly, Klaine et al, 2008 introduced another possibility, namely accumulation in the "surface microlayers of the oceans" where nanoparticles are trapped due to surface tension and viscous properties. This presents a risk to marine birds and mammals as well as the organisms living in the surface microlayer. (Klaine et al., 2008) However, there is no research that studies the different effects of accumulated nanoparticles in the surface microlayer of oceans. **Figure 3.** represents a schematic diagram of the proposed behavior of nanoparticles in marine ecosystems. It shows the nanoparticles and organic matter discharged from coastal runoff and suggests that the NPs are diluted and transported in the open ocean, and can also sink to the bottom and become aggregates. These aggregates are further transported by microbes or settle in sediment causing toxic effects to benthic species. Another mechanism shows that the free nanoparticles that are transported can have a toxic effect on pelagic species through direct contact, as well as become a risk to seabirds and mammals through aerosol formation. (Klaine et al., 2008)

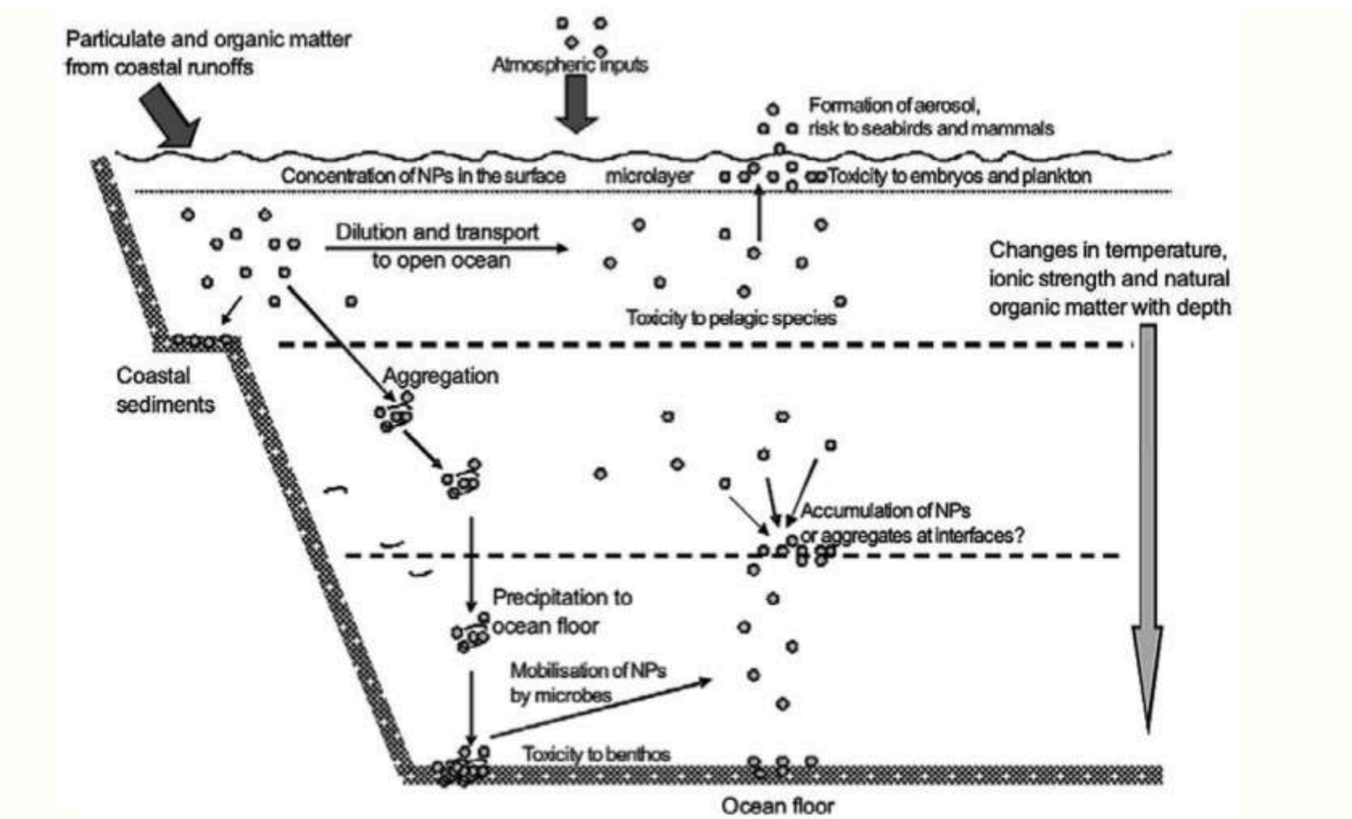

Figure 3. Schematic diagram showing the behavior and effects of NPs on marine environments as well as the organisms at risk of exposure. (Image retrieved from Klaine et al., 2008)

3.3 Bioaccumulation of NMs in Aquatic Ecosystems

Nam et al., have written a comprehensive review, which focused on the bioaccumulation of NPs in aquatic environments. Due to the complexity of nanoparticles and the effects on the environment depending on the size, shape, coating and functionality, they decided to focus specifically on TiO_2 NPs and Ag NPs. A simplified microcosm system was designed to "assess the bioaccumulation of TiO_2 NPs in multiple model species". (Nam et al., 2014) They discovered that a high level of TiO_2 NPs were present in the sediment layer due to settling of NPs as previously mentioned in the proposed schematic by Klaine et al. 2008. However, there was also experimental evidence, which showed a movement of the TiO_2 from the sediment to "water dropwort roots and nematodes living on these plants". (Nam et al., 2014) Nanoparticles were also transferred through different trophic levels as shown by Nam et al., through transfer from "dropwort roots, to nematodes and snails feeding on these roots as well as from biofilm-consuming plankton to ricefish feeding on the plankton". (Nam et al., 2014) This shows that the engineered nanoparticles can travel through feeding patterns of aquatic organisms. There was also direct evidence of trophic transfer of TiO_2 NPs showing NPs "transferred from water fleas to zebrafish, indicating potential biomagnification of TiO_2 via food chain transfer". (Nam et al., 2014) Ag nanoparticles were also studied in order to measure the bioavailability of these nanoparticles to higher trophic organisms. (Nam et al., 2014) Nam et al., show that algal cells concentrate nanoparticles due to adhesion of NPs to the cell wall. They also provide a table summarizing the different uptake endpoints of the Ag NPs in different species. In the earthworm they found that the bioaccumulation depended on the concentration and could be possibly distributed throughout the body. In rainbow trout they propose that the possible adsorption occurs through the gill, which was similar to the Japanese medaka. (Nam et al., 2014) The two tables are presented below of both the bioaccumulation of TiO_2 NPs as well as the Ag NPs.

Table 2. Uptake and bioaccumulation of TiO_2 nanoparticles in aquatic organisms (Table retrieved from Nam et al., 2014)

Species tested		NP size (nm)	Type	NP characterization	Concentration range	Duration	Uptake endpoints
Water dropwort Biofilm Ricefish Mud snail Nematodes Quillworts Algae	*Oenanthe javanica* *Oryzias sinensis* *Cipangopaludina chinensis* *Meloidogyne* sp. *Isoetes japonica* *Spirogyra* spp.	9		TEM	2 mg/L	17 d	Bioaccumulation and trophic transfer
Biofilm, plankton		20	P25	TEM	5.3 mg/L	24 d	Deposition and adsorption to biofilm
nematode	*Caenorhabditis elegans*	50	Anatase	TEM	24-239.6 mg/L	24 h	Possible to adsorption
Earthworm	*Lumbricus terrestris*			XRF	1-100 mg/L	7 d	Unpredictable
Water flea	*Daphnia magna*	< 20 21	Anatase Anatase, Rutile	Microscope SEM	0.5-500 mg/L 0.1-100 mg/L	48 h 72 h	Uptake and distribution in body Bioaccumulation in body
Water flea Zebrafish	*D. magna* *Danio rerio*	21	P25	SEM	0.1-1.0 mg/L	24 h	Trophic transfer and biomagnification
Carp	*Cyprinus carpio*	21	P25	TEM	10 mg/L	25 d	Bioaccumulation in intestine, stomach, and gills
		21	P25	TEM	10 mg/L	25 d	Bioaccumulation in viscera and gill
Rainbow trout	*Oncorhynchus mykiss*	21	P25	TEM	0-1.0 mg/L	0-14 d	Bioaccumulaton in tissues
Zebrafish, trout							Adsorption and translocation in gill and skin
Duckweed	*Lemna minor*	21	P25	TEM	0.01-10 mg/L	72 h	Attachment onto cell wall, but no cellular uptake

TEM: Transmission electron microscopy; SEM: Scanning electron microscopy; XRF: X-ray fluorescence

Table 3. Uptake and bioaccumulation of Ag nanoparticles in aquatic organisms (Table retrieved from Nam et al., 2014)

Species tested		NP size (nm)	Type	NP characterization	Concentration range	Duration	Uptake endpoints
Diatom	*Thalassiosira weissflogii*	10	PVP	TEM, EDS		48 h	Cellular distribution
Aquatic bacterium	*Pseudomonas fluorescens*	30-50		TEM, EDS	2-2000 ppb	24 h	Aggregation by nanoscale film formation
Eastern mud snails, Juvenile hard clams, Grass shrimp, Cordgrass, Biofilms		20-80		ICP-MS		60 d	Bioaccumulation and trophic transfer
Nematode	*Caenorhabditis elegans*	< 100		TEM	up to 0.5 mg/L	24 h	Uptake/adsorption to body
		7-25	PVP, Citrate	TEM	5-50 ppm	24 h	Uptake/transgenerational transfer to body
Earthworm	*Eisenia fetida*	30-50	PVP, OA	TEM, XAS		28 d	Bioaccumulation in a concentration-dependent manner
		10-50	PVP, OA	TEM	< 100 mg/kg	48 h	Unpredictable
			PVP	TEM, SEM	1000 mg/kg	28 d	Possible body distribution
Water flea	*Daphnia magna*	40-50	Carbonated	TEM	up to 5000 μg/L	8 h	Uptake and bioaccumulation
Zebrafish embryos	*Danio rerio*	5-15		TEM	0.71 nM	120 h	Uptake in embryos through chorion pore canals
		11.3		TEM	0.2 nM	21 h	Adsorption to embryos
		20-30		TEM, SEM	10-20 ppt	24 h	Penetrated skin and blood tube as aggregated particles
		20-30		TEM, SEM	0-4 ppm	10 d	Bioaccumulation in muscle and intestine
		20-30	P25	ICP-MS	10 mg/L	48 h	Possible body uptake
Eurasian perch	*Perca fluviatilis*	30-40	PVP	TEM	63-300 ppb	25 h	Possible to adsorb into gill
Rainbow trout	*Oncorhynchus mykiss*		PVP, Citrate	TEM	10-20 mg/L	48 h	Cellular compartmentallization, transport over epithelial layers
Japanese medaka	*Oryzias Latipes*		Citrate	TEM	20 μg/L	7 d	Bioaccumulation in liver and gill
Zucchini	*Cucurbita pepo*	100		ICP-MS	1000 mg/L	12 d	Translocation through shoots
Thale cress	*Arabidopsis thaliana*	20-80		Microscopy			Uptake and accumulation to roots
Common grass	*Lolium multiflorum*	6-25	GA	TEM	0-40 mg/L	24 h	Uptake into roots and shoots

EDS: Energy dispersive X-ray spectroscopy; GA: Gum arabic; ICP-MS: Inductively-coupled plasma mass spectrometry; PVP: Poly vinyl pyrrolidone; OA: Oleic acid

3.4 Impacts on Algae, Fish and Aquatic Organisms

The toxic impact of nanomaterials on fish and aquatic organisms is important to study because most contaminants released in the environment are consumed by aquatic species. Griffitt et al., 2008 have conducted a study in order to assess the toxicity of metallic nanoparticles in aquatic organisms. They used zebrafish, daphnids and *Pseudokirchneriella subcapitata* as models of the varying trophic levels and different feeding strategies. These organisms were exposed to Cu NPs, Al NPs, Co NPs, Ag NPs, Ni NPs and TiO_2 since they are the most commonly engineered nanoparticles with known parent metal toxicity.

The results of the study showed that the nanometals were causing “acute toxicity in multiple aquatic species”, and “filter-feeding invertebrates” having the highest susceptibility to metallic nanoparticle exposure. (Griffit et al., 2008) All the organisms tested were acutely susceptible to Ag NPs and Cu NPs, with the daphnids and algae being more affected by toxicity than zebrafish. (Griffit et al., 2008) Since daphnids are particulate filter feeders, they are more likely to be exposed to larger numbers of nanoparticles compared to larger organisms. (Griffit et al., 2008) These results also concluded that there was no apparent relation between size and surface area, and that the nanoparticles were capable of “causing acute toxicity in multiple aquatic species” regardless of their shape. (Griffit et al., 2008) Ag NPs and Cu NPs were the most toxic in all species yet both the size and shape of these nanoparticles varies greatly, and other nanomaterials with the same sizes have no effect on organisms. Furthermore, silver and copper are the most toxic when presented in a soluble form and the toxicity was confirmed to be partly because of dissolution of particles. (Griffit et al., 2008) These toxic effects however are not only limited to daphnids and MacCormack et al., suggest that the physical dimension of nanoparticles may allow them to “interact with cellular receptors or transport proteins” in aquatic animals. “The respiratory and ion transport surface area can be greater than 60% of the total surface area” of fish posing large health threats when interacting with nanoparticles. (MacCormack et al., 2008) It was found that the interaction of gill ion transport with metal nanoparticles resulted in ionoregulatory failure. (MacCormack et al., 2008) Furthermore, studies have mostly focused on the effects of NPs under “steady-state physiological conditions”, which is not the case in aquatic environments. (MacCormack et al., 2008) Marine ecosystems have changes in pH, salinity, pressure and temperature, which is not taken into account when studying nanomaterial toxicity in animals living under these conditions. (MacCormack et al., 2008) Specifically, cell membrane structure is altered during temperature changes by the “variation of fatty acids and cholesterol” which can be affected by nanomaterials that are specifically engineered to insert "into biological membranes". (MacCormack et al., 2008) Salinity of marine ecosystems also has to be considered, as "fish adapt to changes in salinity" through changes in their gill membrane which also effects organisms "exposure to nanomaterials in the environment". (MacCormack et al., 2008) Lastly, due to limited oxygen exposure in water, "fish exhibit hypoxia" increasing their vulnerability of nanoparticle exposure and toxicity. Hypoxia is exhibited in forms of increased "ventilatory effort" as well as "cardiovascular changes necessary to exploit adjustments". (MacCormack et al., 2008) This increases the risk of toxicity to the respiratory system since many species double their skin capillary surface area in order to improve the efficiency of oxygen uptake, allowing for nanoparticles to enter the system and cause acute toxic effects in aquatic species. (MacCormack et al., 2008) Chronic exposure of nanomaterials to aquatic species is important to study in order to know long-term effects this will have on organisms. Zhu et al., have studied the chronic exposure to sublethal fullerenes aggregates on carp showing that the most susceptible organs were found to be the gills, the brain and the liver. They found that "oxidative stress due to long-term exposure could be the main mechanism for toxicity" for these fish under freshwater conditions. (Zhu et al., 2008) These findings are important in creating regulations that will penalize companies for spills or emissions of nanomaterials in rivers, lakes and oceans.

