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The Potential of Edible Crops in Floating Treatment Wetlands

Gilder, William

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Author: William Gilder
Supervisor: Dr. Christian Dunn
Bangor University
bsu74a@bangor.ac.uk
Student ID: 500261036

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Abstract

Floating Treatment Wetlands provide a space efficient, economically viable alternative to traditional Constructed Treatment Wetlands for wastewater treatment and water purification. Current Floating Treatment Wetland systems are planted with native wetland plants adapted to the conditions in wetlands. These native wetland plants provide wastewater remediation and water quality improvements, but the plants themselves require disposal off-site and provide only limited useable products. The efficiency of Floating Treatment Wetland systems is greatly reduced over winter, due to low temperatures and microbiological activity. In this experiment we show the possibility of growing 'Little Gem' lettuce in Floating Treatment Wetland systems instead of native wetland plants, with similar and sometimes better water treatment properties than *Phragmites australis* planted systems. The most efficient batch (26/04/2019) reduced phosphate loading by 82-89% after 7 days in lettuce planted systems, while *Phragmites* systems reduced phosphate loading by 47-56% after 7 days which represents a significant difference ($p<0.05$). *Phragmites* systems were slow to establish and provide water quality treatment properties. We propose the testing of hybrid systems with intercropping, a system where both plants are grown in one growing season, in order to maximise treatment efficiency and produce useful agricultural products for consumption either by humans or animals.

Introduction

Pollution of freshwater and coastal waters has been an emerging problem in developed and developing countries since the industrial revolution. This pollution originates in two forms, point source pollution and nonpoint source pollution. Point source pollution is defined by the EPA as “any single identifiable source of pollution from which pollutants are discharged, such as a pipe, ditch, ship or factory smokestack” (US Department of Commerce, 2009). Nonpoint source pollution is harder to define but is usually categorised as non-discrete sourced pollution which occurs as a result of water runoff from various sources (Zaring, 1996).

The problem of point source pollution had largely been eliminated in most developed countries by the end of the 20th century, due to the introduction of legislation. In America, the first federal law aimed at combating a rising pollution problem caused by post World War 2 industrialisation was The Federal Water Pollution Control (FWPC) Act of 1948. The FWPC Act stated that the states have the primary responsibilities and rights in water pollution control, and for federal authority to seek judicial orders for the abatement of water pollution in interstate waters. However, this act proved extremely ineffective in reducing pollution, and despite 5 amendments the law never had any substantial effect on lowering pollution (Barry, 1970). Pollution in some American rivers got so bad in these years that rivers in industrial areas routinely caught fire due to the amounts of oil that was discharged into them. A fire in the Cuyahoga River in Cleveland in 1969 caused by oil pollution prompted a review of the FWPC and it received a comprehensive amendment of the law in 1972.

The Clean Water Act (CWA) of 1972 made it unlawful to discharge any pollutants from point sources into navigable water without a permit, to be supplied by the Environmental Protection Agency (EPA). States were federally mandated to carry out water quality standards sampling, in order to meet water quality standards, set by the EPA. The CWA also put in place grants for construction and research of treatment works, so that point sources of pollution could first be treated before being released into waterways (Federal Water Pollution Control Act Amendments of 1972, 1972)

The CWA has undergone several more amendments since its introduction, most notably in 1977, 1987 and 2015 (US EPA, 2021).

In the EU similar legislation at improving the environmental condition of waterways and waterbodies was introduced in 1976 with the creation of the Bathing Water Directive (BWD) and the Waste Framework Directive (WFD). The BWD was more focused on improving water quality to make it safe and more pleasant for human use, with a strong focus on microbiological contaminants that could be dangerous to humans, such as faecal streptococci and Salmonella. The BWD only primarily focused on the protection of public health in public waters; it sought to improve the health of the environment to improve human health, rather than improving the environment for the benefit of the wildlife (The Council of European Communities, 1976). The BWD was updated in 2006 to accommodate advances in science which allowed for better monitoring, and with updated guidelines for pollutant levels (European Union, 2006).

Further EU legislation was enacted in 1991 with the Urban Wastewater Treatment Directive (UWTD), which was to monitor and protect the water quality of rivers and seas not designated as bathing waters by the EU. A part of the European Water framework directive was the Nitrates Directive, which focuses on the development and implementation of best management practices at helping farmers mitigate nitrate run-off from fields into waterways. The Directive states this is done “in order to protect human health and living resources and aquatic ecosystems and to safeguard other legitimate uses of water.”

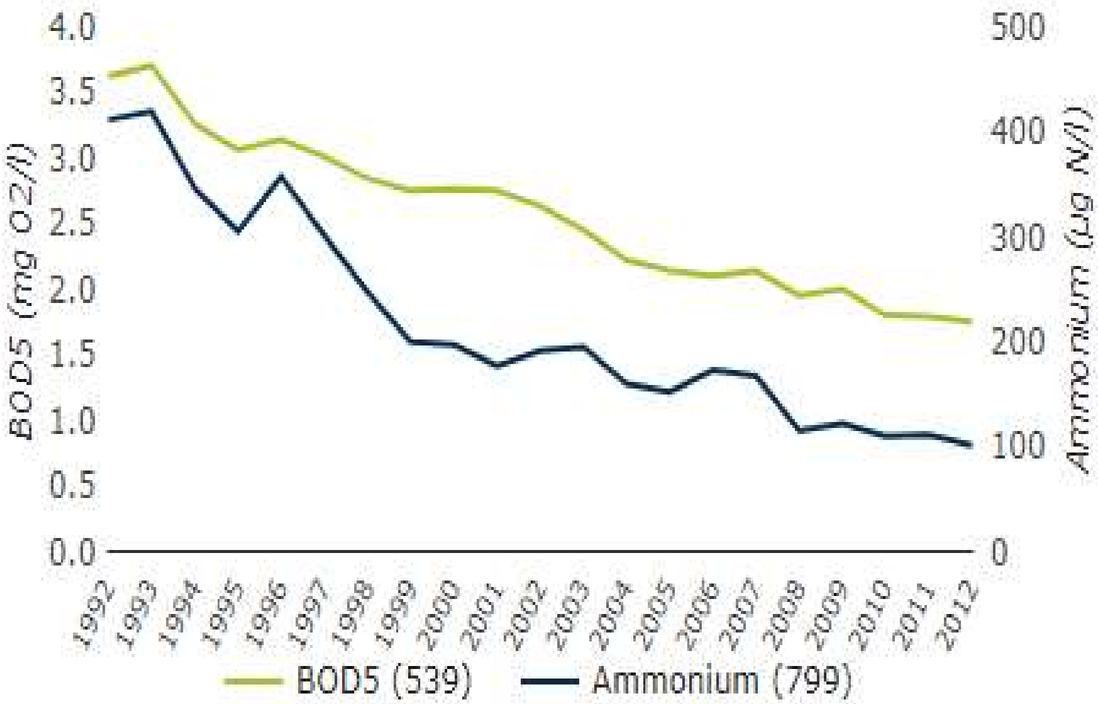

Figure 1: ("Changes in water quality variables during the last two decades — European Environment Agency," 2015).

Figure 1 shows the success of the UWTD, enacted in 1991, with a steady downward trend in the concentration of ammonium found in European rivers over the following decade, and a consistent decrease in Biochemical Oxygen Demand (BOD). BOD is an indicator of the organic pollution in rivers, and a lower value means a lower amount of readily biodegradable fraction of the organic load in water (Jouanneau et al., 2014).

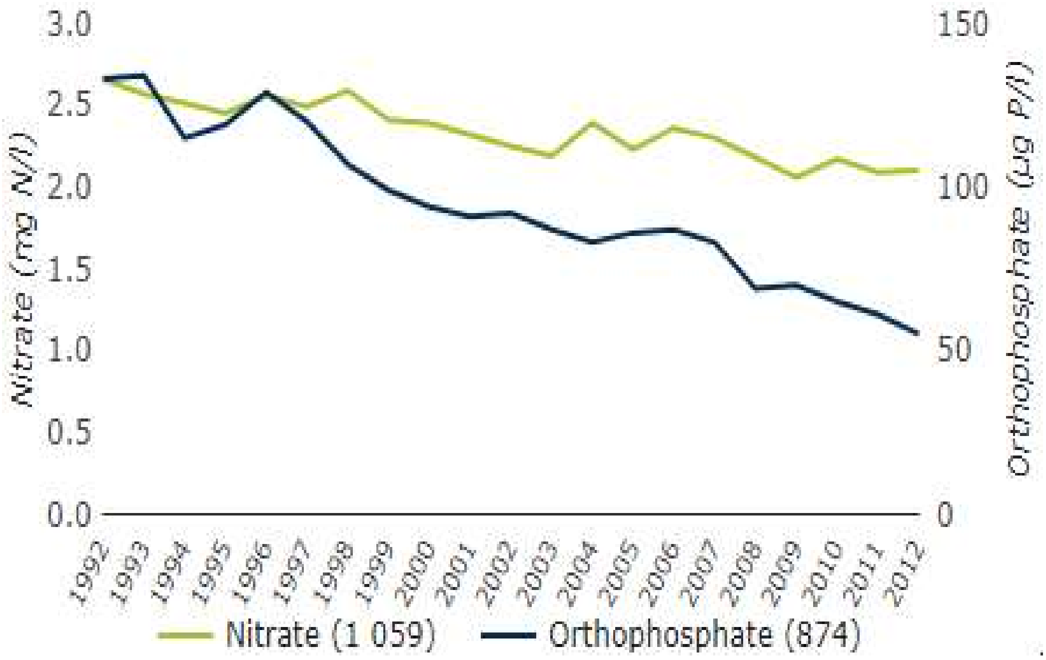

Figure 2: ("Changes in water quality variables during the last two decades — European Environment Agency," 2015).

Figure 2 shows the limited success of the European Nitrate Directive, also implemented in 1991. While dissolved orthophosphate loading in European rivers has over halved in a decade (133µg/L in 1992, 55µg/L in 2012) nitrate loading in the same rivers has only reduced by around 20% (2.66mg/L in 1992, 2.1mg/L in 2012).

This is due to the nonpoint solution being harder to identify and that nitrate addition to soil is required for agriculture. If nitrates were removed from fertilizer, a decline in production would be a result. Point source pollution is often a single large input from a specific source, usually a factory or sewage pipe. Point source pollutions can be reduced by several means, such as shutting down the source factory or introducing legislation that requires the factory treat the water before release into the environment(Cabe and Herriges, 1992). The solutions are less effective against nonpoint source pollution as there are so many different contributing sources.

The problem of nonpoint source pollution has been more pervasive in freshwater and coastal saltwater systems. Nonpoint source pollution is much more difficult to reduce, even with legislation and cooperation from industry and agriculture. The nitrate directive discussed above also in theory reduces the nitrate pollution from nonpoint sources; however it has only had limited success (Ward et al., 2018). The nitrate directive introduces the legal designation of Nitrate Vulnerable Zones, which are designed to limit nitrogen input in environments which are *"being at risk from agricultural nitrate pollution"* (Gov UK, 2018).

NVZs exist in different forms across Europe, but they all have the same aim of reducing nitrate inputs in the environment to lower the nitrate levels in groundwater to below 50mg/L. Measuring the effectiveness of nitrate input reduction strategies on nitrate concentration in groundwater is difficult, due to the delay in results. It takes years, sometimes decades, for differences in input to have an impact on groundwater levels, due to the time it takes for natural filtration of water through the water table. This means that while NVZs were first implemented in 1991, their actual effect may take decades to be seen due to this "time lag" (Vero et al., 2018).

Despite the introduction of NVZs in different forms across the EU in 1991, the status of nitrates in groundwater has failed to improve in many European countries. This is partially due to the lack of enforcement of the Nitrate Vulnerable Zones (NVZ) in European countries. Examples in France show nitrate pollution of groundwater is still a problem in an estimated 64% of French tap water. The problem of France's non-enforcement of NVZs was so bad that the European commission initiated litigation against France for failing to implement sufficient nitrate reduction strategies in NVZs (Richard et al., 2018). A problem also persists of farmers being unconvinced about the effectiveness of the nitrate directive in balancing farm productivity with environmental benefits.

(Barnes et al., 2009) investigated farmer compliance and attitudes towards NVZs in Scotland, which has four NVZ regions, covering around 14% of the farmland of Scotland impacting over 12,000 farmers. The report showed an overwhelmingly negative response from farmers towards NVZs, based on the scientific basis for designations, and the inflexibility of farming practices carried out with NVZs.

The wide opposition and failure to implement satisfactory NVZs since their introduction in 1991 has led to England failing to meet the 50mg/L nitrate concentration in around 44% of the land they are implemented in. This is due to rivers with above 50mg/L loading of nitrate; a further 25% of land is under NVZ designation due to groundwater having loading above 50mg/L of nitrates.

This is a problem for public health in England as groundwater is often a source of drinking water (EA, 2019).

