The Impact of Golf Courses on Nutrient Dynamics from Developed Areas

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Abstract

Proper management strategies aimed at reducing the export of excess nitrogen and phosphorus to water bodies are fundamental for resolving nutrient pollution and eutrophication. Golf courses are more intensely managed than residential expanses of turfgrass, increasing the possibility of their acting as a source of nutrient export to inland water bodies. Experimental turfgrass mesocosms were established and nutrient concentrations of influent, runoff and infiltrate was examined by performing rain and storm simulations. The purpose of this research was to investigate how the variables of two grass species, three seed densities and five fertilizer treatments influence the concentration of phosphate, nitrate, total suspended solids (TSS), total phosphorus and total nitrogen in the infiltrate and runoff. Attempts were made to collect winter snowmelt to examine the effect of seasonality on the concentrations of studied nutrients. Completion of this study did not indicate that grass species, seed density or fertilizer treatment had a significant influence during rain simulations. However during storm simulations significant differences were noticed. Fertilizer treatment significantly influenced infiltrate phosphate concentrations and average runoff phosphate (0.93mg/L) was higher than the infiltrate (0.30mg/L). TSS was the only parameter studied that was lower than the inlet water (78.6mg/L) in both the runoff (16.7mg/L) and infiltrate (5.8mg/L). Winter snowmelt experiments also showed significant differences in phosphate concentrations between grass species with average concentrations of 1.58 mg/L for Creeping bentgrass and 0.85 mg/L for Kentucky bluegrass. Thus, the results of this study suggest that turfgrass can mitigate phosphate exports to inland water bodies.

Lay Summary

The mission statement of Lakehead University's Department of Biology is "faculty and students in the Department of Biology are bound together by a common interest in explaining the diversity of life, the fit between form and function, and the distribution and abundance of organisms." This study focuses on analysing the potential turfgrass has to retain nutrients and aid in the protection of water quality as one of its functional benefits to the environment. As such, the knowledge gained from this research contributes to one of the central research themes in the mission statement, which is explaining the relationship between life forms and their environmental functions. The study advances our knowledge in protecting water quality by contributing to methods of preventing eutrophication with potential in urban land management planning and policies. Three major research questions were investigated; 1. To what extent does turfgrass influence phosphate and nitrate exports based on grass species, seed density and fertilizer treatment? 2. What impact does turfgrass seed density, grass species and fertilizer treatment have on the quality of already polluted water? 3. Are nitrate and phosphate loss higher when soil is frozen? Results showed that fertilizer treatment had an effect on infiltrate phosphate concentrations during storm simulations and grass species (Creeping bentrgrass) had a significant influence on phosphate concentrations during winter sampling. This study was able to contribute to the wider body of knowledge on nutrient pollution of water bodies and combined field-based research with control of specific variables.

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Chapter 1. Introduction

1.1. Nutrient pollution: risks to water quality

Water is a renewable resource and is essential for living organisms as well as domestic and industrial activities. However, it is also considered a finite resource in the sense that the amount present on Earth is all that will ever exist (Karr, 1991). Consequently the issue of water quality is an important one, and efforts are being made that allow its continued and safe use (Suski & Cooke, 2007). One type of water quality impairment is the addition of excess nutrients (nitrogen and phosphorus) to water bodies, which can contribute to eutrophication (King, Balogh, Agrawal, Tritabaugh, & Ryan, 2012). Eutrophication is a process by which a body of water becomes enriched in dissolved nutrients (Eutrophication, 2017). This can be a cumulatively damaging process by which algae are able to grow at accelerated rates often resulting in algal blooms which can be harmful to fish, mammals and avian species (King et al., 2012). Moreover, when the algae decompose dissolved oxygen is utilized to an extent that the viability of other aquatic life can be compromised (King et al., 2012). In addition to the effects eutrophication can have on aquatic organisms, significant economic losses related to social and ecological responses and remediation efforts arise (Dodds et al., 2008). As a result of well water experiencing taste and odor problems potentially linked to eutrophication, it has been estimated that 813 million dollars are spent annually on bottled water (Dodds et al., 2008). Moreover, the potential economic losses associated with reduced recreational water usage, reduced waterfront real estate value, costs associated with the recovery of threatened and endangered species and extra steps necessary to meet drinking water quality standards were estimated to be 2.2 billion dollars annually (Dodds et al., 2008).