3.5 Evidence of Engineered NMs in Terrestrial Ecosystems

Natural nanomaterials have long been recognized to exist in terrestrial ecosystems. Soil contains many materials that are less than 2 micrometers, namely "loosely called colloidal soil". (Videa et al., 2011) The components of colloidal soil contain iron oxide nanoparticles as well as humic acids and phyllosilicates. (Videa et al., 2011) However due to the increase in production of nanotechnology, various engineered nanoparticles are entering the terrestrial environment through direct modes, such as "zero-valent metal for remediation" and indirect modes such as spills and emissions. (Videa et al., 2011)

Emissions are caused mainly through catalytic converters that increase the occurrence of Pt, Pd and Rh nanoparticles in the environment. (Kalavrouziotis et al., 2008) The concentrations of the nanoparticles either natural or anthropogenic are unknown in soil or terrestrial ecosystems due to the complexity of separating and identifying the nanomaterials. (Videa et al., 2011) It is assumed that the fate of the nanoparticles when they are released into such environments depends on their specific physical and chemical characteristics. (Videa et al., 2011) In addition, ionic strength, pH and soil texture also affect the impact and transport of nanomaterials, causing multiple behaviors such as “aggregation, transport, sorption, desorption, stabilization and dissolution” into the soil. (Videa et al., 2011)

3.6 Uptake and Bioaccumulation of NMs in Edible Plants

Rico et al., have reviewed relevant literature on the uptake and bioaccumulation of NM’s in edible plants and the impacts this could have on the food chain. As previously discussed in the report, a lot of studies have gone into researching the uptake of NMs and their effects on cells, however little is known about the effects of NMs in edible plants. (Rico et al., 2011) The only carbon-based nanomaterials “shown to readily accumulate in plants” were fullerols and the C_{70} fullerene, whereas most of the metal-based nanomaterials such as Au, Ag, Cu and Fe were readily accumulated, “although some conflicting data exists”. (Rico et al., 2011) Even though many nanomaterials are different depending on their size, shape, coating and core, data suggests that NP’s can enter the plant cells by “binding to carrier proteins, through aquaporins, ion channels, or endocytosis, by creating new pores (in the case of Carbon Nanotubes (CNTs)) or by binding to organic chemicals in the environmental media”. (Rico et al., 2011) Carbon based nanomaterials such as CNTs are used as novel drug delivery systems and ongoing studies are being conducted to find the mechanisms which allow CNTs to penetrate plant cells. Rico et al. (2011) hypothesize that the CNTs interact with proteins and polysaccharides on “the cell walls and elicit hypersensitive responses mimicking plant pathogens due to their small size", which ultimately leads to "cell mortality". () Metal-based (MB) nanoparticles are also studied, and very little is known on the mechanisms and uptake of MB NP's. Due to the fact that the cell wall pore sizes vary from 2-20 nm the larger nanoparticles have a harder time penetrating the cell walls, which means that only the small NP's, which were found to be more reactive, can accumulate in plants. It is also not clear if the NP's remain the same when they are inside the plant walls or if they form aggregates or colloids, which will change the function and behavior. Below is a schematic diagram of the different methods in which nanoparticles could accumulate in edible plants also showing possible mechanisms, however the data for some mechanisms is still inconclusive, as shown in the diagram.

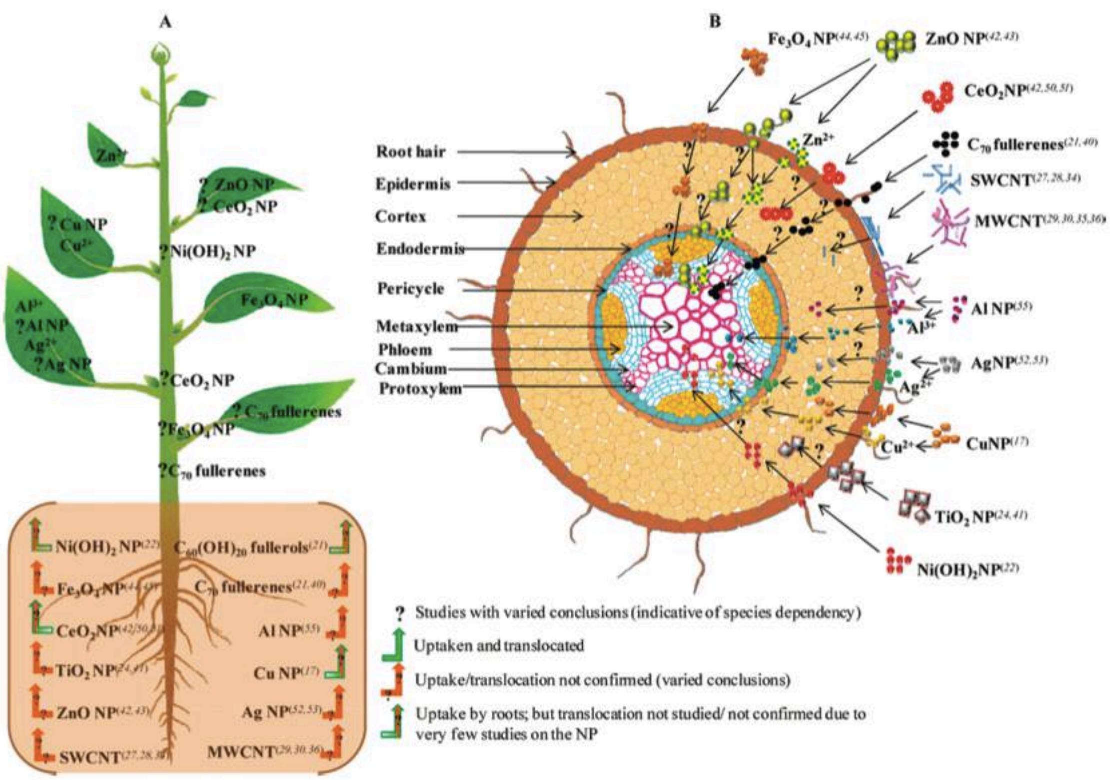

Figure 4. Diagram depicting the uptake, translocation, and biotransformation pathway of various nanoparticles in plants. Part A shows the uptake and the proposed location of the particles whereas part B shows the cross section of the absorption zone in the root, showing different nanoparticle interaction on exposure. (Rico et al., 2011)

Rico et al.(2011) also concluded that the medium in which the plants are grown is important, due to an observed zero intake of NMs in plants grown in soil. Furthermore, there are no studies that show how and where plants store nanoparticles that are accumulated.

3.7 Gene Expression Changes in Plants

The effects corresponding to NM's in edible plants are found to be both positive and negative. Rico et al.(2011) give a comprehensive review of the effects different nanoparticles have on growth and germination of many plants. Van Aken et al. (2015) have also studied the gene expression changes in plants and microorganisms exposed to nanomaterials with mostly negative results. These papers concluded that the TiO_2 nanoparticles on soil bacteria changed the "bacterial community structure", and showed a "reduction of nitrogen fixers and methane oxidizers" through direct toxicity to the soil. They have also concluded that many Ag nanoparticles varying in size from 45 nm to 85 nm show cell death, inhibition of cell growth, association with the cell wall, and antibacterial activity. (Van Aken, 2015) In plants, Kaveh et al., have concluded that Ag nanoparticles increase the growth of *A. Thaliana* at low doses and decrease it at high doses, positively increasing the "pathogen resistance and plant biomass" at low doses. (Van Aken, 2015) The studies suggest that the Ag nanoparticles are affecting the plants partly through the toxicity of the Ag metal and the size of the NP. (Van Aken, 2015) (Mendes et al., 2015) Some photosynthetic pathways have also been affected by NMs in plants, specifically Ma et al., have concluded that high levels of Cerium Oxide negatively "impacted plant growth and chlorophyll production". (Ma et al., 2011) (Van Aken, 2015) Positively, it was found that when *A. thaliana* is exposed to TiO_2 nanoparticles, a light harvesting complex gene is induced and results in increased efficiency of light absorption in the chloroplast. (Van Aken, 2015) Research has also shown that graphene oxide treated with PEG negatively impacts the development of *A. thaliana* seedlings due to the negative effect on genes involved in the development of the roots. Several studies, specifically Lahiani et al., have shown that multi-walled carbon nanotubes (MWCNTs) can have positive impacts on the seeds of barley, soybean and corn. Agglomerates of MWCNTs were found using Raman Spectroscopy and Transmission Electron Microscopy inside the seeds, and activated the expression of water channel genes (aquaporins) producing favorable results in growth and germination. (Lahiani et al., 2013) This study however, was created with the specific intent on increasing yield in these crops and not accounting for nanomaterials found in the environment due to unintentional release. Van Aken has also included that MWCNTs enhance the growth of tobacco over a wide range of concentrations. MWCNTs were however shown to reduce the root length of lettuce and cause cell death, plasma membrane detachment and cell shrinkage in rice. (Rico et al., 2013) Single walled carbon nanotubes (SWCNTs) have also raised concern because they inhibit growth of hair roots in maize plants. (Van Aken, 2015) Rico et al., have shown that Ag nanoparticles show reduced germination and shoot length on ryegrass, flax and barley as well as reduced transpiration and biomass on zucchini. Cu NPs were shown to have reduced seedling growth on mungbean and wheat as well as reduced root growth on zucchini. (Rico et al., 2013) CeO_2 NPs show negative effects on alfalfa, tomato, lettuce, cucumber, maize and soybean through reduced shoot growth, reduced germination and reduced root growth. (Rico et al., 2013) (Doolette et al., 2015) Lastly, it is observed that nanomaterials behave differently in plants than bacteria, affecting more than one transcription factor making it difficult to understand the pathway and mechanisms involved in activation.

3.8 Impacts on Humans Health

Tang et al., have studied the implications that engineered nanoparticles can have on the health of infants and children. They propose that humans have a risk of coming into contact with nanomaterials through consumer products, foods, sunscreens, toys, clothes, medical applications, drug delivery and biomedical imaging. (Tang et al., 2015) Other possible mechanisms would be through bioaccumulation in the food chain through plants and crops, animals and fish as well as through direct contact with the air, water and soil. They propose that through these contact methods, nanoparticles can pose potentially threatening toxic effects for the skin when exposed to metallic nanoparticles such as iron, TiO_2 and quantum dots. (Tang et al., 2015) The nanoparticles were shown to penetrate the skin barrier and cause ROS mediated skin aging, and it can also lead to "systematic exposure and development of lesions". (Tang et al., 2015) A 17-year old patient "developed hepatotoxicity and argyria-like symptoms after treatment with an Ag – containing wound dressing". (Tang et al., 2015) The respiratory system is also very susceptible to airborne nanoparticles, which can be deposited in the alveoli increasing toxic symptoms. Due to the small size of the nanoparticles, they can penetrate the "thin blood-air barrier", and move to other organs increasing the damage. In 2006, 100 German consumers had symptoms including "coughing, sleep disruption, headache and vomiting" after using a bathroom cleaning aerosol product that contained ZnO nanoparticles. (Tang et al., 2015) The illness was later found not to be linked to the nanoparticles however, a 26-year old chemist handling nickel NPs had symptoms of throat irritation, nasal congestion and flushing whereas a 38-year old male died thirteen days after inhaling nickel nanoparticles. (Tang et al., 2015).

The same mechanisms that are shown in aquatic and terrestrial organisms can be expected in humans as well due to the size and shape of specific nanoparticles. Other case studies show adverse health effects to the gastrointestinal tract (GIT) and liver through exposure of engineered nanoparticles after ingesting food and pharmaceuticals. (Tang et al., 2015) Rats exposed to silver nanoparticles in the liver showed a reduced liver weight and "accumulation of granular material". (Tang et al., 2015) The brain, immune and circulation systems as well as the reproductive and developmental systems can also be affected by engineered nanoparticles. Videa et al., also discuss the impacts of engineered nanoparticles in humans and mammals showing that CuO NPs are extremely toxic to lung epithelial cells due to induced DNA and oxidative lesions. CdSe quantum dots with a coating of polyethyleneglycol (PEG) lost their coating when tested in the intestinal cells due to the lowered pH, which increased their nanotoxicity. (Videa et al., 2011) The mechanisms for exposure are the same as previously seen, where the nanoparticle size and shape allows it to enter the cells of the organs and cause toxic effects. High instances of "cardiovascular disease" were seen in workers that are handling ENPs when compared to a control group. (Tang et al., 2015) Lastly, the reproductive and developmental systems are affected "causing early miscarriages and fetal malformations" in pregnant mice after "10 days of SWCNTs injection". (Tang et al., 2015) These observed health effects on humans have large implications for future studies, showing that engineered nanomaterials should be carefully assessed before being made widely available.

4.0 Limitations

There are many limitations when trying to assess the impact that nanomaterials have on the environment. Due to the many differently engineered nanoparticles and their different sizes and properties it is very difficult to make assumptions about all nanomaterials based on the findings of one. Some nanoparticles have shown that they are extremely toxic to bacteria, yet other studies show that nanoparticles can help in the growth of edible plants. (Lahiani et al., 2013) Other limitations include individual species and their interaction with the environment as well as their ability to store or expel small nanoparticles. Furthermore future studies should focus on the long-term impacts on the environment rather than acute toxicity tests on simple species. There are also limitations in understanding how the behavior of the nanoparticles differs in the environment, and creating replicable scientific studies on the mechanisms and the bioaccumulation of nanoparticles on both aquatic and terrestrial species. (Yao et al.,

2013) Very little is also discussed on the biomagnification of these nanoparticles within aquatic and terrestrial organisms and how this will affect food consumption. Moreover, fundamental studies between the structure and function relationships of nanoparticles are lacking which makes it more difficult to assess toxicity. (Rickerby et al., 2007) This study is also very wide and tries to provide a holistic view of the environmental impact of nanomaterials in many species, however in order to have a better understanding each of the aspects have to be extensively researched.

5.0 Conclusion

The increase in nanotechnology poses large toxicological implications because of the release of nanomaterials into the environment affecting bacteria, edible plants, fish and mammals. The size, shape, coating and functionality of nanoparticles are crucial in understanding their specific effects on aquatic and terrestrial ecosystems. Firstly, the small size of nanoparticles increases their surface area and their ability to enter cells causing cell mortality and increased reactivity in microbes. (Suresh et al., 2013) The coating and production process have large effects on the NP characteristics, increasing toxicity and environmental impact. NMs have been shown to enter cells through cell diffusion as well as through endocytosis and adhesion. (Klein et al., 2008) This poses a threat due to the mass production of these materials making them more widely available in the atmosphere increasing the risk for plants, animals and humans to come into contact with them. Moreover, nanoparticles are shown to settle in the sediment of waterbeds as well as soil, causing small organisms living in these systems to have high nanoparticle accumulation. Lastly, human health case studies have shown that humans can be affected through inhaling, eating and coming into contact with nanoparticles affecting all the organs due to their small size and ability to enter the circulatory system. (Tang et al., 2015) However, many of the studies available are only focused on the short-term acute effects of nanoparticles in the environment, yet long-term studies are also crucial due to the fact that health impacts might not be relevant until decades later. (Tang et al., 2015) In order to study this, biomarkers can be used to examine nanoparticles in the environment and not only within the human body. (Tang et al., 2015) Finally, regulation needs to be discussed for companies that are producing nanomaterials and new technology in order to manage risk before seeing adverse health and environmental effects. Without governmental control nanomaterials are likely to accumulate in soil and water through extensive use and production of new technology, spills, runoff and emissions.

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Costs and Benefits of Nitrogen and Phosphate Fertilizer Use In the Lake Erie Basin

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CENTER for DEVELOPMENT and STRATEGY

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In the Lake Erie Basin

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Abstract

This paper explores both the positive and negative externalities associated with nitrogen and phosphate-based fertilizer use. Using 57 scholarly journal articles, government reports, manuscripts, and news articles; a comprehensive review was made on the effects fertilizer use and eutrophication has on ecological, environmental, human health, and economic systems in the western Lake Erie Basin. Negative externalities associated with fertilizer use included species population decline; environmental degradation; increased risks on public health; increased water treatment and maintenance spending; decreased tourism and recreation spending; decreased real estate value; and decreased aquaculture yields. Positive externalities associated with fertilizer use included increased crop yields; decreased food prices; and increased food security. The paper qualitatively examines total costs and benefits accrued on these systems, while making recommendations for further study and investigation in a quantitative manner.

Keywords: fertilizers, eutrophication, Lake Erie, externalities

1.0 Introduction

Over the course of the State of Ohio's history, entrepreneurs and farmers have developed a series of new techniques and tools in order to increase agricultural yields and overall food production. By building prosperous communities backed by the hard working ideals of resource extraction and industrial production, Ohio quickly became a haven for Americans and immigrants looking for opportunities westward. As agricultural production increased though, unintended consequences as a result of those new innovations became apparent.

Over the last few decades, algae growth in western Lake Erie has become a constant concern for various stakeholders invested in the environment, society, and economy. This paper will go through an analysis on the history, technology, and processes by which chemical fertilizers induce increased algal growth and therefore affect ecological, human health, and economic systems across western Lake Erie. In addition, the paper will review the benefits of using chemical fertilizers by farmers in Ohio's Lake Erie basin. By compiling research on the costs and benefits of chemical fertilizer use in the western Lake Erie basin, stakeholders will have access to a comprehensive review of externalities associated with agricultural production and eutrophication. In this paper, it was expected that costs incurred on species, the environment, human-health, and the economy are, on an aggregate basis, greater than the benefits gained out of using chemical fertilizers on Ohioan cropland.

The primary limit of this paper's approach and analysis is the qualitative nature of aggregating effects across the environmental, social, and economic spectrums. Additionally, surveys and other data capturing techniques were not used to produce the analysis. Instead, a literature review and compilation of 57 scholarly journal articles, government reports, manuscripts, and news articles were used to produce an overview of the costs and benefits associated with chemical fertilizer use. Recommendations for areas of future investigation were also made with respect to the surveyed research on this subject.

