A Literature Review of Nutrient Management-Related Best Management Practices Used in the Lake Winnipeg Basin

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1.0 INTRODUCTION

1.1 Purpose

This literature review aims to provide a resource for quantifying reductions in phosphorus (P) and nitrogen (N) that contribute to nutrient pollution in Lake Winnipeg. Water quality has deteriorated in the lake due to multiple sources of excessive nutrients in the watershed that have increased the frequency and magnitude of algal blooms, including toxic Cyanobacteria or blue-green algae blooms. This has both direct and indirect consequences on not only the biological health of Lake Winnipeg, but has economic, recreation and tourism repercussions as well. Targets have been set to restore conditions that are similar to those from the 1990s, before occurrences of algal blooms had doubled in frequency and size (Schindler, Hecky & McCullough, 2012). Optimal targets of annual total P and N concentrations of Lake Winnipeg have been set as 4,850 and 62,140 tonnes, respectively (Manitoba Conservation and Water Stewardship, 2015).

As the tenth largest freshwater lake in the world, Lake Winnipeg has a watershed area of almost one million km^2, spanning over four Canadian provinces and four American states (Wassenaar & Rao, 2012). To achieve the target concentrations, ten percent reductions in both P and N loading in the Winnipeg, Saskatchewan and Dauphin River watersheds is required, with a fifty percent reduction in P loading and a thirty percent reduction in N loading in the Red River watershed (Manitoba Conservation and Water Stewardship, 2015). It will require multi-jurisdictional cooperation and involvement to achieve such targets.

The Lake Winnipeg Basin Initiative (LWBI) is the Government of Canada’s response to address nutrient pollution issues in Lake Winnipeg. The LWBI aims to engage citizens, scientists and domestic and international partners in actions to restore the ecological health of Lake Winnipeg, reduce nutrient pollution and improve water quality. The LWBI provides support for high-impact, stakeholder-based projects that improve the health of the watershed through the Lake Winnipeg Basin Stewardship Fund (LWBSF). Examples of projects funded by the LWBSF include: implementation of Best Management Practices (BMPs) that reduce rural or urban non-point sources of nutrients (e.g. water retention projects, conservation tillage, riparian enhancements, nutrient management and recovery projects); wetland conservation and restoration; development of innovative technologies that reduce nutrient loading from municipal wastewater systems or other point source discharges; cattle fencing and alternative watering systems.

The LWBSF requires successful applicants to provide information as to how effective their project was in reducing nutrients. Ideally that information is based on water quality monitoring of nutrient concentrations conducted before and after implementation of their project. Unfortunately for many stakeholder-led projects water quality data is not produced due to the lack of capacity to conduct such monitoring. In those cases, nutrient reductions are estimated based on values derived from the scientific literature. In the past, estimates derived from the Phosphorus Reduction Calculation Report developed by Environment Canada’s Lake Simcoe

Clean-up Fund were used in the Lake Winnipeg program (Sealock, 2011). The objective of this report is to provide nutrient reduction estimates based on research conducted primarily in the Lake Winnipeg Basin (LWB) that more accurately reflects the hydroclimatic and soil conditions and then can be used to help quantify the impact of the implementation of projects funded by the LWBSF on nutrient loading in the LWB.

1.2 Phosphorus and nitrogen management

Traditional management practices consider aquatic primary production as N-limited in marine waters and P-limited in freshwater systems (USEPA, 2015). This has led to focusing past management of the LWB primarily on reducing P loads, with minimal regard for N loads. According to Gillor, Hadas, Post and Belkin (2010), an ideal N:P ratio exists which reduces instances of exponential population growth of cyanobacteria. This ratio is favored over the occurrence of levels weighted strongly towards excessive P concentrations.

In the past, N was not able to accumulate to a significant concentration in the LWB largely because bacteria present in the water would convert it back to N_2, removing it from the water and releasing it into the atmosphere (Moffat, 1998). In a 19-year study by Messiga et al. (2012), although there is a minimum P-requirement for corn and soybean production, it was found that crop yield was not significantly different dependent on P application, but increased when N-rich fertilizer was applied. Findings such as this has increased usage of synthetic agricultural fertilizers, rich in N. These fertilizers, combined with the increased burning of fossil fuels, has largely accelerated the accumulation of N in freshwater to a point beyond what can be converted by the bacteria (Mayer & Wassenaar, 2012). Schindler et al. (2012) found that while both P and N are reaching critical levels, inputs of P are rising four times faster than inputs of N.

The scientific community currently argues both for and against promotion of N-limitation in freshwater systems, but slowly gaining acceptance are those associated with the consequences of single-element management in recent scientific literature (Finlay, Small & Sterner, 2013). Schindler et al. (2012) found that although controlling solely P inputs will improve water quality, it is the combined monitoring of P and N that will more successfully recover waterways from eutrophication. Increased N-loading has caused decreased efficiency of denitrifying microbes, thus allowing concentrations of N to surpass N fixing algal requirements (Kelly, Rudd & Schindler, 1990). Paerl et al. (2011) found in eutrophic lakes in various areas across the globe, excessive N loads have already corresponded to rapid increases of non-N fixing bacteria, combined with the already succeeding population of N fixing bacteria. This in turn further promotes growth of toxic algal blooms. Nitrate is also known to be a contributing factor to acidification of freshwater (Kelly et al. 1990). USEPA (2015) has outlined a number of reasons against the conventional single nutrient management model, including: nutrient concentrations vary across a landscape as a result of a multitude of factors therefore the relative contribution of and limitation of N and P can change spatially and temporally; aquatic flora and fauna have a diverse set of nutritional needs; N fixation does not fully offset N deficiency; both N and P have a role in protecting downstream waters; and controlling only P may not effectively prevent the occurrence of harmful algal blooms in freshwaters.

Agricultural practices worldwide have increased to keep up with the ever-growing demand for food. The changing crop requirements combined with intensive fertilizer and manure usage has left large portions of the LWB with noticeable changes in soil nutrient availability. In agricultural lands, there has been a 75 percent increase in soil storage of P alone, compared to soils back in the 1850s (Bennett, Carpenter & Caraco, 2001). Located in a region where about 80 percent of annual runoff takes place as spring snowmelt runoff, this foreseeable, annual trend allows for ways in which nutrient loading can be minimized, without destroying agricultural potential (Liu, Elliott, Lobb, Flaten & Yarotski, 2013).

While agricultural practices are not the only contributing factor to eutrophication of Lake Winnipeg, Liu et al. (2013) suggested that agricultural activity is perhaps the dominant shaping agent of the LWB eutrophication problem. In 2011 alone, 82 percent of the annual total P production, 24.87 million tonnes, was used in fertilizers whereas detergents and feed additives accounted for about 12 percent of total P production (Scholz, Ulrich, Eilittä & Roy, 2013). The destruction of wetlands, poor wastewater treatment and other similar actions are contributing smaller amounts of P and N loading, but will typically require a more specialized, engineering-intensive method of approach for each unique situation to reverse these effects.

1.3 Influence of climate

While studies regarding nutrient export have been conducted in numerous regions throughout the world, it is important to note the differences presented by the climate in the LWB region. Winter temperatures fall well below freezing, posing the need to account for variations in processes due to freeze-thaw cycles and the absence of infiltration while soils are frozen. For example, Tiessen et al. (2010) found that over two four-year study periods, although snowfall accounts for a quarter of annual precipitation in southern Manitoba, snowmelt runoff accounts for nearly 80 to 90 percent of annual runoff. A study by Schindler et al. (2012) yielded similar results, where winter snow throughout the LWB accounted for, on average, 30 percent of annual precipitation, yet 80 percent of annual runoff occurred during the spring snowmelt season. Frozen soils therefore alter the movement of nutrients.

In the LWB, spring snowmelt is not only the dominant source of runoff occurring all at one time, but spring snowmelt often lasts for longer periods of time than rainfall-induced runoff events (Li et al., 2011). During that time, the ground is often semi-impermeable, resulting in widespread prolonged saturation of soils. Li et al. (2011) has shown that per-event nutrient export levels were much greater for snowmelt runoff events than for rainfall events.

This literature review will not consider the implications of climate change, as there is much uncertainty in the literature regarding future changes in the LWB. While there has been less than a ten percent increase in precipitation over the last century, increased intensity of flooding is expected (Schindler et al., 2012). This flooding however will fall within the same spring snowmelt runoff period, and therefore will not drastically alter the methods of reducing nutrient export to waterways.

1.4 Methods

This literature review focuses on scientific peer-reviewed articles regarding sources of P- and N-loading into freshwater systems, specifically looking at the LWB. When information directly pertaining to the LWB region was unavailable, searches were directed towards the North American Great Plains region, where similar freeze-thaw cycles are observed. It is important to consider the implications for nutrient cycles in areas which receive snow cover, as the microbial processes occurring underneath the frozen soil will influence nutrient availability (Shi, Lalande, Hamel & Ziadi, 2015). In high-latitude regions, short growing seasons and frozen soils throughout winter limit decomposition and nutrient turnover (Shi et al., 2015). In some cases where information was still absent, northern regions of Europe were substituted, which allowed comparison in areas with similar temperatures and snow-coverage. If research in these geographically similar areas was altogether absent, information from other regions was considered with caution, and the potential for discrepancies was noted.

Various Best Management Practice manuals were consulted such as the Best Management Practices Environmental Manual for Livestock Producers in Alberta or Agriculture and the Watershed Evaluation of Best Management Practices for South Tobacco Creek, Manitoba (Alberta Agriculture and Rural Development, 2010; Agriculture and Agri-Food Canada, 2013). While topics such as flood management, carbon sequestration, and economic feasibility were beyond the scope of this paper, these may hold additional positive benefits in terms of erosion control, water detainment and even wastewater treatment.

This review closely examines the work previously done by Randall (2011) for the Red Assiniboine Project, information provided by Sealock (2011), as well as the South Nation Conservation *Clean Water Program 2009 Annual Report*, created by the South Nation Conservation Clean Water Committee (2010). Calculations for nutrient reduction estimates are formulated to suit the needs of the LWSBF requirements, which are similar to the approach taken by Sealock (2011). This means calculations will be tailored largely to account for annual P and N removal, rather than daily removal. This review does not represent a complete review of all literature available, but rather is focused on water quality and reducing nutrient transport in both urban and rural management practices, with a focus on P- and N- loading.

2.0 BENEFICIAL MANAGEMENT PRACTICES

With rising global concern over the apparent decline of water quality, much research has been aimed at improving water quality, targeting areas with intense agricultural, livestock, land and water management issues. In an effort to minimize such impacts, an assortment of Beneficial Management Practices (BMPs) have been developed as a reference to improve water quality and long-term health of the environment (Gabel, Wehr & Truhn, 2012). These management approaches act to reduce nutrient loading originating in both urban and rural areas, thus improving water quality. Li et al. (2011) found in a study that nutrient export during snowmelt and rainfall events was reduced during the years where various BMPs were implemented. A combination of five BMPs- a holding pond below an overwintering cattle feedlot, riparian zone, grazing access restriction, forage conversion area and nutrient management- concluded in average reductions of 38 percent of P and 41 percent of N.

While BMPs aid in reducing degradation of water quality, they are merely suggestions and offer guidance to ensure preservation of water. It is up to individuals and landowners to employ and manage these economic both economic and environmental components of sustainability. The Lake Winnipeg Basin Stewardship Fund (LWBSF) offers financial aid to various community-based projects throughout the LWB, which all aim to reduce nutrient loading to Lake Winnipeg through initiative projects, research, or community outreach programs. One of the measures of success for each of the projects is determined from the total P and N load reductions. Without pre-monitoring at each site however, a value may be difficult to obtain. After the discussion BMPs are accompanied by a formula which may be used to create an estimate of how much P and N may be removed.

The BMPs discussed in this report focus on influences from Alberta, Saskatchewan, Manitoba and North Dakota. Although sections of Montana, South Dakota, Minnesota and Ontario are all included in the LWB watershed, it is the previous three provinces and one state which comprises the largest physical portion of the LWB. To put this into context, Evans (2000) stated that the Winnipeg River, which is located along the eastern side of Lake Winnipeg is largely fed by the multiple lake systems of eastern Manitoba and north western Ontario. The Winnipeg River has a drainage basin of 126,400 km^2, whereas the Red River flows right along the North Dakota and Minnesota border, through the capital city of Manitoba, with a drainage basin of 287,500 km^2 and contributes over 50 percent of total annual P to the southern basin of Lake Winnipeg (Evans, 2000). The Prairie landscape in this region is largely dominated by agricultural activities and therefore BMPs must have a focus on these activities.

2.1 Crop Management

With the rising global population, the demand for higher agricultural productivity is ever increasing. It has been well articulated that global food demand will at least double by the year 2050 (Keating, Herrero, Carberry, Gardner & Cole, 2014; Tilman, Balzer, Hill, & Befort, 2011). This makes it more critical than ever to find and maintain sustainable agricultural practices without depleting our soil and water supply. A good harvest requires knowledge of how many nutrients are needed by each crop, as well as implementation of these requirements. Simply put, the crops need sufficient food to produce a satisfying harvest, but each type of crop has different requirements.

Throughout the past 40 years, over application of nutrients to ensure proper nutrition in crops has become a common practice in the farming industry across Canada and the United States (Bennett et al., 2001). While the use of fertilizers, manure and pesticides may certainly ensure high productivity, crops are only able to absorb a certain amount. After the limit is reached, all the excess resonates in the soil until it is swept away by flowing water, wind or soil erosion. This section deals with crop management practices, without discussion of using additional inputs, such as fertilizers or manure (see Section 2.2 and 2.4, respectively). The impact of pesticides will not be discussed in any significant depth, but will be briefly visited with regards to how pesticides interact with N-fixing crops.

2.1.1 Crop Types

Although the LWB is comprised of four provinces and four states, the major agricultural area of the LWB in terms of physical area is located in Alberta, Saskatchewan, Manitoba and North Dakota. It is estimated that Manitoba agriculture alone contributes fifteen percent of current P-loads to Lake Winnipeg (Thomsen, Kulshreshtha, Lobb, Flaten & MacDonald, 2010). Recognizing an approximate P and N requirement for successful growth of crops will improve the balance of nutrients available in soils, thus minimizing occurrences of both under-application and over-application of nutrients (fertilizer or manure). This makes knowing the approximate nutrient removal of common crops important.

Table 1. Major Crops Grown in AB, SK, MB & ND from 2010-2014.

	Oilseeds					Legumes/Pulses			
	Canola	Sunflower	Mustard	Flaxseed	Soybean	Bean	Lentil	Dry Pea	Chick Pea
Alberta [1]	**							✓	
Saskatchewan [2]	✓		✓	✓			✓	✓	✓
Manitoba [3]	✓	✓		✓	✓			✓	
North Dakota [4]	✓	✓				✓	✓	✓	

	Cereal					Forage		Vegetable	
	Corn	Wheat	Oat	Rye	Barley	Alfalfa	Tame Hay	Potato	Sugar Beets
Alberta [1]		✓	✓		✓	✓	✓		
Saskatchewan [2]		**	✓	✓	✓				
Manitoba [3]	✓	**	✓	✓	✓			✓	
North Dakota [4]	✓	**				✓		✓	✓

✓ denotes recognition by provincial/state government as a top crop in acreage

** denotes number one crop in acreage in province/state.

[1] Source: Government of Alberta. (2012). Agriculture Alberta. Retrieved from http://www.albertacanada.com/business/statistics/central-agriculture.aspx

[2] Source: Government of Saskatchewan (2015). Agriculture in Saskatchewan. Retrieved from http://www.agriculture.gov.sk.ca/Default.aspx?DN=7b598e42-c53c-485d-b0dd-e15a36e2785b

[3] Source: Manitoba Agriculture. (2014). Manitoba Crop Highlights. Retrieved from http://www.gov.mb.ca/agriculture/statistics/index.html

[4] Source: North Dakota Department of Agriculture. (2010). *Major Crops and Livestock of North Dakota*. Retrieved from www.nd.gov/ndda/files/resource/agbrochure2010.pdf

Wheat is the number one crop in total farmland acreage in Saskatchewan, Manitoba and North Dakota (Table 1), making it very important to monitor the amount of P and N the crop absorbs throughout growth. That will ensure fertilizer and manure are not over applied. Gao and Grant (2012) calculated that approximately two million hectares of cropland in the Canadian Prairies is being utilized for wheat growth and according to the North Dakota Department of Agriculture

(2010), half of all cropland in North Dakota is used for wheat. While wheat is also grown in Alberta, canola is the crop with the highest acreage throughout the province. Both winter and spring wheat composes the large portions of cropland in Manitoba, Saskatchewan and North Dakota.

While Table 1 depicts the major crops (by area of land) for each province or state, it is by no means a complete representative list of all crops. For the purpose of this report, it is important to remember that in looking at the Basin as a whole, one needs to focus on large acreages of land, as it is those areas that will have the largest impact on reducing nutrients. After seeing which crops are most common, it is possible to focus the development of BMPs regarding those specific crops, as they will likely apply to a larger majority of farmers.

Thomsen et al. (2010) pointed out that crops differ in the amount of ground cover they provide. Crops that provide longer cover, such as forages, will have lower rates of runoff and erosion, and therefore potentially lower P and N export. By matching nutrient uptake to nutrient input (e.g. application of fertilizer, manure, etc.), it is possible to reduce P and N loading and even consume excess nutrients. Some crops have lower efficiency for P and N than others.

In knowing which crops are most commonly grown, it is possible to take a look at nutrient requirements and better predict how many nutrients are being put into the ground compared to how many are generally being taken out by crops. For example, wheat requires far less P than other crops due to the lower protein content in the seed (Alberta Agriculture and Rural Development, 2013). It is possible to observe that wheat can be considered inefficient in utilizing P, as it absorbs a relatively small amount of P in comparison to other crops (Vu et al., Armstrong, Sale & Tang, 2010). If an agricultural land is low in P, farmers know that they may plant wheat and use a low-P content fertilizer or manure and still have a successful crop.