While the European Water Framework directive has been successful in stopping degradation of many rivers, in the UK it has failed to improve the quality of many surface waters. In a report in 2015 it was reported that 79% of waters in England were bad to acceptable in ecological status or potential, with only 21% reaching surface water ecological status of "good", while it was predicted it would be 30%.

The same report detailed the pressures preventing waters from reaching good ecological status, with the largest single pressure being phosphate, with a total of 6091 out of 13,911 water bodies failing due to phosphate pressures. Only 121 water bodies failed due to nitrate (Environment Agency, 2015).

The negative effects of nutrient runoff from agricultural fields

Nutrient runoff from agricultural fields causes many negative effects on the environment, the most significant of which is a process called eutrophication. This process has been known about for some time, with the first positive correlation between nutrient input and aquatic productivity being observed in (Weber, 1907). While increasing productivity may sound like a desirable trait for processes in agricultural systems, it is not a positive factor in natural systems.

The two forms of freshwater lake systems are oligotrophic (nutrient-poor) and eutrophic (nutrient-rich). Oligotrophic water systems are characterised by nutrient deprived, deep, clear water with a low level of biological productivity. Eutrophic lakes are characterised by nutrient-rich, shallow water with higher levels of biological productivity, causing the water to be murkier and have less clarity. Due to the shallower nature of eutrophic lakes, the water in these systems is warmer than oligotrophic systems (Smith et al., 2006). This influences the composition of both the micro and macro life of the systems, resulting in a lower total biodiversity in freshwater lake basins due to the dominance of eutrophic systems over oligotrophic. This transition from oligotrophic to eutrophic conditions means the species which rely on oligotrophic conditions fail to thrive once conditions become eutrophic.

These two systems were traditionally thought to have been linked in many geographic situations, with a natural progression in lake basins from oligotrophic to eutrophic as nutrients naturally enter the system from water flow and photosynthesis. This theory was disproven by Engstrom and Fritz (2006) which showed that without human nutrient input, isolated glacial basin lakes grew to be more oligotrophic over time rather than more eutrophic. The driving factor behind this progression was the natural lowering of nitrogen levels in the system over time. With human nutrient input, however, this natural progression does not occur, and the system becomes eutrophic over time as natural cycles are disrupted.

This disruption of natural cycles has shifted the global ratio of eutrophic to oligotrophic natural freshwater basins to extremely favour the former. A result of this is the loss of the idyllic, often sought after "aesthetic" of the tranquil, clear water oligotrophic lakes. These systems are becoming eutrophic and losing the desirable qualities of high water clarity, low water turbidity and low nutrient levels (Goldman, 1988). This loss of water clarity has negative effects on tourism and the economic benefit of water bodies (See Appendices for further information.)

The loss of oligotrophic water systems is not only of negative consequences to the tourism industry, but also to the larger ecosystem. Oligotrophic systems are home to specific species which are adapted to the clear water and low nutrient levels; filter feeders such as freshwater pearl mussels, which rely on extremely clean water, struggle to survive in the turbid, high-nutrient waters of eutrophic systems (Bauer, 1988). It has been reported that reedbeds of *Phragmites australis* severely contract when agriculture is started in nearby fields and nutrient-heavy runoff from agricultural fields made the water becomes more eutrophic (Boar et al., 1989).

The transformation from oligotrophic to eutrophic results in a decrease in total plant biomass but an increase in floating plant biomass. A result of this is more shading and a lower phytoplankton concentration in the water column due to the additional shading that occurs (Feuchtmayr et al., 2009). This lower level of phytoplankton negatively effects the microbial life, which has a knock-on effect on the entire food web. Any animal life reliant on filter feeding, or predating on species which do filter feed, is negatively affected by this change. Predating Yellow Perch in Lake Erie's western basin showed severe stunting in growth after severe hypereutrophic conditions prevailed in the western basin, compared to only mild growth stunting to predating Yellow Perch in the central basin which had only experienced mild eutrophication (Hayward and Margraf, 2011).

The process of eutrophication not only affects lakes but other freshwater systems such as rivers and coastal saltwater. Over the last century organic pollution of coastal waters has become a serious problem worldwide, leading to widespread hypoxia, anoxia, habitat degradation, alteration of food-web structure, loss of biodiversity, increased frequency and duration of harmful algal blooms (Howarth, 2008). One of the major stresses comes from the input of excessive nitrogen and phosphorus which leads to eutrophication. The effects of eutrophication in coastal waters are most easily observed in benthic communities, along with a comparison between benthic (sea bed) primary productivity and pelagic (water column) primary productivity (Smith et al., 2006).

Eutrophication causes a shift from benthic primary production to pelagic primary productivity (Goldman, 1988). The increased nutrients in the water increase the viability for growth of phytoplankton in the water column. Large amount of algal growth results in the water losing clarity and blocking sunlight from reaching the lower depths, lowering production on the seabed. This leads to a decline in benthic algae and biofilm, which in natural coastal waters is the dominant form of primary production (Howarth, 2008).

Harmful Algal Blooms

Algal blooms are naturally occurring events in the ocean and freshwater bodies, usually during summer when there is a large amount of sunlight. They are more common in coastal waters than in open ocean, due to the nutrient influx from rivers releasing nutrient-rich water into coastal waters(Anderson et al., 2008). The nutrient loading in this water has been steadily increasing since the green revolution and the increase of fertilizer application on agricultural fields; nitrogen and phosphorus are the most important nutrients in the rate of algal growth (Anderson et al., 2002). As the loading of nitrogen and phosphorus increased over the century due to human population density increase, intensification of farming and fertilizer usage increase, the occurrence and severity of algal blooms increased. As the intensity of algal blooms increases their damage to the environment also increases until they cause substantial ecological growth and are categorised as Harmful Algal Blooms (HABs).

HABs can be devastating to coastal economies, due to a variety of factors Algal growth consumes oxygen due to the process of respiration, which means where excessive algal growth occurs water becomes anoxic (Watson et al., 2016). Other problems that HABs cause is that of toxin production, and causing water to be deadly to marine life, as well as humans that eat marine life which have lived in or near HABs. On the U.S. west coast, the main toxin-producing algal species are dinoflagellates that cause paralytic shellfish poisoning (PSP) and diatoms that produce domoic acid and cause domoic acid poisoning (DAP). Other harmful diatoms kill fish at aquaculture farms but are not harmful to humans directly(Horner et al., 1997).

Perhaps most famous and visually alarming, HABs are the so called "red tide" blooms caused by Dinoflagellates and similar organisms, which produce an array of natural toxins called *brevetoxins* which turn coastal waters a bright red and are toxic to all marine life. This "red tide" is dangerous to humans, even if they do not go into the water directly, as aerosolized *brevetoxins* can cause respiratory irritation, as well as other adverse health effects (Kirkpatrick et al., 2004).

HABs are not just limited to saltwater, as they can occur in freshwater systems too. It is estimated that Freshwater Harmful Algal Blooms (FHAB) cost the United States between 2.2 and 4.6 billion dollars annually. These costs were incurred in recreational water usage, waterfront real estate, spending on recovery of endangered species, and drinking water (Dodds et al., 2009). Further economic damage is caused by FHABs in the tourism sector. If a particularly popular freshwater lake suffers from a severe FHAB it may kill many of the fish which will discourage fishing tourism (Lopez et al., 2008). Other concerns from FHAB are the aesthetic problems they cause for the tourism industry, as tourists want lakes with clear, clean water. If FHABs are strong enough they can cause strong discolouration of water, turning clear water into a thick green soup-like sludge (Kirkpatrick, 2003).

Algal blooms can also have a more direct effect on human health and economics. In recent years there has been a rise in the occurrence of harmful algal blooms in reservoirs, especially in America where environmental regulations are less restrictive and there is a large amount of high-intensity sunlight. In August 2014 toxic algal blooms caused Toledo, Ohio to shut down their municipal water supply due to the water being toxic to humans (Mitsch, 2017).

A solution water companies are developing to prevent algal growth in their reservoirs is to introduce a layer of black plastic "shade balls" to cover the surface of the water. The theory behind this is that the layer of balls would shade the water and prevent the sunlight from promoting algal growth in the reservoir. The layer of shade balls also had other positive effects, including lowering the temperature of the water in the reservoir due to preventing sunlight from directly shining on the water and heating it up. This effect further reduced algal and other microbial growth (Haghighi et al., 2018).

This solution to HABs suggests that along with a dual-nutrient reduction strategy, the reduction of water surface area exposed to direct sunlight also helps to lower the rate of occurrence of HABs.

A study published in 2017 aimed at replicating this shading effect with the use of floating treatment wetlands, with the rafts replicating the shading effect of the balls used in reservoirs. It showed the rafts have a similar effect on reducing algal growth, lowering chlorophyll-*a* concentrations by around 80%. The rafts in the systems used in these experiments were planted with *Phragmites australis*. However, the study also showed there might be issues with the production of dissolved organic matter leaching from the treatment wetlands (Jones et al., 2017).

The existing role of Floating Treatment Wetlands in pollution control

The use of floating treatment wetlands (FTWs) as augmentation to existing water treatment systems has been becoming more common in recent years. FTWs are comprised of a buoyant floating mat planted with emergent macrophytes, whose roots extrude downwards into the water source. Water settlement ponds retrofitted with floating treatment wetlands have been shown to increase sedimentation rates and removal of nutrients, including the important nutrients of nitrogen and phosphorus (Tanner et al., 2011; Ryan J. Winston et al., 2013).

These floating treatment systems offer many benefits over full constructed treatment wetlands (CTWs) and provide a cheap, viable alternative in situations where CTWs are not available due to local limitations, most commonly lack of available land space and high occurrence of flooding. The primary advantage of FTWs over CTWs is their ability to operate under non-standard waterflow conditions. CTWs fail to function under high waterflow conditions and can become exporters of nitrate due to flooding (Spieles and Mitsch, 1999).

FTWs could augment existing water treatment systems and be relocated if requirements for nutrient removal change. They provide a viable option for retrofitting existing ponds due to a variety of factors. No heavy machinery is required for their installation due to no heavy earth moving being required; no additional land needs to be dedicated to treatment due to the mats being placed on existing settlement ponds; they do not detract from the legislated requirement for storage volume for wet ponds (Ryan J Winston et al., 2013).

Under similar environmental conditions FTWs have been shown to have comparable treatment efficiencies to CTWs under normal nutrient loading levels (Tanner et al., 2011).

In addition to serving as effective tools for removal of excess nutrients, floating islands are useful for providing wildlife habitat, reducing biogas emissions, and improving the visual appearance of treatment areas (Stewart et al., 2008).

Vegetated Floating Rafts for water quality improvement

The process of water remediation by FTWs is complex and still under investigation; current theories are that it is a combination of many different pathways and processes. Several of these processes are understood, but many are not; the effectiveness and efficiency of each pathway is also only vaguely defined, with no fixed value given to each.

It has been previously commented on by (Van de Moortel et al., 2010) that direct comparison between studies as to the removal efficiencies of floating raft systems is difficult, due to the multitude of complicating independent variables between systems. This has led to an extreme variation of results when it comes to reported treatment efficiencies.

Plant growth requires nutrients, most importantly nitrogen as it is needed for protein synthesis. Phosphorus is another key nutrient in plant growth as it is needed for DNA synthesis (Paerl, 2009). This means that a correlation should occur between plant mass and nutrient removal from the water; it has been reported there is substantially higher N removal in the presence of FTWs than could be accounted for by plant uptake alone (Matheson et al., 2010).

This extra nutrient removal is often attributed to the plant root systems which form under the rafts. These systems provide extensive attachment surfaces for microbial biofilms, assimilating nutrients from the water column, releasing bioactive exudates, and modifying environmental conditions beneath the mat (Tanner and Headley, 2011).

It was suggested by Wang and Sample (2014) that the primary nutrient removal system is that of organic matter decomposition. (Hart et al., 2003) somewhat supports this statement, indicating that a combination of microbes and actively growing macrophytes provides the best conditions for removing ammonium in FTW systems.