2.0 History

Shortly after the American Revolutionary War, Americans looking to expand their territory soon populated the area of what is today known as Ohio. As the 17th state in the union, Ohio, like many other Midwestern states, focused its economy on the production of raw materials and common-pool resources, which were both subtractable and non-excludable. These types of goods made it easy for Ohioans to establish local economies that were self-sustaining and efficient. A combination of abundant land, rich soil, and ample water resources made it possible for Ohio to become a hotbed of growth for the early American agricultural industry.

Farming provided the opportunity for Ohio's economy to develop rapidly over the course of the early-mid 19th century. By 1849, Ohio was the largest producer of corn and the second largest producer of wheat in the United States (Knepper, 2003). This also helped contribute towards the increase of Ohio's population, which increased from 42,159 in 1800 to 2,339,511 by 1860 (U.S. Department of Commerce, Census Bureau, 1970).

The diversification of Ohio's economy eventually began to occur after the end of the civil war. Early Ohioan factories were born out of the very agricultural industry that established the state's economy. Industrialized goods were produced in order to complement the existing agricultural economy. For example, iron-manufacturing plants along the shores of Lake Erie were able to create the steel needed for new farming equipment. Advances in agricultural technology also helped increase total crop yields, which helped incur greater profits for farmers. In addition to farmers, investors looking to expand their opportunities were provided with a wealth of both natural resources and human capital, which made newly industrializing cities, such as Toledo and Cleveland, attractive locations to settle in. The economic ecosystem of the region became a flourishing environment for a diverse set of players that had been backed by decades of a developing agricultural industry.

As Ohio became increasingly industrialized, however, the state relied on its agricultural industry less. While specific periods in the 20th century spurred growth for the industry, such as World War I and World War II, agriculture in Ohio had been declining. Despite this decline, farming in the state has remained an essential segment of Ohio's economy during the 20th and 21st centuries. Ohio's agricultural industries today represent over $90 billion USD of the state's total economic output and employ one in seven Ohioans either directly or indirectly (Myers, 2005).

In addition to being a core component of Ohio's economy, the state's geography is also significantly shaped by the agricultural industry. In 1997, approximately 13.6 million acres (52%) of Ohio's 26.4 million acres were agricultural, 7.1 million acres (27%) were forested, and 3.6 million acres (14%) were developed or urban areas (Ohio Legislative Service Commission, 1997). Today, approximately 30% of Lake Erie's surrounding land is cropland, which is significantly greater than any other Great Lake (Ohio Legislative Service Commission, 1997).

2.1 Technology & Innovation

America in the mid-1800s was a time of technological innovation and re-development for the agricultural industry. New farming equipment, such as the steel plough, were a major contributor to the economic success of the Midwest. However, it wasn't until the "Green Revolution", that occurred between 1930 and 1970, when agriculture started to become a truly technologically driven industry. During this period, farming technologies that had already existed in many industrialized countries were spread to developing countries, such as Mexico, India, Brazil, and the Philippines.

The one development that has possibly revolutionized the production of food and shifted its supply more than any other has been the continuous improvement of agricultural fertilizers and pesticides. While prioritization for the management of soil fertility had been in place for thousands of years before, the modern science of plant nutrition didn't develop until the 19th century. Malthusian theories of exponential global population growth concurrent with linear food production growth drove scientists to investigate the mechanics of agricultural production greatly, which included comprehensive study within the botanic sciences. Prominent scientists, such as the Dutch chemist, Justus von Liebig, were now performing research and development on an industry that had experienced little advancement since the Middle Ages.

Throughout the modern historic use of fertilizers and pesticides, there have been a series of discoveries that have made them more efficient to produce, less costly, and more effective. This has caused the use of agricultural supplements to increase greatly over time. Today they're commonly used throughout the majority of North American farms. According to the United States Department of Agriculture (USDA), 78% of corn acreage in the United States received phosphate fertilizer in 2010, compared to 90% in Ohio (National Agricultural Statistics Service, 2010). In the same survey, it was found that 97% of corn acreage in the United States received nitrogen fertilizer in 2010, compared to 100% in Ohio. The widespread use of new farming technologies in Ohio, such as chemical fertilizers, has a deep-rooted history and tradition that dates back over centuries.

3.0 Soils & Eutrophication

Northern Ohio's soil was formed largely by glaciers and weathering of sedimentary rock (Ohio Department of Natural Resources, 2007). Western Ohio soil also exhibits greater levels of lime, which increases soil pH. This allows soils in western Ohio to generally be more productive and fertile for crops, due to increasing natural acidity to soils over time. Soils along Ohio's northwestern corridor were formed in lake and beach sediments and in glacial till associated with glacial lakes. Because of this, soil horizons in the region typically exhibit high levels of silt and sand within the topsoil. Further west, agricultural land in northern Ohio is characterized by near-level crop fields that contain both drainage ditches and subsurface drains. To the east, soils contain greater clay materials, which provide both coarser textured and steeper soil horizons.

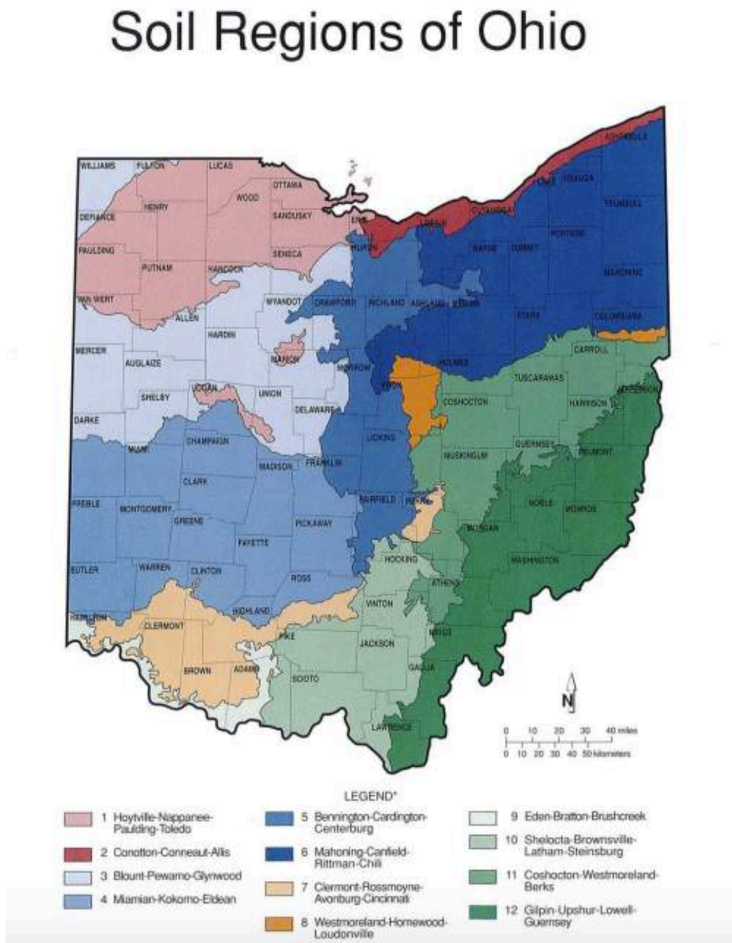

Figure 1 Soil Regions of Ohio (Ohio Department of Natural Resources, 2007)

Due to northern Ohio's high level of sand and silt in its soil, the region is highly prone to experiencing drainage issues and eutrophication. The Natural Resources Conservation Service (NCRS) of the USDA defines eutrophication as, "(1) the degradation of water quality due to enrichment by nutrients, primarily Nitrogen (N) and Phosphorus (P), which results in excessive plant (principally algae) growth and decay. When levels of N:P are about 7:1, algae will thrive. Low Dissolved Oxygen (DO) in the water is a common consequence. (2) The process of enrichment of water bodies by nutrients." (Natural Resources Conservation Service, 2011).

While the use of chemical fertilizers can provide immediate adequate nutrition to crops, they are also highly susceptible to leaching through sand and silt based soils, particularly those in northwestern Ohio. Furthermore, 63% of soils further west experience seasonally high water tables that are less than a foot below the surface (Ohio Department of Natural Resources, 2007). A high water table provides easy transportation for leached chemical fertilizers to flow through these water channels, and eventually into western Lake Erie.

Added chemical fertilizers from crop fields that are leached into local water systems eventually end up in the Lake Erie Basin, which consists of watersheds surrounding the lake. While 80% of Lake Erie's water is captured via the Detroit River, 9% is derived from these watersheds (New York State Department of Environmental Conservation, 2005).

4.0 Algal Blooms

Through rain and irrigation-caused leaching, agricultural fertilizers containing phosphorus and nitrogen provide water systems with an influx of nutrients. These nutrients help spur the rapid growth and multiplication of aquatic vegetation and algae. Western Lake Erie, in particular, experiences the rapid growth of cyanobacteria, which produce harmful toxins known as microcystins.

Microcystins are a class of toxins that are commonly produced by certain freshwater cyanobacteria. With over 60 microcystin toxins known, they pose major threats to ecosystems as well as drinking and irrigation water supplies (Ramsy et al., 2013). Previous research has validated the positive correlation between phosphorus loading and microcystin concentrations in western Lake Erie (Rinta-Kanto, 2009).

4.1 Microcystin Structures

As cyclic peptides, microcystin structures consist of a seven-membered peptide ring that is made up of five non-protein amino acids and two protein amino acids (Schneegurt, 2000). When cyanobacterial cells die, their cell walls burst, releasing the toxins into the water.

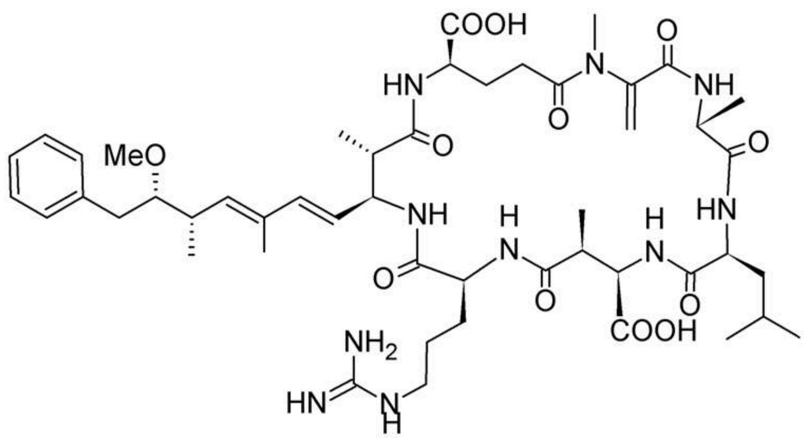

Figure 2 Microcystin Chemical Structure

4.2 Microcystin Remediation

Microcystin structures are exceedingly resistant to chemical breakdown, such as hydrolysis or oxidation. In addition, microcystin toxins are nonvolatile, hydrophilic, resistant to photodegradation, and are stable over a wide temperature and pH range. The structures are therefore extremely stable under most natural conditions. This makes remediation efforts to rid waters from contaminated microcystin toxins difficult and costly. Furthermore, due to the delicate nature of deceased algal cells that are prone to rupturing and contain microcystin, cleanups are usually extremely time intensive (International Organization for Standardization, 2005). The efficacy of drinking water filters to remove microcystin toxins varies significantly by filter type. A study in 2006 found that carbon filters allowed only 0.05-0.3% of the toxin load to pass through, while pleated paper and string based filters allowed more than 90% of the toxin load through (Pawlowicz et al., 2006).

Microcystin-producing cyanobacteria thrive in waters with warmer climate conditions. Because of this, harmful algal blooms (HABs) in western Lake Erie occur more commonly during hot summer months. With anthropogenic climate change, HAB occurrences are expected to increase along with rising temperatures in the northwestern Ohio region (Michalak et al., 2013).

5.0 Overview

With a combination of increased chemical fertilizer use and anthropogenic climate change, eutrophication will have an increasing impact on western Lake Erie's ecosystems, environment, and economies over the next few decades. In particular, algal blooms that produce microcystin toxins, as well as hypoxic water, degrade both ecosystem health and public health. These effects raise significant concerns for key stakeholders including citizens, policymakers, public health officials, and environmental advocates. The following sections of this paper will review those effects by analyzing the role HABs have on individual species; aquatic and terrestrial ecosystems; public health; as well as the economy and environment at-large. In addition to the effects eutrophication has on species, the environment, human health, and the economy; this paper will provide an overview of the benefits gained by using chemical fertilizers.

6.0 Ecological Effects

Large quantities of toxins produced by HABs affect the ecology of both marine and fresh water biomes. For example, 34 phytoplankton species are known causative agents in fish and shellfish mortality events along the U.S. west coast (Lewitus et al., 2012). This section will discuss the mechanisms by which organisms are directly and indirectly exposed to HAB toxins, as well as the impacts they incur as a result of increased algal growth.

6.1 Direct Exposure

Organisms are affected by HABs by way of either direct or indirect exposure. Direct exposure to microalgal cells and their toxins occurs through drinking or ingesting them through various consumption modes, such as filter feeding or predation. Smaller organisms such as zooplankton and shellfish often retain these toxins within their body cavities, and can present issues for bioaccumulation along the freshwater food chain. In calm, summer months, cyanobacteria forms a thick layer of surface scum that is dispersed over a significant portion of western Lake Erie. When wind and wave action is increased, however, this surface scum often concentrates closer towards shorelines. Wildlife and domestic animals that obtain their drinking water supplies from lake shorelines are therefore often directly exposed to large quantities of both algal cells and their toxins. Principal routes by which representative groups of organisms are directly exposed to harmful microalgal microcystin toxins includes the ingestion of cells by zooplankton, molluses, fish, birds, and terrestrial mammals (Landsberg, 2002).

In addition to obtaining direct exposure to microalgal microcystins through the ingestion of cells, organisms may also come into contact with extracellular microcystin toxins. These toxins have often been released from their cell membranes by way of either force or decomposition. If concentration levels are low with respect to water volume, then direct exposure to extracellular microcystin is unlikely. However, organisms that feed in the lake during the expiration of algal blooms are highly susceptible to coming into direct contact with extracellular microcystin. Furthermore, due to microcystin's highly stable amino acid structure, the toxins can persist in lake waters for months before being broken down by natural processes (Landsberg, 2002).

6.2 Indirect Exposure

Aside from direct exposure with HABs, organisms are also susceptible to indirect exposure with algae. When organisms consume other organisms that have previously been directly exposed to microcystin and other toxins, they transfer trophically through the food chain. Worse, toxicity concentration rates increase with each higher trophic level, due to bioaccumulation, bioconversion, and/or biomagnification. One of the most common cyanobacterium in western Lake Erie, *Microcystis aeruginosa*, is known to produce microcystin toxins that readily move through the food chain in this manner (Kotak et al., 1996).

6.3 Impacts from Toxins

Exposure of HAB toxins can lead to mass mortalities of aquatic organisms. Exposure to such toxins usually results in an immediate physiological, pathological, or behavioral change, depending on the species and concentration.