Table 2. Typical P and N Removal by Major Crops Harvested in the LWB.

	Crop	kg/ha	P	N
Oilseeds	Canola	1960	15.7	67.5
	Sunflower	2240	6.9	53.5
	Flaxseed	1492	6.7	51.0
Legumes/Pulses	Bean	3808	26.2	171.0
	Lentil	2016	8.0	61.0
	Dry Pea	3360	14.8	117.0
Cereal	Corn	6272	18.7	97.0
	Winter Wheat	3360	11.0	52.0
	Spring Wheat	2690	10.1	60.0
	Oat	3584	11.0	61.5
	Rye	3450	10.5	58.5
	Barley	4300	14.4	77.5
Forage	Alfalfa	11200	29.7	290.0

Source – Canadian Fertilizer Institute (2001)

Calculations made by the Canadian Fertilizer Institute (2001) differentiate the crop P and N uptake from P and N removal. P and N removal rates are based on values observed in the harvested section of the crop, after the plant has matured (Table 2). Mallarino, Oltmans, Prater, Villavicencio and Thompson (2011) explain that using crop removal is a much more accurate representation of what's left in the soil, and because this report aims to calculate how much P and N is being deterred from entering Lake Winnipeg, it is necessary to use crop removal rates. This is the value which considers the nutrients left in the soil available for runoff, transport or export. Many crops are harvested to a point where some degree of crop residue is left in the ground. This residue contains P and N which is recycled back into the soil and is available for runoff.

Table 2 provides typical values for the major crops grown in the LWB (as mentioned In Table 1), with the exception of mustard, soybeans, chick peas, tame hay, potatoes and sugar beets. Those crops were not listed under the Canadian Fertilizer Institute's calculations for Western Canada and were also classified as major crops in two or less provinces/states. This means they were of low importance in comparison to the other crops throughout the LWB.

A study done by Anthony, Malzer, Sparrow & Zhang (2013) showed that the American guidelines largely overestimated the amount of P actually removed from soils by corn and soybeans in the American Great Plains region. This has ultimately led to a perpetual P storage in the soil. According to Mallarino, Stewart, Baker, Downing and Sawyer (2002), applying P based on numerical standards for how much P is typically removed through crop removal will reduce P-loading in nearby waterways by an average of 0.6 percent, with zero significant impact on annual crop yield. Of course, this assumes optimal soil-test P levels and optimal P uptake by crops. While this may only be a small reduction, it is an easy and convenient method for farmers to cut costs while reducing P and N runoff potential.

2.1.2 Crop Rotation

Crop rotation enhances management of P and N through prevention of export to downstream waterways, as well as prevention of economic loss to farmers. Certain crops may have a higher intake of P or N than others, so planting them in alternating years allows for improved balance of nutrients in the soil, especially in areas where the soil has been perpetually saturated with nutrients. As crop yields are improved due to niche complementarity, this lowers the need for fertilizers and therefore lowers the occurrence of excess nutrients in the soil. This helps minimize runoff into nearby waterways, reducing P and N loading.

Besides the obvious nutrient reduction benefits, crop rotation can also help to reduce diseases and pests. Planting legumes in rotation with cereals tends to help control of weeds, pests and diseases that would normally build up under continuous cereal cropping on large scale farms (Yadav et al., McNeil, Redden & Patil, 2010). Throughout the northern Great American Plains, crop rotation systems have limited pests, enhanced soil N availability and improved soil productivity (Gan et al., 2010). Canadian Prairie farmers have reported a nearly 30 percent reduction in disease where wheat has been in a fallow–wheat–wheat rotation (Kutcher, Johnston, Bailey & Malhi, 2011).

Farmers are heavily influenced by the economy and it is considered cheaper in the short-term by farmers to supplement their soil with additives. Farmers are more likely to plant crops in response to what is economically desired based on predicted market-values than by which would benefit their soil quality most. For example, in the Canadian Prairies, canola holds a high economic value. Although canola harvests after as little as one year of summer fallow tend to be one and a half times greater than when planting continual canola, farmers depend heavily on the immediate economic return seen by yearly canola harvests (Yadav et al., 2010 ; Malhi et al, 2011a).

While it is clear that the benefits of crop rotation ensure less dependence on fertilizer or manure application, farming practices across the globe since the Green Revolution have shifted from crop rotation practises to higher usage of chemical supplements (Dias et al., Dukes & Antunes,, 2015). In response, many farmers tend to plant canola in a two-year rotation, rather than the suggested four-year rotation, with elimination altogether of summer fallows (Yadav et al., 2010). Over half of the canola produced in Manitoba is cropped in a two-year rotation (Yadav et al., 2010).

In the Great Plains where rotation does occur, many cereal-based monoculture systems have been diversified with both oilseed and pulse crops in rotation with cereals (Gan et al., 2010). Throughout Canada, wheat is very commonly rotated with oilseeds or pulse crops (Gao & Grant, 2012). A 4-year study in central Saskatchewan showed that the amount of soil nitrate N throughout different soil layers was greatest with monocultures and the least when canola and peas were in a 4-year rotation with wheat and flax (Malhi et al., 2011a). Canola has a great role in minimizing downward movement of nitrate N in soil and instead draws it upwards for consumption and plant use, however, Malhi et al. (2011a) found that P levels varied depending on crop phase (Malhi et al., 2011a).

Peoples, Gault, Lean, Sykes and Brockwell (1995) found that N-fixation can be one of the most important single sources of N in an agricultural cropping system. When crops are able to form symbiotic relationships with Rhizobia soil bacteria, the bacteria will fix N from the atmosphere solely for the host plant to consume, in exchange for energy from the plant's roots (Fox, Gulledge, Engelhaupt, Burow & McLachlan, 2007). Although, it should be noted that not all plants have the ability to form such symbiotic relationships with Rhizobia soil bacteria, this feature occurs most notably in majority of legumes (St. Luce, 2015 ; Bell, Sparling, Tenuta & Entz, 2012). Brassica crops, along with most oilseeds are not mycorrhizal compatible, meaning they require fertilization, or rotation with mycorrhizal compatible crops due to the very low N mineralization under their roots (Gan et al., 2010).

It was estimated that 50-90 million American dollars annually could be saved by rotating alfalfa and corn throughout the Midwestern United States, simply due to the effective symbiotic N-fixation which would eliminate the need for N supplementation (Fox et al., 2007). For the most common crops grown throughout the Canadian Prairies, Yadav et al. (2010) found that planting fallow will reduce the next year's required amount of N supplementation by 50 to 60 percent. This holds especially true when paired with wheat. The amount of N added into the soil will depend highly on what crops are in rotation, how frequently each crop is planted and the application of synthetic pesticides (Steinshamn et al., 2004).

A study done in the Midwestern U.S. found that pesticides and other synthetic chemicals in the soil will compromise the chemical signaling between host plants and Rhizobia soil bacteria, limiting N fixation (Fox et al., 2007). Typically it is the organochlorine compounds in pesticides which prevent the symbiotic relationship between Rhizobia bacteria and the host plant (Potera, 2007). Little research has been done in the Canadian Prairie region, but similar climate and soils suggests synthetic chemicals would have similar negative effects on N-fixation throughout the Canadian Prairies.

Yadav et al. (2010) predicted that legume rotations in the coming years will become a thriving practice due to the benefits from Rhizobia in soil fertility and ability to minimize environmental impacts. Because the actual reductions of nutrients depend so much on soil quality, moisture content, fertilizer and manure application histories, pesticide use, and so on, it is unrealistic to provide a quantitative reduction formula. Liu et al. (2013) warns that perennial crops may add nutrients to runoff due to the harsh freeze-thaw cycles observed throughout the LWB, as nutrients leak out of the plant during the freezing period. It is however worth mentioning that effective crop rotation will eliminate much of the need for fertilizers and manure application, which in turn reduces the amount of P and N lost to runoff, ending in Lake Winnipeg.

2.2 Fertilizer Management

Intensification of agriculture throughout the 20th century has spiked the usage of synthetic fertilizers (Schindler et al., 2012). Excessive application of fertilizers can cause not only economic loss for farmers, but may cause eutrophication in waterways as they are washed away by overland or subsurface runoff from flooding or heavy rain events. However, P and N remain two of three essential macronutrients for plants which will limit growth if not readily available in sufficient quantities (Chien, Sikora, Gilkes & McLaughlin, 2012).

In Europe, Bennett et al. (2001) found that fertilizers are applied in terms of N requirements rather than considering P levels. This has created a situation where all observed European countries were classified as P accumulators, caused not by natural processes, but by input of fertilizer and manure exceeding crop requirements (Runge-Metzger, 1995). The average P balance of the studied European countries was an annual surplus of 12.8 kg per ha.

The best way to manage fertilizers to reduce both economic loss and environmental degradation is to match specific crop nutrient needs with inputs. This involves managing the rate at which P and N are applied, the timing of application as well as tillage practices. This section will focus on large scale rural operations rather than urban fertilizer usage, however similar practices may be applied to smaller-scale operations with a modified approach.

2.2.1 Application Rate

Excess input of P and N can be avoided by altering how much fertilizer you apply. In the South Tobacco Creek Watershed in south-central Manitoba, the export of excess nutrients represented an annual economic loss of $6.80 per ha, or about 7% of the total annual cost of fertilizer (Khakbazan, Hamilton, Elliott & Yarotski, 2013). Sharpley, Kleinman and McDowell (2001) estimated that only thirty percent of total input of P is actually used by crops. The rest is lost during application, runoff or other mechanisms.

Critical soil-tests for P and N allow farmers to see what nutrients are available in the soil, find the approximate constant P and N concentration for each crop, then find fertilizers which match the needs of that specified crop (Anthony et al., 2013). If critical soil-tests are unable to be performed, having an accurate estimation of how much P and N is removed by a crop (see Section 2.1.1, Table 2) can also ensure the correct amount of fertilizer can then be applied without excess. Fertilizer applications are generally indexed to the N requirement of crops, however it is beneficial to consider P requirements as well (Schindler et al., 2012).

Liu et al. (2013) found reports of a positive correlation between fertilizer rate and stream N concentration. Alternatively, Malhi, Nyborg, Solberg, Dyck and Puurveen (2011b) reported on a long term study in Saskatchewan stating that significant amounts of N had accumulated in the 90-240 cm soil layers. Bennett et al. (2001) found after assessing N application rates compared to potato growth that once the N threshold was reached, applying increased amounts of N did not yield more N in the potato, nor did it increase harvest. It is then reasonable to suggest that once the maximum potential absorption of N for a crop has been reached, the remaining N is simply washed away through runoff processes or seeps into the deeper ground where it is no longer accessible for plant roots.

2.2.2. Application Timing

The freezing winters in the LWB cause a buildup of snow until spring, when temperatures climb and allow for a period of snowmelt. During springmelt, soils are still partially frozen, resulting in flooding, erosion and small losses of particulate P (Liu et al., 2013). As the soils warm up and are moistened, a portion of N is lost due to denitrification (Malhi et al., 2011b).

This period of snowmelt is unavoidable and causes large amounts of surface runoff in the spring, which carries with it many nutrients. This is why it is not a BMP to apply fertilizers in fall or winter in the LWB. Mallarino et al. (2002) found that moving N fertilizer application from fall to spring reduces N in runoff by up to thirty-one percent. Not only is this a source of nutrient loading in streams, but also a costly economic endeavor for farmers when considering how much P and N is lost for such a costly action.

2.3 Livestock Management

Livestock trends since the 1990’s have shifted throughout the Canadian Prairies in response to the removal of the subsidy to farmers for transporting grain to coastal (Schindler et al., 2012). Rather than shipping these grain crops away, farmers are using them to feed livestock, which in response has caused livestock herds to increase dramatically (Schindler et al., 2012). Fadellin and Broadway (2006) stated that the Prairie Provinces, which accounts for majority of the LWB, are responsible for as much as seventy percent of all Canadian raised cattle. Schindler et al. (2012) make the point that one adult hog or cow produces the equivalent of ten humans in terms of excreted P.

According to Statistics Canada (2015), when considering the entire Canadian portion of the LWB, Alberta has the highest beef and sheep inventory, with the second largest hog inventory. Saskatchewan has over 1.2 million beef cows, which accounts for just under 30 percent of the total Canadian beef cows and has about a third of the total Canadian bison herd (Government of Saskatchewan, 2015). In consideration of purely the LWB, Manitoba accounts for the largest hog producing and exporting province, with the third largest cattle operation and ever-growing sheep and goat industries (Statistics Canada, 2015). Even in the North Dakota region of the LWB, we see large amounts of land devoted to livestock, where cattle outnumber humans at a ratio of approximately three to one (North Dakota Department of Agriculture, 2010).

Pasturelands and rangelands are subject to large amounts of degradation through grazing and trampling. Saskatchewan, a province where the economy is largely based upon agriculture, has over six million hectares of pastureland (Government of Saskatchewan, 2015). This makes it important to consider the implications of raising large amounts of livestock within the LWB, which is experiencing already high levels of eutrophication. It is important to note that although feedlots, pastures and rangelands are generally quite different in terms of underlying vegetation, location and purpose, in this section these terms may be used interchangeably, as the key points and calculations are applicable to all three livestock containment structures.

2.3.1 Access restriction

It is becoming increasingly apparent across North America that pastures, rangelands and feedlots should be considered when determining nutrient runoff due to the large volume of excrement left on the soil as a result of these operations (Utah State University, 2012). When open access to a waterway is provided to livestock, this allows the opportunity for contamination through excretion of feces directly in the water, or contamination through runoff processes, due to such close proximity of excreted matter (Larsen, Miner, Buckhouse & Moore, 1994). On average, cattle will spend two to three percent of their total time between 8:30 am to 4:30 pm, from April-October, in-stream (Bond, Sear & Edwards, 2012). Prevention of contamination by restricting livestock access to watercourses using fencing will eliminate direct fecal contamination in a waterway, as well as reduce erosion of streambanks due to trampling (Miller, Chanasyk, Curtis, Entz & Willms, 2010b). While erosion is an additional source of nutrients entering waterways, it will be addressed in another section (see Section 2.6.2).

A study in Southern Alberta by Miller, Chanasyk, Curtis and Willms (2010a), found that adding fencing along a streambank significantly increased vegetation cover and standing litter, while decreasing bare soil by over 70 percent, resulting in P and N load reductions in the stream. Miller et al. (2010a) noted that during their study, the years with higher rainfall observed higher P and N reductions, inferring that rainfall will contribute higher amounts of streambank runoff.

Therefore, it would be worth considering pushing barriers further away from the streambank during those wet years, or in areas prone to intense rainfall events.

Water access restriction methods do not necessarily have to mean fences. Other materials could be used to divert cattle, and depending on the material could be just as effective in deterring cattle from the stream. For example, Rawluk et al. (2014) suggest that dense, thorny hedges or large, rugged boulders serve as an effective barrier, whereas using natural deadfall was not as successful. Cattle eventually maneuvered through the deadfall barrier and ultimately ended up in the stream. Constructed bridges or ramps allow livestock to cross a waterway without directly contaminating it, but of course feces may be deposited on the walkway and eventually be deposited into the waterway (Collins et al., 2007). This method has however been shown in the literature to reduce contamination by some amount.

The amount of P or N in animal excrement is based on an equation which considers total nutrient feed intake subtracted by typical nutrient retention of the animal. This equation, which the American Society of Agricultural and Biological Engineers use, produces rational values to formulate total nutrient content in livestock feces. It is important to note that a beef cow has different nutrient requirements than a lactating dairy cow. Therefore, beef and dairy cows have separate formulas, but will both be considered as 454 kg cattle. It is important to note that values for all types of livestock are based on the average weight of the specified animal and may need to be adjusted if the majority of animals on the specified pasture are larger or smaller than the given weight (Table 3).

Table 3. P and N excretion per day (kg) for the most common livestock found in the LWB.

Total Excretion Per Day	Size of Animal (kg)	P daily (kg)	N daily (kg)
Beef Cow	454	0.04 [1,2]	0.2 [1,2]
Dairy Cow	454	0.03 [2]	0.2 [2]
Swine	200	0.01 [1,2]	0.03 [1,2]
Sheep/ Goat	36	0.003 [1]	0.016 [1]
Horse	500	0.013 [1]	0.091 [1]

[1] Based on USDA (2008)
[2] Based on ASAE (2005)

According to the USDA (2008), a typical beef cow excretes 0.04 kg of P and 0.2 kg of N per day. These numbers were verified by calculations from ASAE (2005) stating that a beef cow excretes 0.04 kg of P and 0.19 kg of N daily. The USDA (1995) calculated that dairy cows excrete 0.02 kg of P and 0.1 kg of N daily, per cow, which is very similar to calculations by ASAE (2005), suggesting that a dairy cow excretes 0.03 kg of P and 0.2 kg of N. It has been argued by many researchers that urea accounts for about three quarters of total excreted N in cattle, and about 80 percent of N in urine is lost by volatilization (Nader, Tate, Atwill & Bushnell, 1998). For the purpose of estimating total reduction in the LWB due to stream access restriction, this potential overestimation will be accepted because in this situation, urine would be directly deposited into the stream with no interception loss from volatilization.

A typical, mature hog weighing 200 kg will excrete approximately 0.01 kg of P and 0.03 kg of N daily (USDA, 2008). These exact values were concluded for hogs as well by the ASAE (2005).

Total manure excretion was stated as 3.8 kg a day per hog, both by the USDA (2008) and ASAE (2005). Sheep excrete a total of 1.44 kg manure a day, with 0.003 kg of P and 0016 kg of N (USDA, 2008). Separate measurements for goats were not available in the scientific literature, but can be assumed to have similar nutrient outputs as sheep based on size and typical diet.