There are numerous factors that can impact nutrient pollution, such as runoff volume and infiltrations rates of vegetated areas, landscape slope and nutrient source. In freshwater systems phosphorus is the limiting nutrient for aquatic plant growth (Blomqvist, Gunnars, & Elmgren, 2004), and is of greater concern than nitrogen because concentrations as low as 0.02 mg/L can result in eutrophication (King, Hughes, Balogh, Fausey, & Harmel, 2006). After measuring suspended sediment and phosphorus concentrations in collected water samples a positive correlation between suspended sediment and phosphorus was determined (Wang, Liu, Miao, & Zuo, 2015). The positive correlation was attributed to the adhesive nature of the particulate form of phosphorus (Wang et al., 2015). Consequently, efforts that attempt to reduce the frequency and amount of runoff would help mitigate the issues of sediment and associated phosphorus inputs to water bodies.

Limiting the amount of excess nutrients and sediment that reaches water as a result of anthropogenic activities is the most common strategy for managing eutrophication. Urban and suburban areas are cited as being important contributors to nonpoint-source pollution (Soldat & Petrovic, 2008). Despite the fact that these areas are growing at a rate of 567,000 ha/yr limited research on nutrient losses from these areas has been undertaken (Soldat & Petrovic, 2008). In the United States, Scotts Limited Liability Company reports that 56% of the 90 million homeowners apply lawn fertilizer (Soldat & Petrovic, 2008). These application rates vary with the products manufacturer, but are typically 195 kg/ha/yr for nitrogen and 7 to 32 kg/ha/yr for phosphorus (Soldat & Petrovic, 2008). Moreover, other aspects of lawn management can impact nutrient inputs from the urban landscape. For instance, removing grass clippings after mowing can remove 2 to 15 kg/ha/yr of phosphorus inputs (Soldat & Petrovic, 2008). Although the application of fertilizer by homeowners may contribute to the problem of nutrient pollution, it is also the compound effect of alterations to the natural landscape that can make urban development a risk to water quality.

1.2. Turfgrass studies

Research on runoff from turfgrass can be sorted into three general categories (i) plot-scale, worst-case scenario research where runoff is simulated on small plots shortly after a fertilizer application is made, (ii) plot-scale research where runoff is collected from natural precipitation or rainfall events, and (iii) watershed-scale research where runoff losses from turfgrass areas are estimated by changes in flow and phosphorus concentration of a water body flowing through a turfgrass-dominated landscape (Soldat & Petrovic, 2008). Nutrient loss from golf courses could be overestimated during large rainfall events and field studies can be limited by the ease with which large precipitation events are sampled compared to small events.

As a result of housing and industrial development the amount of impermeable surfaces preventing natural pathways of the hydrologic cycle have increased. Therefore, in an urban landscape turfgrass associated with golf courses, turf farms, city parks and lawns may help to protect water quality (King, Harmel, Torbert, & Balogh, 2001). Turfgrass is the surface layer of the earth that contains a dense growth of grass with shoots that can be very dense, ranging from 7,500 to 2 million shoots per square meter (Soldat & Petrovic, 2008). In addition to the high density of turfgrass shoots, the associated root systems also provide a dense network for soil stabilization (Beard & Green, 1994). Therefore, a common strategy to protect water quality involves the use of vegetative buffers to border the boundary between human development and water bodies. However, relatively few controlled studies have been undertaken to assess the performance of turfgrass as a buffer, and very few models are available to predict their efficiency (Deletic & Fletcher, 2006).