The following are examples of species that are affected by toxic cyanobacteria in freshwater lakes and reservoirs:

Molluscs:

- *Anabaena circinali*
    - Reduced overall feeding for *Alathyria condola*, species of mussel (Negri & Jones, 1995)

Zooplankton:

- *Anabaena affinis*
    - Reduced overall feeding for *Ceriodaphnia dubia*, species of water flea (Kirk & Gilbert, 1992)
    - Reduced fecundity for *Daphnia galeata*, species of planktonic crustacean (Gilbert, 1990)
    - Reduced fecundity for *Daphnia magna*, species of water flea (Gilbert, 1990)
    - Reduced fecundity and mortality for *Daphnia pulex*, species of water flea (Gilbert, 1990)
- *Anabaenan flos-aquae*
    - Feeding inhibition for *Daphnia hyalina*, species of planktonic crustacean (De Mott, et al., 1991)

  - Reduced feeding for *Daphnia parvula*, species of planktonic crustacaen (Fulton, 1988)
  - Reduced feeding for *Daphnia pulex*, species of planktonic crustacaen (Fulton, 1988)
  - Feeding inhibition for *Daphnia pulicaria*, species of planktonic crustacean (De Mott, et al., 1991)
  - Feeding avoidance for *Diaptomus reighardi*, species of copopod (Fulton, 1988)
  - Feeding avoidance for *Eurytemora affinis*, species of copopod (Fulton, 1988)
- *Anabaena minutissima* var. *attenuate*
  - Reduced feeding, survival, and inhibition of appendage beat rate for *Daphnia carinata*, species of planktonic crustacean (Peter & Lampert, 1989), (Forsyth et al., 1992)
- *Aphanizomenon flos-acquae*
  - Reduced feeding and fecundicity for *Acartia bifilosa*, species of copepod (Sellner et al., 1994)
  - Inhibition of appendage beat rate for *Daphnia carinata*, species of planktonic crustacean (Haney et al., 1995)
  - Feeding avoidance for *Diaptomus reighardi* (Fulton, 1988)
  - Reduced feeding, increased avoidance, and reduced fecundity (Fulton, 1988), (Sellner et al., 1994)
- *Microcystis aeruginosa*

- Reduced feeding and fecundity for *Acartia bifilosa* (Sellner et al., 1994)
- Reduced feeding for *Bosmina longirostris*, species of water flea (Fulton & Pearl, 1987), (Fulton & Paerl, 1989)
- Reduced feeding for *Ceriodaphnia quadrangula*, species of planktonic crustacean (Fulton & Paerl, 1989)
- Reduced feeding and mortality for *Daphnia ambigua*, species of planktonic crustacean (Fulton & Paerl, 1989)
- Feeding inhibition for *Daphnia hyalina* (De Mott, et al., 1991)
- Reduced growth and depressed clutch size for *Daphnia longispina*, species of planktonic crustacean (Stangenberg, 1968), (Reinikainan et al., 1994), (Hietala et al., 1995)
- Feeding avoidance for *Daphnia magna* (Yasuno & Sugaya, 1991)
- Mortality for *Daphnia parvula* (Fulton, 1988)
- Reduced growth, depressed reproduction rate, and clutch size for *Daphnia pulex* (De Mott, et al., 1991), (Reinikainan et al., 1994) , (Hietala et al., 1995)
- Feeding inhibition for *Daphnia pulicaria* (Lampert, 1981), (De Mott, et al., 1991)
- Reduced feeding of *Diaptomus reighardi* (Fulton & Paerl, 1989)
- Mortality for *Eucypris virens*, species of planktonic crustacean (Stangenberg, 1968)

    - Feeding avoidance and mortality for *Moina macrocopa*, species of water flea (Yasuno & Sugaya, 1991)
    - Mortality for *Moina micrura*, species of water flea (Fulton, 1988)
    - Reduced feeding for *Simocephalus serratulus*, species of crustacean (Fulton & Paerl, 1989)
- *Planktothrix agardhii*
    - Reduced growth and fecundity for *Daphnia pulicaria* (Infante & Abella, 1985)
    - Reduced growth and fecundity for *Daphnia thorata*, species of planktonic crustacean (Infante & Abella, 1985)

In addition to the aforementioned species harmed by algal toxins, fish, reptiles, and mammals such as birds, are often affected as well. However, animals that drink contaminated freshwater are by far the largest terrestrial group affected by HABs.

Overall effects of HABs on food webs and ecosystems are often difficult to study. This is particularly due to the complexity of such food chain systems. The set of long-term implications of released algal toxins currently requires further investigation.

6.4 Impacts from Hypoxia

In addition to HAB toxins, algal blooms also produce hypoxic (low-oxygen) water over time. This is due to bacterial decomposition of algae that has grown from added nutrients (phosphorus and nitrogen from chemical fertilizers). Due to bacterial respiration during this process, temperature differences between poorly oxygenated water and oxygenated-water helps stratify water columns to prevent mixing from occurring.

Hypoxic water zones typically do not kill fish populations by way of suffocation (Almeida, 2015). Instead, by decreasing the amount and quality of habitat available, fish become physically constrained to habitable zones that do provide adequate oxygen, light, and temperature levels. However, with other species inhabiting these environments already, fierce competition between species often occurs.

During mid-late summer, water stratification becomes more intense, which prevents fish from occupying the cooler, poorly oxygenated bottom waters, where benthic prey are abundant.

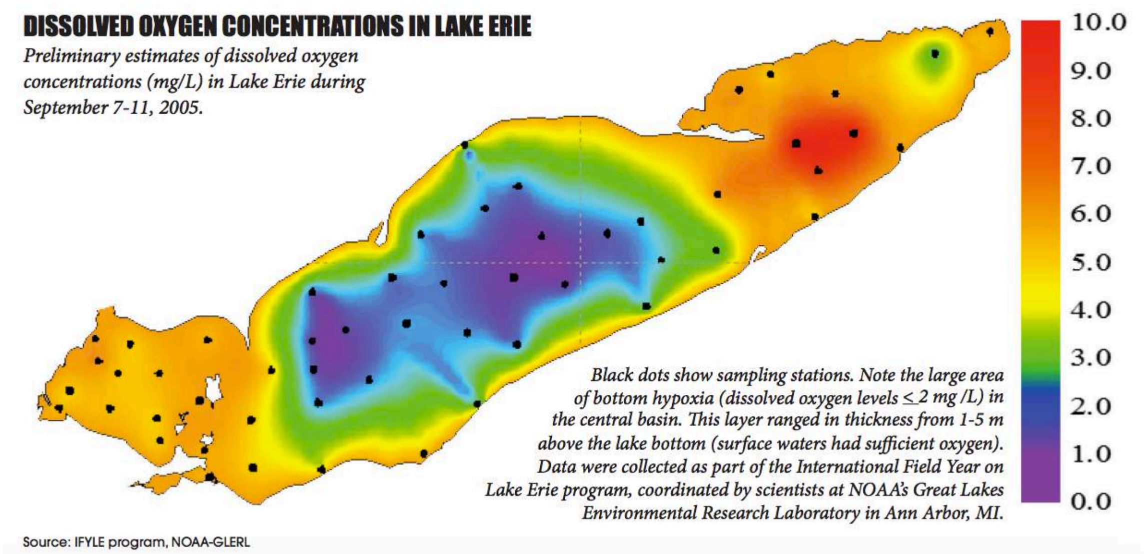

Figure 3 (Hawley et al., 2006)

Due to overall warmer temperatures year round, Lake Erie becomes more productive, and thereby allows the increase of amount of prey available and habitat suitability early in the year. However, these benefits don't cancel out the negative effects that occur during late summer.

Fish species that are most affected by hypoxic water zones in Lake Erie include the yellow perch (*Perca flavescens*), rainbow smelt (*Osmerus mordax*), emerald shiner (*Notropis atherinoides*), and the round goby (*Neogobius melanostomus*). These fish primarily feed on zooplankton and benthic organisms. Due to the decreased habitat availability during late summer months, zooplankton becomes the primary source of food for these fish during this time. However, the availability of zooplankton in warmer, upper waters becomes constrained over time. Furthermore, algal toxins directly affect many species of zooplankton, as cited previously. With decreased overall availability of zooplankton prey, significant populations of fish along the food chain become malnourished. These effects strongly increase the risk for mass death among aquatic species.

7.0 Human Health Effects

Huynh et al., 1998 notes that at the biochemical level, microalgal toxins are genotoxic and can affect DNA adducts (Huynh et al., 1998). At the cellular level, though, Huynh et al. also notes that toxins can be cytolytic, hemolytic, antieoplastic, or tumor inducing. At the organ level, toxins can be neurotoxic, dermatoxic, or hepatoxic. Neurotoxic paralytic shellfish toxins (PSTs) are also produced by the following cyanobacteria: *Anabaena circinalis*, *Aphanizomenon flos-aquae*, *Cylindrospermopsis raciborskii*, and *Lyngbya wollei*. PSTs present significant concerns for public health officials as well as drinking water treatment specialists. As extremely potent neurotoxins, PSTs are highly lethal. Symptoms of paralytic shellfish poisoning (PSP) include paresthesia and numbness, particularly around the face and neck.

Effects of microcystin on organisms can also be lethal when administered in high doses. In humans, symptoms of microcystin poisoning include diarrhea, vomiting, piloerection, weakness, and pallor (Bell & Codd, 1994). Microcystin toxins specifically target the liver and can cause significant cytoskeletal damage, necrosis, and pooling of blood, which can increase liver weight by up to 100% (Hooser et al., 1989). Microcystin disrupts the liver's cytoskeleton, which leads to loss of cell morphology, loss of adhesion from cell-to-cell, and cellular necrosis. At especially toxic doses, microcystin causes disorganization of tissue, which leads to massive hepatic hemorrhage, which can be lethal within a few hours after administering the dosage (Hoosner et al., 1989). In addition to liver damage, microcystins also promote tumor growth in humans by inhibiting the protein enzyme's phosphatase type 1 and 2A activities (Eriksson et al., 1990).

In February of 1996, a hemodialysis center in Caruaru, Brazil exposed 116 patients to acute levels of microcystin toxins via water supplied from a nearby-contaminated reservoir by cyanobacteria. Of the 100 patients who were affected, 52 died (Jochimsen et al., 1998). Although there have been reports of human deaths caused by inadvertent injection of microcystins, there have been no reports caused by the direct ingestion of such toxins (Butler et al., 2009).

7.1 Drinking Water Quality

Aside from the direct health risks cyanobacteria pose, toxins can also leave communities vulnerable to a variety of indirect consequences. One of the foremost concerns public officials have on HABs is in the area of drinking water quality and treatment. In August of 2014, approximately 500,000 residents of the Greater Toledo, Ohio region were advised not to drink water via tap and were recommended to use bottled water for showering, bathing, brushing teeth, and washing dishes. In addition to emergency warnings on drinking water supplies, microcystin toxins forced local restaurants, universities, and public libraries to close. The National Guard was also deployed in order to deliver cases of bottled water from Akron, Ohio, to residents who were especially vulnerable.

The EPA currently does not have standards and regulatory limits associated with cyanobacterial toxin concentrations provided for public water systems. However, many states have implemented guidelines that apply to cyanotoxins and cyanobacteria in drinking water. Many of these states utilize standards set forth by the World Health Organization (WHO) of the United Nations, which has a standard of 1 μg/L for microcystin-LR (Rao et al., 2002).

8.0 Economic Effects

HABs present biological systems with an array of challenges. Often, these stressors not only affect wild species, but also human society as well. Estimating the economic impacts HABs create is important for policymakers and environmental advocates to determine the level of effort that mitigation and remediation treatments should yield. However, it should be noted that estimating the social and environmental costs associated with HABs is difficult and values should be exercised with caution. Nevertheless, the provision of a comprehensive economic analysis can be highly valuable for agriculturally driven communities that are in the process of weighing the costs and benefits of using chemical fertilizers that contain phosphates and/or nitrogen.

8.1 Water Systems

In 2013, the city of Toledo, Ohio allocated $4 million USD for water treatment chemicals, which was doubled from what it spent in 2010 (Henry & Writer, 2015). In 2014, blooms around Maumee bay and areas of western Lake Erie were especially concentrated and thick, and prompted the city to deliver its drinking water emergency advisory. This resulted in total spending to increase to $4.7 million USD for water treatment chemicals (Henry, 2014). Monitoring tasks by municipal, state, and federal governments include the testing, treatment, and management of infrastructure and facilities that are designed to sustain large populations with a consistent supply of water. Monitoring programs that are designed to look for PSTs in particular, had an average annual monitoring and management cost that totaled $2.89 million USD in the United States in 2000 (adjusted for 2015) (Hoagland et al., 2002). However this figure only factored in 12 total states and did not account for HAB occurrences in any of the Great Lakes.

8.2 Tourism and Recreation

In addition to water treatment and monitoring costs, a number of industries along Ohio's Lake Erie coastline are directly affected by HABs. In 2013, *Tourism Economics* calculated that total sales from tourism were valued at $12.9 billion USD across the Ohio coastline (Winslow, 2015) [48]. The industry helps employ over 119,000 people and generates over $1.7 billion total in tax revenue for federal, state, and local governments. Increasing HAB occurrences primarily threaten the tourism industry by limiting land area recreational usage. However, the tourism industry in particular, may be the most significant industry affected by HABs. It was estimated in 1975 that economic damage to the tourist industry of a summer 1971 *Gymnodinium breve* red tide event amounted to over $93 million (Habas & Gilbert, 1975) (adjusted for 2015). Future research should estimate similar impacts to the tourism industry along Ohio's coastline.

8.3 Real Estate

HABs also can affect property values and the real estate industry along Ohio's shorelines. This is due to both decreased recreational and aesthetic values associated with algal bloom sights and smells. While several news reports reference property value losses as a result of algal blooms along western Lake Erie's shores, there is currently limited scholarly research in this area.

8.4 Fisheries

The commercial fishing industry is the most directly affected industry by HABs in western Lake Erie. The industry is especially important both socially and economically for both Ohio Buckeyes and Ontarians. Overall, the industry is worth approximately $50 million CDN and employs around 715 people (Hill, 2015). The fish processing industry in Lake Erie has an economic impact of approximately $194 million CDN and employs around 755 people. In total, the industry amasses over $244 million CDN and employs over 1,490 people either directly or indirectly.

Several news reports have suggested decreased earnings during algal blooms in western Lake Erie, although scholarly surveys are currently limited and should therefore be investigated. HAB events harbor the potential to shift the supply of fish inwards, which simultaneously increases cost and decreases quantity demanded. Both producer and consumer surpluses are therefore shrunk and total economic losses are incurred on society.

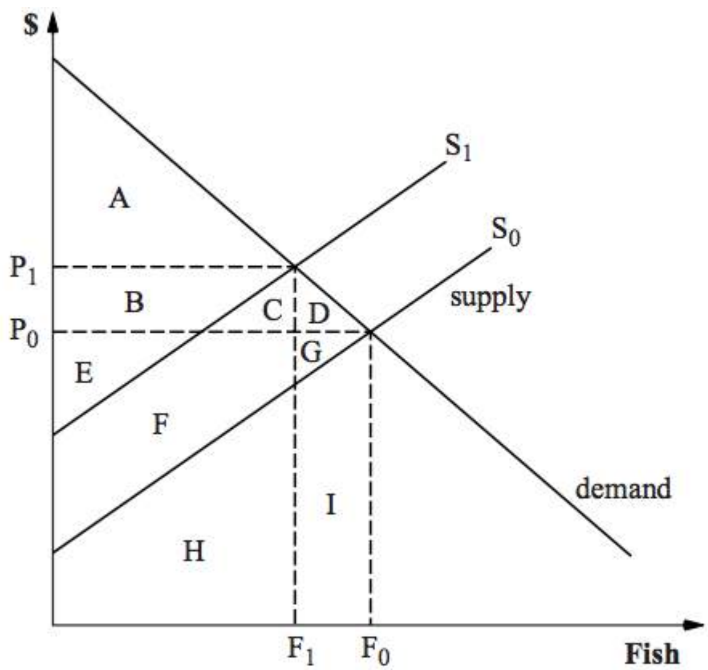

Figure 4 Economic Effects of Decreased Supply (Anderson et al., 2000)

8.5 Public Health

Lastly, human sickness and death from exposure to cyanotoxins in Lake Erie has the potential to significantly increase both the economic and social cost of HAB events. Costs of widespread medical treatment due to an algal outbreak in drinking water could pose significant economic concerns for communities. While relatively few cases have thus far been reported, the region's population of over one million residents raises the risk of such an outbreak to occur in the future. With expected increasing temperatures in the region due to anthropogenic climate change, these risks may be further elevated.

Overall, HABs pose a variety of costs and risks related to the health of Lake Erie's economy. Furthermore, due to the frequent recurrence of such events, annual costs are multiplied over the long run and threaten the existence of industries and communities reliant on the lake's resources. While few data are available to complete a comprehensive overview on the economic effects of eutrophication-induced HABs in Lake Erie, researchers have been able to build framework models that can quantitatively estimate overall impacts. A joint study commissioned by the Woods Hole Oceanographic Institution in September of 2000 estimated the average total annual economic impacts from HABs in the U.S. amounts to be over $67 million USD (adjusted for 2015), which included public health, commercial fishery, recreation/tourism, and monitoring/management costs (Anderson et al., 2000). Future research should develop a framework that estimates the total balance between both the benefits and costs incurred by using chemical fertilizers that contain phosphorus and nitrogen.

9.0 Benefits of Chemical Fertilizer Use

Fertilizers consist of a wide array of materials of either natural or synthetic origin that are used to supplement the growth of plants through their application to soils or plant tissues directly. In the context of this paper, fertilizers will reference only those materials that contain phosphates and nitrogen.