The amount of feces produced by a horse depends largely on its function for the farmer. Sedentary horses will eat less and have lower nutrient requirements, and according to the equation (nutrients taken in subtracted by nutrients retained by the animal), will therefore excrete less. However, for the purpose of estimating nutrients reduced from direct in stream deposition, we will assume all horses to be sedentary as to not overestimate the amount of nutrients diverted from the stream. Based on information given by the USDA (2008), horses excrete about 25 kg of feces daily, with 0.013 kg of P and 0.091 kg of N. While bison are becoming a popular livestock animal throughout Saskatchewan, literature was unavailable regarding the defecation habits of bison, resulting in an absence of values for this livestock (Government of Saskatchewan, 2015).

According to a study by Kohl, Krausman, Kunkel and Williams (2013), cattle spend 45-49 percent of their daily time grazing. This finding was consistent with the grazing pattern of cattle observed in a northeastern Oregon study, where grazing occurred about 50 percent of the time (Ballard & Krueger, 2005). However, Bond et al. (2012) found that per unit time, there is a higher rate of defecating within streams than anywhere else on the pasture. Similar observations were reported by Ballard and Krueger (2005), who saw cattle spending a higher proportion of their time in streams defecating than performing any other activity in comparison to other habitats, such as adjacent to the stream or in the meadow area of the study site.

Bond, Sear and Sykes (2014) found on a pasture in England that cattle produced an average of 10.1 kg of feces each, per day. These results are comparable to a study in the Western United States, which found that cattle could produce on average 13.2 kg of feces daily (Nader et al., 1998). Of the total 10.1 kg of feces produced each day, about 1.2 kg, per cow, was deposited directly in-stream between the grazing months April to October (Bond et al., 2014). It is likely that this number includes feces deposited directly adjacent to the stream, but ended up in the stream as run off, as the methods of data collection did not differentiate the two. Using the North American data for total cattle excrement provides a total of up to nine percent of feces deposited in stream. In the first year of a study by Ballard and Krueger (2005), three percent of total feces were deposited directly in stream, with four percent directly adjacent to the stream. In the second year of this study, two percent was deposited directly in stream with none deposited adjacent (Ballard & Krueger, 2005). Throughout the entire study period, ninety-six percent of defecation occurred in terrestrial habitats, with at least two percent of total feces deposited directly in stream. An older study by Gary, Johnson and Ponce (1983) suggested that five percent of total cattle fecal matter is deposited directly in stream.

The South Nation Conservation *Clean Water Program Annual Report* uses a recently updated value of three percent total defecation deposition occurring directly in stream (Conservation Ontario, 2003). Upon consideration of the difference between each of the abovementioned studies for feces produced by cattle, it is reasonable to accept a value of four percent total daily feces is deposited in stream, or directly adjacent, when direct access is provided. The importance of incorporating feces deposition directly adjacent to the stream comes from the knowledge that fencing or barriers would not be located far enough back to restrict livestock access in these areas, and therefore, through both surface water runoff and through transfer from physical contact of animals stepping on the feces, contamination of water would still occur. This allows justification for claiming total reduction of feces in stream (through access restriction) equal to four percent of the total feces, per animal.

There is a lack of quantitative scientific literature regarding behavior of other livestock (i.e.: swine, chicken, sheep, etc.) in reference to time spent in streams when direct access is provided. Kohl et al. (2013) observed during a study that cattle spent a higher proportion of time at a water source than other livestock. Collins et al. (2007) found that other livestock (i.e.: not cattle) were far less attracted to water bodies compared to alternative water sources, especially sheep and goats. As Bond et al., (2014) mention, cattle feces is highly soluble due to its high water content. While cattle feces contains nearly ninety percent water, sheep feces contains only forty-five to a maximum of seventy percent water (Bond et al., 2014). This makes cattle feces much more susceptible to mixing and transport, causing contamination in waterways rather than settling out. Because of this, the value given by the Lake Simcoe Phosphorous Reduction Calculation Report will be used for all other livestock. This means two percent of total manure from all livestock, excluding cattle, will be assumed to be deposited directly in a stream (Sealock, 2011). Due to the lower solubility of feces and lack of scientific literature on the topic, no additional percent will be allocated due to adjacent deposition in an effort to reduce overestimation and uncertainty.

Table 4. Total P and N deposited in stream, daily, per animal, when direct access is provided.

Animal (typical kg)	P (kg)	N (kg)
Beef Cow	0.0016	0.008
Dairy Cow	0.0012	0.008
Swine	0.0002	0.006
Sheep	0.00006	0.00032
Goat	0.00006	0.00032
Horse	0.00026	0.00182

* Values are based on information provided in Table 3.

The amount of P and N reduced from the waterway will ultimately depend on how many animals are being restricted, how many days they are being restricted for and how much fecal matter is being diverted from the waterway onto the pasture.

Cattle:

Annual P reduction due to restricted cattle access

=

# of cattle x days of restriction x P daily x 0.04
(# of animals) (total days per year) (Table 4) (% in stream)

Annual N reduction due to restricted cattle access

=

# of cattle x days of restriction x N daily x 0.04
(# of animals) (total days per year) (Table 4) (% in stream)

Example calculation:
100 beef cattle restricted for 50 days would reduce P by 8 kg and N by 40 kg.

Other livestock:

Annual P reduction due to restricted livestock access
=
# of livestock x days of restriction x P daily x 0.02
(# of animals) (total days per year) (Table 4) (% in stream)

Annual N reduction due to restricted livestock access
=
# of livestock x days of restriction x N daily x 0.02
(# of animals) (total days per year) (Table 4) (% in stream)

Example calculation:
100 horses restricted for 50 days would reduce P by 1.3 kg and N by 9.1 kg.

2.3.2. Alternative water sources

Providing an alternative water source to livestock minimizes occurrences of contamination in streams. A study by Miller et al. (2010b) showed that by simply adding an alternative water source, without adding fencing, did not immediately reduce the amount of P and N loading to the stream, but did significantly reduce the amount of time cattle spent along the streambank. It was noted that the precipitation pattern over the study period was higher than normal, which could account for the absence of P and N reduction (Miller et al., 2010b). Lardner, Kirychuk, Braul, Willms and Yarotski (2005) reported that cattle avoided water sources which were contaminated with manure when given a choice. This finding was confirmed by Lardner et al. (2013), who compared behaviors of cattle in Saskatchewan when presented the choice between eight different water sources.

Throughout the LWB, dugouts and water troughs are common water storage and access points for cattle, however variability in water quality is a great concern to livestock health and productivity. In the Canadian study by Lardner et al. (2013), it was observed that direct access water sources contained sixty-six times more E. coli than treated dugouts, untreated dugouts, well water within the study site and well water pumped from one km away from the study site. Water from a direct stream source tends to have higher amounts of P than both treated and untreated pumped wells, aerated and coagulated sources (Lardner et al., 2005). Eliminating direct water access can improve livestock performance and weight gains by up to nine percent (Lardner et al., 2005).

A study by Rawluk et al. (2014) found in two southern Manitoba pastures that cattle would use alternative water sources when available, however the usage of an alternate source did not significantly decrease usage of the natural stream. Similarly in Iowa, Haan, Russell, Davis and Morrical (2010) found that cattle would not significantly decrease their time spent in the stream when also using an alternative water source. Viera and Liggins (2002) suggest that cattle do however show some preference to alternate water sources over natural streams. When Miller, Curtis, Bremer, Chanasyk and Willms (2010c) compared groundwater for P and N levels in a Southern Alberta pasture with both a water trough and a natural stream, nutrient enrichment had shifted towards the trough, indicating cattle were spending more time defecating near the trough than in the stream. This suggests lower levels of nutrients may be deposited in the stream.

When alternative water sources are strategically placed in close proximity to shade and shelter, literature shows a significant response by cattle to move away from streams, reducing the time spent at streams and natural waterways (Collins, 2007). Reducing time at these waterways not only reduces direct fecal deposition in the stream, but adds distance between droppings and the stream, which may allow for greater filtration into the soil and vegetation. A study on a pasture in Oregon showed that upon installation of a water trough, the average time for cattle spent in the natural stream was reduced by four minutes daily, or 53 percent (Clawson, 1993). In a different Oregon study, Godwin and Miner (1996) found that cattle reduced their time spent at a stream on average by fifty percent, in favor of the watering tank, located the exact same distance away as the stream. This occurred in a milder climate which typically receives slightly more precipitation than the LWB and less frequent freeze-thaw cycles. Greater reductions were found upon implementation of water troughs in a West Virginia study site through a study by Sheffield, Mostaghimi, Vaughan, Collins and Allen (1997). In the West Virginia pastures, cattle were found using the water troughs and defecating near these troughs, resulting in an 81 and 54 percent reduction in P and N, respectively (Sheffield et al., 1997). The study by Godwin and Miner (1996) suggested alternative water sources reduce the number of defecations directly in stream from once daily to once every four days, with some consideration to rainfall, distance of water source from stream and runoff conditions.

While some studies claim no difference in stream usage when an alternative water source is provided, it is fair to conclude based on the above studies that, assuming location of the alternative water source is at least halfway between the feeding area and stream, it is likely that in-stream defecation will be reduced by fifty percent (Clawson, 1993; Godwin & Miner, 1996). This reduction can be ensured by placing the alternative water source near a shaded area. At the present time, this equation is only to be used for cattle due to the absence of any literature on behavior of other livestock (i.e.: swine, sheep, etc.) when alternative water sources are present.

The amount of P and N reduced from the waterway will then rely on how many animals have access to the alternative water source, how many days they have access to the source and how much fecal matter typically is deposited in the stream. This means a similar equation will be used as the equation for calculating reductions caused by access restriction (fencing). This equation is only to be used when there is no access restriction to streams in place.

Annual P reduction due to alternative water source provided

=

# of cattle	**x**	**days of access to alt. water source**	**x**	**P daily**	**x**	**0.04**	**x**	**0.5**
(# of animals)		(total days per year)		(Table 4)		(% in stream)		(50 % reduction)

Annual N reduction due to alternative water source provided
=
# of cattle x days of access to alt. water source x N daily x 0.04 x 0.5
(# of animals) (total days per year) (Table 4) (% in stream) (50 % reduction)

Example calculation:
100 beef cattle with access to alternative water source for 50 days would reduce P by 4 kg and N by 20 kg. (note: in this scenario, there is no restriction to the on-site stream)

2.3.3. Grazing rotation system

Livestock has a large impact on the plant and soil profiles due to trampling, eating and defecating. The weight of the animals will compact the soil, especially when the plants have been removed and eaten (McLaughlin & Mineau, 1995). Such compaction can potentially increase water runoff, leading to higher nutrient loading. (Nader et al., 1998) found that on average, soil N is affected within an area of 0.1 m^2 of each livestock defecation event. When large congregations of cattle concentrate on one area with favorable vegetation, soil quality and vegetation is largely impacted.

There are two main grazing systems found in the LWB: continuous and rotational grazing. Continuous grazing systems allow cattle to roam over a large area, usually the entire pasture or rangeland, and consume vegetation in varied locations of the animal's choice and leisure (Godwin & Miner, 1996). The selective manner of cattle in a continuous system is often undesirable by farmers as cattle tend to selectively graze favorable vegetation, leading to lowered rates of regrowth and scattered pockets of vegetation with disconnected patches of degraded soil (Bailey & Brown, 2011). Rotational grazing systems use fences, or some other restricting material, to confine cattle and control what areas they can feed on. These systems allow certain areas to be undisturbed for part of the time, creating controllable sections of uniform vegetative health (McLaughlin & Mineau, 1995). Possible benefits of this method might include higher availability of plants to uptake nutrients, as well as minimal soil disturbance in areas left undisturbed for periods of time, allowing for increased infiltration and reduced erosion.

A study by Ellis, Kothmann and Phillips (1994) compared continuous grazing, frontal grazing and rotational grazing for equal period lengths to observe differences in steer gains and herbage left at the end of the study period. Continuous grazing involved letting the steer graze freely throughout the entire grazing pasture whereas rotational grazing allowed steer to only graze in a restricted section for a period of time before moving into a different restricted section. Frontal grazing involved a steer-operated piece of equipment in which their heads would push forward a movable fence to access fresh forage. It was found that there was no significant difference in steer gains at the end of the period, however there was a difference in herbage left. Both frontal and continuous grazing left less total herbage standing at the end of the period than rotational grazing left. McLaughlin and Mineau (1995) also found that using a rotational grazing system slightly improved the soil quality and vegetative health compared to a continuous grazing system, as rotations left time for uniform areas to regrow and filter nutrients.

received rainfall simulations of identical intensity at multiple locations over two different slope heights. What Haan et al. (2006) found was that the mean total P load was greatest from the conventional grazing area.

There is much debate in the literature as to whether rotational grazing truly holds any significant benefit over conventional grazing. Experimental trial results continuously conflict with practical management results and much are focused on benefits to the farmer in terms of animal size, rather than environmental health (Roche, Cutts, Derner, Lubell, & Tate, 2015). Bailey and Brown (2011) make the suggestion that cattle should simply be managed around certain areas during critical points in the season. This would be a form of rotational grazing, where movement of cattle would coincide with seasonality and weather predictions.

Grazing near streambanks and waterways should be restricted during and immediately prior to the wet period of the grazing season, or even simply prior to and during rainfall events (Bailey & Brown, 2011). This idea is explained by Collins (2007), who says because these soils are already prone to saturation during rainfall events, removing the vegetation through grazing would significantly increase runoff into waterways, thus increasing nutrient export and fecal runoff into the water. This idea was confirmed by Jacobo, Rodríguez, Bartoloni and Deregibus (2006) through a study in an Argentine area prone to flooding. Haan et al. (2006) also agreed with this idea through a study in Iowa, suggesting avoiding grazing in areas prone to saturation or surface water runoff in the spring, as this is the wettest season in the LWB. Haan et al. (2006) makes the valid point of considering slopes as well when developing grazing systems, as surface water runoff will be greater along a slope, increasing the need for vegetation to intercept nutrients.

Due to a lack of scientific exploration regarding P and N runoff in areas of different grazing systems, no quantitative reduction to nutrient loading in waterways can be formulated. It should be noted that rotational grazing based around seasonality, in areas where soils are typically saturated during rainfall events, has the greatest potential of all grazing systems to reduce P and N in nearby waterways. Careful consideration when selecting grazing systems, in combination with the two previously mentioned livestock management BMPs, will ultimately ensure the greatest reduction in P and N. Olson, Kalischuk, Casson and Phelan (2011) found that by adding restrictive fencing, implementing off-stream watering and relocating a winter feeding lot for cattle, significant amounts of P and N were reduced from entering the waterway over a four year period in Southern Alberta. This suggests that implementing multiple livestock management BMPs will have the greatest success in reducing nutrient loading to nearby waterways.

2.3.4. Overwinter feeding systems

Because the highest percentage of runoff throughout the LWB occurs in the spring from snowmelt (Elliott, 2013 ; Cade-Menun et al., 2013), winter feeding within pastures poses the risk of increased nutrient export, as defecated matter can be carried away as the snow underneath it melts. While feeding directly on the pasture allows higher nutrient recycling in a more cost-efficient manner and can pose a greater benefit than artificially spreading manure and soil enhancements, this can mean higher export rates than the conventional winter feedlot system (Jungnitsch, Schoenau, Lardner & Jefferson, 2011). Rather than spending money on overwinter manure storage facilities, which are typically associated with some amount of loss due to failure or malfuncation of the structure, as well as spending money on application of the manure come spring time, the in-field overwintering feeding system is being adopted by many cow-calf farmers (Smith, Schoenau, Lardner & Elliott, 2011).

While economic benefits are apparent through overwintering feeding systems, the environmental effects regarding allocation of nutrients come spring time is less clear (Smith et al., 2011). In each Canadian province within the LWB, spring snowmelt export rates will, on average, account for at least 70 percent of total annual runoff (Elliott, 2013). Allowing cattle to defecate directly on the frozen surface throughout winter poses the risk of adding an additional nutrient source to the highest runoff event of the year. Smith et al. (2011) indicated that initial research regarding the fate of nutrients through overwintering feeding systems had high rates of retention within the field and very little export through runoff when compared to storage of manure in containment facilities followed by spring application. However, through a study of their own, it was found that nutrient concentrations in the snowmelt runoff were significantly higher where winter fed livestock had defecated compared to a control site without overwinter feeding.

An experiment by Jungnitsch et al. (2011) in Lanigan, Saskatchewan compared winter feeding hay in bales directly on a pasture with conventional feeding in a drylot pen. P and N were then measured the following spring on both the control pasture and the winter fed pasture. N was 3 to 3.7 times greater on the winter fed pasture with 117 kg ha^{-1} more on average on the winter fed pasture than the control pasture. 20 percent of P and 30 percent of N was recovered in forage throughout the winter fed pasture, but only 3 percent of P and 1 percent of N was estimated to be recovered in the forage from the winter-excluded grazing system. While this study did not examine nutrient losses due to snowmelt runoff, it does highlight the high amounts of nutrients deposited on the surface from overwinter feeding.

A study by Smith et al. (2011) in Saskatchewan divided a study area into 5 sections- 1 control site and 4 overwinter feeding sites. Water samples were collected from a total of 10 basins surrounding the overwinter and control sites daily until the spring snowmelt was complete. Results showed that orthophosphate-P concentrations from snowmelt runoff in the spring time were over 10 mg L^{-1} higher in all 4 overwinter feeding sites than the control site and ammonium-N concentrations were nearly 40 mg L^{-1} higher. There was no significant difference in water extractable P nor was there a significant difference in nitrate-N. The absence of a difference in nitrate-N was likely due to the cool temperatures which limited microbial conversion of ammonium to nitrate (Smith et al., 2011).

Smith et al. (2011) recommended further research to focus studies on long-term fate of nutrients, as much of the previous studies were extremely short term, not evaluating the composition of snowmelt runoff water. Further research would help to conclusively determine whether overwinter feeding systems for livestock on pastures truly aid in reducing nutrient export into waterways. To combat the problems associated with snowmelt runoff, yet still capture the beneficial nutrient recycling, Jungnitsch et al. (2011) suggested a reduced stocking density of livestock on the pasture throughout winter to allow better control of defecation. Smith et al. (2011) recommended using sites with crop species that utilize nutrients in the extremely early spring to reduce losses from late spring snowstorms and rainstorm events. Smith et al. (2011) also noted that sites should be placed away from sensitive water bodies, or containment basins should be used to trap runoff prior to entrance into a main water body. While using overwinter feeding for livestock, caution should be used to ensure nutrients are contained on the pasture, rather than being swept away in the spring snowmelt.