It has been proposed that when runoff occurs various degrees of soil erosion also takes place and dissolved nitrogen and phosphorus are transported to water bodies (Balogh, Leslie, Walker & Kenna, 1992). Moreover, it has been shown that nutrient loads in runoff are the product of concentrations and runoff volume (Soldat & Petrovic, 2008). Hence, reductions to runoff volume from urban areas will lower sediment loss and nutrient loads. One reason turfgrass can be a viable option for buffer areas is due to the efficient removal of water from the soil which lowers runoff and leaching potential by reducing the soil moisture (Easton & Petrovic, 2004). It has also been suggested by Moss et al. (2006) that turfgrass is able to reduce sediment transport because of two complimentary mechanisms; sediment capture and the provision of a physical barrier to slow surface water flow. For instance, when sediment loss from creeping bentgrass and perennial ryegrass was measured there was no detectable sediment in 83% of 237 runoff samples (Soldat & Petrovic, 2008). Additional studies on sediment loss from cool-season turfgrass species ranged from undetectable, to very low (3.2 to 16.2 kg/ha) (Soldat & Petrovic, 2008).

Furthermore, Kentucky bluegrass has been found to sequester up to 50% of applied nitrogen and 88% of applied phosphorus depending on the amount of fertilizer applied (Easton & Petrovic, 2004). Strategies that have been identified for minimizing nutrient losses include the use of phosphorus only when soil tests indicate it is needed, watering-in fertilizer after application, irrigating to avoid runoff, withholding fertilizer application before expected rain events and the development of vegetative buffers (Soldat & Petrovic, 2008). Research continues to be carried out in order to confirm the effectiveness of known strategies that mitigate nutrient loss from urban areas and to enable the discovery of additional efforts.

1.2.1. Impact of slope

When considering methods to limit nutrient inputs to water bodies a factor that can impact nutrient transport due to runoff is the slope of the landscape (Easton & Petrovic, 2005). This is because the soil on the slope of a hill can be shallow; meaning the depth from the soil surface to a restrictive soil layer is less than what is experienced on a more even surface (Easton & Petrovic, 2005). Shallower soil is prone to runoff because it becomes saturated quickly with excess precipitation, and once the soil has been saturated soluble nutrients are more easily transported (Easton & Petrovic, 2005). In a study conducted by Easton and Petrovic (2005) the effect of hill slope on runoff volume was analyzed. It was reported that as turfgrass became more established from 2001 to 2002 the average precipitation that generated runoff increased from 17.9 millimeters (20% became runoff) to 35.2 millimeters where 1.7% became runoff (Easton & Petrovic, 2005). Other research has proposed that the root channeling caused by turfgrass creates large interconnected pores in the soil thereby increasing the infiltration rate with subsequent reduced runoff (Lee, 1985). Similar findings have also been reported by Dunne and Black (1970) where no overland flow was generated because the infiltration capacity of the soil exceeded the rainfall intensities that occurred. However, the capacity for turfgrass to perform functional benefits is dependent on various factors that impact the overall health and resilience of the turfgrass.

1.2.2. Effect of fertilizer on turfgrass

Fertilization is one of the major turfgrass management practices and without an effective turfgrass fertilization program turfgrass will not respond sufficiently to other management strategies (Carrow, Waddington & Rieke, 2001). Fertilizer also plays an important role in the stress tolerance of turfgrass (Carrow et al., 2001). To investigate the impact of fertilizer use on groundwater Cohen, Nickerson, Maxey, and Senita (1990) conducted a study where 19 monitoring wells were installed on golf courses in Cape Cod, USA. The monitoring wells were positioned in such a way as to collect water that was not affected by nearby sources of contamination (upgradient) and water that was influenced by fertilization (Cohen et al., 1990). Over the period of the research it was found that different nitrogen sources did tend to influence nitrate leaching into groundwater (Cohen et al., 1990). One of the golf courses that applied the highest amount of slow release nitrogen fertilizer had the lowest concentrations of nitrate in groundwater, while another golf course that utilized more water soluble compounds had the highest nitrate concentration (Cohen et al., 1990). When the same golf course reduced nitrogen application in the following year, the groundwater concentrations of nitrate decreased (Cohen et al., 1990). Overall the nitrate concentrations were below 10 mg/L (Federal maximum concentration level) (Cohen et al., 1990). Therefore, alterations to fertilizer practices can reduce nitrate loss and overall it was not of concern.