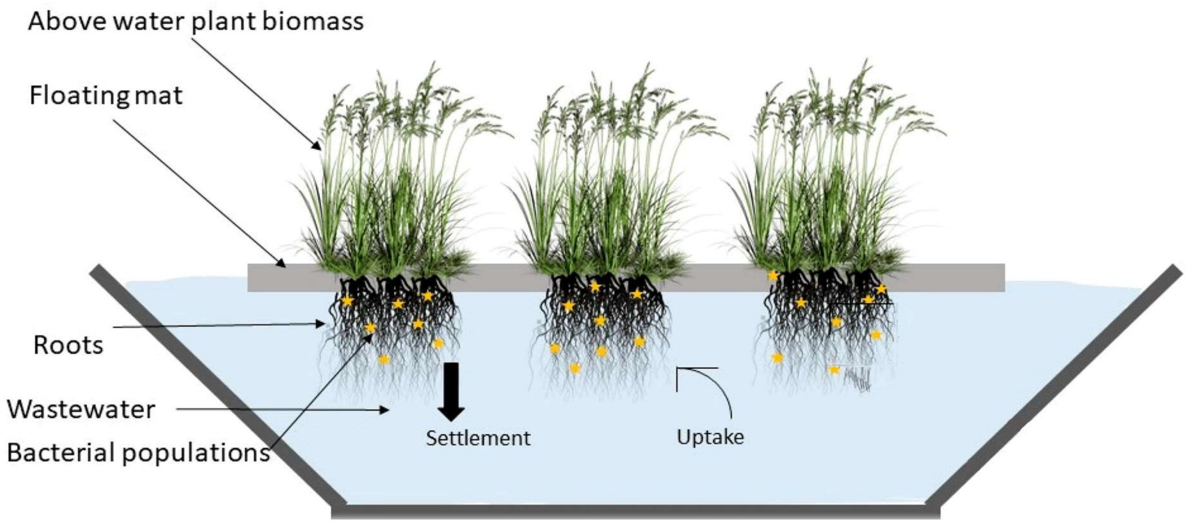

Figure 3: Overview of a standard Floating Treatment Wetland system Adapted from (Wei et al., 2020)

Nutrient removal

In laboratory and field tests FTWs have been shown to remove a wide variety of nutrients; this is due to their multi-faceted and multi-pathway treatment methods. Experiments have been carried out to test the treatment viability of many different pollutants using FTWs.

Table 2: Examples of studies showing the pollutant removal potential of FTWs.

Pollutant	**Treatment Pathway**	**Treatment Efficiency**	**Reference**
Nitrogen	Phytoremediation	90.4% removal	(Xian et al., 2010)
Nitrogen	Phytoremediation, Plant Growth	58-85% average removal over two years	(White and Cousins, 2013)
Phosphorus	Phytoremediation, Plant Growth	45.5-75% average removal over two years	(White and Cousins, 2013)
Phosphate	Sedimentation	83.4% removal	(Xian et al., 2010)
Chemical Oxygen Demand	Oxidation by plant root systems	83.4% removal	(Xian et al., 2010)
Sulfadiazine	Abiotic and Biotic Transformation Phytoremediation	99.2% removal after 15 days	(Xian et al., 2010)
Sulfamethazine	Abiotic and Biotic Transformation Phytoremediation	91.% removal after 15 days	(Xian et al., 2010)
Sulfamethoxazole	Abiotic and Biotic Transformation Phytoremediation	99.5% removal after 15 days	(Xian et al., 2010)
Cadmium	Sedimentation, organic matter production	5% average removal over 9 years	(Gill et al., 2017)
Copper	Sedimentation, organic matter production	60% average removal over 9 years	(Gill et al., 2017)
Copper	Sedimentation, Phytoremediation	3.8-6.4mgm^{-2}	(Tanner and Headley, 2008)
Lead	Sedimentation, organic matter production	31% average removal over 9 years	(Gill et al., 2017)
Zinc	Sedimentation, organic matter production	86% average removal over 9 years	(Gill et al., 2017)
Cadmium	Phytoremediation	88% over 60 days	(Wang et al., 2021)
Copper	Phytoremediation	56% removal over 15 days	(Wang et al., 2021)
Zinc	Phytoremediation	89% removal over 60 days	(Wang et al., 2021)

Table 2 details previous studies on FTWs and their ability to remove a wide array of pollutants. Most commonly they are used to treat runoff wastewater (White and Cousins, 2013), or contaminated floodwaters which are high in fine particulates such as copper and zinc (Headley and Tanner, 2006; Tanner and Headley, 2008). They have also shown success in removal of sulfonamide antimicrobials, which are common antibiotics (Xian et al., 2010). The impact of pharmaceutical waste on the environment is a relatively under reported topic, but it is an emerging pollutant which governments must monitor closely in the future. Regulation should be implemented if it emerges that it is damaging to the environment. The accumulation of excess sulfonamide antimicrobials in the environment can cause damage to common wetlands plants like *Phragmite australis* which can lead to reduced root activity and reduced production of leaf chlorophyll (Liu et al., 2013).

Table 3: Overview of the different types of polluted water that have been treated by FTWs.

Input Origin	Removal amount	Planted Fauna	Reference
Agricultural Runoff	90% of Phosphorus 84% of Nitrogen	*Pontederia cordata* *Juncus effusus*	(Jonathan T Spangler et al., 2019)
Stormwater Runoff	80% of Total Suspended Solids 17% of Nitrogen 53% of Phosphate	*Carex appressa*	(Nichols et al., 2016)
Domestic Wastewater	50 to 60% reduction in Nitrogen, Ammonia and Phosphate	*Juncus Effusus* *Scirpus Validus* *Typha Latifolia*	(Coleman et al., 2001)
Swine Wastewater	43% of Nitrogen, 35% of Phosphate	*Typha Latifolia*	(Hubbard et al., 2004)
Swine Wastewater	52% of Nitrogen 41% of Phosphate	*Panicum hematomon*	(Hubbard et al., 2004)
Simulated Urban Storm water Runoff	86.29% removal of Total Phosphorus. 67.0% removal of Total Nitrogen.	*Canna indica* *Thalia dealbata* *Lythrum salicaria*	(Ge et al., 2016)

Shown in Table 3 are some samples from the existing literature of studies in a similar field, focused on treatment of runoff or wastewater. They have a focus on pollutants that cause eutrophication like nitrogen and phosphate.

Plant selection for use in Floating Treatment Wetlands

During their establishment FTWs are planted with emergent macrophytes to establish root complexes and secure the bedding sediment used in the floating mat. Currently, the most commonly used plants in FTWs are locally sourced, indigenous wetland plants. In the Americas the preferred plants vary due to the large geographical size of the continent, with species preference varying by location. Table 4 details briefly the species of plants that have been used in previous studies; the most common are *Typha latifolia* (Cattail) and *Pontederia cordata* (Pickerelweed). In Europe the *Phragmites australis* (Common Reed) is the most widely used native wetland plant in FTW systems, but many other species are also used, including *Typha* species (Headley and Tanner, 2006). However, the dominant deciding factor should be the suitability of a plant species to the specific environmental conditions that the FTWs will be placed in. It is not acceptable to introduce invasive species into an environment for the aim of water quality improvement.

Table 4: Examples of the species of plants which have been used in published studies of FTWs.

Plant Species	Mean Nitrogen Removed (%)	Phosphorus Removed (%)	Reference
Agrostis alba	41.5	29.8	(Jonathan T Spangler et al., 2019)
Canna x generalis	43.7	26.1	(Jonathan T Spangler et al., 2019)
Carex strica	38.9	28.3	(Jonathan T Spangler et al., 2019)
Iris ensata	50.4	48.6	(Jonathan T Spangler et al., 2019)
Panicum virgatum	82.4	64.7	(Jonathan T Spangler et al., 2019)
Iris pseudacorus	98	92	(Keizer-Vlek et al., 2014)
Typha angustifolia	57	23	(Keizer-Vlek et al., 2014)
Juncus effusus	40	48	(Lynch et al., 2015)
Pontederia cordata	18.2	8.2	(Wang and Sample, 2014)
Canna flaccida	58	45.5	(White and Cousins, 2013)
Phragmites australis	91.5	Not Measured	(Li and Guo, 2017)

Plant selection is also affected by the type of wastewater which the FTWs are placed on to treat. Some plants are more suited to different type of wastewater.

There is evidence that floating mats are effective at lowering nutrient loading levels in piggery effluent. While *Typha latifolia* (Cattail) and *Panicum hematomon Schult* (Maidencane) colonised floating rafts and thrived, a common wetland plant, *Juncus effuses* (Soft Rush), failed to thrive. This was attributed to the low dissolved oxygen levels in the piggery effluent; soft rush naturally grows in poorly drained soil, rather than fully submerged like the other two wetland species (Hubbard et al., 2004). This problem could be resolved by artificial aeration, such as by solar powered aerators. Aeration would increase the dissolved oxygen content, potentially making it a viable environment for soft rush to establish and thrive (Chang et al., 2014).

Phragmites australis was chosen for our experiment due to its local availability and previous use in traditional wetland systems. Its use in FTW systems is somewhat limited, with reports of it being unable to survive in hydroponic conditions (Barco and Borin, 2017). However, we had no issue establishing it in hydroponic systems and it thrived well in our growth systems. Li and Guo (2017) also reported very promising results in their study, which showed that *Phragmites australis* provided effective treatment of simulated eutrophic water in cold climates which suited the conditions for our experiment.

Plant Selection for Food Production for Human Consumption

Given that FTWs have been shown to be effective remediators of wastewater when planted with wetland plants, due to the plant growth and microbial communities of the root systems submerged in water, it is of interest whether the FTW systems have the same effectiveness when planted with non-wetland plants. The interest in growing non-wetland plants in FTW systems is for several reasons, primarily that while adapted to wetland environments, wetland plants have very little economic value. If it was possible to substitute the wetland plants in the system for plants with agricultural value, such as edible food crops or fodder, FTWs could be of economic benefit as well as an ecological one.

Floating raft systems have been used to cultivate food in many civilisations that have needed to farm on marginal farmland; the ancient Aztecs reclaimed flooded marginal farmland with the use of floating rafts to grow crops called Chinampas (Coe, 1964). The existing literature of the growth of plants in floating treatment wetlands uses systems planted with native wetland plants (Headley and Tanner, 2006). These plants have limited agricultural or economic value, as they cannot be used as food for people or fodder for animals. They also require periodic harvesting to remove the foliage from the floating raft systems. If foliage is not removed the decaying plant matter will return the nutrients absorbed by the plant to the water, thus reverting any prior nutrient removal through plant matter growth.

Native wetland plants are adapted for life in wetlands, the primary factor being the anoxic conditions of the submerged soil. This leads to low levels of dissolved oxygen which agricultural crops are not adapted to, while adapted native wetland crops are (Barclay and Crawford, 1982).

Floating Treatment Wetlands are essentially large outdoor hydroponic deep water culture systems. When looking for ideal plants to be grown in FTWs the best available options are those which are known to grow well in hydroponic systems. One of the most commonly and successfully grown crops in hydroponics systems is lettuce, with a long history of cultivation going back to at least the 1970's in America. There have also been many studies into the productivity and viability of lettuce in hydroponic, aquaponic and aeroponic cultivation (Li et al., 2018).

Method

An experiment was carried out to determine if agricultural plants, specifically Little Gem lettuce, had the same beneficial effects on water being treated by FTWs as the native wetland plant, P*hragmites australis*. This plant was chosen due to its widespread use in traditional CTW systems, as well as its proven viability and efficiency in FTW systems as shown in Figure 6. Past experiments have shown that FTWs offer a comparable performance to traditional CTWs when planted with native wetland plants. A series of experiments was performed to determine if comparable treatment performance is observed when the native wetland plants are replaced by non-wetland species.

Our hypothesis is that FTW systems planted with lettuce show an equal-to, or non-significant ($p<0.05$) decrease in water treatment quality when compared with FTW systems planted with traditional wetland plants. Water treatment quality will be measured in overall conductivity as proxy for total ion concentration, with a focus on the key nutrients of phosphate and nitrate.

A secondary hypothesis will be tested; the same FTW systems planted with lettuce will provide the same, or non-significantly less algal growth reduction in the treatment water as FTW systems planted with traditional wetland plants.

Experimental Design

24 hydroponic systems were constructed in a heated greenhouse (average 5 °c above ambient) with 200W LED purple spectrum grow lights with an average coverage of $2m^2$ running between 16:00 and 19:00 each day to simulate a longer growing day. The experiment ran from 01/03/19 until 01/06/19; the location

was Bangor, Gwynedd, Wales; table 5 details the local weather conditions from the nearest weather station (Valley,-4.53524, 53.25238).

Table 5: Weather data at study location for the duration of the study (MetOffice.gov.uk)

	03/2019	**04/2019**	**05/2019**
Mean Daily Maximum Temperature	10.3°C	13.7°C	14.3°C
Mean Daily Minimum Temperature	5.9°C	6.5°C	8.1°C
Total Hours of Sunlight	103.5	138.8	204.5

Each system comprised of a 9L box with external dimensions**:** L395 x W255 x D155mm; internal dimensions**:** L335 x W210 x D140mm.

Floating rafts were created using expanded polystyrene sheets with a thickness of 2.5cm cut to fit flush inside the boxes. Circular holes were cut through the polystyrene sheets with a radius of 4cm, to fit aerator cups with the same radius. They had cut-out sides to allow for the flow of water through the root systems. Aeration was provided by aquarium air pumps, rated at 90 litres per hour of air flow.

Young dormant *Phragmites australis* plants were grown in these systems, with 2 plants per box. The roots of the young plants were washed to remove all soil, then transferred directly into aerator cups filled with expanded clay aggregate pebbles.