2.4 Manure Management

It has been estimated that 7,000 tonnes of P per year is spread on agricultural lands through manure application in the southern LWB (Schindler et al., 2012). When excess manure is applied to the surface, there is great risk of nutrient export and runoff into nearby waterways. To prevent this economic loss and environmental harm, it is important to consider how much manure is actually needed by the soils and vegetation. Other variables to be considered as well include things such as the slope of the surface, the saturation of soils, and the season or weather conditions, as these variables will increase the amount of nutrient runoff (McKenzie, Dormaar, Adams & Willms, 2003).

Because the growth of crops is so dependent on N availability in soils, although P and potassium (K) are also macronutrients required by plants, most fertilizers and manure are measured for their N properties. In fact, Sharpley et al. (2001) explain that manures are applied at rates which meet N requirements to ensure optimal plant growth, which results in P build-up far beyond required amounts, therefore increasing P runoff potential. If N was already available due to earlier applications in the season, or if the present soil contents of N are underestimated, manured crops will receive too much N with improper N:P ratios (Schröder, Uenk & Hilhorst, 2007). When excess levels of P and N accumulate in soil, these excess nutrients will seep downwards into the groundwater and potentially contaminate nearby waterways. McKenzie et al. (2003) suggested that excess nutrient availability in soils can potentially attract non-native plant species, or ‘pest’ species (weeds), at the cost of the native or desired plant.

Only forty percent of all livestock excrement is recovered and applied to crops as manure, with the remainder lost to volatilization, failure to collect or contain all excrement, or other nearly uncontrollable events (Oenema, Oudendag & Velthof, 2007). (Nader et al., 1998) explains that up to eighty percent of N in recovered manure may be lost due to volatilization. Through a simulated feedlot study, (Nader et al., 1998) actually found that after twenty four hours, half of the initially measured N was lost from the total supply of manure. McKenzie et al. (2003) replicated these results, also observing a fifty percent loss to volatilization. This is why it is critically important to ensure manure is stored properly and incorporated in a timely manner into the soil to reduce losses.

To prevent pollution of water in Canada, each provincial government is responsible for setting guidelines and legislation regarding proper manure management. Manitoba, for example, has set a ban on applying livestock manure on agricultural land between November 10 and April 10 unless otherwise noted, and requires all manure to be incorporated within 48 hours of application (Khakbazan et al., 2013). Manure management requires consideration of a multitude of factors. While it is important to recognize the differing nutrient content within manure from different types of livestock, it is important to recognize the BMPs for storing manure, the rate and frequency of application, the seasonal timing of application, and the method in which manure is applied to the desired area. This section will simply discuss practices to reduce P and N runoff into water and provide quantitative measures where applicable, but relevant legislation must be considered upon implementation.

2.4.1. Livestock source

2.4.1.1 Dry vs. wet

Livestock excrete manure in both liquid and solid form. It is important to recognize that the terms ‘dry’ and ‘wet’ may very slightly when discussing manure. ‘Dry’ manure consists of all solid manure, while the term ‘wet’ consists of all slurry or urine (Oenema et al., 2007). Whether dry or wet, the manure types potentially hold different nutrient ratios, and require different storage methods. Liquid manure has a significantly higher concentration of ammonium-N due to high urine content, and therefore N losses in liquid manure are much more economically costly to a farmer (Alberta Agriculture and Rural Development, 2010).

Beef cattle excrete thirteen kg daily, two kg of which is dry content, with the remaining eleven kg at a moisture level which classifies as wet manure (Tate & Bushnell, 1998). Nader et al. (1998) found that of all cattle excrement that was produced in a study, on average, between fifteen to seventeen percent was dry, which agrees with results from Tate and Bushnell (1998). Nader et al. (1998) found that many studies had shown urea, which largely composes wet manure, accounts for seventy-five percent of total excreted N. Over ninety-five percent of total excreted P is found in solid manure (Nader et al., 1998).

Oenema et al. (2007) estimate that throughout the European Union, sixty to seventy percent of all livestock excrement is collected in housing systems, whereas the remaining thirty to forty percent falls from grazers in pastures or rangelands and is left unmanaged by farmers. More than half of the excrement collected in housing systems comes in liquid form, whereas virtually none of the liquid manure is collected from a pasture, rangeland or feedlot (Oenema et al., 2007).

2.4.1.2 Animal source

Depending on which livestock the manure originates from, not just the quantity of manure produced will vary, but the levels of P and N will differ based on what the animal has consumed and absorbs. For example, a growing pig only absorbs and utilizes about thirty percent of total ingested P, leaving the rest to be excreted and used as manure (Huang, Ackerman & Cicek, 2013). Cattle typically excrete seventy-five percent of the total ingested N as a mixture of solid and liquid manure (Nader et al., 1998). When considering the countries in the European Union, it is interesting to note that out of all the manure used in the year 2000, fifty-seven percent came from cattle, eighteen percent came from poultry and seventeen percent came from swine (Oenema, Witzke, Klimont, Lesschen & Velthof, 2009).

Table 5. Daily feces weight breakdown per animal, per day in kg.

Animal	Typical Animal Size (kg)	Total Feces Produced (kg)	P (kg)	N (kg)
Beef Cow	454	13.2 [1]	0.04 [2]	0.2 [2]
Dairy Cow	454	13.2 [1]	0.03 [2]	0.2 [2]
Swine	200	3.8 [2,3]	0.01 [2]	0.03 [2]
Layers	2	0.03 [4]	0.0003 [4]	0.0008 [4]
Broilers	1	0.02 [4]	0.0002 [4]	0.0005 [4]
Sheep	36	1.44 [2]	0.003 [3]	0.016 [3]
Goat	36	1.44 [5]	0.003 [5]	0.016 [5]
Horse	500	25 [2]	0.013 [3]	0.091 [3]

[1] Based on Nader et al. (1998)
[2] Based on USDA (2008)
[3] Based on ASAE (2005)
[4] Based on USDA (1995)
[5] Based on calculations for 'sheep'

2.4.1.3 Diet modification

Typically, any given cereal grain feed for livestock contains about fifty to eighty percent phytate (McGrath et al., 2005). Phytate is a form of P which is indigestible to monogastric animals, meaning only ruminants are able to break down this form of the nutrient (Sharpley et al., 2001). All indigestible P is then excreted, causing P buildup in manure and litter. Pig, poultry and horse feed is commonly supplemented with more readily digestible forms of P to allow the animal full intake of required nutrients for growth (Sharpley et al., 2001).

Supplementing feed with additional nutrients to ensure animals receive an optimum amount of nutrients means large amounts of P are being wasted. The high concentration of P in this manure is causing higher P runoff rates when the manure is applied on fields (Yan, Kersey & Waldroup, 2001). Rather than using P additives in feed, an effective way to reduce the amount of undigested P excreted in litter in these monogastric animals is to add phytase enzymes to cereal grain feed (McGrath et al., 2005). This enzyme is effective in releasing thirty to fifty percent of otherwise unavailable P for monogastric animals (McGrath et al., 2005). This not only ensures animals are getting a sufficient supply of P, but reduces excess P in manure and over application of P to fields.

Yan et al. (2001) suggested that addition of the phytase enzyme to bird feed (poultry feed, turkey feed) will reduce P in fecal matter by thirty percent on average. Yi, Kornegay, Ravindran, Lindemann and Wilson (1996) found a twenty-five to fifty percent reduction fecal P content with addition of phytase and reduction of supplementary P to swine feed. Harper, Kornegay and Schell (1997) found an estimated P decrease of 21.5 percent in swine fecal matter after adding five hundred units of phytase per kilogram of feed. When Smith, Moore, Maxwell, Haggard and Daniel (2004a) supplemented swine feed with phytase, it was calculated that soluble reactive phosphorus in manure was reduced by seventeen percent. Smith, Moore, Miles, Haggard and Daniel (2004b) found adding phytase to poultry feed reduced dissolved P in manure by ten percent.

Maguire, Crouse and Hodges (2007) reviewed literature and found that by modifying livestock diets to more closely follow specific P requirements and including a phytase supplement, manure P could be reduced by forty percent for poultry, by fifty percent for swine and by twenty-five percent for dairy cattle, compared to typical USDA manure P values. By using a combination of phytase and aluminum chloride in swine feed, P runoff was reduced by over fifty percent (Smith et al., 2004a). A seventy-four percent reduction of dissolved P runoff was observed after using a combination of aluminum chloride, phytase and a high available P corn diet.

Simply feeding livestock a lower-P diet has been shown to reduce fecal P content, however it also results in an average fifteen to twenty percent reduction in growth and efficiency (Harper et al., 1997). In an American study, it was reported that implementing diet modification for livestock reduced the number of counties experiencing a large manure P surplus from eleven percent to eight percent throughout the country (Maguire et al., 2007). Other diet modification methods exist which have been shown to reduce P in fecal matter, but they may not be as effective when considering economic needs. Maguire et al. (2007) suggest using corn and soybean diets instead of cereal grain feed because corn and soybeans have high amounts of digestible P for monogastric animals. Smith et al. (2004b) found that feeding poultry high available P corn resulted in a reduction of dissolved P in manure by thirty-five percent, compared to manure from a control group. Typically however, in the Canadian Prairies cereal grain is the cheapest and most available form of livestock feed (Schindler et al., 2012). Adding phytase may be a more beneficial option in terms of balancing economic and environmental needs in the LWB.

2.4.2. Storage

Huang et al. (2013) indicated that typically across the Canadian Prairies, fresh manure is seldom applied on land directly, but is held in storage and applied once or twice per year. Across the Prairies, where snow covers land for up to half the year, storage facilities must be large enough to contain all of this manure. Studies in Alberta have suggested that due to the large cost associated with building storage facilities, some farmers elect to repeated application of large quantities of manure or simply stockpile manure in an unused corner of land for usage at a later time (Olson, McKenzie, Larney, & Bremer, 2010). Of course when manure is not stored in a sheltered area, it can be degraded and washed away, resulting in high levels of nutrients in the run-off. Interestingly enough, Oenema et al. (2007) calculated that throughout the European Union, typically only fifty-two percent of total N excreted by livestock is actually properly contained and available for re-application back onto the farmyard.

The South Nation River Conservation Authority estimated four percent loss of P can be prevented from run-off through containing manure by using sufficient storage (Sealock, 2011). Changes in P solubility occur throughout the storage period, so when manure is to be stored, it is best to keep it as dry as possible to limit dissolved P loss from runoff after manure application to fields (McGrath et al., 2005).

A study Shah, Groot, Oenema and Lantinga (2012) compared N loss in piles of solid cattle manure in three different storage methods; covered, composted and stockpiled. During the one hundred thirty day storage period, ten percent of the initial N was lost in the covered pile, but losses were over thirty and forty percent, respectively, for composted and stockpiled methods. It is assumed that the airtight plastic covering on the covered pile acted to reduce volatilization and eliminate run-off of N (Shah et al., 2012). Volatilization during storage accounted for over twenty percent of total N loss in a study by Oenema et al. (2007), with an additional four percent loss due to leaching and run-off.

Considering the various storage methods and the losses associated with improper storage, it is reasonable to say that four percent of P and N can be prevented from entering nearby waterways if sufficient storage space is available and utilized. Because these nutrient loss reductions depend on manure avoiding contact with rain, the storage method must involve a roof to eliminate run-off and reduce potential for additional moisture added to the manure. Based on higher water content in wet manure, it is reasonable to assume higher reductions will occur when proper containment is available to prevent leaching into soils and runoff, however separate quantitative data in the literature varied greatly. To prevent overestimation of nutrient reduction, the same equation will be used, regardless of the moisture content of the manure.

The amount of P and N reduced from the waterway through run-off while in storage will ultimately depend on how many animals there are and how many days of manure is being stored. It is important to note that there are regulations in place in each province and state which determine whether or not manure storage facilities should be granted access. This equation in no way stands to overrule the regulation, but serves as a reference in calculating reductions of P and N.

Annual P reduction due to properly stored manure

=

# of livestock	**x**	**days of collection**	**x**	**P daily**	**x**	**0.04**
(# of animals)		(total days per year in which manure is contained/ stored)		(Table 5)		(% eliminated from run-off)

Annual N reduction due to properly stored manure

=

# of livestock	**x**	**days of collection**	**x**	**N daily**	**x**	**0.04**
(# of animals)		(total days per year in which manure is contained/ stored)		(Table 5)		(% eliminated from run-off)

Example calculation:
Proper storage of manure from 50 beef cattle for 100 days would reduce P by 8 kg and N by 40 kg.

2.4.3. Application rate

Manure application rates should follow uptake needs of the plant (see Section 2.1, Table 2). It can be difficult to estimate the physical nutrient content of manure if the instruments or tools are not available. For example, in Europe, surplus P in soil based on agricultural crop activity and manure application averages from 5 kg P ha^{-1} in the eastern half to upwards of 20 kg P ha^{-1} in parts of the western half (Schröder, Smit, Cordell & Rosemarin, 2011). Manure transportation is extremely expensive for farmers, and in an Alberta survey, Olson et al. (2010) found that many farmers would apply manure to land at a minimum distance from the source. This meant some areas receive repeated surplus amounts of P and N easily available for runoff transport.

When significant excess manure is applied to an area, it will seep past the water table into the groundwater (McKenzie et al., 2003). If the excess nutrients do not move downward with gravity, those extra available nutrients not used by plants are then left in the soil for any rainfall event to potentially carry away these nutrients as surface runoff (McKenzie et al., 2003). While a study by Wen, Schoenau, Charles and Inanaga (2003) observed increased N uptake in canola in response to increases in rate of manure application, many other studies have found a positive relationship between rate of manure application and P in runoff (Volf, Ontkean, Bennett, Chanasyk & Miller, 2007).

In the Canadian Prairies, the short growing season and low precipitation act to slow decomposition of manure after application into the soil (Wen et al., 2003). This allows much of the N to be retained in the soil which means some of it is still available for the next growing year (Wen et al., 2003). This is true for P as well. By considering residual P and N levels in the soil, less P and N fertilizer and manure is actually required in following years to meet the needs of crops. This can reduce occurrences of P and N runoff.

Table 6. Residual P and N % availability after manure application for three consecutive years.

	Year 1	Year 2	Year 3
Phosphorus	70	20	10
Nitrogen	25	13	7

Source: Adapted from Olson, McKenzie, Larney and Bremer (2010).

Olson et al. (2010) performed a literature review and found that after application of manure, the residual available P and N content in soil for three years after application was just less than half what was available the previous year (Table 6). This of course takes into consideration an initial fifty percent loss of N due to volatilization during collection of manure and application procedures which Olson et al. (2010) suggested. Nader et al. (1998) found in a simulation study that as much as ninety percent of N was lost in liquid manure, however accepted a fifty percent loss as this is more realistic to occur under typical conditions. An experiment by Angers, Chantigny, MacDonald, Rochette and Côté (2010) measured measured N movements after manure application and traced soil retention of N ranging from fifteen to fifty percent, whereas total forage uptake accounted for thirty to fifty percent. The significant unaccounted amount of N was assumed to have been lost due to volatilization and runoff.

2.4.4. Application timing

Winter application of manure is not effective as soils are typically frozen and manure is washed away with the spring snowmelt runoff (Alberta Agriculture and Rural Development, 2010). Because the entire LWB area is affected by snow cover for the winter season, management strategies have restricted or eliminated the allowance of spreading manure throughout the winter season. In Manitoba, for example, manure generated from large-scale livestock operations cannot be applied between November 10 and April 10 of the following year (Khakbazan et al., 2013). In many other Canadian provinces, areas have been deemed 'safe' for winter spreading, or less prone to runoff based on proximity to a natural flood plain (Srinivasan, Bryant, Callahan & Weld, 2006). Because livestock continually produce excrement, this restriction on spreading throughout winter is why many livestock farms must have manure storage containers (see Section 2.4.2). It is best to apply manure when soils are less prone to compaction due to snowmelt or heavy precipitation, as this will result in runoff of nutrients; an economic loss to farmers and an environmental concern as well (Alberta Agriculture and Rural Development, 2010).

2.4.5. Application method

There is much debate in the literature regarding tillage practises of crop land and whether they are more detrimental or beneficial to nutrient movement into waterways. Incorporating manure is similar to tillage, however the benefit lies with the movement of large amounts of supplementary P and N, which would otherwise stand on the surface of the soil as manure. Incorporation of fresh manure has been shown to reduce N losses and improve nutrient uptake by crops (Volf et al., 2007).

Traditionally, BMPs have included manure incorporation through some method within the first few days of application to minimize N losses and reduce the odor (McKenzie et al., 2003). In Manitoba, it is required that any surface applied manure is incorporated into the soil within 48 hours of application (Khakbazan et al., 2013). Manure incorporation has the added benefit of prolonging time it takes for P and N to runoff, which minimizes water contamination during spontaneous rainfall events (Daverede et al., 2004).

Daverede et al. (2004) found that injecting swine liquid manure in a northwestern Illinois study site caused ninety percent reductions in immediate runoff and concentration of dissolved reactive P, total P and algal-available P when compared to surface-applied swine liquid manure runoff. Sommer and Hutchings (2001) found that injection of liquid manure and incorporation of solid manure both reduce ammonia loss compared to broadcast spreading by ninety percent. Van der Weerde, Luo, Dexter and Rutherford (2014) indicated that ammonia (a form of N) loss is greatest after the first twenty four hours of application. In a longer-term, whole farm simulation where manure was incorporated, total P runoff reductions were not as substantial at only eight percent, as greater erosion occurred where fields were previously under a no-till system (García, Veith, Kleinman, Rotz & Saporito, 2008).