Similarly, the use of phosphorus-containing fertilizers has been restricted by implementing policies in some areas (Lehman, Bell, & McDonald, 2009). Ann Arbor in southeast Michigan is one area that has put this strategy into action and Lehman et al. (2009) carried out sampling a year after such a policy was put in place to determine its ability to meet the predicted 25% reduction in total phosphorus. Water samples were collected weekly for a year from locations that were both inside the boundaries and outside of the area affected by the ordinance (Lehman et al., 2009). The results found that no reductions were occurring in the area outside of the area affected by the policy but where it was in place a 28% reduction was recorded which was consistent with the predicted value (Lehman et al., 2009). However, as a result of multiple changes taking place at the same time it is difficult to isolate a single cause for phosphorus reduction because broader efforts to reduce phosphorus were also carried out. These additional changes included public education efforts about yard waste discharges into storm drains, more diligence regarding buffer strips along stream banks and more environmental awareness in general (Lehman et al., 2009).

In Duluth, Minnesota another study examined watershed scale changes in phosphorus concentrations and loading after alterations were made to phosphorus management at a golf course (King et al., 2012). Over the duration of the study fertilizer management went through a transition from large applications of commercial fertilizer in the years of 2003 to 2006 to a frequent low dose application of organic blends for the 2007 to 2010 study years. This reduced the application rates by >75% (King et al., 2012). Inorganic phosphorus forms used were mono-ammonium phosphate, di-ammonium phosphate, ammoniated normal super phosphate, triple superphosphate, and calcium metaphosphate (King et al., 2012). The majority of the phosphorus was applied from April through June to boost root development and accelerate greening of the course. Large applications of commercial fertilizer were used for this purpose; whereas reduced levels of phosphorus were applied from July through October at rates to maintain a level of desired turfgrass quality (King et al., 2012). When fertilizer treatment changed to organic formulations it was applied at a reduced level for the months of April to October with slightly higher amounts used during May to August because this was the active growing season of the grass (King et al., 2012). Primary organic fertilizer sources included fish extract, liquid seaweed concentrate, yucca and black strap molasses and compost growers' tea (King et al., 2012). The results showed a smaller percentage (20%) of the samples exceeded the reference value when organic formulas were used with a larger percentage (37%) of exceedances when commercial fertilizers were used (King et al., 2012). Similarly, monthly total phosphorus concentrations exceeding the 0.05 mg/L threshold were much more frequent during the inorganic commercial practices compared to the organic formulations (King et al., 2012). However, irrigation amounts also decreased during the second study period and King et al. (2012) speculates that increased tile flow following irrigation in period 1 may have contributed to the majority of phosphorus.

Therefore, since multiple changes were made from one study period to the next a single method for reducing phosphorus inputs cannot be suggested.

Other research similar to King et al. (2012) has found that the source of fertilizer can influence nutrient loss from turfgrass because of differences in solubility. Shuman (2003) conducted a study that evaluated the effect of eight fertilizer sources on nitrogen and phosphorus levels in the leachate from simulated golf greens. Fertilizer was applied four times to greenhouse plots (40 x 40 x 15cm deep) throughout the study at a rate of 11 kg/ ha for phosphorus and nitrogen was added separately at 24 kg/ha (Shuman, 2003). When the results of the samples collected were analyzed it was found that most fertilizers are the same as far as the leaching of phosphorus with only the very soluble sources of 20-20-20 fertilizer and the 16-25-12 starter fertilizer resulting in significantly more leaching (Shuman, 2003). These soluble formulations had the highest concentrations of phosphate in the leachate with 43% of that added leaching through the soil while the other formulations varied from 15% to 25% of the amount added being leached (Shuman, 2003). During the analysis of nitrate leaching in the greenhouse samples the highest cumulative mass leached out was for the soluble 20-20-20, the agricultural grade granular 10-10-10 and the liquid N source (Shuman, 2003). However unlike what was found for phosphate leaching, coated materials were able to significantly lower the loss to 1.4% and 0.7% of that added (Shuman, 2003). Nitrate was found to leach through the soil more rapidly than phosphate, but the percent of applied material that leached out was lower for nitrate than phosphate. This may be due to the more efficient use of nitrate than phosphate by the turfgrass and the coated sources tend to keep the nitrate in the root zone for a longer time than the uncoated granular and liquid sources (Shuman, 2003). In addition to the source of fertilizer, the age of turfgrass can also affect the amount of nutrient loss.