Little Gem lettuce seeds were sown in small rockwool cubes and watered with a nutrient-water mix. The mix was Baby-bio houseplant food (NPK: 10.6-1.9-1.4) mixed to label strength; 10ml per litre of water. Once established, the Little Gem cultivars were transplanted in their rockwool cubes to the aeration cups; the aerator cups were partially filled with clay aggregate pebbles to ensure the rockwool cubes were not entirely submerged. Each aeration cup housed one lettuce cultivar with 4 lettuce plants per system. In the *Phragmites australis* systems there were only 2 plants per system; this was because the 9 litre boxes were not large enough for the root systems of 4 fully grown *Phragmites australis* plants to thrive in.

Sampling Scheme

In total 6 test groups were created, with 4 systems in each group (n=4).

1. A control group without floating raft systems
2. A group with floating raft systems with aeration pots filled with clay aggregate pebbles with no emergent macrophyte vegetation planted. No aeration.

3. A group with floating raft systems with aeration pots planted with *Phragmites australis* and no aeration.
4. A group with floating raft systems with aeration pots planted with Little Gem lettuce
5. A group with floating raft systems with aeration pots planted with *Phragmites australis* and aeration.
6. A group with floating raft systems with aeration pots planted with Little Gem lettuce and aeration.

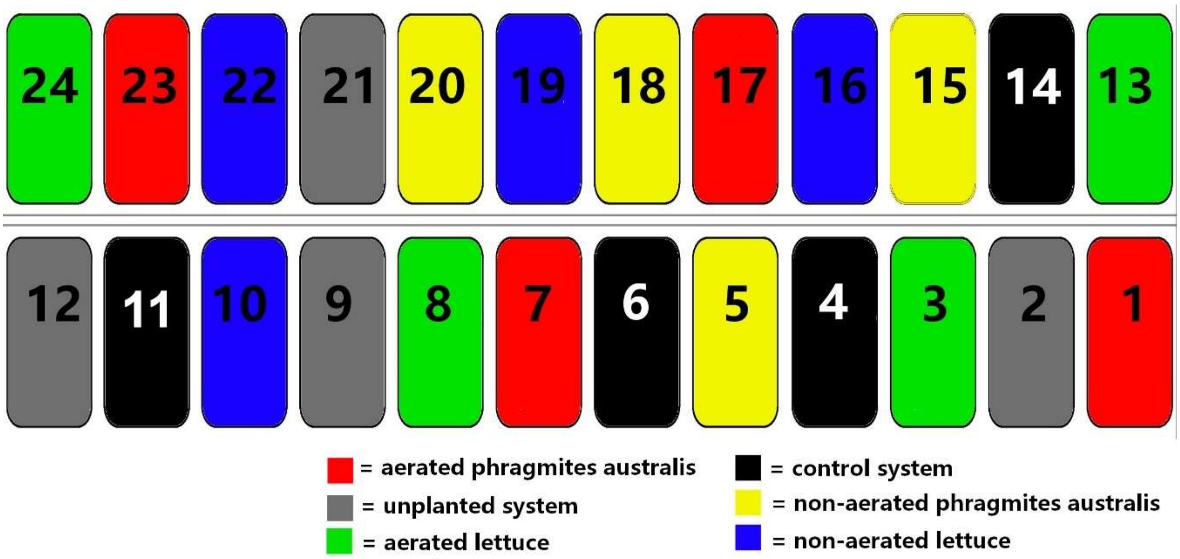

Figure 7: Experimental layout with as much randomisation as possible, to lower the factors of local conditions.

To create simulated stormwater for each run 200L of water was mixed with 20g of *Miracle-Gro All Purpose Soluble Plant Food* with NPK ratio of 24-8-16. This mixture produced simulated stormwater with an average loading of 10.6mg/L of phosphate ions. This loading was based on Hubbard's (2004) study which showed 50/50 diluted swine effluent with average loading of 15mg/L of phosphate gave best viability to plants in FTW systems. The undiluted effluent of 30mg/L proved to be toxic to some plant species. Water was mixed thoroughly to ensure uniformity of nutrients in the solution. Once mixing was finished 6 water samples were taken from the mixing tank as samples for input water.

Nutrient Concentration Sampling

Samples were collected over 2 months after the *Phragmites australis* plants had been given enough time to establish after their winter dormancy and the lettuce cultivars had established. Our test was a batch sample method; 8 litres of water were loaded into each system for each 7-day run. 50ml water samples were taken after initial mixing and on the 3rd, 5th and 7th day after loading. Each system was emptied after 7 days and refilled with fresh simulated wastewater. This was following the literature on batch mesocosm FTW systems, as shown in (Jonathan T Spangler et al., 2019). The study ran over 2 months to establish the effect of seasonal change on the effect of treatment efficiency. Samples were stored in an incubator at 20°C for 3 hours in order to standardize temperature for reading (See method development). Conductivity and pH were measured using a seven-easy meter (Mettler Toledo, Ohio, USA). Samples were filtered through GF/A filter paper (Fisher, Leicestershire, UK) and again through 0.2 μm cellulose acetate filters and the solution stored at 4°C until analysis. Anion and cation concentrations were determined with the use of a 850 IC Anion MCS (Metrohm AG, Herisau, Switzerland); anion concentration was determined with the use of a “metrosep A Supp 5 – 150/4.0” column, and cation concentration with the use of a “metrosep C4 250/4.0” column. Calibration curves were created using samples of known concentration, provided by fluka standards (Fisher, Leicestershire, UK). 5-point calibration curves were created with relative standard deviations of x<2.

Chlorophyll-a Concentration Sampling

For chlorophyll-a sampling 4 test groups were created with 4 replicates in each group (n=4), with these systems created in the same 9L boxes as the nutrient concentration experiment. 8 litres of water were loaded into each box. The same strength simulated wastewater was used as detailed above. The systems were loaded with simulated wastewater on 26/04/2019 and final results taken on 17/05/2019. Before sampling, water was thoroughly mixed to equalise distribution of the chorophyll-a, as in its natural grow distribution it favoured growth on the surface of the water. Samples were taken using Fisher Scientific 50ml tubes. Chlorophyll-*a* concentration was determined following protocol using 90% Acetone (Talling and Driver, 1963). The equation to transform light absorbance into chlorophyll-*a* concentration is presented below.

$$\mathrm{Chlorophylla}\left(\mu \mathrm{gL}^{-1}\right) = 11.9\left(\mathrm{Abs}_{665} - \mathrm{Abs}_{750}\right)\frac{v}{V_p}$$

In this experiment V is the volume filtered in mL (50ml), v is the volume of extract in mL, p is the light wavelength (cm) and 11.9 the specific absorbance coefficient of chlorophyll-*a* in 90% acetone as specified by Talling and Driver.

Plant Mass Sampling

After 42 days of growth the lettuce plants in their aerator cups were removed from the FTW systems. After this duration of growth substantial root systems had developed and much care had to be taken to remove the lettuce plants from the polystyrene rafts without damaging the roots or the rafts. The root systems had grown into and around the rockwool plug that it was planted into. As much rockwool was removed as possible but some remained trapped in the root complexes. The taproot was thoroughly cleaned of rockwool with a stiff brush.

Phragmites weight could not be determined due to the difficulty in removing the reeds from their floating raft systems (Ryan J Winston et al., 2013). Reed systems are also designed to be permanent fixtures, with the root systems being firmly established in the floating mat systems. The lettuce in our systems was Little Gem lettuce which can be harvested after 6 weeks; this means during this experiment one harvest of lettuce was performed on 12/04/19. This is discussed in our results section.

Statistical analysis

For this study the p value determined to be significant was 0.05. Independent factor t-tests will be used for data involving only two groups. Paired Samples t-tests will be used for comparison of individual groups performance over time. ANOVA tests will be performed to detect significant difference between groups, when there is more than one test group. If results are not normally distributed Welch's ANOVA will be used; comparison between groups will be done with Dunnett's T3 post-hoc analysis, as our groups have <50 samples per group. IMB SPSS statistics package will be used to perform this statistical analysis.

Results

Conductivity in the control systems showed minimal change over the 168-hour treatment period, 200µS > 195µS; this corresponded to a non-significant change to conductivity over 168 hours from input levels. *Phragmites*, aerated *Phragmites* and raft systems all reported significant ($P<0.005$) reduction in levels of conductivity compared to controls. They all also reported significant ($P<0.005$) difference compared to input levels.

Both Lettuce samples, non-aerated and aerated, reported a significant (P<0.001) difference in conductivity levels when compared with *Phragmites*, aerated *Phragmites* and raft systems, as well as in comparison to input levels.

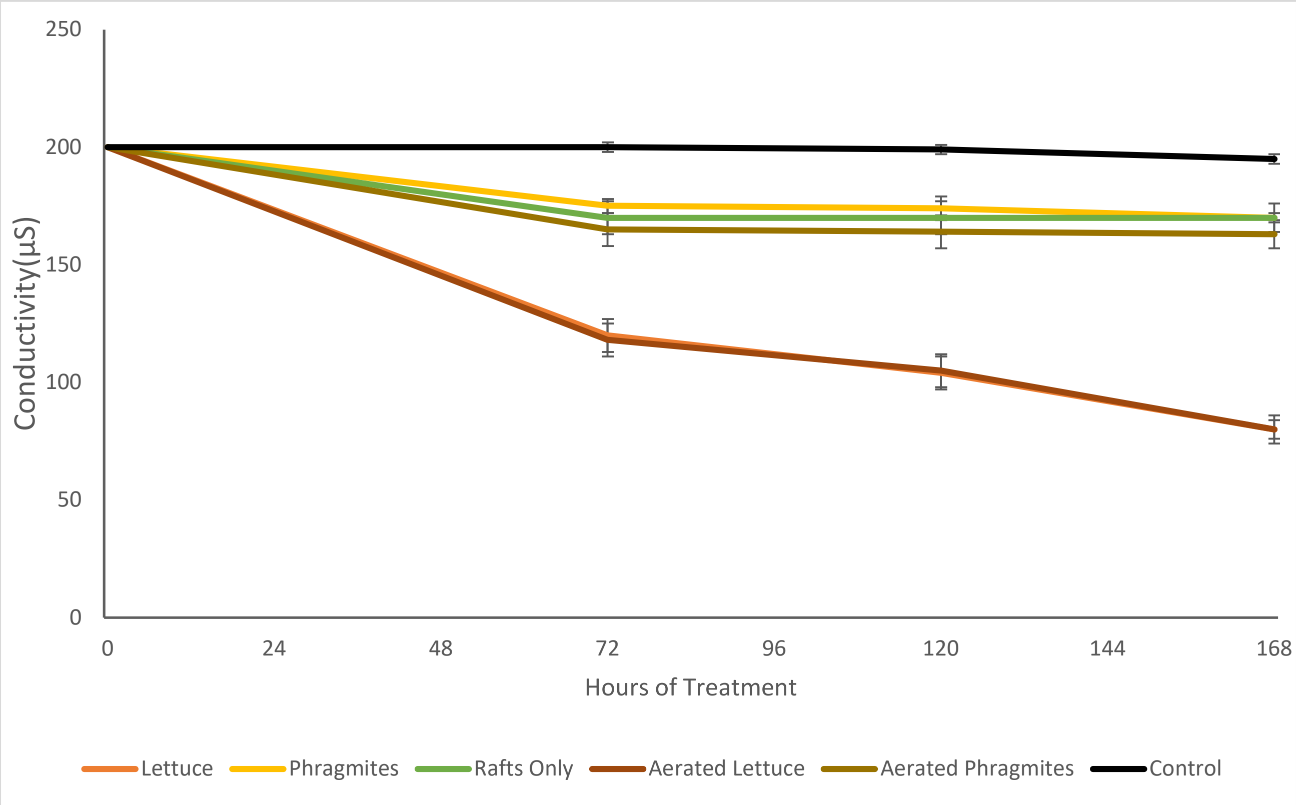

Figure 8: Trendline of conductivity over time. Averaged data from 8 batch runs of the 7-day experiment. Error bars represent the standard error of the means (n=4)