2.5 Land Management

Water quality throughout the LWB is impacted largely by the activities that occur on the land, including agricultural and urban activities, as well as natural events. A portion of whatever nutrients, chemicals and substances that are stored or applied on land within the LWB can be transported by water in surface and subsurface runoff, ultimately reaching Lake Winnipeg. Because the land in the LWB has such high agricultural potential, many original wetlands have been converted into agricultural land (Gleason, Euliss, Tangen, Laubhan & Browne, 2011). In the Canadian Prairie provinces, land use has changed the greatest in Saskatchewan over the past 40 years, where up to 20 percent of total land area has been converted from summer fallow and forage cropland, where vegetation was left untouched for a minimum of one year, to intensive cropping with annual harvests (Betts, Desjardins, Worth & Cerkowniak, 2013). A significant portion (80 percent) of stream ecosystems in the western U. S. have been damaged due to livestock grazing and contamination, with over 89 percent of riparian areas damaged in Alberta (LaRocque, 2014). Improper management of land has led to the degradation of these streams and riparian ecosystems.

In the U.S. portion of the LWB alone, over half of the original wetlands have been eliminated (Schindler et al., 2012). It has been suggested by Dong et al. (2014) that in the American Ozarks, areas in close proximity to agricultural catchments have been modified and now have a significantly lower P retention ability compared to their prior states. In areas adjacent to agricultural land, much of the native vegetation is being removed and the runoff catchment areas are becoming increasingly urbanized. This trend is not exclusive to the Ozarks. On a smaller scale, this is what is occurring along the Red River where we see extreme P and N loading. Especially within suburban and urban settings, riparian corridors are becoming increasingly fragmented, with intense modifications caused by increasing presence of concrete and other impermeable surfaces (Timm, Wissmar, Small, Leschine & Lucchetti, 2004).

Understanding the regions in the LWB where these nutrients may be absorbed or retained is of great benefit, as we can create areas that absorb and extract P and N to intercept runoff (Wang, Li, Zhu & Zhang, 2012). According to Koontz, Pezeshki and DeLaune (2014), wetlands throughout the world have the potential to filter and retain 80 percent of incoming nutrients, however due to wetland destruction and draining over the past two decades, instead we are observing increased nutrient loading in our waterways. Conversion of cropland to perennial cover surrounding a wetland in the Prairie Pothole Region reduced P and N runoff rates on average by a significant amount, according to a study by Tangen and Gleason (2008). This study suggested a loss of up to 75 tonnes yr^{-1} of P and 5,622 tonnes yr^{-1} of N annually were prevented by restoring 2,200,000 ha of wetlands. Recognizing the benefits of replacing impervious surfaces with wetlands and native grasses is becoming more common, especially once formulas are made available to make an approximation of how much P and N will be reduced. As the value of a healthy ecosystem is being more widely recognized by society, it is becoming quite clear that improved land management will help to restore the declining quality of our valuable freshwater systems (Johnson, Polasky, Nelson & Pennington, 2012).

2.5.1. Tillage

There are three main approaches with altering levels of soil manipulation when it comes to tillage: no-till systems, conservation tillage and conventional tillage. Under no-till systems, the soil is not worked or ploughed between harvest and planting season, but is left as is following harvest. Conservation tillage involves assimilation of less than 30 percent of the surface residue into deeper soil layers (Lal, 2003). Conventional tillage consists of the most vigorous mixing of soil layers, as more than 30 percent of crop residue is worked into lower layers following harvest, or prior to planting season (Lal, 2003). Across western Canada, no-till systems have become increasingly prevalent (Kutcher et al., 2011). In 2005, no till systems accounted for half of the farms throughout the Canadian Prairies, while conservation tillage accounted for 26 percent (Shrestha et al., 2014). Recently, there has been much debate in literature as to whether no-till practices or conventional practices are responsible for greater reductions in nutrient transport.

Tillage affects the management of crop residue, which has various influences on the soil below. Many studies suggest greater accumulation of snow occurs in areas of no tillage compared to conventional tillage, as snow drifts are more likely to pile near the clumps of untouched crop residue. Higher snow accumulation acts to insulate the surface layer, causing a temperature gradient throughout subsequent layers (Shi et al., 2015). This would also mean higher rates of surface runoff come spring time due to the higher amount of snow to contribute to snowmelt. However, a study in Southern Manitoba by Tiessen et al. (2010) shows that snow does not always accumulate in higher amounts around crop residue.

What has been shown by multiple studies is that no-till systems hold large amounts of crop residue along the soil surface in which P leaches from and accumulates in the top layers of soil (Khakbazan et al., 2013). This residue also acts to increase accumulation of C and N in the surface layer as the plant matter decomposes throughout the winter (Shi et al., 2015). This process at the surface is responsible for increased nitrous oxide emissions in the spring once the snow has cleared (Shrestha et al., 2014). The layer of crop residue creates stratification of both nutrients and microbial biomass in the soil. Liu et al. (2013) found in a cold climate region where multiple freeze-thaw cycles frequently occur, the contribution of nutrients in plant residue increases when the plant freezes. This means tillage may be an important consideration to incorporate residue deeper within the soil to prevent runoff from carrying these excess nutrients released from crop residue after spring thaw.

Messiga et al. (2012) found that under no-till practices, soil experienced a shortage of P in lower soil levels where plant roots develop. This was due to the stratification caused by the absence of mixing, which lead to an accumulation of P near the surface layer rather than at a depth useful to plant roots. In order to ensure plants receive enough P, fertilizer or manure must be applied, adding to the amount of P available for runoff. Without mixing the surface layer into deeper layers, soluble P builds up near the surface, available for runoff transport. Schindler et al. (2012) found on the Canadian Prairies that up to 80 percent of P in small tributaries is in soluble form, which is the form most readily available for usage by cyanobacteria. This means when considering tillage systems as contributors, no-till systems are potentially responsible for higher proportions of P runoff than conventional or conservation till systems.

Of course, conservation and conventional tillage has been shown in some studies to provide benefits such as reduced soil erosion, conservation of soil moisture as well as maintaining optimal soil structure in layers (Shrestha et al., 2014). However, Tiessen et al. (2010) found in a study conducted in Manitoba that although conservation tillage may reduce N runoff by 68 percent, P exports were actually 12 percent higher than conventional tillage. This study considered effects of both snowmelt and rainfall-induced runoff conditions. While reducing tillage intensity may have been shown to significantly reduce total P loss by up to ninety percent in various studies in the southern United States, the reduction of mixing causes soluble P to build up in the surface soil, which is readily available for transport as runoff during rain events (Mallarino et al., 2002 ; Tiessen et al. (2010) ; Schindler et al., 2012).

The South Nation Conservation Program determines conservation tillage to be a BMP with a P-load reduction of 0.75 kg per ha of land. However with such vast discrepancies in the literature, it is reasonable to suggest the benefits of one tillage practice do not consistently outweigh benefits presented with a different tillage practice. Further, more comprehensive and conclusive studies need to be examined in the LWB to conclude whether no-till, conservation tillage or conventional tillage provides the largest reduction of nutrient loading to nearby waterways.

2.5.2. Wetland areas

Vast acerages of original wetland areas in the LWB have been drained for conversion to agricultural land, making land management of wetlands a critical part in reducing P and N loads to Lake Winnipeg. This section will evaluate the influence of wetlands in retaining and sequestering P and N from agriculturally-dominated landscapes. It will exclude constructed wetlands in urban wastewater treatment centers because urban wastewater has entirely different rates of loading, as well as different nutrient concentrations, and therefore would have different nutrient contribution rates than that of an agricultural area (see Section 2.7).

Wetlands are not dependent on proximity to a waterway like riparian areas are, but they do indeed encapsulate large quantities of water and vegetation for varying amounts of time. According to Hilliard and Reedyk (2014), a riparian area is the area of transition between terrestrial and aquatic zones. Wetlands may or may not be adjacent to an aquatic zone and therefore may or may not be fueled by flooding. According to the USEPA (2005), a wetland is saturated by surface or ground water, whereas the soils in a riparian area are typically saturated from the influence of the adjacent water body. Wetlands have a specifically adapted set of soils, substrates and biota which thrive under conditions of frequent flooding and waterlogged soils (Vymazal, 2011). Due to the low costs of implementing a constructed wetland, they are becoming increasingly popular for catching agricultural runoff in rural areas (Gottschall, Boutin, Crolla, Kinsley & Champagne, 2007).

Constructed wetlands can reduce soil and sediment loss from erosion by a significant amount, which in turn reduces transport of particulate P and N (Gleason et al., 2011). Wetlands allow large volumes of water to seep into the porous ground and will act as a storage area for water (Halabuk, 2006). Because much of the LWB experiences a large, annual spring snowmelt runoff event, water storage helps slow the movement of water off the land, reducing nutrients ending up in waterways in the LWB. This allows time for nutrients to settle to the bottom and infiltrate out of the water and into the wetland, rather than flowing into a tributary in the LWB (Gleason et al., 2011).

There is debate in the scientific literature as to whether wetlands truly are a sink or are a source of P and N. Often, research regarding the effectiveness of wetlands in reducing nutrient loadings, especially P and N, shows some inconsistencies (Palmer-Felgate et al., 2013). The minimal research in cold climates also throws into question the effectiveness of wetlands in reducing P and N loadings. Because wetlands depend on vegetation and soils to remove nutrients, during the winter months, the removal process is significantly reduced. Some farmers in cold climates will store livestock wastewater until spring when it can be treated by such wetlands (Smith, Gordon, Madani & Stratton, 2006). This is very costly for operators, so not all farmers practice this option. There is a need for further research to develop more conclusive answers regarding the effectiveness of wetlands in cold climates.

As absorption and release of nutrients is one of the main ecological services provided by wetlands, seasonality and the recurring hydrologic shifts can have some of the largest effects on whether a wetland is a source or a sink for P and N. Lane, Madden, Day and Solet (2011) found that along a tributary of the Mississippi River, where seasonal flooding impacts the area annually, P was both a sink and a source depending on discharge volumes. N was consistently a sink, with highest exports occurring during peak discharge.

The rate of nutrient uptake will also depend on the type of vegetation present, as well as soil saturation (Dong et al., 2014). The age of the wetland will determine the ability of the macrophytes to remove P and N, or whether the roots may instead release nutrients into the soil (Anderson et al., 2013). Macrophytes have root systems which will absorb nutrients as they are developing. In the long term however, as roots decompose and mature, nutrients will be released back into the soil, especially P (Morari & Giardini, 2009). This makes the age of the wetland, or at least age of the vegetation, important to consider.

Vymazal (2013) suggested that the macrophytes present in a constructed wetland will largely control whether P and N is stored or released. Koontz et al. (2014) disagreed, but found that soils largely account for whether P is stored or released in a wetland. Regardless, the type and quality of soil will determine how productive vegetation will be in utilizing P. In a typical wetland, about 10 percent of P removal occurs through sedimentation and burial (Beutel, Morgan, Erlenmeyer & Brouillard, 2014). Under conditions of lower influent water, plant uptake is largely responsible for P removal from water (Vymazal, 2013). Results from a study by Smith et al. (2006) agreed that high loading rates correspond with lower P absorption rates. This means floodplains may frequently act as a P source if the vegetation is unable to quickly utilize the available P, or if vegetation is too old (Koontz et al., 2014). This would suggest that during the annual spring runoff event in the LWB, plant uptake is not at an optimal level, and therefore likely may not act as a sink to the same degree. Other variations in reducing P loads could be due to different vegetation, soils, wastewater characteristics, climate, as well as size and capacity of the wetland (Wu, Zhang, Li, Fan & Zou, 2013).

N removal occurs through denitrification, settling in soils or sediment and minimally by uptake by plants (Sutton-Grier, Wright & Richardson, 2013). Ronkanen and Kløve (2009) found that numerically, denitrification should be considered the main method of N removal in most wetlands. Similarly to P removal, as vegetation ages, it holds great potential for releasing N into the soil rather than absorbing it. As previously mentioned however, although P removal may decrease with time, denitrification ensures that wetlands remain a sink for N even after other removal methods of N are not as prominent (Morari & Giardini, 2009).

Smith et al. (2006), who performed a study in Atlantic Canada, explained that typically P and N removal is most effective in the early stages of operation when plant uptake is high due to vigorous growth and expansion. Morari and Giardini (2009) conducted a study in northern Italy and found that after two years, the wetland was fully established and nutrient removal would decrease because the growth cycle for macrophytes had been completed. This area however does not experience a freezing event as temperatures remain above zero degrees Celsius year-round. Gottschall et al. (2007) conducted a study in Ontario and found that within the first five years of wetland development, substantial nutrient reductions occurred. As wetlands become well established, the processes in which P and N are removed change and soils will become increasingly saturated with nutrients from settling and release from roots. After those five years, the annual rate of reduction started decreasing. Cameron, Madramootoo, Crolla and Kinsley (2003) agreed and stated that wetlands may not be fully effective at removing P after 2 to 5 years of life.

Smith et al. (2006) referenced a study by Warner, Jenssen and Maehlum (1995) where a wetland was being used to filter out nutrients at a study site in Norway. Because this area observes freezing temperatures throughout winter, the wetlands were insulated with straw and snow to help maintain hydraulic movement. While this study saw large reductions of P, insulation may not be a realistic method in widespread areas across the LWB, as this can be difficult to maintain.

Gottschall et al. (2007) found many case studies which resulted in a 35 to 96 percent P removal rate and a 48 to 98 percent N removal rate in livestock wastewater, depending on the wetland's age and the rate of loading the wetland experienced. When Gottschall et al. (2007) performed their own study in Ontario, they created two wetland sites where wastewater would run through a manure pile, into the first wetland, then into the second wetland after exiting the first. P removal in the first wetland was 45.9 kg ha^{-1} yr^{-1} with an even higher removal rate of 120.1 kg ha^{-1} yr^{-1} for the second wetland. 1160 kg of N ha^{-1} yr^{-1} was removed in the first wetland cell, with 521 kg of N ha^{-1} yr^{-1} removed in the second wetland (Gottschall et al., 2007). Substantial retention of nutrients occurred within the first pile, but an even higher percentage of N removal was observed at the second wetland where loading rates entering the wetland were lower.

A 17-month study in Atlantic Canada by Smith et al. (2006) found that 92 to 96 percent of P was removed and 98 to 99 percent of N was removed from a side-by-side constructed wetland study. The wetlands were planted with the macrophyte *Typha latifolia*, which is a common cattail found throughout the LWB (Grace & Harrison, 1986), and average monthly temperatures were below freezing for 7 of the 17 months. Dairy washwater was loaded continuously into the wetlands at a rate of about 300 L d^{-1} for the entire study period, winter included. The highest daily P reductions were observed where the inlet water contained an average of 44.4 mg L^{-1} of P, and the measured output was 4.0 for one wetland and 2.2 mg L^{-1} for the other. This means there was a minimum 40.4 mg L^{-1} P reduction. At the inlet, an average of 147.1 mg L^{-1} of N was flowing into the wetland, and after a 98 to 99 percent reduction, less than 10 mg L^{-1} of N was flowing through the outlet, indicating a removal rate of at least 137 mg L^{-1}.

In Quebec, Yates (2008) created a wetland to determine the capability of removing P and N from agricultural wastewater as a BMP. The two main macrophytes present were cattails (*Typha latifolia L.*) and reed canary grass (*Phalaris arundinaceae L.*). A continuous simulated flow of wastewater containing 0.3 mg L^{-1} of P and 10 mg L^{-1} of N was poured on the simulated wetlands. The wetlands retained 41 percent of P and between 40 and 63 percent of N.

In a Southern Manitoba study comparing three watersheds, the Soil and Water Assessment Tool (SWAT) was used by Yang et al. (2014) to evaluate how land use changes affected nutrient and sediment transport into nearby waterways. The three watersheds compared were the La Salle River watershed and the Boyne River watershed, which drains directly into the Red River and the Little Saskatchewan River watershed, which drains into the Assiniboine River and then into the Red River. Preliminary research using the SWAT model found that restoring historic wetlands in the area could decrease P loading by 9 percent and N loading by 21 percent (Yang et al., 2014). Through the study, it was found that in all three watersheds, the highest P load reductions occurred within wetland and forested areas compared to residential areas, agricultural areas and pastures. Even though water yield reductions were not the greatest from a wetland, the highest reduction of N was observed when historic wetlands were restored. N reductions in the forested areas were harder to conclusively determine as the biogeochemical processes that occur in forested areas are not well represented by the SWAT model (Yang et al., 2014). P was reduced by 3.4 to 34.1 percent and N was reduced by 1.9 to 18.9 percent.

Another study in Southern Manitoba by (Yang et al., 2010) used the SWAT model to determine the quantity of nutrients that could be reduced by recreating historic wetlands within the Broughton's Creek Watershed. This watershed drains into the Red River, which is a direct tributary of Lake Winnipeg. Under the area's typical nutrient flow of 1.1 mg L^{-1} of P and 5.9 mg L^{-1} of N in an agriculture-dominated watershed, the SWAT model showed that for each hectare of restored wetland, 1.3 kg of P and 6.8 kg of N will be suppressed annually, eliminating it from entering the Red River. These results agreed with the study by Yang et al. (2014) with an average P and N reduction of between 1.2 to 11.7 percent (Yang et al., 2010 ; Yang et al., 2014).