A large portion of the research that has been conducted to analyze how turfgrass management affects nitrogen leaching has focused on younger stands and it has been suggested that with the age of turfgrass the nitrogen requirement decreases (Barton & Colmer, 2006). For instance, it has been suggested that cool-season turfgrass nitrogen requirements remain the same for the first 10 years after establishment, and then continue to decline for up to 60 years (Barton & Colmer, 2006). In addition, the amount of nitrogen that should be applied to established turfgrass varies depending on the species, but typically ranges from 100 to 300 kg/ha/year (Barton & Colmer, 2006). At these application rates nitrogen leaching is not significant from established turfgrass when irrigated at a rate that maintains the soil water in the rooting zone. However, if clippings are not removed from turfgrass during mowing practices nitrogen needs of the turfgrass can be altered because returning grass clippings has been shown to reduce fertilizer requirements by 30%

to 75% (Starr & DeRoo, 1981; Barton & Colmer, 2006). Consequently if the amount of nitrogen applied is not adjusted adequately, nitrogen leaching could become a problem (Barton & Colmer, 2006). Another strategy to minimize nutrient loss from turfgrass is to apply fertilizer during active growing times, which can vary depending on grass species (Barton & Colmer, 2006). Turfgrass species can be classified in two general groups; being warm or cool-season species. When warm-season species are being considered it is suggested that fertilization should take place during the warmer months of late spring to early autumn. In comparison, cool-season grass species should receive fertilizer in early spring and late autumn (Fig.1) (Barton & Colmer, 2006). Consequently the impact of different management practices on nutrient inputs to water bodies can vary with grass species.

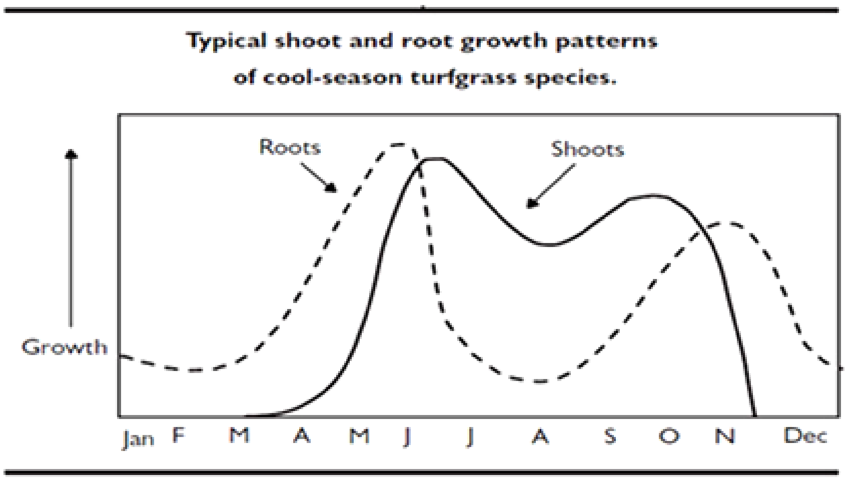

Figure 1. Cycle of active growing times for cool-season grass species (Owen, Lanier, Ebdon, & Spargo, 2013).

1.2.3. Effect of turfgrass species

Despite the fact that turfgrass species have been shown to directly influence nitrate leaching a limited number of studies comparing turfgrass species have been carried out (Barton & Colmer, 2006). One study that compared warm season species determined that nitrate leaching was greatest for Meyer' zoysia grass (*Zoysia japonica* 55 kg/ha/yr) and lowest for St Augustine grass (*Stenotaphrum secundatum* 3 kg/ha/yr) (Barton & Colmer, 2006). Differences between these two warm season species were attributed to differences in root length density at soil depths, with greater root length densities improving nitrate uptake (Barton & Colmer, 2006).