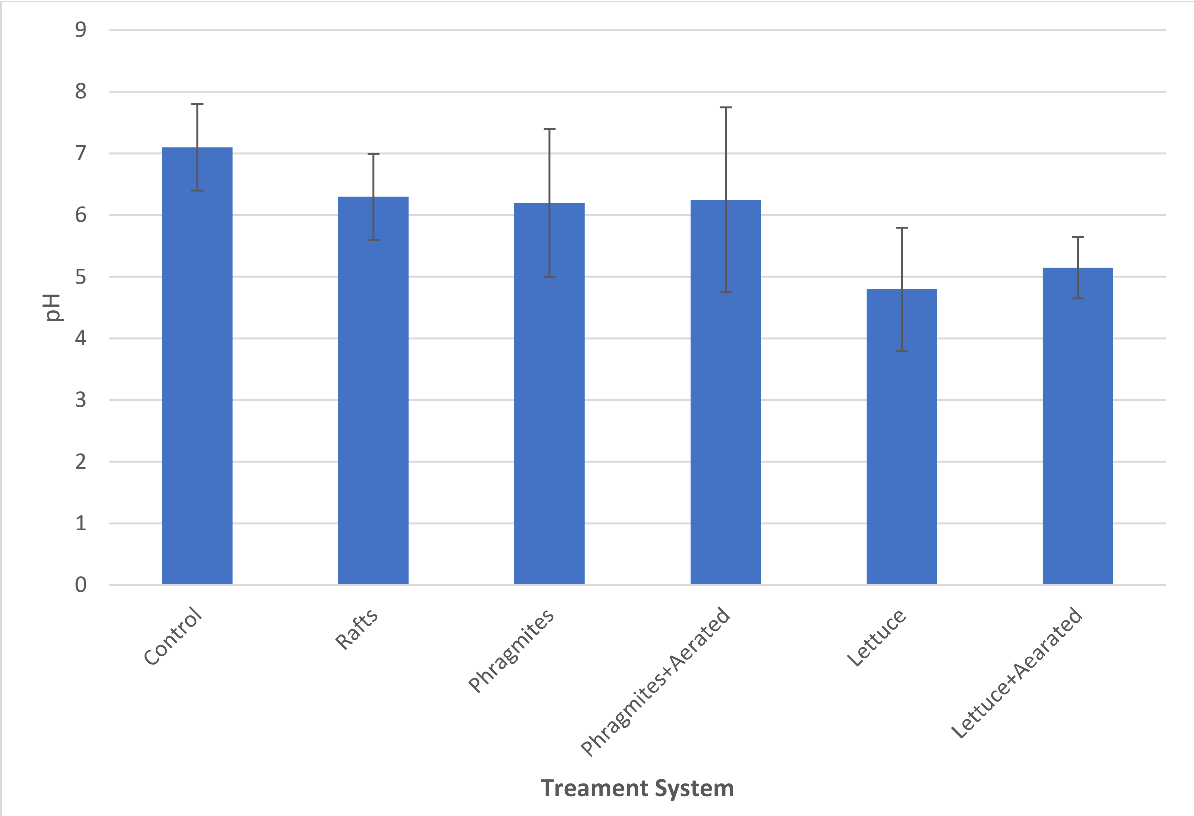

Figure 9: pH of water samples after 7 days of treatment. Averaged data from 8 batch runs. Error bars represent the standard error of the means (n=4)

For statistical analysis of acidity pH values were converted to H^+ ion concentration using the formula pH = - log [H+].

There was significant difference ($p<0.05$) between the H+ ion concentration of effluent from systems planted with lettuce plants, and those planted with *Phragmites*. The systems planted with established lettuce plants had significantly greater H^+ concentration than those planted with *Phragmites* ($p<0.05$). There was a lower concentration of H^+ ions in aerated lettuce samples, but the difference between means was non-significant($p<0.05$).

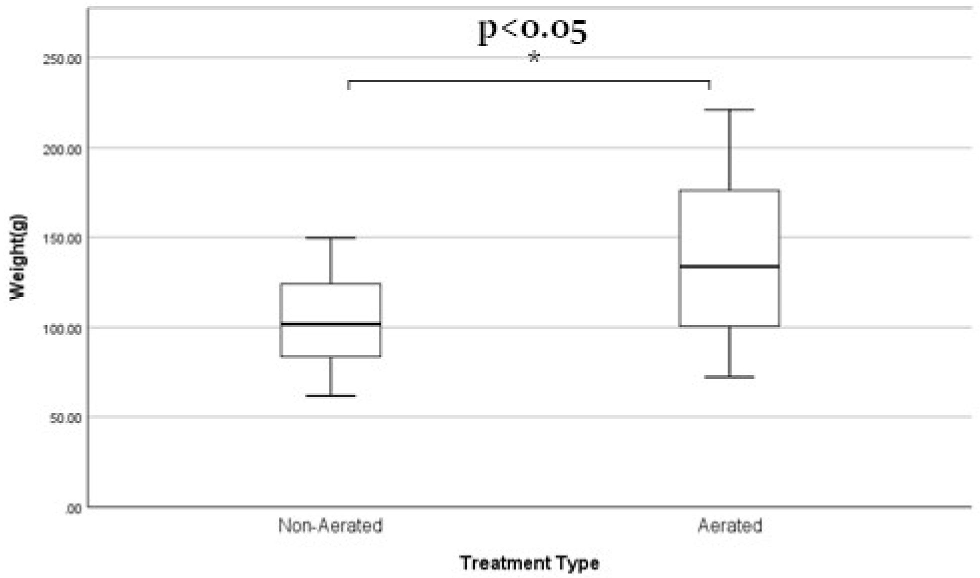

Figure 10: Wet weight of lettuce plants in each treatment group after 6 weeks of growth. (n=16)

A t-test was carried out between the weighed wet mass of the aerated lettuce test group and the non-aerated lettuce test group and a statistically significant difference was returned ($p<0.05$).

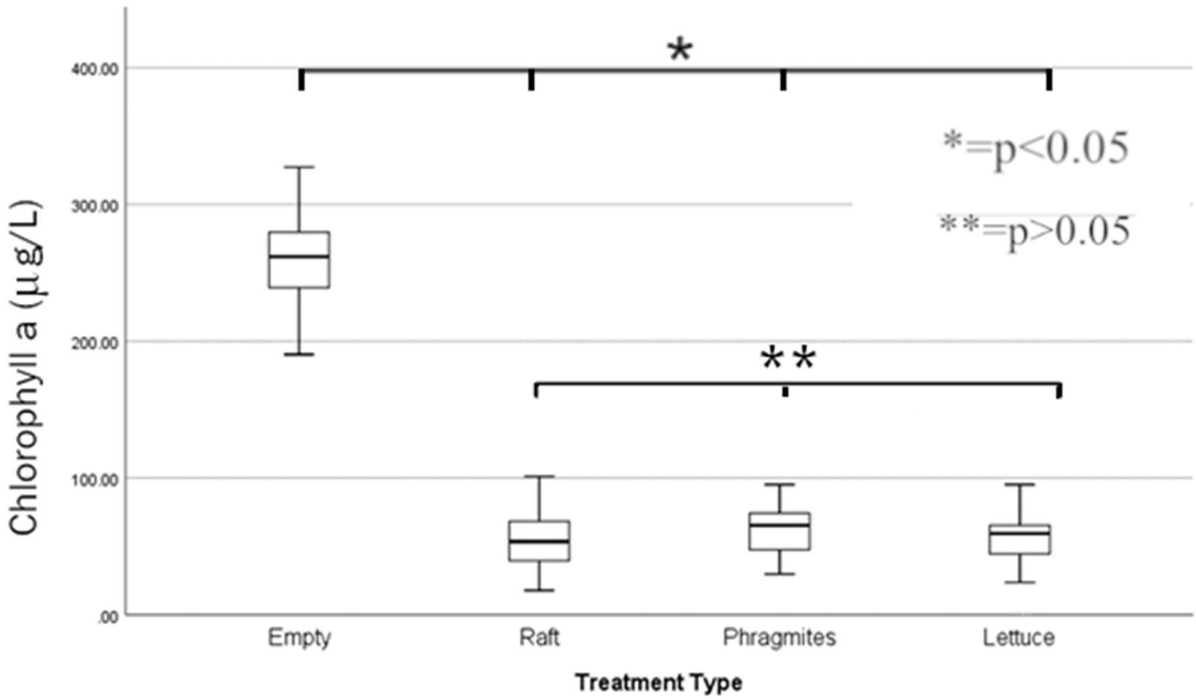

Figure 11: Chlorophyll-*a* concentration of water samples after 7 days of treatment. (n=4)

An ANOVA was performed and returned a significant result ($p<0.05$). Post-hoc analysis showed there was a significant difference ($p<0.05$) between the algal growth in the empty control system and all the other groups. There were no significant differences between the raft, *Phragmites* and lettuce groups. The treatment system with a floating raft had an average of 260μg/L of chlorophyll-*a* after 7 days of growth, compared to an average of 60μg/L for the test group with unplanted floating rafts.

While both planted raft systems showed mean higher concentrations of algae compared to raft-only systems, it was statistically insignificant ($p>0.05$). The mean average of raft-only systems = 52μg/L, *Phragmites* planted systems = 62μg/L, and lettuce planted systems = 59μg/L.

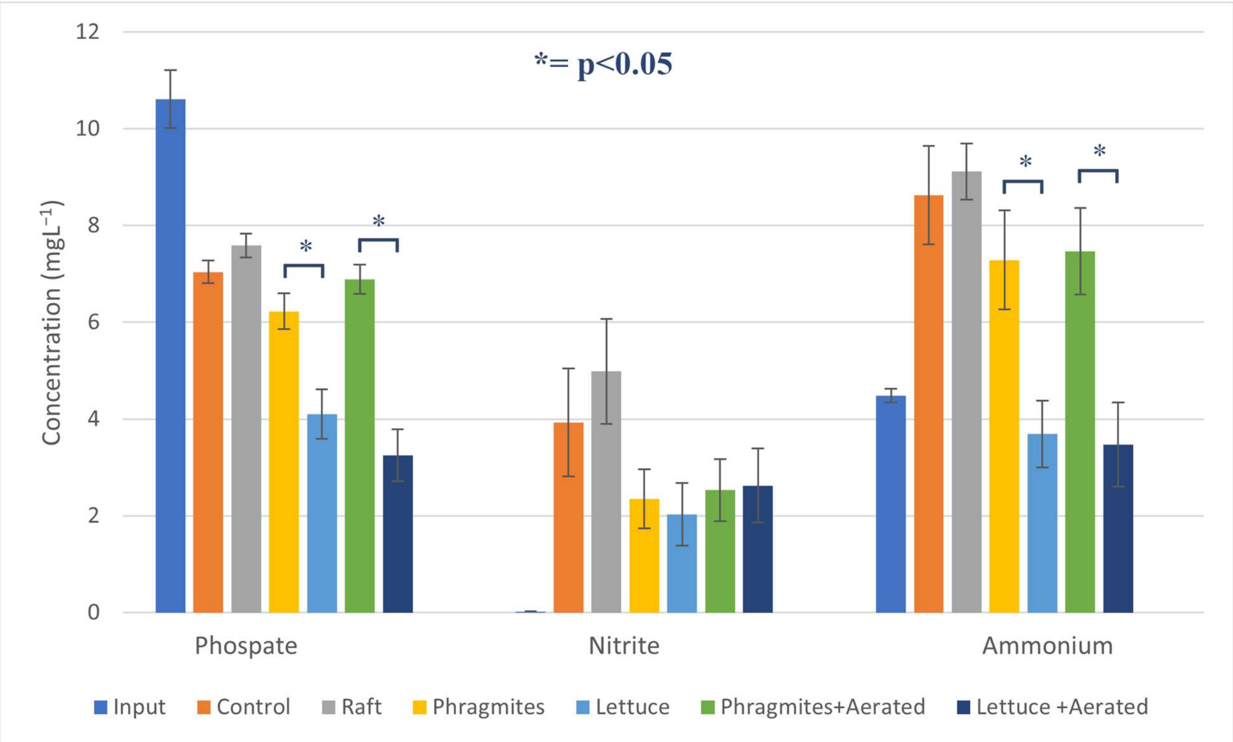

Figure 12: Concentration of nutrients important to plant growth and eutrophication in effluent after 7 days of treatment. Averaged data from 8 batches. Error bars represent the standard error of the means (n=4)

All systems removed a significant amount of phosphate when compared with the input ($p<0.05$). However, when compared with the control system, the raft, *Phragmites* and *Phragmites*+aerated all reported non-significant differences in phosphate concentration ($p>.05$). Both lettuce systems reported significant differences with the control system, but while the aerated lettuce system lowered phosphate by more than the non-aerated system, it was not a significant difference ($p>0.05$).

All systems showed a significant ($p<0.05$) increase in the quantities of nitrite over the input. All the planted systems showed a significant reduction of nitrite when compared with the control, but insignificant differences between planted groups ($p>0.05$).

An ANOVA was performed between values of ammonium in output water; a significant result was returned ($F=7.512$, $df=5$, $p<0.05$). Post-hoc analysis was performed to determine the non-aerated lettuce systems, which showed statistically significantly lower amounts of ammonium in output water compared to non-aerated *Phragmites* systems($p<0.05$). There was no significant difference in ammonium levels in output water between aerated lettuce systems and non-aerated lettuce systems ($p>.05$). There was also insignificant difference between ammonium levels in output water between aerated *Phragmites*, and non-aerated *Phragmites* ($p>0.5$).