9.1 Nitrogen Fertilizers

Nitrogen fertilizers are most often derived from ammonia (NH_3). Ammonia itself is produced through the Haber-Bosch process ($N_2 + 3\ H_2 \rightarrow 2\ NH_3$), which is highly energy-intensive. In 2009, Sarah Simpson published in the sustainability section of *Scientific American*, “Nitrogen Fertilizer: Agricultural Breakthrough--And Environmental Bane” (Simpson, 2009). While the article gives a nod to Chemist Fritz Haber’s discovery of ammonia’s synthesis, which has enabled the widespread fertilization of croplands globally, Simpson notes that nitrogen-based “nutrients often spur harmful algal blooms as they flow into the ocean, and hundreds of estuaries around the world suffer from so-called seasonal dead zones as a result.” (Simpson, 2009). The article goes on to cite some of the more beneficial products nitrogen fertilizers have been able to yield, such as biofuels.

The total consumption of nitrogen fertilizers in the United States has increased significantly over the last 50 years. Increased consumption has largely been driven by scholarly research, which has shown a strongly significant and positive effect on total crop yields by increased inputs of nitrogen fertilizers (Lawlor et al., 2001), (Jagadamma et al., 2008). However, with the rise of environmental concerns with respect to nitrogen leaching and eutrophication, many scholars and practitioners have developed management techniques to maximize the use of nitrogen inputs, while limiting external effects on the environment.

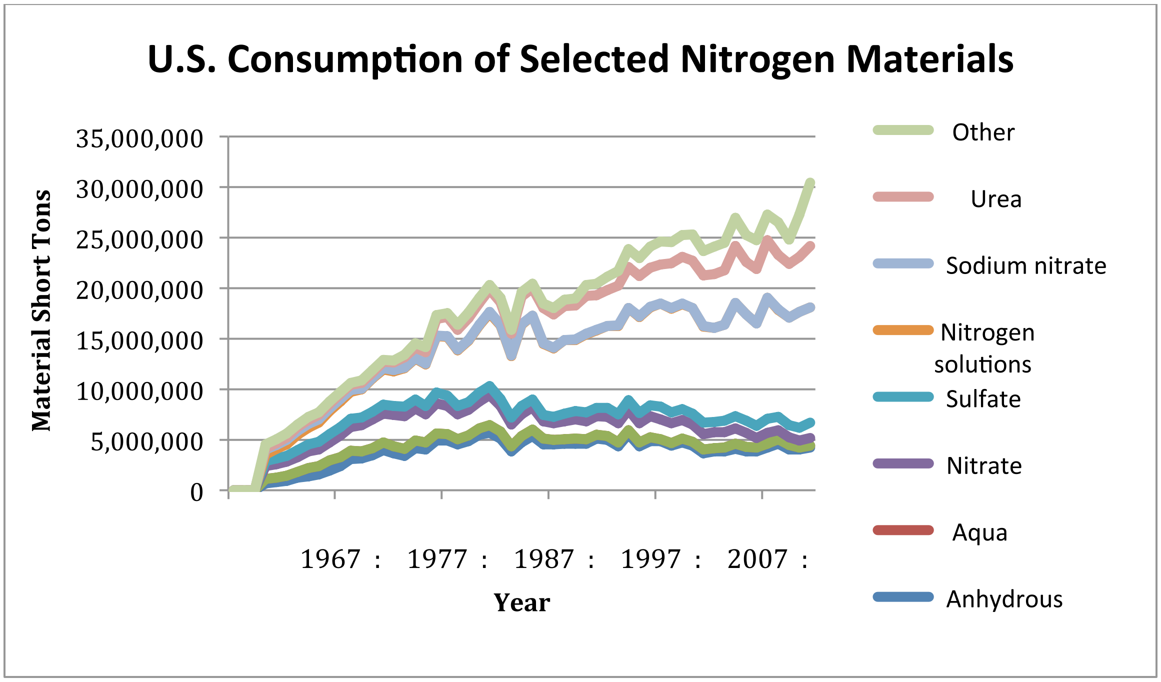

Figure 5 (Cordell, 2009)

A joint study by Texas A&M University and the Tennessee Valley Authority investigated the impact that chemical use reduction had on yields for eight major agricultural crops in the U.S (Smith et al., 1990). Overall, results showed that U.S. corn yields would decline by 41%, cotton by 37%, rice by 27%, barley by 19%, sorghum by 19%, and wheat by 16% when grown without any added nitrogen. Economically speaking, this would result in similar figures according to pricing of each respective crop. Lowered crop prices are important for the provision of local and regional food supplies, due to the variable elasticity of demand for food by food type.

In addition to decreasing prices associated with nitrogen use, there is also considerable evidence that nitrogen fertilizers have significantly helped support the supply for food globally, which is important for communities to support food security (Tilman et al., 2002). In total, evidence is clear that the benefits to society, by using nitrogen fertilizers for increased cropland yields, are high.

9.2 Phosphorus Fertilizers

The second main chemical that helps spur cyanobacterial growth through fertilizer eutrophication in western Lake Erie is phosphorus. Phosphate fertilizers are primarily extracted from minerals that contain the anion PO_4^{3-}. Like nitrogen fertilizers, phosphate fertilizers use has also increased over the last 50 years. Phosphorus is vital for plants to grow, as the chemical is used to transfer and store energy within their cells. An adequate supply of phosphate nutrients enables plants to grow rapidly and mature earlier than those plants without such a supply. Specifically, phosphorus is most abundant in plants in the early stages of tissue growth. Plants without an adequate supply of phosphorus become stunted and can turn shades of purple or brown.

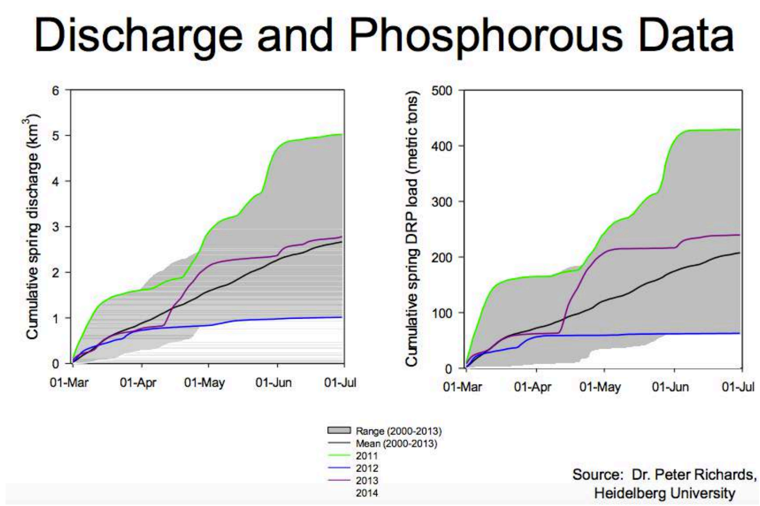

Figure 6 Annual Phosphate Loading (Winslow, 2015)

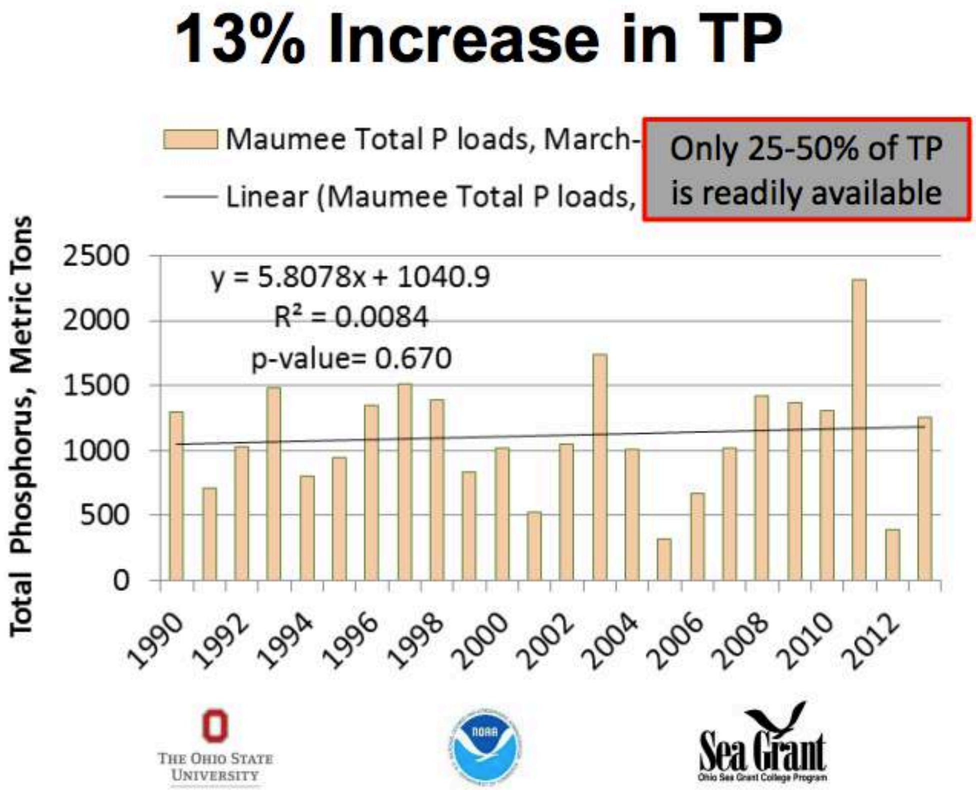

Figure 7 Total Phosphorus Usage Over Time (Winslow, 2015)

Agronomic experiments with phosphate fertilizers have provided evidence for increasing crop and plant yields with increasing phosphate application (Pasda et al., 2001). However, the agronomic effectiveness of phosphate fertilizers is largely attributed to the capacities of the soil to retain and release the chemical (Ozanne & Shaw, 1967). As the phosphorus retention capacity of the soil increases, however, a larger amount of phosphate fertilizers are needed in order to produce the same output yield.

Total benefits of phosphorus are significant when considering total agricultural output. Like nitrogen, phosphate fertilizers promote the opportunity to further food security by increasing crop yields and decreasing food prices.

9.0 Discussion

Completing a qualitative analysis on chemical fertilizer use in the Lake Erie basin required the development of a decomposition framework. By decomposing both direct and indirect effects, it's possible for practitioners and stakeholders to assign weighted values per each of the variables analyzed, and therefore make recommendations for remediation efforts and policies going forward.

Overall, effects of eutrophication via uses of chemical fertilizers on agricultural land surrounding Lake Erie are substantial. Additionally, the benefits Ohio communities gain out of using chemical fertilizers are sizable as well. It is generally reasonable to argue net costs outweigh net benefits. However, due to biased weighting, assigning greater value towards particular variables over others is discriminatory in nature.

When considering total costs and benefits of chemical fertilizer use in the Ohio Lake Erie basin, it is important to note the qualitative nature many variables exhibit. This is largely due to the difficulty in setting social prices according to many of the externalities that affect ecological, environmental, and human health outcomes. However, it is possible to set economic prices according to the effects chemical fertilizer use has on industrial, public health, governmental, and agronomic variables. Future research should therefore investigate total aggregate effects of chemical fertilizer use in the Lake Erie basin, with regards to a full quantitative analysis of its costs and benefits. Deciding whether the costs of chemical fertilizer use outweighs its benefits presents many issues underlying variable ambiguity.

10.0 Conclusion

Eutrophication-induced HABs present a diverse range of issues for ecosystems and communities in the western Lake Erie region. Microcystin and other toxins produced by cyanobacteria harm many organisms along the food chain, human health, and economic viability in core industries that are reliant on the lake's resources. Providing Lake Erie stakeholders with an overview of these effects, how they are caused, and the mechanisms by which they transpire is useful for creating pragmatic and enduring solutions for the future.

With a long history of agricultural innovation and entrepreneurship, Ohio has the capability to further develop and grow its communities and industry. While chemical fertilizers provide farmers with the opportunity to increase yields and reduce costs of agricultural products, they present unintended consequences to the environment, society, and the economy at-large. Cyanotoxins produced by eutrophication-induced algae growth directly harms numerous species, including various zooplankton, fish, reptiles, and birds. Additionally, microcystins harm human health by damaging liver processes and inducing tumor growth. By reviewing both these consequences and gains, stakeholders will have access to a comprehensive vantage point on the effects of chemical fertilizer use in Ohio.

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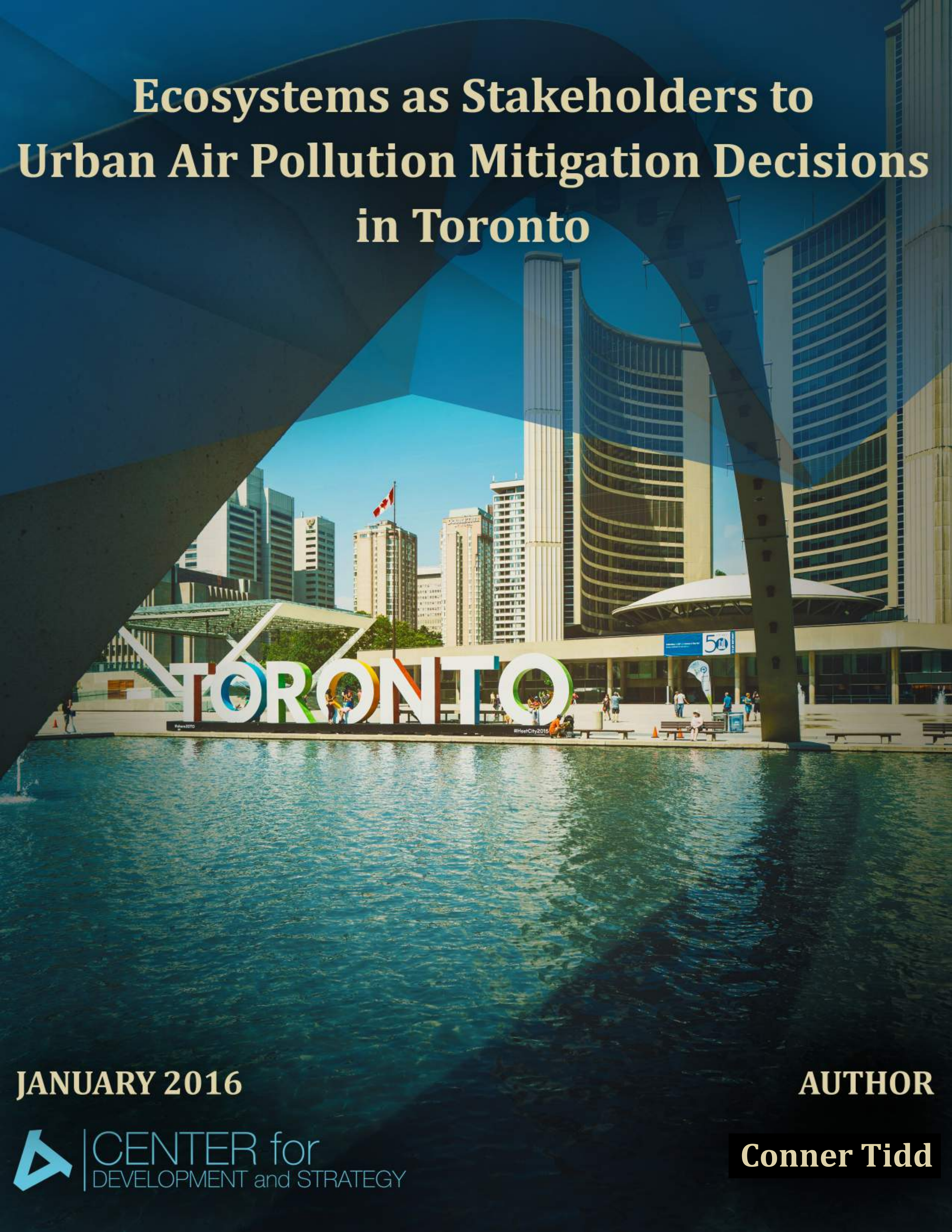
Ecosystems as Stakeholders to Urban Air Pollution Mitigation Decisions in Toronto



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Ecosystems as Stakeholders to Urban Air Pollution Mitigation Decisions in Toronto

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Abstract

This paper explores the role that ecosystems can have in the decision making framework for urban air pollution mitigation in Toronto, Ontario, Canada. The focus is on management planning of green roof implementation to mitigate the effects of urban air pollution. The importance of Toronto's impact on surrounding ecosystems, conservation areas and their fragile habitats is often overlooked. Through a literature analysis of primary sources, the need for consideration of these nearby ecosystems in management planning is shown. Toronto's proximity to aquatic and terrestrial ecosystems causes the dual effects of the urban heat island effect and urban air pollution to carry over to these systems. The city should address the impact on these systems in terms of mitigation options. Toronto is currently building many new residences, commercial properties and infrastructure projects. Investigating the effects of green roofs in Toronto and the effects that this could have on the city and surrounding ecosystems provides a better framework for decision makers in the city to decide whether green roofing has a place in Toronto. By providing a sound scientific background on the topic relevant stakeholders and decision makers will be able to properly gauge the need and effects the implementation of green roofs in the city could have on surrounding ecosystems.