Rather than different turfgrass species being compared, comparisons of turfgrass species cultivars appear to be more frequently carried out. For instance a study analyzed the nitrate uptake rate of six different cultivars of Kentucky bluegrass (Jiang, Sullivan, & Hull, 2000). Results of the study found that total nitrate uptake was positively related to total nitrogen recovered and the cultivar with the highest uptake rate differed when analyzed based on root length, root weight and temporal variation (Jiang et al., 2000). Furthermore when two genotypes of Creeping bentgrass (a cool-season species) that had shallow and deep roots were compared the shallow root genotype leached 38% of applied nitrogen while the deeper root genotype leached 18% (Barton & Colmer, 2006). It has been suggested that when differences in the root depth of different turfgrass types are known nutrient leaching from shallow root types can be mitigated by lowering the irrigation rate and delaying irrigation after applications of fertilizer (Barton & Colmer, 2006). Although different grass species have biological factors that impact their nutrient use, physical factors such as shoot density can also be a factor.

1.2.4. Effect of seed density

Research has been performed where turfgrass was seeded at different rates to produce a range in shoot densities for studying the influence on runoff volume (Soldat & Petrovic, 2008). After simulations were carried out to force runoff to be generated differences in the amount of runoff volume were not detected, even when shoot densities were low (867 to 5,692/m^2) compared to what is commonly observed for turfgrass densities (7,500 to 2 million/m^2) (Soldat & Petrovic, 2008). Another study found that infiltration increased from 7 to 21 cm/hr as turfgrass shoot density increased from 60,000 to 120,000 shoots/m^2 and plots that received fertilizer had higher shoot densities than a no fertilizer control (Easton, Petrovic, Lisk, & Larsson-Kovach, 2005). Moreover, it has been found that when Kentucky bluegrass density and soil organic matter content are high fertilizer applied in late autumn does not increase the risk of nitrogen leaching (Barton & Colmer, 2006).

In comparison, some research has reported results that indicate shoot density is of less influence. For example, a study was performed without controlling the soil moisture between two turfgrass species. Creeping bentgrass was found to generate less runoff than Perennial rye grass; which was attributed to a higher shoot density for Creeping bentgrass

(Linde, Watschke, Jarrett, & Borger, 1995). However, when the study was repeated and the soil moisture was controlled, no difference was observed indicating water management is more important than shoot density (Linde et al., 1995). Maintaining turfgrass at taller heights has been suggested to reduce shoot density, but taller turfgrass has also been observed to reduce runoff volume (Soldat & Petrovic, 2008). Nutrient loss from turfgrass systems can also be influenced by the time of year.

1.2.5. Effect of seasons

Some studies suggest seasonality is a factor that affects nitrogen and phosphorus exports from turfgrass. This is due to changes in plant physiology that result from decreasing temperatures (King et al., 2006). King et al. (2006) while studying a golf course in Austin, Texas reported substantial losses of nitrate and dissolved reactive phosphorus during the months of October through March. Nitrate losses were the highest in December and January while maximum dissolved reactive phosphorus concentrations occurred during October and November (King et al., 2006). It was suggested that because bermudagrass has optimal growing temperatures between 27°C and 38°C the temperatures during fall and winter cause the grass to enter dormancy resulting in reduced nitrate utilization (King et al., 2006).

Lloyd, Soldat and Stier (2011) stated that research on nitrogen uptake during cold temperatures is sparse and environmental concerns exist regarding nitrate leaching. To address this they performed a study comparing cool-season turfgrass species Creeping bentgrass, Kentucky bluegrass and Annual bluegrass in three climate regimens corresponding to the months of September, October and November (Lloyd et al., 2011). Results of their study found that all turfgrass species had an increase in shoot growth in response to nitrogen application in the September regiment, but not in October or November regiments (Lloyd et al., 2011). Moreover, nitrogen uptake was significantly lower in the November regiment compared to September with an average of 73% of fertilizer recovery in September compared with 57% and 38% in October and November, respectively. The results of this study indicate that nitrogen uptake capacity is greatly reduced as average daily temperatures approach 0 °C.

Other studies have also found that seasonality can influence nitrogen and phosphorus concentrations because of the difference between non-frozen and frozen soil conditions (Bierman et al., 2010). Kussow (2008) conducted a six-year study that found the majority of annual surface runoff occurred during winter months from December to March. In one of the study years 99% of annual runoff was collected over the winter and contributed 97.8% and 98.5% of the nitrogen and phosphorus, respectively (Kussow, 2008). This was thought to be a result of twelve separate runoff events that occurred during the winter months from repeated temporary thaws and two rainstorms in March (Kussow, 2008). Furthermore peaks in total nitrogen have been observed in February to March (Winter & Dillon, 2006) and Bierman et al. (2010) also recorded the greatest phosphorus losses in runoff during times of frozen soil. Hence, there is a need to investigate the effect of Canadian winters on nutrient export because of the combined effect of temperature changes on plant physiology and soil conditions that contribute to high levels of nutrient loss.