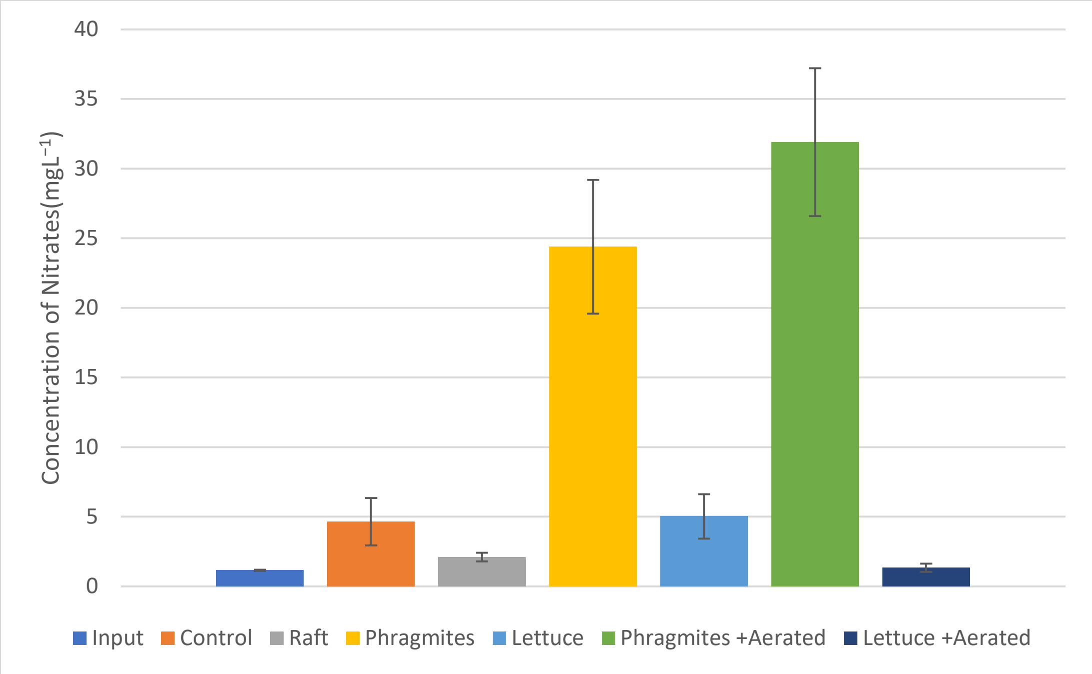

Figure 13: Concentration of nitrates after 7 days of treatment. Averaged data from 8 repeats of the 7-day run. (n=4).

Both *Phragmites* systems showed significant different values for nitrates, from the other test groups in an ANOVA test with post-hoc analysis ($F=17.19$, $p<0.05$). There were no significant differences returned between the other groups (control, raft, lettuce, lettuce+aerated).

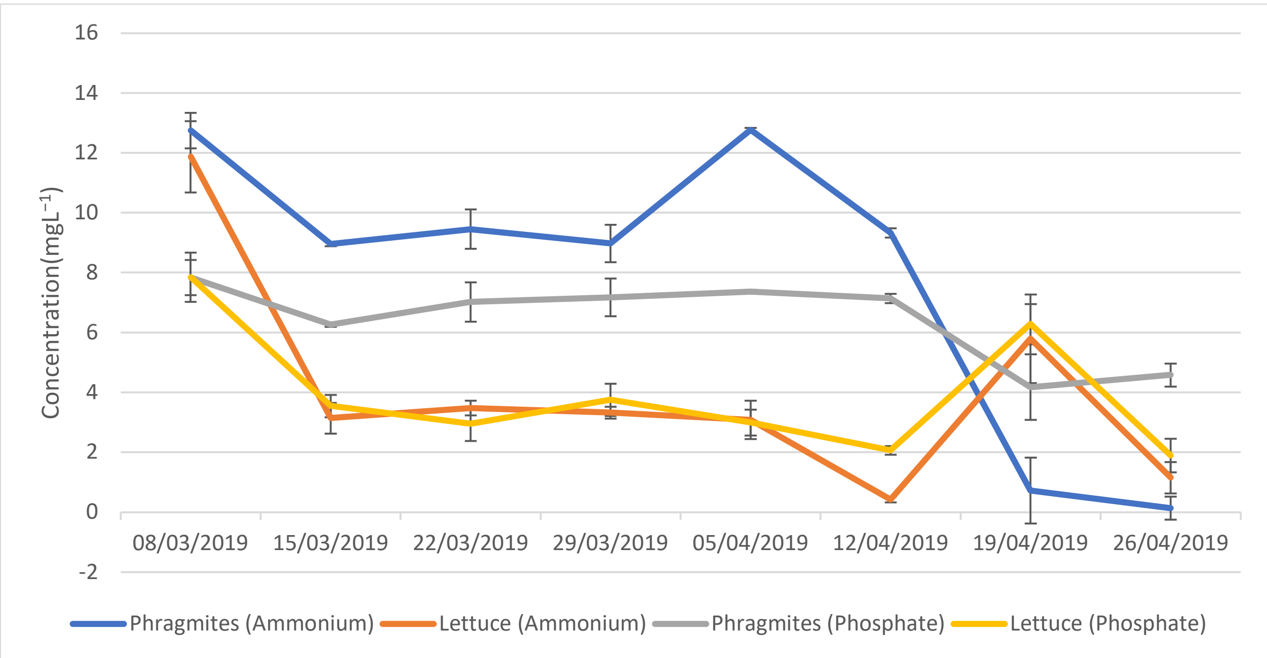

Figure 14: Ammonium and Phosphate concentrations in samples at the end of each 7-day treatment period, over the course of the 2 month experiment(n=4).

There was a trend over the course of the 2-month experiment for an increase in the removal of phosphate and ammonium in each 7-day batch run. There was insignificant change in phosphate for both lettuce ($t=.06$, $df=6$, $p>0.05$) and *Phragmites* ($t=-1.96$, $df=3.2$, $p>0.05$) between 15/03/2019 and 12/04/2019).

There was a significant difference between the removal amounts in the 08/03/2019 batch run and the removal amounts in the 26/04/2019 batch run in all test groups for both phosphate and ammonium ($p<0.05$).

Table 6: Comparison of the efficiency of treatment between the first batch and last batch. Negative numbers indicate an increase in concentration of nutrient in output water

Treatment Type	**Phosphate Removal 08/03/19**	**Phosphate Removal 26/04/19**	**Ammonium Removal 08/03/19**	**Ammonium Removal 26/04/19**
Control	.31g m^{-2}	.415g m^{-2}	-.409g m^{-2}	-.61mg m^{-2}
Raft only	.136 m^{-2}	.263g m^{-2}	-.409g m^{-2}	-.503g m^{-2}
Phragmites	.220g m^{-2}	.481g m^{-2}	-.353g m^{-2}	.347g m^{-2}
Phragmites with aeration	.151g m^{-2}	.403g m^{-2}	-.204g m^{-2}	.82g m^{-2}
Lettuce	.220g m^{-2}	.696g m^{-2}	-.276g m^{-2}	.266g m^{-2}
Lettuce with aeration	.275g m^{-2}	.758g m^{-2}	-.300g m^{-2}	.310g m^{-2}

Discussion

It has been previously discussed by (Stewart et al., 2008) that direct comparison between effectiveness of FTW systems is difficult due to many compounding factors. Examples of variability in the set-up of experiments are: batch vs continuous; nutrient loading; time span; the use of a control situation with or without a floating mat; the use of bottom substrates or not; and the use of soil media on the floating mat or not. All of these factors can have a large influence on experimental outcomes (Keizer-Vlek et al., 2014). However, despite these concerns raised we will try to compare the efficiency of our experimental systems with comparative systems. It is difficult to find experiments with identical setups in every category listed above, so we will try to draw comparisons mostly with batch-fed, substrate-less systems.

While this variation amongst FTW systems is a difficulty when it comes to direct statistical comparisons, it should be noted that it is also one of their biggest strengths in the field as it allows for a high amount of adaptability to local conditions and requirements

Hypothesis

Our hypothesis was that lettuce planted systems would perform water quality improvement at a comparable rate as phragmites planted systems, with a focus on phosphate and nitrate levels. Our secondary hypothesis was lettuce planted systems having the same effects on reduction of phytoplankton growth as traditional wetland systems.

As shown in Figure 14 our results show that lettuce planted systems reduce phosphate levels in treatment systems by a greater amount that traditional floating treatment wetland systems planted with traditional wetlands plants. The difference was significant ($P<0.05$) for the first 6 weeks, after which the phragmites sytems improved in efficiency and surpassed the lettuce systems.

Nitrate concentration is a more complex situation as shown in Figure 13. While lettuce planted systems had lower nitrate levels that phragmites planted systems, the levels in the lettuce planted systems were comparable to the control systems. However the nitrate concentration was significantly lower in the both the aerated and non-aerated lettuce systems when compared to their equivalent phragmites systems.

Figure 11 shows that lettuce planted systems had the same effect in reduction of chlorophyll-a production as the phragmites planted systems. This represented a non-significant different ($P>0.05$). This means we can accept our secondary hypothesis of lettuce planted systems having a similar reduction effect on chlorophyll-a concentration as traditional floating treatment wetlands.

Acidity

As discussed in our introduction, acidification of agricultural land is a problem faced by many farmers. As shown in Figure 9 the phragmites planted systems had negligible effect on the pH level of the treatment water, while the lettuce planted systems increased H^+ ion concentration by a significant amount($p=0.05$). As reported in our results section, aeration of the lettuce systems did reduce the increase in acidity, but not a significant amount. This would be a good area for further investigation as aeration is a common method in industry for treatment of wastewater(Rosso et al., 2008). Reductiuon of acidification caused by lettuce planted FTW systems would be desireable in our systems. The increase in H^+ concentration could be caused by active transportation in lettuce root systems which takes places during nutrient uptake.

Conductivity

The Little Gem lettuce FTWs in our experiment showed the ability to lower the conductivity of treatment water to the same amount, and sometimes lower levels, than the *Phragmites australis.* Conductivity levels have been used to measure nutrient quantity in field runoff and their origin fields (Heiniger et al., 2003). Both lettuce-planted FTW systems reduced conductivity by 60% over 7 days on average; aeration did not affect conductivity after 7 days. *Phragmites* planted systems showed conductivity reduction of between 10-15% compared to control systems, but the same levels as raft systems.

Phragmites australis is one of the wetland plants most adapted to high nutrient levels, and it is shown to dominate wetland environments with a mild increase of available nutrients (Bedford et al., 1999). *Phragmites australis* is viable in substrates with nitrogen levels between .05mg N g^{-1} and 2.43mg N g^{-1} (Meyerson, 2000; Ruiz and Velasco, 2010). Lettuce species have been shown to be viable in substrates with 27.2mg N g^{-1} and often much more, with sometimes nitrogen levels as high as 60mg N g^{-1} (Gonzalez et al., 2016).

This provides the potential for the use of agricultural FTWs for the treatment of more heavily polluted water; for instance, tertiary treated sewage water as used in other hydroponic systems in arid countries with very limited amounts of water (Al-Karaki, 2011). Agricultural FTWs may provide an alternative to traditional FTWs in water which is too nutrient-rich to support native wetland plants.

Nitrite

The concentration of Nitrite between plant groups was negligible, while lettuce planted systems did have a lower concentration of Nitrite on average, the difference was not significant. An important thing to note is that all planted systems had lower concentration of Nitrite on average than both the control and raft systems. This means that while the lettuce planted systems do not show a benefit over the phragmites planted systems they still have a positive effect for the removal of Nitrite. This is an important factor for our hypothesis as it shows that phragmites sytems can be replanted with lettuce and it would not have a negative effect on water treatment efficiency with regards to the removal of Nitrite contimants.

Nitrate

Nitrate was the ion with the biggest concentration difference between groups, with the nitrate ion concentration in the *Phragmites* treatment groups being many times higher than those found in the lettuce systems and the unplanted system. While nitrate levels were also increased in the lettuce systems when compared to input and controls, the mean average was much lower than in the *Phragmites* systems (5 mgL^{-1}/1.3 mgL^{-1} for lettuce systems vs the 24 mgL^{-1} /32 mgL^{-1} for reed systems.) When compared with the non-aerated *Phragmites* treatment group, the aerated *Phragmites* group had higher levels of nitrates. This large increase in the concentration of nitrate in the aerated reeds test is likely due to the introduction of oxygen. Aeration, either intermittent or constant, has been shown to increase nitrate concentration in effluent from CTW systems; a suggested process for this is the bacterial nitrification of ammonium into nitrate (Stewart et al., 2008). The result of increased nitrate concentrations in effluent from aerated treatment systems is consistent with previous literature on the use of aeration to treat wastewater (Maltais-Landry et al., 2009; Uggetti et al., 2016). However, the Uggetti (2016) experiment had constant aeration, whereas our systems had intermittent aeration which Uggetti recommended for improved nitrate removal. A suggested method for this build-up of nitrates is that removal of nitrate requires dissolved organic carbon; this carbon needs to be released from plant root systems. The *Phragmites* in our experiment may not be large enough in quantity or size to provide enough dissolved organic carbon to promote sequestration of nitrate (Zhu and Sikora, 1995). Few studies exist with *Phragmites* and they offer contradictory results for the effectiveness of nitrate removal by *Phragmites* planted FTW systems. While (Abed et al., 2017) showed poor removal of nitrate, they suggested it was due to the lack of biodegradable organic matter. (Li and Guo, 2017) reported *Phragmites* offered good nitrogen and nitrate removal qualities in cold conditions, especially when compared with the *Acorus calamus* planted systems used in their experiments.