Keywords: ecosystems as stakeholders, frameworks for mitigation decision making, green roof implementation, economics of green roofs

Ecosystems as Stakeholders to Management Decisions of Green Roofs as a Mitigation Tools for Urban Heat Islands and Air Pollution in Toronto

Introduction

As rapid urbanization occurs globally the issues of ensuring urban health of humans and the natural environment is receiving increasing attention (UNEP, 2015). From an ecosystems perspective there are many different facets to be considered: air quality, water quality, urban infrastructure, biodiversity, emissions levels, human population levels, waste levels, energy use, etc. These all have significant impacts on the populations and surrounding areas. Cities increasingly are turning to mitigation measures to help negate the harmful effects of city life. Two significant impacts of large urban spaces are UHI and urban air pollution (UNEP, 2015). Combatting these dual effects is often a costly process and as such a well-established decision making framework needs to be put in place and all stakeholders must be considered. Green roofs have come in to use throughout the world to help mitigate some effects of urban life. In Toronto however there has been a failure to recognize the impact the green roofs can have in helping mitigate the UHIs amplification effect on urban air pollution and the effects that has on surrounding ecosystems.

For the purposes of this analysis the focus will be on the place of surrounding ecosystems in a decision making framework for the implementation of green roofs as a mitigation tool for the effect of Toronto as an urban centre on surrounding ecosystems through the UHI effect and urban pollution.

This is an important topic of research as Toronto is currently going through large increases in the construction of residential, commercial, and infrastructure projects throughout the city. As these projects are built both vertically in the downtown core to increase density and horizontally to expand the city they are increasing the dual UHI and urban air pollution effects (Gough and Rozanov, 2001). This is also putting more pressure on the few intact ecosystems surrounding the city. This is a key time for decision makers to consider mitigation efforts. The city of Toronto has worked to explore the efficacy of green roofs as mitigation tool. However, the decision making framework has always been based on an anthropocentric view point. This takes away from the relevance of surrounding ecosystems' worth and the various ecological goods and services that they provide. As such the effects that UHI and urban air pollution have on surrounding ecosystems needs to be illustrated and the potentially affected areas and species identified. This will give decision makers a better understanding of the true magnitude that the UHI and air pollution have beyond the human level.

In order to investigate the place of ecosystems in the decision making framework, a review of relevant history and past research on the topic of the heat island effect, urban pollution, and their typical implications for human populations and surrounding ecosystems was carried out. Drawing from primary sources and grey literature based around technical reports from the municipal government of Toronto, the city will be situated as an urban heat island, its levels of urban pollution will be examined, an illustration of the effects that this has on the people of Toronto and surrounding ecosystems specific to Toronto will be explored. This provides a framework to analyze the current decision making framework for green roofs as a management tool and their efficacy in combatting the effects of heat islands and urban pollution. The current state of research and the planning framework for potential green roofs as a management tool localized to Toronto and the predicted effects that it would have on surrounding ecosystems is illustrated. This will be concluded by recommendations for management.

However, this approach is limited in that as a preliminary report showing the linkages, it lacks the quantitative data and models needed to show the more exact effects that varying levels of urban air pollution and the UHI effect have on surrounding ecosystems over specific spatial and temporal scales.

Background

Urban Heat Island Effect

The UHI effect causes higher temperatures in urban areas compared to surrounding rural areas. This is a well-founded effect that has been observed and documented globally (Santamouris, 2007; Tran, et al., 2006; Yang, et al., 2015). The UHI effect increases with the population density and size of cities (Gough and Rozanov, 2001). Toronto is a rapidly expanding city set to face continually growing UHI effects. UHI effects globally have been observed to cause as much as an 11 degree Celsius difference by mid-morning (Aniello, et al., 1995). The UHI effect and the elevated temperatures lead to a rise in energy usage, particularly for cooling. Due to Toronto's seasonal variance in weather, this can cause cooling systems to have to work much harder than they already do. In Toronto the UHI effect has been observed to be as much as 3.25 degrees Celsius (Gough and Rozanov, 2001). The UHI in turn then exacerbates problems of urban air pollution by placing additional needs on the cooling systems throughout the city leading to more air pollution (Gough and Rozanov, 2001).

Urban Air Pollution

Air pollution is a well-recognized problem globally and the United Nations Environmental Programme (UNEP) estimates that "more than 1 billion people are exposed to outdoor air pollution annually" and "Urban air pollution is linked to up to 1 million premature deaths and 1 million deaths each year" (United Nations Environmental Programme [UNEP], 2015). Not only does this cause loss of human lives and livelihoods, but it is also an economic burden as "Urban air pollution is estimated to cost approximately 2% of GDP in developed countries and 5% in developing countries" (UNEP, 2015). It is evident that this a problem needs to be addressed. In addition Toronto as a rapidly urbanizing city faces particular risks as "Rapid urbanisation has resulted in increasing urban air pollution in major cities" (UNEP, 2015). Toronto is no exception to these problems and the municipal government has been actively considering mitigation measures such as green roofs.

Implications of UHI and Urban Air Pollution (UAP)

The combination of UHI and UAP causes significant problems for human populations such as increased heating costs, premature death, and aggravating illnesses (Zupanic et al., 2015). Surrounding ecosystems have also experienced negative effects of nearby urban air pollution and animals have been found to be early indicators of damage (Newman & Schreiber, 1984). Even though it has been noted that urban air pollution controls are in place and that "most developed countries have put in place measures to reduce vehicle emissions, in terms of fuel quality and vehicle emission reduction technologies" (UNEP, 2015), there are still significant emissions of pollutants. Further, these two effects have a complementary effect on one another and it has been noted by Sarrat (2006) that "both nocturnal and diurnal urban effects have an important impact on the primary and secondary regional pollutants, more specifically ozone and nitrogen oxides (NO_x)" This serves to illustrate the relationship between the UHI and urban air pollution. In addition “the spatial distribution and the availability of pollutants are significantly modified by the urbanized area mainly due to enhanced turbulence.” (Sarrat, 2006). The dual effect of UHI exacerbating urban air pollution and its effect on surrounding ecosystems is well documented.

The effects on ecosystems can be quite severe. Looking at cities globally it is found that “Sulphur dioxide (SO_2) and coal smoke formerly dominated the developed world and remain a growing problem in the developing countries. [However] In both regions, the ‘modern’ pollutants, in the form of nitrogen oxides (NO_x), ozone (O_3) and fine particulate matter (PM_{10}), are also major problems for urban vegetation.” (Bell, et al., 2006). This shows the need for management strategies for urban pollution levels. The study continued to show that “despite generally lower pollutant levels in the developed world, there is evidence that both crops and wild species are adversely impacted.” (Bell, et al., 2006). This contributes to the idea that in a management framework for adaptation and mitigation of these effects surrounding ecosystems need to be considered a stakeholder and should be given due consideration.

Toronto as an Urban Heat Island

Toronto’s heat island effect has been established and studied in the past. Gough and Rozanov (2001) did a comparison study of the UHI effect by looking at temperature differences between the downtown core, Pearson airport and Vineland. Observed temperature differences show a maximum difference of +3.25 degrees Celsius during the day time between Pearson and Toronto. There are also significant diurnal temperature differences observed year round. Though this is a smaller figure than in other larger global cities it still leads to a significant effect on the surrounding ecosystems and the people of Toronto. The range in diurnal temperature changes can be seen in the following two figures.

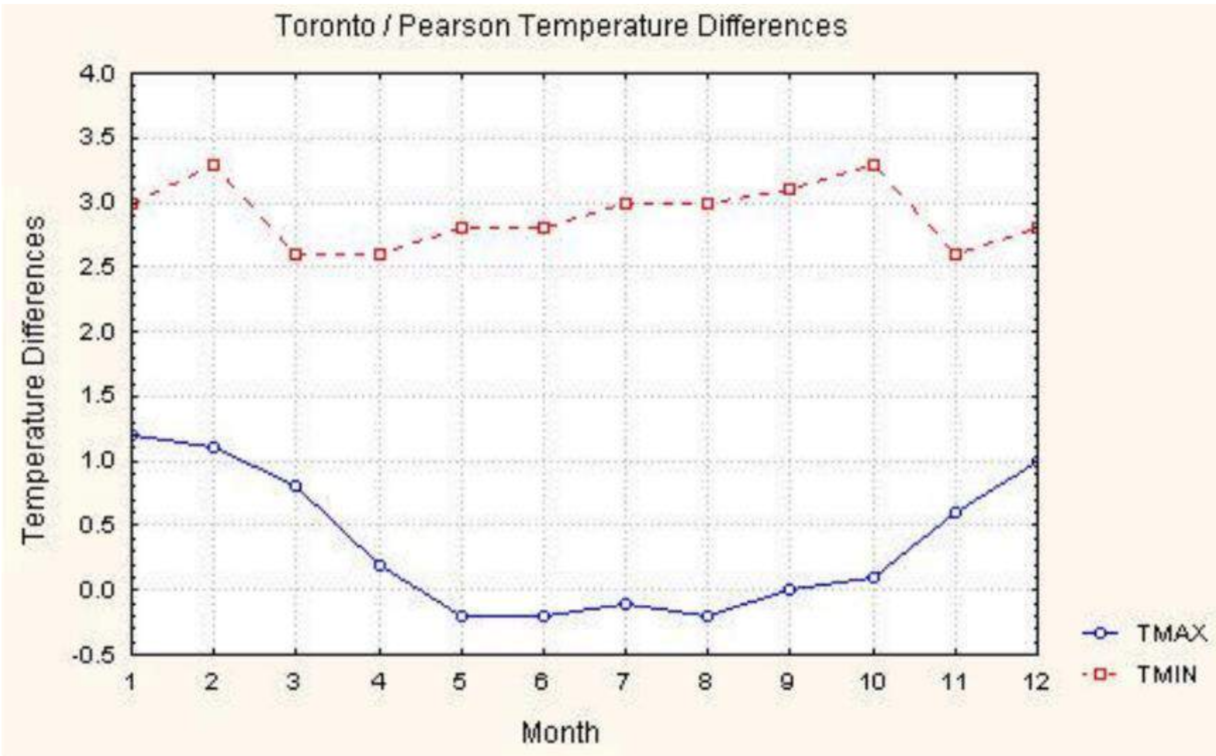

Figure 1 Difference between Toronto (downtown) and Toronto (Pearson airport) (Gough and Rozanov, 2001)

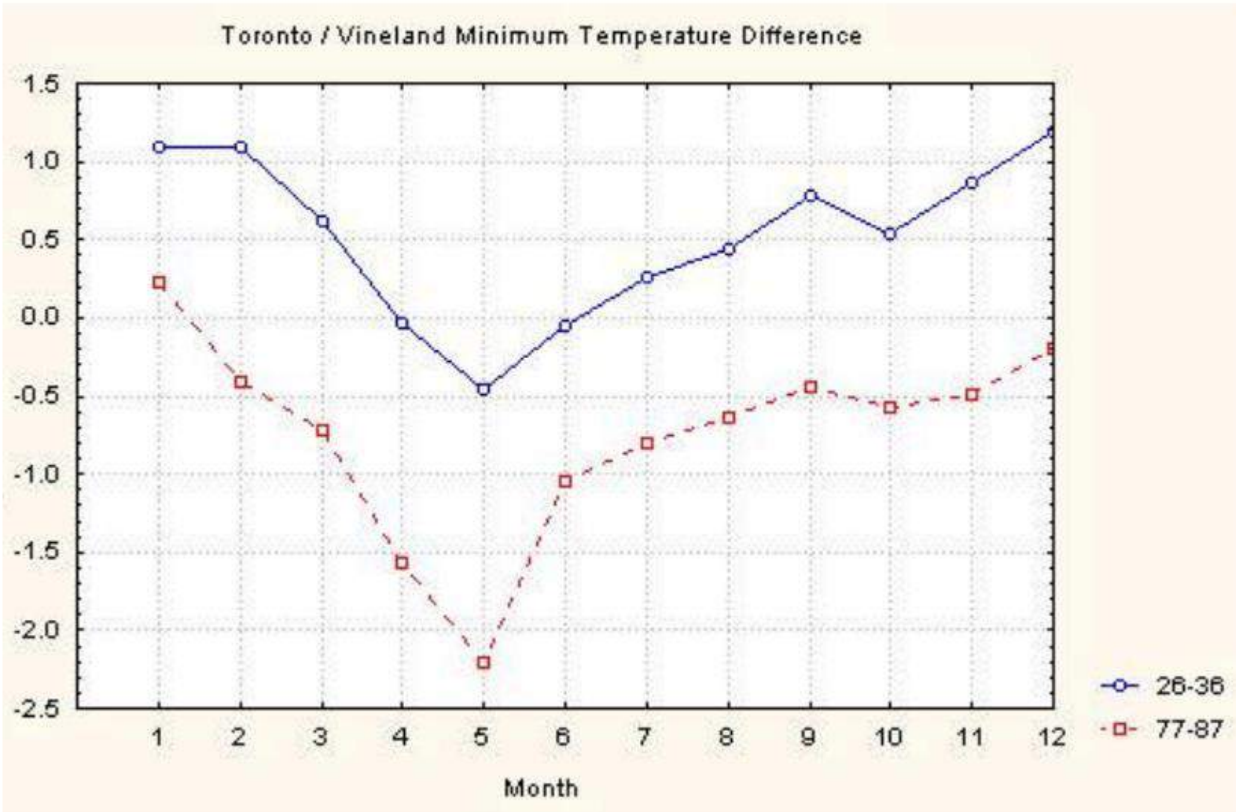

Figure 2 Difference of monthly mean minimum daily temperatures between Vineland and Toronto for 1926-1936 and 1977-1987 (Gough and Rozanov, 2001)

What is important to note about these figures is how significant the UHI is in terms of proximity to the city. As a direct measurement Pearson airport lies 19.31 km from the city centre through mostly urban centres. This shows the difference in temperature due to the UHI the further distance travelled from the city centre. Vineland lies 56.32 km from the city centre mostly over Lake Ontario. However, Vineland is on the Niagara Peninsula and has too many different geographical features and is too far from Toronto to establish a causal link with UHI.

Urban Air Pollution in Toronto

Urban air pollution in Toronto is considered to be when air contains "one or more substances in amounts harmful to people and plants" (Clean Air Partnership, 2015). The air pollution levels in Toronto are monitored by Public Health Toronto and Environment Canada and measured on the Air Quality Index (Pengelly and Sommerfreund, 2004). Three pollutants are considered when calculating the burden of illness on the population. These are ground-level Ozone (O_3), fine particulate matter ($PM_{2.5}$), Nitrogen Dioxide (NO_2). “When the proportion of the burden attributable to each individual pollutant is considered, NO_2, $PM_{2.5}$, and O_3 contribute the most to cardiovascular and respiratory ill health. They account for about 13%, 69%, and 14% of premature mortality and about 35%, 33%, and 29% of hospitalizations, respectively” (Campbell & Gower, 2014).

The main sources for these pollutants can be broken down into the following categories: traffic, industrial sources, residential and commercial sources, and off-road mobile sources such as rail, air, and marine sources.” (Campbell & Gower, 2014)

Implications of Urban Air Pollution to Human Health

The effects on Urban Air pollution in Toronto have been broken down by the city. A summary view can be seen in the figure below.

Air Pollution Source		Health Outcome	
		Premature Deaths	**Hospitalizations**
	All Sources Combined[1]	***1,300***	***3,550***
Sources in Toronto	Traffic (Cars and trucks)	280	1090
	Mobile off-road (eg.,rail, air, marine sources)	80	280
	Industrial	120	200
	Residential/Commercial	190	400
Sources outside Toronto	Transboundary from United States	390	870
	Transboundary from Ontario	270	740

Figure 3 Burden of illness attributable to air pollution from sources inside and outside Toronto (Campbell & Gower, 2014)

Of these sources, traffic has the greatest impact on health, contributing to about 280 premature deaths and 1,090 hospitalizations each year, or about 20% of all premature deaths and 30% of all hospitalizations due to air pollution (Pengelly and Sommerfreund, 2004). When only pollutants emitted within Toronto's boundaries are considered, the proportions of premature deaths and hospitalizations attributable to traffic are 42% and 55%, respectively" (Campbell & Gower, 2014). Additionally pollution in Toronto comes from transboundary sources with 21% out of province and 25% out of country (Pengelly and Sommerfreund, 2004).