Based on scientific literature in relevant geographic areas, it can be concluded that wetland areas typically are sinks for P and N in the LWB, even when considering freeze-thaw cycles. Sutton-Grier et al. (2013) suggested careful consideration of plant species when constructing or restoring a wetland, as vegetation must be capable of utilizing large quantities of P and N. While it has been mentioned that wetlands absorb the highest nutrients when they have been recently established, Gottschall et al. (2007) suggest mitigation measures such as harvesting and re-planting vegetation as it gets older to ensure plants are always absorbing, rather than releasing P. Smith et al. (2006) indicated that their 17-month study never experienced a decline in nutrient removal because the cattails in their constructed wetland were just nearing the end of their growth stage. If cattails are not harvested upon maturation, between 35 to 75 percent of the previously stored P is released upon commencement of tissue death (Smith et al., 2006). In recent literature, studies have started exploring the potential harvesting of cattails to use as a form of bioenergy (Williams, Robertson & Wuest, 2015). A study by Hurry and Bellinger (1990), found that harvesting cattail shoots from a wetland every 3 months was responsible for removing 50 g m^{-2} $year^{-1}$ of N. The study however did not elaborate on how many cattails were required to be harvested for such a reduction. Jeke, Zvomuya, Cicek, Ross and Badiou (2015) also found that after 86 days of growth, cattails should be harvested to ensure the highest efficiency of nutrient removal. The LWBSF has funded a few recent projects investigating the practicality of harvesting cattails as a method of reducing P loading in waterways and producing a valuable biofuel product. This however is still a relatively new process which requires much more conclusive research.

Calculations for estimating P and N reductions will assume the wetlands are within the first 5 years of creation, and if they are older, it will be assumed that there will be regular harvesting of the macrophytes to ensure P is continuously absorbed, rather than released.

Annual P reduction from constructed or restored wetlands (kg)
=
 # ha of wetland x 1.3
(total # of hectares created or restored)

Annual N reduction from constructed or restored wetlands (kg)
=
 # ha of wetland x 6.8
(total # of hectares created or restored)

Example calculation:
Creating or restoring 100 ha of wetland would reduce P by 130 kg and N by 680 kg.

2.5.3. Native vegetation

The widespread replacement of native vegetation with agricultural cropland may shoulder some blame for the declining quality of nearby waterways (Christensen, Lee, McLees & Niemela, 2012). Less than 1 percent of historic Tallgrass Prairies are left in North America (McLachlan & Knispel, 2005). Agricultural areas have significantly higher rates of erosion than areas planted with native prairie vegetation (Christensen et al., 2012). The lack of land cover early in the season when crops have not broken through the top soil surface layer has been linked to high nutrient leaching and runoff during rainfall events (Gutierrez-Lopez, Asbjornsen, Helmers & Isenhart, 2014). This adds to the problem caused by springtime flooding in the LWB, as there is little vegetation to absorb water, and therefore it simply passes over the land and floods the already saturated nearby waterways (Johnson et al., 2012). Even if the soils weren’t still frozen, there would be very little infiltration of water.

Land retirement is a term used when highly sensitive agricultural land is removed from crop production and planted with a permanent, perennial vegetative cover (Luo, Li J., Huang & Li H., 2006). The goal of land retirement is to improve water quality through reduced erosion and nutrient transport to streams (Christensen & Kieta, 2014). Not to mention, replacing agricultural land with native vegetation gives the added benefit of eliminating the typical over-application of fertilizers, pesticides, manure, etc. on an agricultural area. The conversion of cultivated land to perennial, native grassland through land retirement processes reduces soil loss and sediment transport by significant amounts (Christensen et al., 2012). Land retirement does not solely consider converted cropland, but includes the conversion of rangeland, pastures or feedlots to native vegetation (Luo et al., 2006).

In the Minnesota River Basin, which is just southeast of the LWB, agricultural land has been retired through various conservation initiatives, and has since been replanted with native grasses. Studies have shown that this land retirement was directly correlated with stream quality improvements when considering rates of nutrient loading (Christensen et al., 2012). A total of 9,800 hectares of agricultural land in Renville County, Minnesota and 1,100 hectares in the West Fork Beaver Creek Basin, Minnesota were retired from 1987 to 2012 through various conservation programs (Christensen & Kieta, 2014). Through observations made from 1998 to 2008, it was found that there was a significant downward trend in annual mean P concentrations in waterways adjacent to areas of land retirement (Christensen & Kieta, 2014). When looking at the last 3 years of this study, a similar study within the Minnesota River Basin found that these conservation programs had resulted in lowered N concentrations, however no significant relationship was found between P concentration reductions and land retirement (Christensen, Lee, Sanocki, Mohring, & Kiesling, 2009). Currently, the LWBSF is supporting various projects involving the buyback of land from private owners to restore native vegetative cover. These projects may include wetland restoration and buffer strip formation, but also include the planting of trees and other native grasses.

Establishment of native vegetation can be difficult to commence as most native prairie species are very vulnerable to invasive species throughout the early stages of growth (Vink et al., 2015). Diallesandro, Kobiela and Biondini (2013) found that nutrient-rich fields in North Dakota have shown higher succession of invasive species than growth of native grassland species. Typically, invasive species in a grassland prairie will reduce diversity, reduce productivity and change the ecosystem's overall functioning and are therefore generally thought of as unwelcome (Diallesandro et al., 2013). Native grasses are more acclimated to low concentrations of soil N and may therefore have lowered success in a field with prior agricultural activity (Diallesandro et al., 2013). This means the soil legacies from previous activities in the area have an impact on how successful planting native species will be (Vink et al., 2015). Very few studies have been done to determine the capabilities of invasive grassland vegetation compared to native species in removing nutrients from runoff. Herr-Turoff and Zedler (2005) indicated that not enough is known yet to determine whether there is a performance difference between the capabilities of native or invasive vegetation in reducing P and N, as both have contributed to equally significant reductions in various studies.

A Tallgrass Prairie restoration project in Beaudry Provincial Park, Manitoba showed that compared to a control group, the soils with high P and N concentrations compromised the growth of various forb species, favoring the competitive, dominant species (McLachlan & Knispel, 2005). Although this study was not targeting nutrient reductions, it was noted that P and N soil concentrations had retreated from a high level to an intermediate level with respect to the control sites by the end of the project. This suggested nutrients in the soil were being absorbed by the prairie grasses, ultimately leading to lowered levels of nutrients available for export into nearby waterways.

Christensen et al. (2012) found that there was a strong negative relationship between N in a waterway and the amount of retired agricultural land adjacent to the waterway. Native grasses had been planted in an area of retired land, which was found to have captured large quantities of sediment from runoff, minimizing total erosion. However, it is extremely difficult to quantify a numerical reduction as there are many different variables to consider. Not only does the amount of nutrients reduced depend on the type of vegetation, but it depends on the agricultural practices occurring in the area prior to retirement. Some crops may retain more nutrients than others, some farmers may not apply as much fertilizer or manures, similarly to the fact that some species of native vegetation may have lower uptake tendencies. In the literature, it seems that quantities values from restoring perennial vegetation is typically analyzed based on an economic impact, rather than nutrient reductions (Johnson et al., 2012).

Consideration for the potential of native perennial vegetation to enhance the hydrologic balance of agriculturally-dominated areas could mean a reduction in surface flow which would lead to lower nutrients in the LWB (Gutierrez-Lopez et al., 2014). It is known that native grasses and vegetative cover through the LWB has a high capacity for the uptake of nutrients, especially N. While is it becoming an accepted practice to retire agricultural land back to native perennial vegetation, hydrologic improvements in terms of nutrient loading have been observed, but a complete understanding of the altered processes are not yet well understood (Hernandez-Santana et al, 2013). As Christensen & Kieta (2014) explained, short-term studies are unlikely to indicate adequate P or N reductions needed to justify land retirement conservation programs, which make it extremely important for further investigation. Land managers have a need for a complete understanding of the potential benefits created by such a practice.

2.5.4. Urban development

The increasing prevalence of impervious surfaces within urban areas is leading to higher volumes and velocities of surface runoff (Cheng, McCoy & Grewal, 2014). Urban development relies largely upon storm sewer systems to prevent flooding across parking lots, roads and all other impermeable areas. As a result, this means all runoff, including that from lawns, gardens and other permeable surfaces is also collected into this concentrated supply of water and released into a main body of water. Lawns and gardens have the ability to reduce erosion, sediment and nutrient transport, but in order to maintain these features in an esthetically pleasing way, quite often in urban areas there is a large dependence on fertilizers, insecticides and herbicides. The runoff then carries these excess chemicals into our rivers and lakes, thus creating a large source of nutrient contamination. In fact, in the U.S., urban runoff has been ranked the third highest source of water pollution in rivers and the second highest source of water pollution in lakes (Cheng et al., 2014).

This section will not discuss sewage wastewater in any length as this is an important consideration all on its own (see Section 2.7). It will briefly discuss the effects of increased runoff due to high volumes of concrete, asphalt, or other impervious surfaces, as well as the contribution of nutrients due to the modified soil and vegetative profiles from levels of anthropogenic modification. While common strategies for mitigation against high levels of runoff include the usage of retention ponds, this specific management practice will be discussed in the proceeding section (see Section 2.6.1). This section will provide a discussion on the sources of nutrients occurring through contact with surface water runoff. Because the runoff produced in an urban area is created by the way we develop and plan our urban areas, the inclusion of parks, ponds and legislation monitoring chemical usage is a useful consideration regarding the way we manage our land and urban areas.

In an urban setting, surface runoff can come in contact with nutrients through a variety of sources. Some examples of such contributions include pet waste, fertilizers, pesticides, or herbicides. A study by Groffman, Law, Belt, Band and Fisher (2004), observed that 75 percent of total N runoff from an urban setting in Baltimore, Maryland was composed of lawn fertilizer at 14.4 kg of N ha^{-1} y^{-1}, followed closely by atmospheric deposition. Pet waste contributed 17 kg of N ha^{-1} yr^{-1} in that suburban watershed. That study found that total average annual N runoff exports in the urban and suburban watersheds fell between 2.9 to 7.9 kg of N ha^{-1} upon consideration for both imports and exports of N. There was no available data regarding P levels.

In a Minneapolis-Saint Paul, Minnesota experiment, it was found that although N exports were dominated by fertilizers, it was pet waste which was responsible for the highest percent of P inputs into the landscape (Fissore et al., 2012). It was assumed however that this was simply because Minnesota state law has restrictions against the use of P fertilizers as a mitigation method to improve water quality. Although Minneapolis-Saint Paul does not fall within the LWB, the climate is quite similar and has a comparable population compared to Winnipeg or Edmonton, which are two major urban areas within the LWB. Upon consideration of all 360 households, 30 percent of households had at least one dog, and it was dog waste which contributed 84 percent of total P input and 10 percent of total N input in the urban watershed.

Groffman et al. (2004) indicated that very little research has been conducted to evaluate the contributions urban areas have in terms of long-term nutrient loading to nearby waterways. There is an extremely complex combination of considerations to be made when attempting to evaluate nutrient contributions from an urban area (Groffman et al., 2004). While impervious surfaces and application of chemical enhancements for landscaping are largely to blame for high nutrient loading, within an urban setting there are permeable surfaces which do take in some percentage of nutrients. It is extremely difficult to quantify, however, as research is based upon urban watershed models rather than trials and physical experiments. As Wollheim, Pellerin, Vörösmarty and Hopkinson (2005) stated, as urban and suburban sprawl continue to occur throughout the LWB, it is likely that the increase of impervious surfaces will negatively impact aquatic quality, which will ultimately add to the degradation of Lake Winnipeg.

2.6 Water Management

The LWB has a climate conducive to the accumulation of snow and ice throughout winter. This built up solid water eventually melts, causing an annual saturation of streams and waterways throughout the entire LWB (Corriveau, Chambers, Yates & Culp 2011). Managing large volumes of water is essential, as majority of this water is surface runoff. When this water moves across the land, it has the potential to erode the surface, scraping up whatever nutrients it touches. Because such a large portion of the LWB is designated agricultural lands, it is expected that any surface water will contain significant amounts of P and N (Montreuil, Merot, & Marmonier, 2010). Slowing the transport of this water allows biogeochemical processes to filter the water, removing significant quantities of P and N (Liu et al., 2013). Whether this be through detainment structures, uptake of P and N through vegetative buffers, or some other method, it is important to consider the nutrient contribution of such a large annual influx of water entering Lake Winnipeg

2.6.1. Surface water detainment

Throughout the LWB, the majority of nutrient transport enters waterways during spring snowmelt (Elliott, 2013). Snow accumulates along the surface throughout the winter months and come springtime, soils are still frozen which means little to no infiltration of this melt water (Li et al., 2011). Although summer storm events may be a significant source of nutrient transport due to soil erosion and flash flooding, these events are extremely localized (Cade-Menun et al., 2013). Snowmelt runoff events typically last longer than rainfall runoff events and are far more widespread across the entire LWB than storm events (Li et al., 2011). It has been estimated that in Saskatchewan, more than 80 percent of surface water recharge occurs annually due to spring snowmelt and more than 90 percent of total annual runoff in Alberta is recorded during spring snowmelt (Elliott, 2013). Corriveau et al. (2011) estimated that spring runoff in a Southern Manitoba watershed contributes 64 to 98 percent of total P and 71 to 95 percent of N runoff to streams annually.

In the Canadian Prairies, the annual spring snowmelt causes streams to flood extensively, contributing over 80 percent of the total annual volume of runoff (Corriveau et al., 2011). Concentrations of P and N within the melt water may vary greatly, especially depending upon the underlying surface (Stutter, Langan & Cooper, 2008). Snowmelt runoff originating in South Tobacco Creek, Manitoba has been measured to contain P concentrations of anywhere between 36 and 91 kg km^{-2}, whereas N concentrations have reached values between 133 to 307 kg km^{-2} during peak flow (Corriveau et al., 2011). Estimations of P and N reduction in a southern Manitoba watershed study by Li et al. (2011) suggest as much as 57 and 63 percent, respectively, were removed through storing runoff in a holding pond.

Detaining excess water during peak periods of flow, specifically during the spring snowmelt in the LWB, not only helps mitigate against economic losses from flooding, but helps improve water quality by altering nutrient dynamics and allowing P and N to filter out prior to entering a Lake Winnipeg tributary (Reckendorfer et al., 2013). The idea behind a detainment basin or surface water retention pond is that holding a portion of the spring melt water allows the frozen or already saturated ground to prepare for absorption of more water. This minimizes nutrients ending up in the tributary. Retention ponds have the potential to replicate the runoff water storage component of a wetland for P and N. This includes any biogeochemical processes which are apparent in a wetland, however on a much reduced scale of course, as the main goal of a retention pond is to maximize residence time to allow for settling and reduce surface erosion caused by overland flooding (Rosenzweig, Smith, Baeck & Jaffé, 2011). The success of retention ponds relies upon the conditions present. With variable water levels and saturated soils, reduction of P and N are significantly higher when typical wetland vegetation, including an array of macrophytes, are present (Rosenzweig et al., 2011).

The LWBSF is currently funding multiple projects aimed at creating surface water retention ponds to catch excess flood water. One example of a funded project involves the creation of a retention pond to minimize the outflow of nutrients from Pelly's Lake. Another funded project involves creating water retention structures which will aid farmers in irrigation. Using this water for irrigating crops minimizes the need for fertilizers, as this water is already quite P- and N-rich. While numerical nutrient reduction estimates are currently not available for these projects, scientific literature also fails to provide sufficient quantitative data. In the near future, with the current projects underway, values should start appearing in literature, giving a clearer picture as to how effective surface water retention ponds are in reducing nutrient loading to Lake Winnipeg.

2.6.2. Streambank erosion prevention

Streambank erosion is a process that fuels natural development of a stream which can never be fully eliminated. Streambank erosion occurs via two main processes: undercutting hydraulic action and vertical gravitational pull (Simon, Pollen-Bankhead, Mahacek & Langendoen, 2009). Undercutting from hydraulic movement can be difficult and hard to control, whereas it is much easier to reinforce a streambank to minimize erosion. When sediment is eroded into a waterway, it carries with it numerous nutrients, including P and N. When large sediment particles are carried into a waterway, particulate P and N already in the water may be remobilized within the stream due to the turbulence created from the mixing of sediment (Stutter et al., 2008). A study in Denmark has suggested that twenty to sixty percent of P loading into a waterway was delivered by bank erosion (Stutter, Chardon & Kronvang, 2012). This however seems like quite a large and possibly unrealistic contribution of P solely from streambank erosion. That same study found that the presence of natural vegetation was inversely related to the high rate of P loading (Stutter et al., 2012).

In a previous section of this report, livestock access restriction has been discussed regarding the reduction of nutrient loading through the elimination of defecation in a waterway (see Section 2.3). Access restriction does help reduce erosion from trampling under the weight of the livestock, however this will not be discussed in great depth in terms of erosion control, as reduction rates would greatly vary depending on site-specific parameters such as slope, soil composure, stream velocity, etc. As Miller et al. (2010b) mention, while some scientific research shows evidence of reduced erosion rates, some researchers have found no significant difference when cattle were restricted. It should be noted that infiltration from vegetation will also not be considered as a method of reducing nutrients in this section, as this will be touched upon in the next subsection (see Section 2.6.3).

The focus in this section will be on using vegetation to enhance streambank strength, thus reducing erosion. Nutrient reductions will occur based on the ability of vegetation to retain soil and to scatter surface flow through the physical features of vegetation (i.e. roots, dense coverage, etc.). Of course however, it is important to consider the fact that vegetation would be unable to establish with constant grazing and trampling by livestock, so access restriction is a vital part in streambank erosion control (Nellesen, Kovar, Haan & Russell, 2011).

Elliott (2013) has suggested that practices which use vegetation to prevent erosion have been shown to effectively reduce transport rates. However, in some situations, depending on factors such as the type of vegetation, density of the vegetation and soil condition, these practices can act to both reduce and enhance streambank erosion rates (Langendoen, Lowrance & Simon, 2009). Miller et al. (2010) stated that *Populus deltoides* (cottonwood), *Populus tremuloides* (aspen), *Populus spp.* (poplar), *Salix spp.* (willows), and *Cornus stolonifera* (red-osier dogwood) have extensive root systems which act as excellent bank stabilizers. As the roots twist around the bank, they retain pockets of soil which allows for less sediment erosion from both surface runoff and bankside contact.