The planting substrate in our experiment was clay aggregate pebbles, which does not provide biodegradable organic matter and the *Phragmites* in our study were not developed enough to drop organic matter which could degrade. The lack of biodegradable organic matter may result in high concentration of nitrates in the output water of our *Phragmites* planted systems. Experimental FTWs have been used to treat nitrate successfully, but they are mostly planted with T*ypha* or J*uncus* species (Lynch et al., 2015).

The lettuce planted systems showed lower amounts of nitrate in the output water than the *Phragmites* planted systems, but still greater than the input concentrations. While there was a statistically insignificant difference between the groups, the nitrate concentration in both groups was higher than the input amounts. Using further data from figure 13, it is likely that it was nitrate being produced by denitrification of ammonium which resulted in this production of nitrate, as mentioned in Stewart et al., (2008).

Phosphate

The concentration of phosphate in our input effluent was on average 10.6mg/L^{-1}. This was considerably stronger than the input effluent used in a similar experiment with floating treatment wetlands, run by Keizer-Vlek (2014) who had a phosphate concentration of .25mg/L^{-1}. An experiment with similar input phosphate loading was run by Hubbard et. al (2004) which used undiluted effluent, with a phosphate loading of 30mg/L^{-1} and a diluted effluent with a loading of 15mg/L^{-1} of phosphate.

In Hubbard (2004) the systems planted with Rush and Cattail removed 37% and 54% of total phosphate entering the diluted system.

Table 7 shows the average percentage removal of phosphorus in our systems, with similar input values of phosphate loading in input effluent (10.9 mg/L^{-1} in our experiment vs 15mg/L^{-1} in Hubbard).

Table 7: Comparison of percentage phosphate removal in our systems between first batch run and final batch.

System Type	Phosphate Removal 08/03/19	Phosphate Removal 26/04/19
Control Systems	3.7%	49.0%
Raft Systems	16.5%	31.1%
Phragmites	26.1%	56.8%
Phragmites + Aeration	18.0%	47.6%
Lettuce	26.0%	82.0%
Lettuce + Aeration	32.5%	89.4%

Our systems showed comparative removal rates of phosphate to Hubbard's experiment; however, as shown in Table 6 only the aerated lettuce systems exceeded the Rush-planted system in Hubbard's experiment during the batch run between 01/03/19 and 08/03/19.

Later in the year nearly all our planted systems showed higher treatment efficiencies than even the Cattail systems planted in Hubbard's experiment. This difference in treatment efficiency in our experiment over time is likely due to changes in the weather, an increase in temperature and the establishment of plant stocks in our systems.

Hubbard's data is averaged over 16 months, between June 2001 and September 2002. As shown in Table 2 of Hubbard (2004), the temperature at his experiment varied between the lowest of 10.6 °C in February 2002, and the highest temperature of 27.9 °C in July 2002; while our experiment was carried out over only 2 months, with average temperatures of 10.3 °C and 13.7 °C. However, our experiment was carried out in a heated greenhouse which averaged 5°C above ambient. This is likely to have increased the efficiencies of our system in comparison to Hubbard's, which was outside and unheated year-round. Kadlec and Reddy (2001) reported that temperatures between 20°C and 35°c were the optimal temperatures for treatment wetlands efficiency.

Comparing phosphate removal rates in Table 7 to examples shown in the literature in Table 4, our *Phragmites* systems removed a lower percentage than most of the other experiments in the literature, only exceeding the *Pontederia cordata* and *Schoenoplectus tabernaemontani* from (Wang and Sample, 2014). The *Phragmites* systems in our study removed 4.08 P mgL^{-1} d^{-1} in the 07/03/19 batch, compared to the 1.18 P mgL^{-1} d^{-1} and 0.25 P mgL^{-1} d^{-1} removed by P*ontederia* and *Schoenoplectus* respectively in Wang et al. (2014). The low removal rates of phosphate in the Wang study are likely due to their much lower loading of bioavailable phosphate (0.15mg/L TP, compared to the 10.6mg/L TP in our experiment).

The relative inefficiency of our systems in the 08/03/19 batch is likely due to our study being performed so early in the year when conditions were still wintery and our plants still establishing, whereas the above studies were performed on better established plants in a more suitable season.

However, by the final batch on 26/04/2019 the removal efficiency of our *Phragmites* systems had improved to the point of being only behind *Panicum virgatum* from (Jonathan T Spangler et al., 2019) and the *Iris pseudacorus.* Our reed systems removed 6.2 P mgL^{-1} d^{-1} of phosphate compared to the *Iris* removing 9.32 P mgL^{-1} d^{-1}.

Chlorophyll-a Concentration

As the experiment described in Jones et al., (2017) was performed in the same location as this experiment, at a similar time of year resulting in similar weather conditions, this would show consistent algal growth and growth reductions between the two experiments. (Jones et al., 2017) reported reduction of 80% of chlorophyll-*a* (used as a proxy for phytoplankton biomass) over 4 weeks while our experiment showed a 77% reduction of chlorophyll-*a* over 3 weeks (Jones et al., 2017).

In Jones et al. (2017) they suggest that the reason for the reduction in chlorophyll production is due to the nutrient uptake by the *Phragmites australis* plants. It is interesting to note that as shown in our Figure 12, the addition of the unplanted floating mat had the most effect of reducing chlorophyll-a concentration, the addition of adding plants to the floating mat having no positive effect. In fact the algal concentration was in fact higher in the planted mats, but not by a significant amount ($p>0.05$).

This difference may be caused by our mats covering a much larger % of the total surface area of the water, when compared with the Jones experiment.

This difference suggests the most likely explanation for the reduction in chlorophyll-a concentration in our experiment is due to the floating mats blocking sunlight, which is one of the major factors in algal growth (Kim, 2018).

Seasonal Variation on Treatment Efficiency

This experiment was a batch experiment, with each batch taking 7 days to complete. Runs were repeated to produce more results, and to detect change of treatment efficiency over time. Figure 15 shows a timeline of ammonium and phosphate concentrations at the end of each 7-day run. There was an overall trend for the systems to remove more phosphate and ammonium the longer the experiment continued. This is likely due to the further establishment and growth of our plants, as well as the growing season progressing and there being more hours of sunlight and a higher average temperature (refer Table 5).

The decrease in treatment efficiency in both lettuce systems in the 19/04/2019 sample was likely due to this being the week which harvested the first crop of lettuce and replaced them with new lettuce seedlings. These new lettuce seedlings needed time to adapt to their new environment and to establish growth. Normal treatment efficiency had resumed by the following week.

Another trend to observe is how quickly the lettuce systems reach high treatment efficiency in both ammonium and phosphate. By batch 2, 15/03/019, they have almost reached the same level of treatment efficiency as they have in the 26/04/19 batch (3.54 mg/L^{-1} vs 1.8 mg/L^{-1} for phosphate), while *Phragmites* planted systems took 7 weeks to begin to show treatment. In the 15/03/19 batch the *Phragmites systems* were producing effluent with 8.92 mg/L^{-1} of ammonium, and 6.26 mg/L^{-1} of phosphate. By the final run on 26/04/19, these values had been reduced to only 0.13 mg/L^{-1} of ammonium, and 4.58 mg/L^{-1} of phosphate.

In Table 7 a comparison is made between the percentage phosphate removal in the first batch which finished on 08/03/19, and the percentage phosphate removal in the final batch which finished on 26/04/19. The large increase in phosphate removal in the control systems (3.7% to 49%) cannot be due to plants or rafts, as our control system had neither. A suggested reason is probably due to the growth of algae in the unshaded water present in these systems; one of the primary nutrients responsible for algal growth is phosphate. It has been shown that phosphate levels decrease as algae growth occurs (Jones et al., 2017).

All systems showed a greater efficiency at removing phosphate and ammonium in the 26/04/19 batch, than in the 08/03/19 batch. The greatest change was seen in the treatment of ammonium in *Phragmites* systems, an improvement from effluent with 12.743mg/L^{-1} in the 08/03/19 batch to only 0.13 mg/L^{-1} in the 26/04/19 batch. This meant that while lettuce was better at removing ammonium than *Phragmites* for the first 5 batches, by the final 2 *Phragmites* had overtaken in treatment efficiency and were better at removing ammonium than the lettuce systems.

While the trend of improving over successive batches was true for *Phragmites* systems with treatment of phosphate, it was not nearly as much an improvement as that of ammonium. In the 26/04/19 batch the lettuce planted systems still removed more phosphate from the effluent than the *Phragmites* systems.

Plant Mass

There was no significant difference in the nutrient levels of the aerated and non-aerated lettuce after 168 hours (Figure 9), which means that more plant growth had occurred while using the same amount of nutrients. This suggests that while aeration of FTW systems does not increase the total amount of nutrients removed during treatment, it does increase the efficiency at which plants utilize nutrients to grow mass. We propose this is due to the coexistence of both aerobic and anaerobic pathways in the aerated systems, due to our systems being of intermittent aeration, providing both oxygenated and anoxic conditions for nitrifying bacteria (Bodelier et al., 1996).

The lettuce grew for 42 days and the aerated lettuce group had an average mass of 143g and the non-aerated lettuce group had an average mass of 101g. Each system was planted with 4 lettuces, so each one of our aerated systems produced 572g in 42 days, while our non-aerated systems produced 404g in 42 days. Each system had a surface area of $0.1m^2$; this corresponds to a growth rate of 123g m^{-2} d^{-1} for the aerated systems, and 96g m^{-2} d^{-1} for non-aerated systems.

Suggested Further Research Options

This study shows that lettuce-planted FTWs offer comparable water treatment abilities to P*hragmites* planted FTWs with regards to phosphate, ammonium, conductivity and algal reduction. It should be taken into consideration this study was performed in a controlled climate of a greenhouse and within a small timeframe of a few months.

Further studies should be undertaken with equipment that was not available to us. For example, monitoring and analysis of dissolved organic carbon, dissolved oxygen levels and root mass analysis would also be advantageous to the understanding of the performance of lettuce planted FTWs.

The viability of replacing traditional FTWs with agricultural FTWs is dependent on more factors than their ecological benefits. A cost-base analysis should be undertaken to determine the ecological feasibility of agricultural FTWs as they require a larger amount of labour and nutrient inputs to be viable.

More extensive experiments should be undertaken into studies over a longer time frame, designed to measure the nutrient uptake over an entire growing season. Our study is only over a short period of time, in which the *Phragmites* were still getting established. This may affect results for total nutrient uptake over a 12-month period, as the *Phragmites* may have lower nutrient uptake during their establishment period than they do during full-size growth.

This study may not be representative of the long-term water quality effects of replacing wetland plants in FTWs with lettuce plants. In our study, on average lettuce-planted FTWs lowered the nutrient quantities in the wastewater by more than *Phragmites*-planted FTWs; this may not be the case over a full growth cycle. *Phragmites* take longer to establish and grow to a much larger size than fully grown lettuces. This study was undertaken in a greenhouse, which offered increased temperature and protection from the wind. Traditional FTWs do not offer such protection, and plants may suffer from wind damage or buffering from turbulent waters. The negative effects of this may be lowered by raft designs with wind breaks and strong support for nesting plant pots.

Proposal for Agricultural Hybrid Floating Treatment Wetland

The treatment efficiency of existing FTW systems is well proven; planted with native wetland plants like *Juncus latifolia* and *Phragmites australis,* they have shown in real world situations that they provide treatment options for wastewater. A limitation that FTW systems have, just like the CTWs they are derived from, is their limited effectiveness during cold months (Yan and Xu, 2014). It is likely not the temperature itself which is the major cause of slowing down of the treatment systems, but the winter conditions when macrophyte and microbiological activity is at its lowest. Metabolic processes of plants and microbial activity are the major limiting factor during winter months (Nsenga Kumwimba et al., 2021).

Treatment efficiency of FTW systems increases throughout the growing year, peaking in the summer months and declining as the growing season comes to an end and the colonised macrophytes die off or are harvested in nutrient removal strategies. *Phragmites australis* has aerial shoots that die during late autumn to early winter; the roots of the plant remain dormant in the wetland soil over winter. This leads to a growth period in spring where these aerial shoots have to regrow. Shoot height continues to regrow throughout summer, from May until early August (Haslam, 1972); however, maximum stem density is reached in early summer, so there is a period when no new shoots emerge but existing shoots continue to grow (Boar et al., 1989; Gibson and Rodwell, 1995).

This regrowth and establishment of new emergent biomass over several months each year means there is a substantial amount of time each year when FTW systems work at reduced capacity (Tharp et al., 2019). With the improved performance of the lettuce-planted systems earlier in the growing season as shown in Tables 6 and 7 and Figure 15, we propose the idea of an Agricultural Hybrid Floating Treatment Wetland (AHFTW) which would be planted with both native wetland plants like *Phragmites* or *Typha,* and lettuce crops. This system would essentially be a hydroponic version of intercropping, which is the process of cultivating two or more crops in the same space at the same time, in order to maximise production efficiency (Lithourgidis et al., 2011).