The number of deaths and hospitalizations shows the need for additional management strategies within the city for air pollution and UHI which complements air pollution and aggravates the problem. Campbell and Grower (2014) elaborate that "Residential and commercial sectors are the next most important local contributors to health impacts from air pollution, accounting for about 190 premature deaths and 400 hospitalizations (or 28% of deaths and 20% of hospitalizations arising from pollution emitted in Toronto). The main source of emissions from residential and commercial properties is combustion of natural gas to heat homes and buildings, as well as heating water" (Campbell & Gower, 2014). Additionally "based on emissions reported to the National Pollutant Release Inventory (NPRI), industrial sources account for about 120 premature deaths and 200 hospitalizations (or 18% of deaths and 10% of hospitalizations arising from pollution emitted in Toronto)." Finally the least harmful source of air pollution in Toronto is "mobile non-road sources such as emissions arising from rail and air traffic contribute about 80 premature deaths and 280 hospitalizations (or 12% of deaths and 14% of hospitalizations due to pollution emitted in Toronto)." (Campbell & Gower, 2014; Pengelly and Sommerfreund, 2004). Currently there are insufficient management strategies to internalize the negative externalities generated by these polluters. The distribution of urban air pollution within the city boundaries is visible in the figures below.

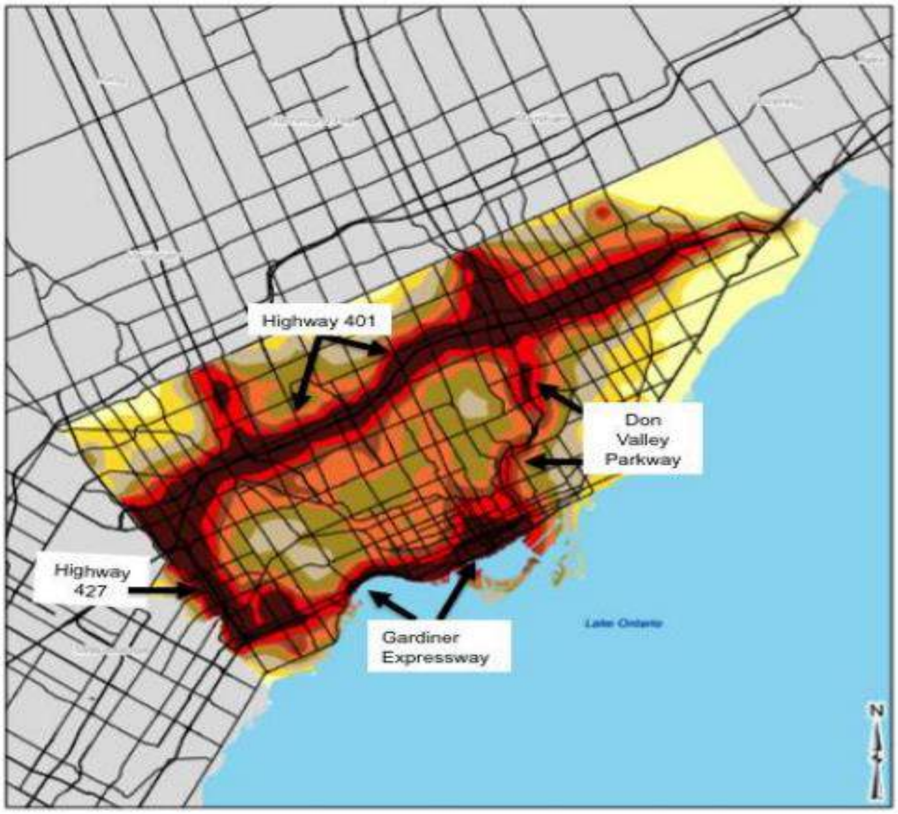

Figure 4 NOx levels across the City of Toronto, 2006 (Campbell & Gower, 2014)

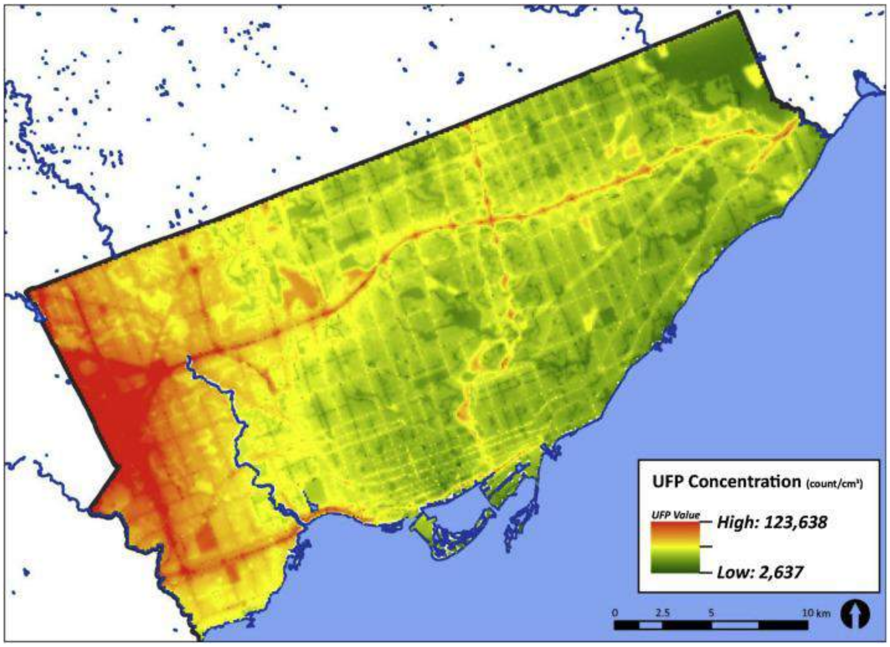

Figure 5 Predicted spatial distribution of ambient Ultra Fine Particles in Toronto, Canada (Weichental, 2016)

Implications of Urban Air Pollution on Ecosystems

In looking at impacts on the non-human environment in surrounding ecosystems, a review of studies looking at animals as indicator species in response to urban air pollution was consulted. It was found that "animal species and populations can act as important indicators of biotic and abiotic responses of aquatic and terrestrial ecosystems." (Newman & Schreiber, 1984). Further it was found that "These responses can indicate long-term trends in ecosystem health and productivity, chemical cycling, genetics, and regulation." As well as that "for short-term trends, fish and wildlife also serve as monitors of changes in community structure, signaling food-web contamination, as well as providing a measure of ecosystem vitality." (Newman & Schreiber, 1984). This serves to frame the argument of incorporating ecosystems as stakeholders in management decisions because they not only are affected but can serve as an indicator to monitor the progress of mitigation efforts.

Green Roofs as a Management Tool

A green roof is defined as “a vegetated roof or deck designed to provide urban greening for buildings, people, or the environment.” (Dvorak & Volder, 2010). These are starting to become more common across North America but are noted for being originally “Made popular across Europe over the past few decades” (Dvorak & Volder, 2010) and it has been observed that “green roofs are now becoming more familiar to North Americans as some cities have built green roof pilot projects and adopted incentives for using green roofs or even require their use.” (Dvorak & Volder, 2010).

There have been several pilot projects in Toronto but so far, they have not been made mandatory as they have been in some European countries such as France (Agence French-Press, 2015). The city has the opportunity to explore this legislation as “Green roof standards and guidelines are also emerging to be used for governance and project specification.” (Dvorak & Volder, 2010). However, before this is to happen more needs to be understood about their application in the context of Toronto. As it currently stands “much is known about the application of green roofs across Europe [however] much less is known about their application across North America's diverse ecological regions. When considering the many decisions required in applying green roof technology to a specific place, there are few choices more critical to their success than the selection of appropriate vegetation.” (Dvorak & Volder, 2010). In the application of this study a review of green roof research was conducted in order to “investigate what is known about the application of plants on green roofs across North America and their ecological implications.” (Dvorak & Volder, 2010). Results of this study indicated that plant survival rates and thus the effectiveness of the green roofs varied across ecoregions and as such they must be tailored to the unique circumstances present. As the study indicates “as green roofs continue to become regulated and adopted in policy, further development of standards and guidelines is needed" (Dvorak & Volder, 2010).

Green Roofs as a Management Tool for UHI and Urban Air Pollution

Green roof usage in city centres can actively work to combat the UHI. This has been studied and examined throughout the world and is well documented. It has been noted that "Green roofs are a passive cooling technique that stop incoming solar radiation from reaching the building structure below. Many studies have been conducted over the past 10 years to consider the potential building energy benefits of green roofs and shown that they can offer benefits in winter heating reduction as well as summer cooling" (Castleton, et al., 2010). This is accomplished because "Green roofs present the opportunity to expand the presence of vegetated surfaces by replacing impermeable surfaces in urban areas, providing for a reduction in peak summer urban heat island temperatures." (Banting, et al., 2005).

Green Roof's benefits in combatting urban air pollution is also well examined. This is due to the way in which smog forms when Nitrogen Dioxide (NO_x) reacts with volatile organic compounds. This process is accelerated by higher ambient air temperatures (Banting, et al., 2005). Three main ways that green roofs work to combat air pollution besides lowerering ambient air temperatures are: reducing the demand on polluting energy production plants from a reduced demand for cooling (Konopacki & Akbari, 2001), the trapping of particulates in foliage (Johnston, 1996), and dissolving or sequestering gaseous pollutants, particularly CO_2 through the stomata of their leaves (Nowak and Crane, 1998).

Discussion

Green Roofs as a Management Tool in Toronto

Green roofs as a management tool in Toronto have been examined in parts through various studies (Banting et al., 2005; Bass et al., 2002; Gough and Rozanov, 2001; Zupancic, et al., 2015.. A 2005 study conducted by Ryerson University in partnership with the City of Toronto attempted to quantify the value added to the city based on a 100% coverage rate. The parameters for the study were as follows: "The benefits on a city-wide basis were calculated based on the assumption that 100% of available green roof area be used. The available green roof area included flat roofs on buildings with more than 350 sq. m. of roof area, and assuming at least 75% of the roof area would be greened. The total available green roof area city-wide was determined to be 5,000 hectares (50 million sq. m.)." (Banting et al., 2005) The determination of available green roof space was based on the following land use study.

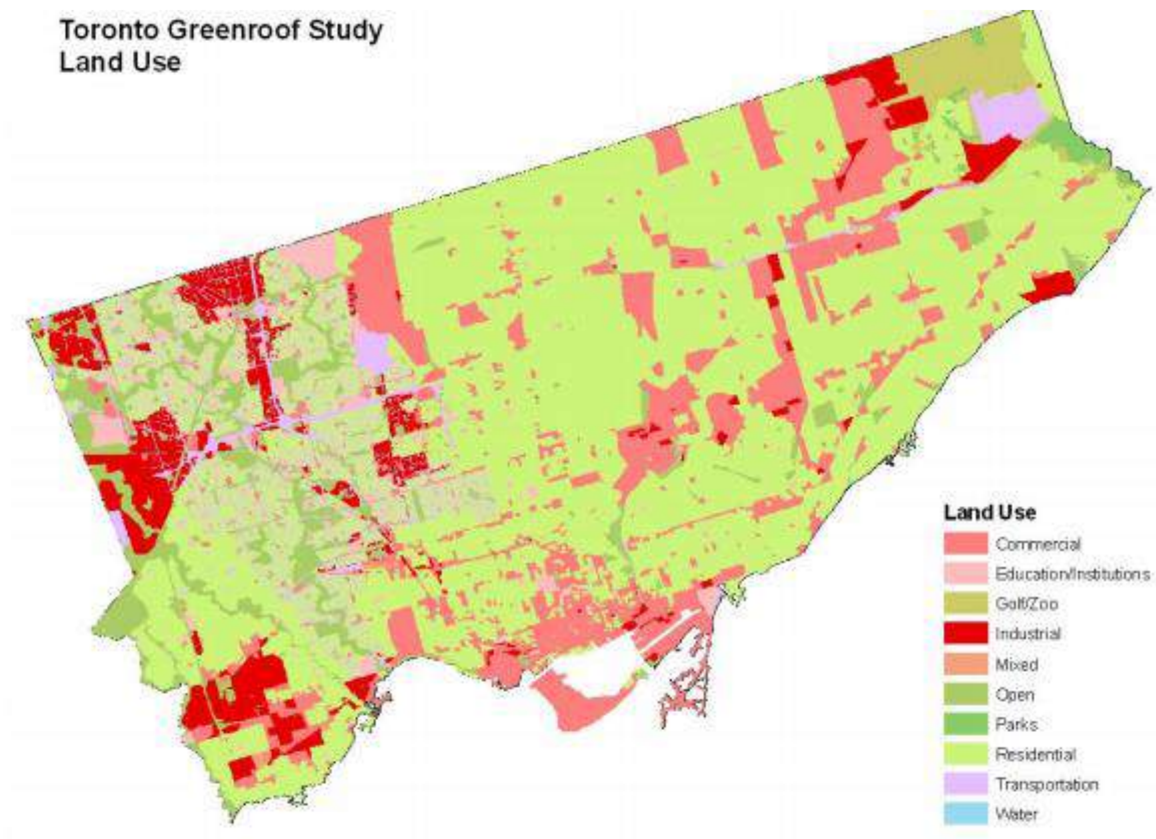

Figure 6 Determination of Land use in Toronto (Banting et al., 2005)

"The benefits were determined as initial cost saving related to capital costs or an amount of annually recurring cost saving" (Banting, et al., 2005).

One limitation acknowledged in that study was the assumption of a 100% coverage rate. This is not a feasible solution and does not represent the best solution. It is known through research that "Older buildings with poor existing insulation are deemed to benefit most from a green roof" (Banting, et al., 2005). As such it may be the most feasible place to start implementing green roofs. Bass (2002) found that with a 5% coverage of the city, starting at the most feasible sites, green roofs "reduce(s) UHI and with sufficient moisture for evapotranspiration, a limited green roof coverage decreases UHI and will enhance the greening strategy in core". (Bass et al., 2002). Further work has been done to advance the argument for green roofs even in a limited area. In a review analysis of the implications of green roofs Zuapncic (2015) found that "Emerging evidence also suggests that closely spaced and connected smaller green spaces can provide greater cooling effects to adjacent urban areas than large individual parks with open grass areas." (Zupancic, et al., 2015). This indicates that green roof coverage need not be 100% but instead can be smaller in scale when dispersed favourably.

Green roofs can also be used in Toronto in order to connect fragmented habitat. In working on this issue for Toronto city planning Bass (2002) found that "green roofs can be used to connect fragmented habitats when installed in aggregation especially if located near fragmented ground-level habitats and that "Where technically feasible, green roofs should be designed to protect sensitive biological communities and avoid aggressive species. By designing green roof scapes that include important habitat forming and forage species into planting designs Toronto will encourage the proliferation of biodiversity across urban green roofs. In addition to strengthening existing green corridors, green roofs represent an opportunity to create new green space in areas that are otherwise unsuitable for natural restoration" (Bass et al., 2002). The enhancement of habitat corridors specifically for migratory species would have an impact on the surrounding conservation areas which are currently well noted for their importance as a habitat for migratory species.

Category of benefit	Initial cost saving	Annual cost saving
Stormwater	$118,000,000	
Combined Sewer Overflow (CSO)	$46,600,000	$750,000
Air Quality		$2,500,000
Building Energy	$68,700,000	$21,560,000
Urban Heat Island	$79,800,000	$12,320,000
Total	**$313,100,000**	**$37,130,000**

Figure 7 Cost savings benefits of green roof implementation (Banting et al., 2005)

In assuming the 100% coverage rate of green roofs for the city of Toronto it was found that there would be approximately $313 million Canadian dollars in initial savings due to a combination of: stormwater, combined sewer overflow, air quality improvements, building energy savings, and mitigation of the UHI (Gough and Rozanov, 2001). In addition an expected 37 million dollars in annual savings could be made (see Figure 7). In estimating the costs of the installation of the green roof project for 5000 hectares, it was estimated that costs would be $15 per square foot due to the operation of economies of scale. This leaves a total of $8 073 000 000 in installation (costs calculated through figures presented in (Banting et al., 2005). This figure includes all the available area on flat rooftops in the city. However, if costs and benefits were to be optimized through retrofits of older buildings, which would derive the most benefit from the installation of green roofs, costs could be minimized while benefits could be maximized.