1.3. Golf course management practices

In the United States there are approximately 18,331 golf courses and 2,390 in Canada with an average size being 61 hectares (ha) for an 18 hole golf course (Baris, Cohen, Barnes, Lam, & Ma, 2010). Since golf courses use pesticides and fertilizers regularly, it is reasonable to assume that they will be sources of nutrient export to surface and groundwater (Hindahl, Miltner, Cook, & Stahnke, 2009). Moreover, the perception that golf courses are a source of nutrient export is often reinforced by information posted on the internet and public interest newsletters (Hindahl, et al., 2009) and research that is carried out under worst case conditions. Such as the study of Linde & Watschke (1997) where runoff was forced to occur 8 hours after fertilizer application and phosphorus leaching was reported to range from 1.7 kg/ha to 2.2 kg/ha. These losses were high compared to other research that observed phosphorus leaching of 0.2 kg/ha to 0.7 kg/ha (Soldat & Petrovic, 2008).

Despite the possible negative effects the golf course maintenance could have on water systems they are also promoted as a tool to improve water quality (Ryals, Genter, & Leidy, 1998). For instance, the research carried out by Ryals et al. (1998) stated that all analyses of the samples collected for pesticide and nutrient (nitrogen and phosphate) testing from the outflows of three golf courses in North Carolina were below detectable levels. This is because while golf course maintenance poses a threat to water quality, it also has the potential to improve surface water quality by providing increased turfgrass health and resulting natural pathways of infiltration for polluted water during precipitation events (Baris et al., 2010).

1.3.1 Fertilizer application

When field research is performed at golf courses, the amount of nitrogen and phosphorous fertilizer they apply can vary. In the research conducted by King et al. (2006) at a golf course in Texas, typical management practices for golf courses in the southern United States were carried out. Average annual mass applications of nitrogen for the study area were 103.3 kg/ha and phosphorus applications totaled 21.8 kg/ha (King et al., 2006). Fertilizer was applied by both dry broadcast and spray techniques throughout the year as a combination of organic, bio-stimulant, slow release, and fast release formulations (King et al., 2006). Table 1 is a summary of nitrogen and phosphorus application for a golf course studied by King, Balogh, and Harmel (2007).

Reported annual average commercial fertilizer application at Morris Williams Municipal Golf Course (MWMGC).

	Nitrogen			Phosphorus		
Year	**Greens**	**Tees**	**Fairways**	**Greens**	**Tees**	**Fairways**
	kg ha^{-1}					
1998†	612.8	115.1	219.0	69.3	0	39.8
1999	309.9	47.3	49.0	196.2	10.0	0
2000	505.2	206.2	82.9	161.0	63.8	27.6
2001	292.8	132.4	37.4	126.5	35.0	12.5
2002	220.1	195.6	48.9	84.0	48.9	0
2003‡	97.3	0	0	39.1	0	0
Mean annual	407.6	139.3	87.4	133.2	32.0	16.0

† 1 Apr. to 31 Dec. 1998.
‡ 1 Jan. to 31 Mar. 2003.

Table 1. Table from King et al. (2007). Annual average commercial fertilizer application rates for greens, fairways, and tees.

Further research of King et al. (2012) focusing on phosphorus stated that depending on the degree to which phosphorus is used by a facility 31 to 66 kg/ ha for low phosphorus use and 62 to 132 kg/ha for high phosphorus use can be applied (King et al., 2012). However, once soil phosphorous levels have accumulated, the phosphorus needs of the turfgrass are reduced and applications should be eliminated or decreased (King et al., 2012). Conversely, phosphorus is frequently present in fertilizer formulation being used to meet nitrogen demands (King et al., 2012). In comparison, annual average commercial fertilizer application rates for nitrogen application can be 36.5 kg/ ha (King et al., 2007). Thus, it can be concluded that a wide range of fertilizer amounts are applied to turfgrass and as indicated in Table 1, can vary depending on the location (greens, tees or fairways) within a golf course.