A study in Mississippi by Langendoen et al. (2009) found that a deciduous tree stand had significantly reduced rates of streambank erosion compared to a grass buffer. The effect of the grass buffer was actually found to be negligible in this study, however this is likely because the grass buffer was too young to have established any type of extensive root system, which drastically reduced its ability to stabilize the bank (Langendoen et al., 2009). Slowing the velocity and minimizing the concentration of water is another method in reducing erosion. Yuan, Bingner and Locke (2009) noted that studies have shown tall grass hedges are efficient in dispersing runoff creating lower concentrations of overland flow, thus successfully reducing bank erosion. Those studies would have waited until the grass hedges had fully established.

Zaimes, Schultz and Isenhart (2008) conducted an experiment in Iowa to measure streambank erosion along different land-use areas such as continuous cropped fields, rotational-cropped fields, livestock pastures both with and without fencing, as well as areas with a forested riparian buffer zone and a grassed buffer zone adjacent to the waterway. Soil losses from the streambank were measured, but it should be noted that these values were based purely on the weight of sediment loss, not nutrient content. This value is significant however, because as previously mentioned, this sediment holds some proportion of P and N, and can cause settled particulate P and N to be disturbed from the stream bed (Stutter et al., 2008). In agreement with the above studies, the forested riparian buffer zone saw the lowest streambank erosion rate, followed by the grassed buffer zone (Zaimes et al., 2008). The pastures and agricultural fields had highest rates of erosion as they had no vegetative reinforcement along the streambank, even though in some cases livestock were provided with restrictive fencing deterring them from approaching the streambank.

An experiment was performed by Frey et al. (2015) to observe rates of P and N loading from streambank erosion at three study sites within an agriculturally intensive area in eastern Ontario, Canada. Through rainfall simulations, the unstable streambanks eroded various amounts of sediments and soils, which were measured and analyzed for nutrient content. The research found that concentrations of P were highly variable among each site, with P exports ranging between 0.2 and 1.5 kg ha^{-1}. N export concentrations declined throughout the rainfall simulation and varied between sites with rates of 0.5 to 1.7 kg ha^{-1}. The focus of this experiment was to show how unstable streambanks can be a source of nutrient transport into a waterway, so values were not determined to indicate how stabilizing the streambank would have minimized nutrient export. The researchers suggested that vegetation would likely reduce runoff levels through canopy coverage and increased surface roughness.

There is a great need for additional published research and information regarding numerical nutrient transport due to erosion, especially in the LWB or comparable areas. Because the mechanical process of streambank erosion is so highly influenced by soil porosity, combined with the added fact that most surface overland flow occurs in the spring during snowmelt, it is extremely important that evidence of nutrient reductions is observed in an area which experiences freeze-thaw cycles similar to that of the LWB (Pollen-Bankhead & Simon, 2010). Quantitative calculations based on streambank erosion are unrealistic at this time, as scientific literature suggests a variance in effectiveness which largely depends on site-specific features (Simon et al, 2009). Pollen-Bankhead and Simon (2010) suggested that although previous research has shown that the mechanical process of root-reinforcement on streambank soil is significant, the root networks are extremely complex and intricate, which makes them difficult to quantify.

2.6.3. Vegetative and riparian buffers

Vegetative buffers are a common BMP used throughout the LWB which act as a zone of separation between a waterway and a terrestrial area high in nutrients (Stewart et al., 2011). There are many interchangeably used terms for 'vegetated buffer strips' across the literature. The term refers to strips or bands of land planted with permanent vegetation, with the purpose of decreasing pollution loads entering an adjacent waterway (Abu-Zreig, Rudra, Whiteley, Lalonde & Kaushik, 2003). Vegetated ditches can act as a small-scale storage basin for runoff, where the storage creates time for P to be removed through sedimentation (Flora & Kröger, 2014). Vegetated ditches also allow infiltration of nutrients, where dissolved N and P can be taken up by vegetation and soils (Stewart et al., 2011). All vegetated buffer zones require minimal upkeep to ensure sediment is removed and they can conveniently be added to already existing edge-of-field or ditch systems, meaning they take up no additional space from agriculturally productive land (Flora & Kröger, 2014).

Numerous studies have been conducted regarding efficiency of various types of buffers with multiple widths and types of perennial vegetation, and for the purpose of this report, all types of buffers will be considered to have approximately equal reductions. Although there are two distinct types of buffers found in the literature- vegetated buffer strips and riparian buffer zones- the biogeochemical processes are quite similar. Vegetated buffer strips are typically planted in strips parallel to a natural or artificial waterway, such as a ditch or a stream channel (USEPA, 2005 ; Yuan et al., 2009). According to the USEPA (2005), these buffer strips maintain soil aeration, whereas riparian buffer zones typically consist of anaerobic, saturated soils. Riparian buffer zones are also usually large in size, appearing as more of a transition zone with close proximity to a natural waterway, such as a stream channel (Christensen et al., 2012).

Vegetated buffer strips can consist of any sort of vegetation, including perennial tall grass, and are planted in a manner which increases surface roughness (Yuan et al., 2009 ; USEPA, 2005). They may be planted in strips as narrow as 0.75 meters, or reach widths as high as 90 meters or more (Yuan et al., 2009). Surface roughness helps to slow the movement of water, increasing infiltration of nutrients into soil (USEPA, 2005). In fact, Sheppard S., Sheppard M., Long, Sanipelli and Tait (2006) suggested that infiltration may be considered the most effective process for retention of P. P retention in a buffer strip can be increased with the removal of plant tissue followed by replanting of perennial plants after maturity (Stewart et al., 2011). Similarly to the processes occurring in wetlands, the largest mechanism in a buffer strip responsible for N reduction to stream loading is denitrification (Stutter et al., 2012).

Riparian areas are the zones located between terrestrial and aquatic areas, which greatly influence the health of aquatic zones (Dong et al., 2014). They act as a transition region, interacting with species from both aquatic and terrestrial origins, moving nutrients and water between the regions. Riparian buffers are accepted across multiple countries as a method for protecting water from nutrient loading, as well as providing numerous other ecological benefits (Dong et al., 2014). Water travels through riparian areas via vegetative uptake and release and by sorption and exchange from sediments and soil (Dong et al., 2014). These biogeochemical processes are responsible for the shifting of P and N in an ecosystem. In this report, riparian areas will be considered under the classification "vegetated buffer strips", as many studies show comparable nutrient reductions, regardless of being classified as a vegetated buffer or a riparian buffer Christensen et al., 2012 ; Lee, Isenhart & Schultz, 2003).

There is some contradicting evidence in the literature regarding the effectiveness of grassed or vegetated buffer strips in removing P and N (Stewart et al., 2011). This is largely because while N is consistently removed through the use of vegetated buffers, it is found that once soils become saturated with P, then the buffer may act as a source rather than a sink for P (Habibiandehkordi, Quinton & Surridge, 2015). The absorption of P in soils means these buffers are only effective in reducing nutrient loads when the prominent water source is surface flow, rather than subsurface or groundwater flow (Habibiandehkordi et al., 2015).

Elliott (2013) has found evidence suggesting that preventing nutrient transport through vegetation is less effective in areas where snowmelt runoff is the dominant surface water movement. Elliott (2013) performed a study evaluating the release or absorption of crops and crop residue under a laboratory setting simulating the typical conditions found through Saskatchewan and the rest of the Prairies. It was found that while soils are still frozen during snowmelt, high amount of nutrients were released from the vegetation that was actively growing prior to snowfall, compared to crop stubble and residue. Elliott (2013) suggests that in the non-erosive conditions observed throughout the LWB, riparian buffers which are actively growing up until snowfall potentially release nutrients into the snowmelt runoff, rather than absorb them as many other researchers have suggested. Elliott (2013) does make the note that the soils used in the study did not contain the excess nutrients which would be typical of soils within an agriculturally-intensive plot of land. This poses a need for further research with more representative soils.

When buffer strips have been found successful, their ability in retaining nutrients depends largely on buffer width, flow rate and sediment size (Stewart et al., 2011). Sheppard et al. (2006) argue that the key component to ensuring success of a vegetated buffer strip in retaining P is the width of vegetation. While vegetative buffer strips are seen in various widths, Sheppard et al. (2006) suggest the strip must be at least 10 to 90 meters in width to ensure P retention. In a study by Abu-Zreig et al. (2003), as the width of the strip is increased, larger P retention was seen. Effectiveness will also be dependent upon the qualities of the contributing area in terms of the nutrient loading and rate of flow (Mallarino et al., 2002). Abu-Zreig et al. (2003) agreed and found that when the rate of flow was lowered, higher P retention occurred. Sheppard et al. (2006) found in a study that when velocity was decreased, infiltration was promoted and higher

P and N removal rates were observed. These findings clarify the concept that rate of flow will alter the amount of nutrient absorption.

A study in southern Quebec by Duchemin and Hogue (2009) considered the effectiveness of vegetative buffer strips on the runoff from a typical corn field. Vegetative strips were planted in 5 meter wide strips, and included a control with no buffer, a perennial grass buffer, as well as a perennial grass and poplar tree mix. The field was treated as a typical corn field in the area would be treated in terms of tillage practices, manure spreading, fertilizer application and fall harvests. These researchers found that the grassed buffer strip had the biggest reduction in total water runoff volume, total P, and total N. While the grassed/tree buffer mix reduced total P by 85 percent and N by 30 to 47 percent, the grassed buffer reduced total P slightly more at 86 percent and N by 33 to 57 percent.

A study in south-central Ontario by Abu-Zreig et al. (2003) compared different slopes of drainage, different rates of watering and different widths of vegetative buffers, and found that regardless of the angle or width, P reductions were significant on all 20 trials in the study. The lowest P reduction was 31 percent of P retained with a 5 meter buffer strip, whereas the highest was an 89 percent P reduction with a 15 meter buffer strip. Abu-Zreig et al. (2003) found that in general, the wider the vegetative strip was, the more P was retained. On average, 61 percent of influent P was retained when considering all the trials (Abu-Zreig et al., 2003)

Due to the absence of quantitative data regarding typical P and N exports per hectare of agricultural land in the LWB, values will be adapted from different regions. The value from the South Nation Conservation District's report, which is adapted from a study in Iowa by Lee et al., (2003), suggests an annual P runoff of 1 kg per hectare of cropland. A study in Norway shows that P runoff values actually have an average of 2.2 kg ha^{-1} yr^{-1} (Øygarden & Botterweg, 1998). A study by Elrashidi et al. (2008) in an agriculturally-dominated region of West Virginia calculated the average P runoff 1.98 kg ha^{-1} yr^{-1} from pastureland and 5.51 kg ha^{-1} yr^{-1} from cropland. Based on these values, it is reasonable to increase the value suggested in the South Nation Conservation District report's to conclude that every hectare of cropland contributes 2 kg of P runoff per year.

Values used in a Wisconsin study, which is located just southwest of the LWB, will be used to gain an understanding of the typical N loads in runoff entering a buffer zone. In 2013, the total annual N export throughout the entire Wisconsin study area was 5.6 kg per hectare of watershed (Fehling, Gaffield & Laubach, 2014). However, agricultural watersheds typically contribute much larger loads of nutrients, this value is a minimum of 6.7 kg ha^{-1} yr^{-1} and can reach values upwards of 50 kg ha^{-1} yr^{-1} (Fehling et al., 2014). Because the LWB is largely composed of agricultural land, it is acceptable to then assume that one hectare of land in the LWB contributes at least 6 kg ha^{-1} yr^{-1} of N.

A study by Miller, Curtis, Chanasyk and Reedyk (2015) was conducted in Alberta focusing on the effects of mowing a grassed buffer zone and came to a similar conclusion stating that the width of the buffer has the greatest influence on reduction in nutrients. The higher rate of P and N removal was attributed to the higher residence time within the buffer, which meant greater infiltration opportunities (Miller et al., 2015). To make calculations as accurate as possible, because buffer width alters the rate of nutrient reduction so greatly, it is reasonable to create two different equations: one equation when the buffer strip is 5 to 15 meters in width, and one equation for when the buffer strip is over 15 meters wide (Sheppard et al., 2006 ; Abu-Zreig et al., 2003).

Although Yuan et al. (2009) report reductions occur when a strip is under 5 meters in width, quantitative data is largely unavailable as majority of relevant studies use a buffer with a minimum 5 meter width. The literature states as you increase width, retention of nutrients increases (Abu-Zreig et al., 2003), however when reviewing multiple studies such as the ones by Abu-Zreig et al. (2003), Lee et al., (2003), and by Duchemin and Hogue (2009), after 5 meters, rates of reduction vary greatly. Therefore, using a generalized rate for buffers over 15 m wide will reduce inaccuracy leading to overestimation. Because quantitative values relatable to the LWB were absent, values from elsewhere in Canada will be accepted as an alternative, with consideration from findings by Lee et al., (2003) in Iowa.

The South Nation Conservation District uses an equation which assumes a 70 percent P load reduction based on the typical effluent discharge per how many hectares are buffered. However, as previously mentioned, the width of the buffer strip also highly influences reduction rates, not just how many hectares of cropland are being filtered by the buffer. Based on values taken from Abu-Zreig et al. (2003) and consideration for the lower limit findings of N reduction by Duchemin and Hogue (2009), a vegetated buffer zone of 5 to 15 meters will reduce both P and N by at least 30 percent. Lee et al., (2003) found that by increasing the buffer width from 7 m to just under 17 increases nutrient reductions by over 20 percent. Therefore, when considering reduction values for P found by Abu-Zreig et al. (2003) and considering a 20 percent increase on the upper limit N reductions found by Duchemin and Hogue (2009), the 70 percent reduction rate used by the South Nation Conservation District is reasonable to use when the buffer strip is a width of 15 meters or greater.

Buffer strip width of 5-15 meters:

Annual P reduction due to buffer strip (5-15 m)

=

 # ha of cropland x 2 kg x 0.30

(determinant of how much P enters the vegetative buffer) (% eliminated from runoff)

Annual N reduction due to buffer strip (5-15 m)

=

 # ha of cropland x 6 kg x 0.30

(determinant of how much N enters the vegetative buffer) (% eliminated from runoff)

Example calculation:
100 ha of cropland buffered by a 10-meter wide vegetative strip would reduce P by 60 kg and N by 180 kg annually.

Buffer strip width of over 15 meters:

Annual P reduction due to buffer strip (15+ m)
=
 # ha of cropland x 2 kg x 0.70
(determinant of how much P enters the vegetative buffer) (% eliminated from runoff)

Annual N reduction due to buffer strip (15+ m)
=
 # ha of cropland x 6 kg x 0.70
(determinant of how much N enters the vegetative buffer) (% eliminated from runoff)

Example calculation:
100 ha of cropland buffered by a 20-meter wide vegetative strip would reduce P by 140 kg and N by 420 kg annually.

Much of the available scientific literature regarding vegetative buffers has been conducted in plot-scale experiments under highly controlled conditions, with information extrapolated to account for the entire study area (Stewart et al., 2011). As Stutter et al. (2012) mention, these studies typically involve a single or limited number of rainfall events and are therefore unrepresentative of the typical processes occurring throughout the year. While studies have been done regarding seasonal or short-term efficiency of vegetated ditches, further research is required in regards to their long-term potential in removing nutrients from effluent discharge (Flora & Kröger, 2014). This long-term monitoring which accounts for climatic variability is necessary, as precipitation and temperature have large influences on the intensity of runoff, which may affect the usefulness of vegetation in withdrawing nutrients from water (Yuan et al., 2009).

Nutrients may be intercepted successfully where erosion is the contributing factor to nutrient loading in streams, but Elliott (2013) questions the ability for vegetative buffers to effectively reduce nutrient loads on gently sloping landscapes. It is important to recognize that this practice may not always be effective in reducing nutrient loading. Because agricultural practices are so dominant within the LWB, soils are likely to have high nutrient concentrations which Elliott (2013) recognizes is largely underrepresented by current research. Although snowmelt runoff is the dominant surface flow process in the LWB, available evidence that vegetative buffers contribute nutrients to snowmelt runoff was not truly representative of soils found in the LWB. Until more conclusive evidence proves vegetative buffers are not effective, the above equation will be accepted, but with great caution. Future research will be closely monitored to determine if this practice truly can be considered a BMP within the LWB.

2.7 SEWAGE AND WASTEWATER TREATMENT

Direct release of municipal wastewater from a treatment facility is a major source of nutrient loading to the surface water across the LWB (Waiser, Tumber & Holm, 2011). This makes it important to ensure the effluent has been treated properly to remove as many nutrients as possible. Sustainably managing waste has become a large challenge, especially for growing cities. Approximately 60 percent of biosolids produced in the U.S. and Canada are spread on agricultural fields (Cogger, Forge & Neilsen, 2006). With growing concerns over the safety of this practice, a variety of wastewater treatment practices have been developed to minimize the need for spreading biosolids.

Carlson et al. (2013) emphasized the need for improved wastewater management throughout the LWB to ensure nutrient loadings are minimized as efficiently as possible. Waiser et al. (2011) explained that sewage treatment may successfully minimize organic pollution, but the technology available for excess P and N is extremely limited. Of course, cost will always play a significant role in the quality of wastewater management. In order for the City of Winnipeg's north end wastewater facility to remove quantities of P and N which would show significant improvements to downstream water quality, it would cost an estimated $400 million, whereas current removal of P and nitrifying ammonium, which currently meets provincial standards, costs less than $100 million (Schindler et al., 2012). Smaller, rural municipalities have even more financial limitations when it comes to wastewater treatment technologies, which has contributed to the growing popularity of constructed wetlands or lagoons as a method for purifying water (Cameron et al., 2003).

While wastewater management systems such as wetlands or lagoons are more prominent than many other systems due to their affordability (Cameron et al., 2003), many other technologies are appearing throughout the world. Some of these alternative systems for cleansing wastewater that are currently being funded by the LWBSF include bioaugmentation systems and biological floating islands. Those technologies are relatively new and have little available research, but are worth mentioning simply due to their great potential. Because wastewater treatment is a concern throughout the world, information on these new technologies may be available in regions other than the LWB. Therefore, when research originating in the LWB is unavailable, research from other geographic areas will be considered.