Predicted Effects on Surrounding Ecosystems

Most research on green roofs as a management system in Toronto stops at the human boundary and at the city limits. This ignores the important environmental aspect of management considerations. The existing research touches on a few key issues. The first of which is the benefits to biodiversity and migratory species through the city of Toronto. As it currently stands

"The City of Toronto has a number of policies and programs directed at migratory and breeding bird conservation and these are described in more detail in the Migratory Birds reports for the City of Toronto and Birds of Toronto (2007)." (Currie & Bass, 2010). These programs and their goals could be augmented by the installation of green roofs. It has been found that "While habitat created by green roofs will typically not provide the same quality of food or shelter found in a natural area, green roofs do provide vegetation where there would otherwise be none and thereby create potential habitat for local and migratory birds." (Currie & Bass, 2010). The authors went on to further elaborate that "Green roofs could be used as part of a strategy to provide or enhance stopover habitat for migratory birds and foraging, nesting and mating needs of breeding birds Urban development and loss of habitat have impacted travel distances, expended energies, and reduced the availability of food sources for migratory birds passing through Toronto."(Currie & Bass, 2010). This work also further shows that 100% green roof coverage is not needed but a well-placed interconnected system as it has been found that "A matrix of well-distributed aggregations of diverse green roof habitats may become attractive for migratory birds that view green roofs as possible "stepping stones" in a search for more suitable and larger habitat patches at ground level" (Currie & Bass, 2010). In addition the selection of appropriate green roofing material can serve to enhance the effects on surrounding ecosystems as "Diverse green roofs established with grasses and herbaceous plants mature each season to produce numerous seed heads that can provide invaluable energy sources for newly arriving migratory birds particularly those who are exhausted by a lengthy migratory journey over Lake Ontario." (Currie & Bass, 2010). This again illustrates the impact that green roofs can have on surrounding ecosystems and migratory species going to the conservation areas surrounding the city. Specifically in Toronto the following birds could benefit from habitat opportunities on green roofs: the Northern

Cardinal, Downy Woodpecker, Black-Capped Chickadee, White-Breasted Nuthatch, Rock Pigeon, European Starling, House Sparrow, American Robin, Red-winged Blackbird, Song, Eastern Meadowlark, and Bobolink. All of these species have been observed in or around the City of Toronto (Currie & Bass, 2010). Particularly species derive great benefit from habitat provided by green roofs if aggregations of biodiverse green roofs provide habitat and food for breeding pairs.

In a literature review of green roof implementation Banting et al., (2001) reported that "adding green space in the form of green roofs to densely populated urban environments provides eco-restorative habitats for displaced creatures. Green roofs provide food, habitat, shelter, nesting opportunities and a safe resting place for spiders, beetles, butterflies, birds and other invertebrates." and "Studies report that this elevated urban ecosystem affords unique protection from grade level predators, traffic noise and human intervention. Studies reveal that butterflies can access green space on the 20th floor of a building" (Banting et al., 2005). The mounting evidence for the benefits of green roofs outside of anthropocentric values furthers the need for environmental stakeholder consideration.

What all studies have neglected to examine are the effects on the surrounding ecosystems specific to Toronto. For the purposes of this analysis the surrounding systems will be based round the analysis of the ecoregion. Toronto and the surrounding areas lie in Ecoregion 7E (Lake Erie-Lake Ontario). This is visible in the maps shown below.

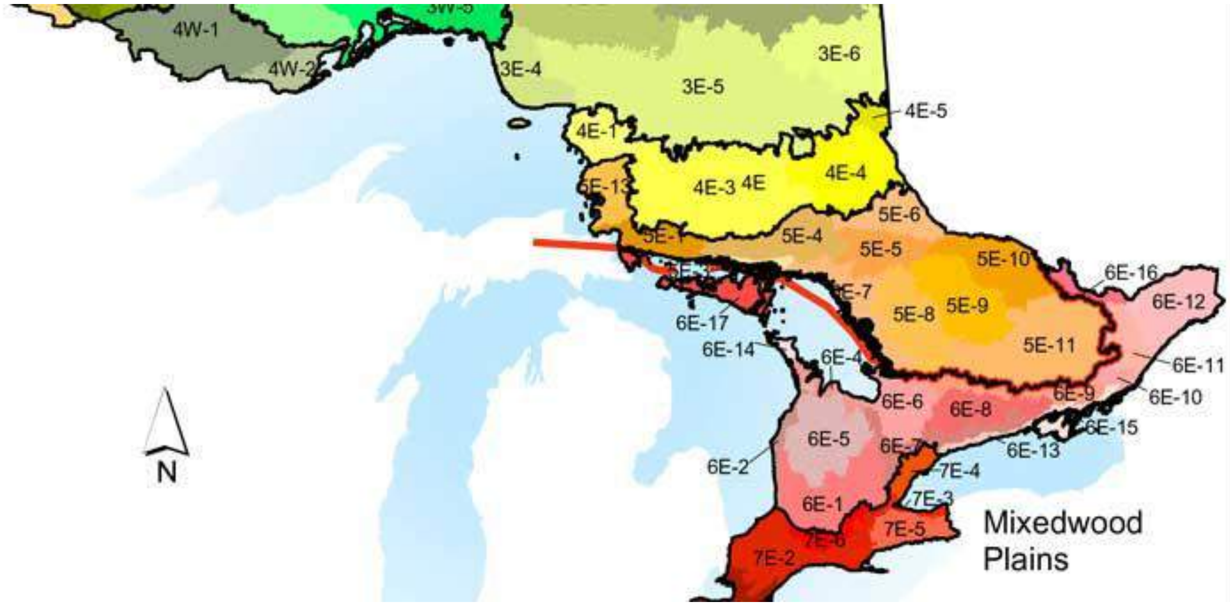

Figure 8 Ecoregions of Southern Ontario (Crins et al., 2009)

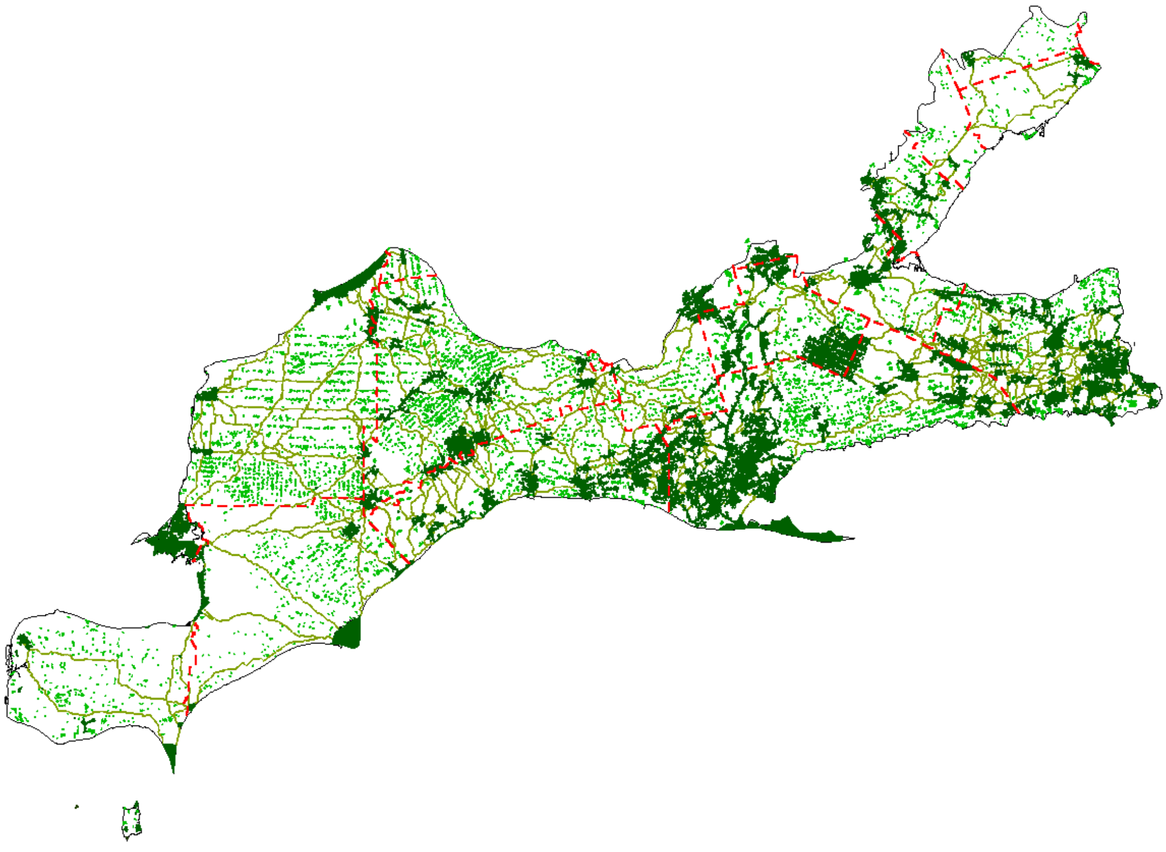

Figure 9 Ecoregion 7e (Crins et al., 2009)

“This most southern ecoregion encompasses 2.2% (2,185,845 ha) of Ontario, and extends from Windsor and Sarnia east to the Niagara Peninsula and Toronto, with shoreline on Lakes Huron, Erie, and Ontario” (The Ecosystems of Ontario Part 1). This ecoregion is an area with fragile and sparse land cover necessitating extra diligence in management considerations. Currently “About 78% of the ecoregion has been converted to cropland and pasture, and developed land (e.g., urban areas and road networks) encompasses more than 7% of the ecoregion. Of the remaining forest remnants, dense deciduous forest covers 10.3%, sparse deciduous forest covers 1.0%, and mixed deciduous forest covers 0.8% of the ecoregion.” (Crins et al., 2009). Due to the low levels of remaining forest cover and the large levels of urban and farmland there is a reduced level of resilience in these ecosystems. This necessitates extra consideration to the effects from the UHI and urban air pollution resulting from Toronto and as such should be considered in the management decisions of the city.

In addition, water features and their interplay with urban air pollution and the UHI need to be considered. As it currently stands "Ecoregion 7E is located in the Great Lakes Watershed. Several rivers have created incised valleys perpendicular to the shores of Lakes Huron, Erie, and Ontario, which add to topographic variation. The Grand, Thames, Detroit, and Humber Rivers, (all designated Canadian Heritage Rivers) are managed through activities of the Grand River Conservation Authority (CA), the Upper Thames River CA, the Essex Region CA, and the Toronto and Region CA. Other large rivers in the ecoregion include the Credit, Niagara, and Sydenham Rivers, and Big Creek. There are a few small lakes and drainage in this ecoregion is poor. There are hundreds of small aquifers in sand and gravel deposits throughout this ecoregion. Although most wetlands have been eliminated, some coastal marshes, deciduous and coniferous swamps, and open fens remain scattered throughout the ecoregion. The Lake Erie coastal marshes (e.g., Point Pelee, Rondeau Bay, Long Point, and Turkey Point) support the largest diversity of flora and fauna in the Great Lakes" (Crins et al., 2009). Often water quality issues are not associated with UHI or urban air pollution but they are certainly affected directly through deposition of air pollution into bodies of water of urban air pollution and through disturbances in thermal properties and regular thermal regulation which can change the composition of water bodies.

Perhaps the most significant impact green roofs could have on UHI and urban air pollution is on the surrounding flora and fauna. It has been identified that "Ecoregion 7E is contained within the Deciduous Forest Region, Niagara Forest Section. The flora and fauna of this ecoregion are the most diverse in Canada. For example, remnants of Carolinian forests contain species such as the tulip-tree, black gum, sycamore, Kentucky coffee-tree, pawpaw, various oaks and hickories, and common hackberry, in addition to the more widespread sugar maple, American beech, white ash, eastern hemlock, and eastern white pine. This ecoregion also supports the largest remnants of tall-grass prairie in the province. Typical mammals inhabiting this ecoregion include white-tailed deer, northern raccoon, striped skunk, and the Virginia opossum which has increased its distribution and abundance since the latter half of the 20th century. Characteristic birds include green heron, Virginia rail, Cooper's hawk, eastern kingbird, willow flycatcher, brown thrasher, yellow warbler, common yellowthroat, northern cardinal, and savannah sparrow. Wild turkey has been re-introduced into the ecoregion. Herpetofauna, is diverse, including several provincially rare species (e.g., spiny softshell turtle), as well as more frequent species such as eastern red-backed salamander, American toad, eastern garter snake, and Midland painted turtle. Longnose gar, channel catfish, smallmouth bass, yellow perch, walleye, northern hogsucker, banded killifish, and spottail shiner are among the fish species found in the lakes and rivers in this ecoregion. This ecoregion is the most imperiled in Canada because of the amount of natural habitat that has been drained, cut, and converted into agricultural and suburban land uses. Many of Ontario's species at risk occur here, including Acadian flycatcher, king rail, prothonotary warbler, hooded warbler, spiny softshell turtle, blue racer, and smallmouthed salamander." (Crins et al., 2009). The diversity and sensitivity of the flora and fauna in this ecoregion require extra attention when considering the calculations of the need and effects of green roofs.

In addition there are various conservation areas surrounding the city that could be at risk from unmitigated air pollution exacerbated by the UHI effect. An example of this is Rattray marsh conservation area to the west of the city. Rattray marsh is the last remaining wetland between Toronto and Burlington. Further "Rattray Marsh is an important habitat with many sensitive and significant species that cannot be compromised" (Credit Valley Conservation, 2015). Under current management plans at Rattray Marsh air quality, which is largely a consequence of Toronto's urban activities, is actively considered and measured (Harrington and Hoyle ltd, 2009). An additional concern for Rattray Marsh is the high heat sink capacity of waters. If urbanization continues in the surrounding areas and the diurnal temperature range continues to increase Rattray Marsh could be especially affected due to its unique properties as a heat sink.

Additional areas similar to Rattray Marsh such as Cootes Paradise which is located at the west end of Lake Ontario and is a National Historic Site, Nationally Important Bird Area (IBA), and a Nationally Important Reptile and Amphibian Area (IMPARA). This area represents over 99% of the remaining unaltered lands along the Lake Ontario shoreline. (Royal Botanical Gardens, 2015). Again risks to this ecosystem needs to be considered in management plans for urban air pollution and UHI.

To the east of Toronto lies the Lynde Shores Conservation Area which "provides excellent habitat for nesting birds and acts as an important stopover point for waterfowl and shorebirds migrating along the north shore of Lake Ontario" (Conservation Ontario, 2015). Again this represents an ecosystem that is sensitive to the increasing nearby UHI and air pollution levels. In order to decide on a management plan the needs of conservation areas such as this need to be taken into account.

Figure 10 Distribution of conservation sites surrounding Toronto (Conservation Ontario, 2015)

Overall there are clear impacts to surrounding ecosystem and natural elements in regards to management decisions made by the city of Toronto. Figure 8 shows the many conservation sites surrounding the city. In this regard it is clear that relevant environmental stakeholders beyond the limits of the city of Toronto need to be considered in management decisions.

Conclusion

The effects that the UHI and air pollution stemming from the city of Toronto have been illustrated. Drawing from this it is evident that there are some effects that the UHI has exacerbating urban air pollution that then harms surrounding ecosystems. This means that there is a need for their proper representation as a stakeholder in management decisions. As the city of

Toronto continues to grow proper mitigation measures need to be taken to address the growing problems of UHI and air pollution.

Now that it has been identified that there is also potential to reduce the impacts of the UHI and urban air pollution on surrounding ecosystems action needs to be taken. Future management decisions should include representation from environmental stakeholders due to the fragility of surrounding ecosystems and the existing land usage patterns in the ecoregion. Specific considerations in future management decisions would be the fragility of surrounding ecosystems, migratory patterns of animals, and highly fragmented habitats.

Further Work

It is recommended that further work is carried out to examine the implications of the UHI and urban air pollution from Toronto across spatial and temporal scales for environmental stakeholders. Starting from the most fragile ecosystems most vulnerable to change based on the proximity to the city would be a good place to start.

A further recommendation would be looking at the carbon sequestration capacity of green roofs and any applications that they may have in the emerging carbon economy that is expected within Ontario in the next 5 – 10 years.

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Authors Note

This paper was inspired by the need to look at new inputs into decision making that reflect the current state of the world. Namely the need to start including non-human factors in decision making frameworks. Often these non-human factors have effects on human life and are a few steps removed from a direct relationship which can cause decision makers to avoid incorporating them in the process. Further, non-human factors should be considered for the intrinsic value they have beyond their use to humans. In Toronto, Ontario the management framework that decides what policies are put through and which aren't are primarily guided by economic and social factors. This paper serves to show the true interrelationships between the human factors of economy, environment and social well-being that is ignored in the current decision making framework. Further, it is shown that there is a value in including surrounding natural areas in the decision making framework especially when these areas can be impacted by these decisions.