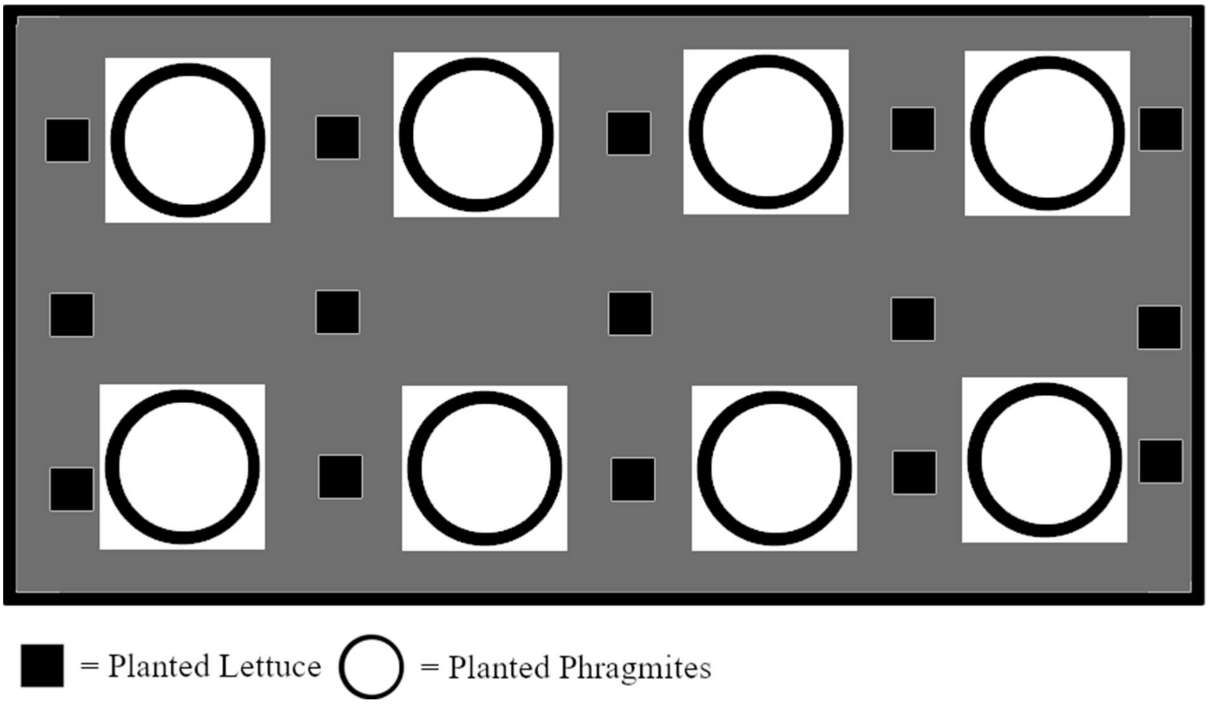

Figure: 15: A proposed layout for a planting regime of an AHFTW

The use of lettuce-planted systems may prove prohibitively expensive in real-world conditions, considering the cost in bi-weekly planting and harvesting, as well as potential stock-loss to the exposed conditions.

The AHFTW system proposed would combine the positives of lettuce-planted systems, the efficient treatment in early spring months, with the high yearly efficiency of *Phragmites*-planted FTWs. There is also the potential for lettuce crops to grow and treat wastewater in early spring months, from March to May, while the *Phragmites* are still establishing themselves after their winter die-off. Once the native wetland plants have established and are operating at full capacity the sowing and harvest of lettuce can stop. Once established, native wetland plants require almost no labour input, apart from one harvest at the end of the growing season in the autumn to prevent plant matter from re-entering the water system.

As discussed in the plant mass section, the lettuce systems in our experiment had a growth rate of at least 96g m^{-2} d^{-1}. A moderately sized retention pond, about 0.5 a hectare, could theoretically produce 44,640kg of lettuce over the 3-month growth period at the beginning of the year before *Phragmites* have fully established, between the dates of 01/03 and 01/06. It is doubtful that the results from our systems would scale at 100% efficiency into real-world scenarios as this experiment was run in ideal greenhouse conditions, with heating and augmented growth lights. Assuming a loss of 50% efficiency of growth due to this change in conditions, these systems would produce 22,320kg of lettuce in a 0.5 hectare retrofitted retention pond. The mass we weighed in our experiment was total wet mass, including root systems which usually comprise between 10-20% of mass of lettuce (Frantz and Bugbee, 2005). Accommodating for this 20% loss for root weight, our systems would produce around 17,000kg of lettuce per 0.5 hectare pool; this is a production rate of 34,000kg per hectare. It was reported that in California between 2007-2009 leaf lettuce production per hectare averaged 33,604kg (Smith et al., 2011). Our systems show similar production rates to standard agricultural conditions, while providing water treatment services.

After this 3 months' growth of lettuce, which should aim to include 3-4 harvests, cultivation of lettuce stops so that wetland plants can fully colonise the floating raft systems.

These AHFTW systems could be fitted into more scenarios than just retention ponds. As shown in Table 3, there is the potential for AHFTWs to be used for road run-off, agricultural wastewater and other types of wastewaters. Figure 16 shows an approximate layout for such a system. While it is being established *Phragmites* plants take up very little surface area of the floating mat surface, as most of their growth is in their roots while they establish in size. This means that lettuce plants can be cropped close together, as the lettuce will not grow to any significant size above the mat in the two to six weeks it has to grow, depending on local conditions and size of lettuce wanted for harvest. Lettuce planting can be stopped once *Phragmites* has established itself to such a size that lettuce no longer fits on the surface of the raft. After a final harvest their planting cups can be replaced with cups filled with an inert substrate like clay aggregate pebbles, and the raft left to be a traditional FTW system.

Conclusion

Not only did lettuce establish and thrive in our FTW systems, but they also provided good treatment efficiency of phosphate and ammonium. Averaged data from all 8 batches showed the non-aerated lettuce system removed 63% of total phosphate from input water and the aerated lettuce system removed 71%. These values compare favourably to a similar experiment ran by (Hubbard et al., 2004) with FTWs which also had input water with phosphate loading similar to this experiment. The non-aerated *Phragmites* systems removed 44% of phosphate from input water, while the aerated systems removed 38%. The lettuce systems also showed the ability to greatly reduce conductivity after 168 hours when compared with P*hragmites* systems; 80 μS in both lettuce systems, 163 μS and 170 μS in *Phragmites* systems.

Phragmites showed a significant lag period in treatment efficiency which increased as the experiment went on, and P*hragmites* systems eventually became the best system for ammonium removal, although its ability to remove phosphate always lagged behind the lettuce systems, despite showing improvement over time. Due to this lag in the efficiency of P*hragmites*-planted FTWs and the labour-intensive nature of lettuce-planted systems, we propose that they be intercropped in a single system over one growing season, with lettuce providing the bulk of treatment efficiency early in the year, and *Phragmites* later in the year when they have established.

Aeration improved treatment efficiency in lettuce-planted systems by a small amount but decreased the efficiency of treatment in the *Phragmites* systems by a small amount. Aeration was shown to have a significant effect on lettuce mass production, with aerated systems producing significantly more above-mat and below-mat biomass.

The potential benefits of our proposed AHFTWs are twofold: to increase the efficiencies of treatment systems in cold conditions, particularly early periods of the growing year; and to provide a crop which is consumable by humans, or if the quality is not good enough for human consumption it could be used for animal fodder (Al-Karaki, 2011; Asadullah Al Ajmi Isam Kadim, Yahia Othman, 2009).

There is much potential for harnessing our wastewaters as a potential resource, replacing their current identity as a waste product. FTW systems evolved from CTW systems to accommodate changing water levels. AHFTWs may prove to be another evolution to accommodate for the increasing need for food and conserving water.

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Appendices

Discussion of the effect of water pollution on tourism

Having lakes with aesthetically pleasing, extremely clear water is of positive benefit to the tourism industry; however, if pollution decreases water quality then this factor disappears and is no longer a benefit to the tourism industry .

Table 1: Comparison of the water quality of lakes in the Fraser Dune Lake systems (Hadwen et al., 2004)

Location	**Lake McKenzie**	**Lake Wabby**
Tannins	100 ($\mu g\ L^{-1}$)	1000 ($\mu g\ L^{-1}$)
pH	4.82	6.72
Secchi Depth(clarity)	8.6m	1.5m
Tourist Pressure Index	61.2	15.7

In Hadwen (2004) correlation is drawn between the clarity of lakes and tourists' interest in visiting them. Lake McKenzie had the highest tourist pressure index, indicating that tourists had the highest interest in visiting the site. 70% of surveyed tourists identified clear lakes as their preferred swimming location. Data in the report backs up these statements, as Lake McKenzie had the lowest concentration of algae (monitored by chlorophyll-*a* concentrations) and the lowest concentration of tannins.

There is a strong indication shown in Table 1 that the cleaner the water, the higher the interest there is in tourists visiting. Lake McKenzie has some of the cleanest and clearest water in the world, due to its geography as a perched lake. Lake Waddy is another lake in the Fraser Dune Lakes but is far less in demand by tourists due to the lesser water quality. Hadwen (2004) also reported than an increase in tourists led to an increase in autochthonous carbon entering the littoral food webs.

Chapter 5 of Hadwen (2004) details the increase in chlorophyll-*a* concentration between 1990 and 1999 in the perched dune lakes of Fraser Island and they discuss the possibility of human activity causing the increase in chlorophyll-*a*. While they propose that a drop in water level may have caused an increase in nutrients and chlorophyll-*a*, due to a decomposing of plant matter, they also state that human input contributed to the increase. They cite (Outridge et al., 1989) who performed similar experiments in lake systems and came to the conclusion that human inputs were adversely effecting water quality.

In a 2002 report, it was estimated that Fraser Island Lakes generated an estimated direct and non-direct tourism income of 277 million AUD and were responsible for almost 3000 jobs (Kleinhardt-FGI, 2002). Adjusted for inflation by the Reserve Bank of Australia’s inflations calculator (rba.gov.au), this is 418 million AUD in 2020. Tourism to Fraser Island remains strong, with between 350,000 and 700,000 international visitors a year (Council, 2021).

If the quality of the water at Fraser Island Lakes continues to be lowered by tourism, the location may lose many tourists if it no longer has the perfectly clear water they want. As 70% of the tourists to Fraser Island Lakes said their primary reason for visiting was the clearness of the water, if the clearness of water gets degraded so far as to no longer be aesthetically pleasing, the Fraser Lakes area may lose the 418 million AUD annually it receives from tourism and lose the 3000 jobs which are supported by the lakes.

An extreme example of the benefits and loss of tourism due to degradation of aesthetically pleasing water body by pollution is the history of the Salton Sea, an artificial lake formed accidentally between 1905 and 1907 when part of the bank of the Colorado river burst, and escaped water drained into the Salton Basin. In the 1950’s and 1960’s, during the post-war expansion of the United States, the Salton Sea was seen as the next “big thing” with it being proposed as an equivalent to Palms Spring. Millions of dollars were poured into the area in development funding. Golf courses, resorts, yacht clubs and extensive fishing clubs were created around this new artificial lake (Boyle, 1996). At its largest size, the lake was the largest water body in the United States, and it was home to almost 600,000 migratory birds in winter months in 1999. This made the Salton Sea a popular destination with birdwatchers, as many rare and endangered birds used it as a winter stop during their migration (Shuford et al., 2002). However, the quality of the water has been steadily deteriorating in the water basin since its creation. Since its cut-off from its original source in the Colorado river, the Salton Sea has only been fed by drain water flushed from surrounding agricultural fields; this water is heavy in pollutants such as nitrogen and phosphate fertilizers, as well as the highly toxic selenium. Steady evaporation has led to an increase in salinity, killing off most of the fish-stocks. Combined with a build-up of pesticides and selenium, the once idyllic waters of the Salton Sea are now a poisoned chalice to the wildlife that live in it. Mass die-offs of birds occur, with an estimated 200,000 birds dying in these events since 1992 (Cohen J.I., Glenn, E.P., 1999).

There is no estimated cost to conserve or restore the Salton Sea. It had been largely left to deteriorate until in 2021 the Salton Sea Management Program was enacted, a 670 million USD project aimed at stabilising sedimentation beds and preventing air pollution caused by exposed lake beds (Sevrens and Sea, 2021). The programme is pushing for a further 220 million USD in funding to help complete the project and prevent further damage to public health from the polluted water and sediment. This brings the total cost of the project to around 891million USD and that does not even include a long-term solution to water quality and evaporation problems in the Salton Sea. The cost of fixing the pollution in the Salton Sea will probably be in the billions of dollars (Metz, 2021).

This is one of the more obvious economic benefits of preserving currently oligotrophic water systems. The cost of prevention will likely be cheaper than the cost of fixing the problem. The current income from tourism, either traditional or ecological tourism, could be used to help conserve current nutrient-deprived system.

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