2.7.1 Water retention and lagoons

Water retention systems, such as lagoons and wetlands, have been used to treat wastewater for over 35 years (Cameron et al., 2003). Sewage lagoons are carefully engineered to perform the cleansing processes of a natural wetland (Vymazal, 2011). Although lagoons are essentially a form of constructed wetland in terms of functionality and majority of biogeochemical processes (see Section 2.5.2 for discussion on biogeochemical processes), the influent water is highly controlled and stored for a set amount of time before release into receiving waters. The influent water also contains higher concentrations of pharmaceuticals than one would typically see enter a wetland from surface water runoff (Carlson et al., 2013)

Where lagoons are a part of the wastewater treatment system, when wastewater exits a municipality, it enters a large retention basin called a lagoon. During the residence time spent in the lagoon, the water undergoes a multitude of processes, including oxidation, sedimentation, fermentation, photolysis, biodegradation, and disinfection (Carlson et al., 2013). Factors such as high plant productivity, high sediment deposition followed by breakdown and extremely active microbial populations make lagoons successful in treating municipal wastewater (Cameron et al., 2003).

The three most important considerations in terms of impact on effectiveness in removing nutrients from sewage effluent include the lagoon's hydrology, macrophytes and flow path (Vymazal, 2011). When designed, operated, and maintained appropriately, these lagoon systems can produce effluent that will meet current regulatory guidelines. In communities where resources are limited, lagoons are the preferred method of wastewater treatment as they are much less expensive than many other treatment practices (Cameron et al., 2003). However, as community populations, nutrient loadings and presence of organic micropollutants to the receiving waters increase, regulations to reduce nutrient loading is becoming more stringent. It may become necessary to improve the efficiency of current lagoon systems to meet demands for higher nutrient removal, whether that means increasing the storage capacity or through the use of complementary treatment methods.

In rural Manitoba, wastewater from the communities of Morden and Winkler enter sewage lagoons, which are then deposited into Dead Horse Creek twice annually, which empties into the Red River and into Lake Winnipeg (Carlson et al., 2013). This is a typical wastewater management system that can be found across much of the rural areas in the LWB. The nutrient contributions per person in the lagoon effluent upon exiting the Morden and Winkler lagoons were found to be 0.3 kg of P and 2.4 kg of N annually (Carlson et al., 2013). Although Carlson et al. (2013) states that the contributions per person are lower from Morden and Winkler than per person in Winnipeg, research found that the annual P and N contributions from the rural lagoon systems were still a significant contribution to nutrient input to Lake Winnipeg. Carlson et al. (2013) also mentions that because the lagoon effluent is released in the early summer when water levels are already quite high, this could cause even higher productivity leading to heightened eutrophication events.

In a typical lagoon system, the removal of P largely occurs through biological uptake by vegetation, microbiological assimilation, sedimentation, adsorption and precipitation (Ronkanen & Kløve, 2009). N is removed through various processes such as ammonification, adsorption, ion exchanges, sedimentation, volatilization and biological uptake, but typically it is nitrification–denitrification which is responsible for the highest N removal (Ronkanen & Kløve, 2009). As previously mentioned, increasing populations and higher pressures to remove larger quantities of P and N have created a need for lagoon treatment systems with higher efficiencies.

Wetlands are most effective in P and N removal within the first 2 to 5 years of life (Smith et al., 2006 ; Morari & Giardini, 2009 ; Gottschall et al., 2007 ; Cameron et al., 2003). Lindstrom and White (2011) stated that wastewater treatment wetlands are especially prone to declining rates of P removal over time due to the high accumulation and deposition of organic matter. Because it is not realistic to construct a new wetland every 5 years for the purpose of wastewater treatment, it is necessary to find some way to prolong the life of the lagoon to ensure municipalities are continually having proper wastewater treatment. There are many additional technologies available to ensure lagoons or wetlands remain active and efficient in removing nutrients (Lindstrom & White, 2011). Most lagoon treatment systems use a combination of treatment processes rather than relying solely on the biogeochemical processes of a wetland (Vymazal, 2011).

After a quick search in the literature, slag filters were found to be extremely popular with great success in their ability to draw P out of the water. Once the filter is no longer performing with high efficiency, it can be removed and replaced by a new filter (Cameron et al., 2003). In a New Zealand study examining the effectiveness of slag filters in a wastewater retention pond by Pratt and Shilton (2010), it was found that the filters removed an average of 1.23 g of P kg^{-1} each through their first 5 years of life. After 5 years of continually treating 2,000 m^3 of wastewater daily, with an average water residence time of 3 days, the filter effectiveness declined. Upon economic and environmental consideration, Pratt and Shilton (2010) recommended replacement of the slag filters every 5 years.

A simple, yet less convenient way to improve P removal in a lagoon is to remove the accreted organic material from the water. This would eliminate the chance for P to be released from the soil back into the water as it starts to degrade, thus increasing volume and capacity of the lagoon for wastewater (Lindstrom & White, 2011). This method is less convenient because it would require the cessation of wastewater effluent into the lagoon for some period of time while the accreted material is removed. This method also disqualifies the benefit of lagoons being convenient and easy for smaller municipalities to maintain as regular extensive maintenance would be required to remove settled sediments and organic materials (Muga & Mihelcic, 2008).

Because the climate observed within the LWB shifts dramatically throughout the seasons, it is necessary to consider the functionality and challenges presented by a lagoon during the cold season. There are numerous studies quantifying P and N reductions in lagoons throughout the world, however these studies are largely performed in areas where there are no freeze-thaw cycles and there is an altogether absence of any distinct freezing period. One example includes the study by Kim, Park, Lee and Kang (2011) in Korea which found that the wastewater effluent experienced average removal efficiencies for P and N to be over 70 percent through a one-year study, where winter typical winter highs fail to fall below freezing.

Some observations have been suggesting a decrease in functionality of a lagoon throughout cold months compared to conventional wastewater systems. Werker, Dougherty, McHenry and Van Loon (2002) however stated that bacterial populations in natural wetlands are able to acclimate to cold temperatures. Those researchers also stated that especially observable in older lagoon systems, the adaptation of bacteria is indicative of the success of lagoons year-round. This means as long as the bacteria is able to thrive throughout the cold temperatures, the lagoon will be successful.

Werker et al. (2002) evaluated an assortment of studies regarding the effectiveness of various biological wastewater treatment systems and found that each exhibits variability in the level of effectiveness in removing P and N. Lagoons across North America were found to remove anywhere between 20 to 90 percent of P and 30 to 98 percent of N (Werker et al., 2002). P and N removal depends so greatly on an assortment of factors that it is not realistic to create a general reduction calculation. The amount of wastewater entering the lagoon, retention time, concentration of nutrients and pharmaceuticals, hydrology or flow, macrophytes present, bacteria colonies, the presences of any additional treatment methods as well as any other aforementioned factors will all influence how much P and N is removed. With the basic assumption that a lagoon has sufficient macrophytes, healthy bacteria colonies year-round and consideration for the lagoon capacity, based on the evaluations presented by Werker et al. (2002), it would be safe to say that a well-maintained and functioning lagoon wastewater treatment system will remove at least 20 percent of influent P and 30 percent of influent N. Physical kg removed annually however will depend on how much P and N enters the lagoon as well as retention time.

2.7.2. Bioaugmentation

Bioaugmentation is the practice of adding specialized microbial cultures to wastewater in an effort to enhance the biogeochemical processes of the microbial community and increase the speed of nutrient removal (Herrero & Stuckey, 2015). The added cultures act as a catalyst for reactions that already naturally occur by enhancing the speed at which they would typically occur. Bioaugmentation is favored for its rapid success in removing a variety of pollutants in the water simultaneously in a relatively inexpensive manner (Hu, Li, Fan, Deng & Zhang, 2008). There is concern over this method of wastewater treatment as there is potential for harmful byproducts to be created by the addition of chemicals, which scientists still know little about.

Herrero and Stuckey (2015) explain that although these microbial processes have been used extensively in water treatment systems, bioaugmentation is a less predictable and less controllable nutrient removal technique due to the complex interactions between a multitude of microorganisms used in wastewater treatment. There are several considerations in determining how effective bioaugmentation will be in removing nutrients from wastewater. Major considerations include oxygen exposure, temperature, pH, chlorination concentration and the hydrology of the site (Aziz, Wymore & Steffan, 2013). The dynamic behaviors can be altered by the smallest of changes in one of the above considerations. Bioaugmentation may effectively degrade P and N in wastewater, but because wastewater can contain pollutants at such variable concentrations, it can be harder for smaller municipalities to monitor how much of a chemical is needed when there are financial budgets to consider (Hu et al., 2008).

The focus of most literature is not on the effectiveness of bioaugmentation regarding reduction of macronutrients such as P or N, but rather how effective the method is in reducing pharmaceuticals and numerous toxins. Bartrolí, Carrera and Pérez (2011) reviewed multiple studies and found the results to be quite contradicting. Oerther et al. (1998) found no significant start-up removal of organic P through bioaugmentation of wastewater. This makes quantifying the reduction in P and N from bioaugmentation unrealistic at the current time. Future research may provide more conclusive evidence towards a quantitative reduction of P and N from wastewater through bioaugmentation.

2.7.3. Floating islands

Floating islands, sometimes referred to as biological islands, are used around the world as a means to reduce nutrients from some reservoir of water. Excess P and N is removed from water through a combination of plants, which use nitrate as a nutrient source, and microbes (Wang, Gao, Xie, Zhang, & Hu, 2015). They are not necessarily used exclusively in wastewater treatment systems, but can be used in stormwater retention basins as a means for removing nutrients from urban sources, such as fertilizers, pet waste, etc. (Chang, Islam, Marimon & Wanielista, 2012). The LWBSF is currently funding a project in Alberta which will test the effectiveness of floating islands in a wastewater lagoon. Constructed islands are especially useful in wastewater lagoon systems due to their ability to succeed in a water system of any depth or shape (Wang et al., 2015).

Floating islands can be created in a variety of sizes and may contain a variety of biological components, but typically are classified as large floating mats of organic material (Yeh N., Yeh, P. & Chang, 2015). Floating islands may or may not contain soils, but underneath the rooting system of the present macrophytes, decaying roots create a layer of porous peat (Yeh et al., 2015). When soils are absent, these islands are called ‘artificial floating wetlands’, but serve the same function as typical biological floating wetlands (Yeh et al., 2015). Lu, Ku and Chang (2015) explained that not only do these islands remove nutrients, but because they reduce light penetration, they may also inhibit harmful algae growth.

While the macrophytes in a floating island are stationed along the surface, a wetland involves removal of nutrients in the sediment and deeper waters as the vegetation is rooted along the bottom. While the biogeochemical processes responsible for sequestering and removing P and N are similar to that of a constructed wetland, floating islands remove nutrients in a top-down manner, opposite that of a wetland which removes nutrients from the bottom-up (Chang et al., 2012). While floating wetlands may prove to be effective on their own, an important consideration comes from the suggestion that the highest success of nutrient removal would be in a situation where a constructed wetland is paired with floating islands (Chang et al., 2012).

A study by Wang et al. (2015) conducted in China, examined a constructed floating island in a wastewater storage basin containing synthetic wastewater, typical of what would be found in the area. Influent water contained 1.41 mg L^{-1} of P and 7.13 mg L^{-1} of N. Effluent water was collected every second day and was found to have reduced P and N by 56 and 70 percent, respectively (Wang et al., 2015).

Chang et al. (2012) performed a study in Florida, where 10 different cylindrical pools each containing 14 m^3 of water tested the ability of floating islands in removing nutrients. The chosen vegetation was native to Florida wetlands, and conditions such as pH, turbidity, temperature and dissolved oxygen were closely monitored to examine the effects they may have on the results. Interestingly enough, the vegetative mats contained 100 percent Canadian peat, showing some applicability to the LWB, as soil porosity and microbial processes would be comparable. Two sizes of floating mats were compared, one at 5 percent surface volume coverage, and one at 10 percent surface coverage. Chang et al. (2012) determined that while both percentages involved significant nutrient reductions, the 5 percent surface volume coverage removed nutrients in the most cost-effective manner.

Yeh et al. (2015) examined the functionality and effectiveness of artificial floating islands in various scenarios upon reviewing a variety of previously performed studies. Rivers were found to have significant reductions of both P and N, as well as other contaminants, such as metals and toxins. Yeh et al. (2015) found evidence through their review that *Salix babylonica*, *Gypsophila sp*. and *Oenanthe javanica*, provided the highest efficiency in reducing P and N from rural wastewater. In a sewage overflow containment basin, floating islands were responsible for removing between 4 and 31 percent of P and between 19 and 44 percent of N. Yeh et al. (2015) pointed out that in combination with other wastewater BMPs, such as slag filters or constructed wetlands, P and N removal was highest.

Similarly to wetlands, the efficiency of P and N removal depends on the types of macrophytes and microbes present on the island (Wang et al., 2015). This means similarly to wetlands, the P removal is limited based on plant uptake. Matured biomass can be harvested and used in biofuels or even as animal feed, increasing the economic viability of this BMP (Yeh et al., 2015). Because N removal is largely conducted by microbes through denitrification, water temperature plays an important role in the success of the microbial community (Wang et al., 2015). Cooler temperatures are likely to slow the removal process of both P and N. The previously mentioned study by Wang et al. (2015) did not include any temperatures below freezing and therefore, it is realistic to expect a smaller reduction in the LWB.

While the LWBSF is currently providing funding towards the implementation of floating islands as a BMP, research is still underway. With floating islands being a relatively new research topic, quantitative data within the LWB is largely unavailable at the current time. The lack of data in relevant geographic areas makes it unrealistic to provide a quantitative reduction estimate at this time. However, this technology has proven successful in multiple studies, such as the studies mentioned above, and should be monitored closely in the upcoming future.

3.0 SUMMARY

It is the goal of the LWBI to achieve targeted reductions of P and N entering Lake Winnipeg in the coming years. This report serves to assist in achieving those results by providing information from published studies on how effective a BMP is at reducing nutrient loading. Ultimately, a well-designed field monitoring program is the best way to determine the effectiveness of an activity or project in reducing nutrient loading. In the absence of such technology, the estimates presented here can be used as surrogate information to determine the nutrient reduction potential of a BMP. While the most common BMPs throughout the LWB have been evaluated in the aforementioned sections, this report is not an exclusive list of all effective measures that could be implemented. They are, however, the most commonly practiced BMPs in the LWB. For some of the identified BMPs, further research is required to validate the nutrient reduction effectiveness of the action, in light of the hydro-climatic conditions of the LWB. With ongoing and future research, paired with the cooperation of those within the LWB, restoration of the once optimal health of Lake Winnipeg can be reached.

The following chart can be used as a quick reference for determining approximate reductions associated with each specific practice. For practices where a formula is not available at this point in time, the reader is encouraged to review the relevant section of the report to determine the scientific basis for implementing the BMP.

Practice	**Formula (P)**	**Formula (N)**
Livestock Management **2.3.1 Access Restriction** (preventing livestock access to open water)	Cattle: # cattle * days restricted * P daily (Table 5) * 0.04 = *reduction (kg)* Other Livestock: # animals * days restricted * P daily (Table 5) * 0.02 = *reduction (kg)*	Cattle: # cattle * days restricted * N daily (Table 5) * 0.04 = *reduction (kg)* Other Livestock: # animals * days restricted * N daily (Table 5) * 0.02 = *reduction (kg)*
2.3.2 Alternative Water Source (providing a watering source away from a natural waterway within the confined livestock lot) ** note: cattle only*	# cattle * days of alternate source access * P daily (Table 5) * 0.04 * 0.5 = *reduction (kg)*	# cattle * days of alternate source access * N daily (Table 5) * 0.04 * 0.5 = *reduction (kg)*
Manure Management **2.4.2 Storage** (properly storing manure eliminating possibility of loss to runoff)	# animals * days of collection * P daily (Table 5) * 0.04 = *reduction (kg)*	# animals * days of collection * N daily (Table 5) * 0.04 = *reduction (kg)*
2.4.3 Application Rate (retention of nutrients in typical prairie, agricultural soils in years following manure application)	Application = based on manure content values Year 1 = 70% of application Year 2 = 20% of application Year 3 = 10% of application	Application = based on manure content values Year 1 = 25% of application Year 2 = 13% of application Year 3 = 7% of application
Land Management **2.5.1 Wetland Areas** (restoring and constructing wetland areas to absorb and filter water in areas with high surface runoff)	ha of wetland restored / created * 1.3 = *reduction (kg)*	ha of wetland restored / created * 6.8 = *reduction (kg)*
Water Management **2.6.3. Vegetative buffers** (vegetative or riparian buffer zones which water flows through prior to entering a waterway)	5-15 meters wide: ha of cropland entering buffer * 2 * 0.3 = *reduction (kg)* 15+ meters wide: ha of cropland entering buffer * 2 * 0.7 = *reduction (kg)*	5-15 meters wide: ha of cropland entering buffer * 6 * 0.3 = *reduction (kg)* 15+ meters wide: ha of cropland entering buffer * 6 * 0.7 = *reduction (kg)*

Reference tables (taken from text) for above summary:

Adaptation of Table 5. Daily feces weight breakdown per animal, per day in kg.

Animal	Typical Animal Size (kg)	Total Feces Produced (kg)	P (kg)	N (kg)
Beef Cow	454	13.2	0.04	0.2
Dairy Cow	454	13.2	0.03	0.2
Swine	200	3.8	0.01	0.03
Layers	2	0.03	0.0003	0.0008
Broilers	1	0.02	0.0002	0.0005
Sheep	36	1.44	0.003	0.016
Goat	36	1.44	0.003	0.016
Horse	500	25	0.013	0.091

* See page 24 for original Table 5 with sources

4.0 ABBREVIATIONS

BMP – Beneficial Management Practice

P – Phosphorus

N – Nitrogen

LWB – Lake Winnipeg Basin

LWBI – Lake Winnipeg Basin Initiative

LWBSF – Lake Winnipeg Basin Stewardship Fund

SWAT – Soil and Water Assessment Tool

U. S. – the United States of America

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