1.3.2. Inlet and outlet water from golf courses

The majority of the reviewed research has collected water samples generated from precipitation events. Accordingly the original nutrient state of the water would have been quite low, whereas collecting samples to examine nutrient levels in water before and after it passes through a golf course may provide a more realistic way to determine the impact of golf courses on water quality. These types of studies allow watershed scale processes to be incorporated into the research. King, Balogh, Hughes, and Harmel, (2007) took samples from an inflow and outflow location of a stream that transected a golf course in Austin, Texas for five years. The mean nitrate concentrations at the outflow location (0.44 mg/L) were significantly greater than the concentrations measured at the inflow location (0.30 mg/L) but stated as being small in magnitude (King et al., 2007). The maximum nitrate concentration measured in this study was 3.52 mg/L and approximately 3.3% of the applied nitrogen (36.5 kg/ha over 29.0 ha) was lost (King et al., 2007).

The same study found that phosphorus concentrations in the outflow water accounted for 6.2% (8.2 kg/ha over 29.0 ha) of applied phosphorus (King et al., 2007). Both the mean inflow (0.12 mg/L) and outflow (0.15 mg/L) measurements exceeded the EPA recommended limit of 0.10 mg/L for streams not discharging into lakes (King et al., 2007). The maximum concentration of phosphate measured in surface discharge was 0.99 mg/L (King et al., 2007).

In a similar study conducted by King, Balogh and Harmel (2007) nitrate and dissolved reactive phosphorus concentrations were measured in water as it entered and left the golf course. The results of the nitrate samples collected gave a range of concentrations entering the course from 0.0 to 2.3 mg/L compared to a range of outflow concentrations 0.01 to 3.5 mg/L (King et al., 2007). For the dissolved reactive phosphorus the range of concentrations measured at the inflow site was 0.01 to 0.90 mg/L compared to a range of 0.0 to 0.99 mg/L at the outflow location (King et al., 2007). During storm events the median concentrations of nitrate and dissolved reactive phosphorus were greater at the exit than at the entry with 11% of the applied nitrogen being lost and 8% of applied dissolved reactive phosphorus (King et al., 2007). The results of the two studies carried out by King et al. (2007) suggest that nitrate levels are often not of concern, while phosphorus concentrations were above the recommended level to guard against eutrophication even before entering the golf course.

Hindahl et al. (2009) performed a study at a golf course in the Pacific Northwest to investigate the impact of fertilizer applications on surface water quality. Samples of surface water were collected monthly for two years from a creek that entered and left the golf course (Hindahl et al., 2009). Inflow and outflow samples were analyzed for nitrate and orthophosphate (Hindahl et al., 2009). Nitrogen was applied to 11 individual areas (24.8 ha total) and the total amount applied during year one was 3,204 kg and 3,183 kg in year two. The amount of phosphorus applied in year one was 407 kg and 777 kg in year two (Hindahl et al., 2009). Results of this analysis showed that the exit point nitrate and orthophosphate samples had concentrations that were equal to or less than the corresponding entry samples (Hindahl et al., 2009). Therefore, the review of research that has been performed suggests monitoring of nitrate needs to continue but is frequently not exceeding recommended levels. However, it has been strongly indicated that phosphate persists to occur at concentrations that result in ecological and financial consequences.

1.4. Purpose

This study has been conducted to address some of the gaps that have been identified following a literature review on water quality impacts of turfgrass. For instance there appears to be a limited amount of research in a Canadian context and as mentioned before changes in plant physiology and soil conditions over the winter months can impact nutrient exports from turfgrass (King et al., 2006; Kussow, 2008; Lloyd et al., 2011). In addition, experiments were performed to examine the nutrient exports from golf courses under "worst case scenario" climatic conditions such as storm events (Linde & Watschke, 1997; King et al., 2007) and winter melt. Moreover, comprehensive studies addressing the impacts of fertilizer controls, type of turfgrass species and seed density on water quality are sparse and information on these important turfgrass management conditions would help turfgrass managers implement strategies that preserve water quality. Therefore, the objectives of the performed reserach were to determine the impact of fertilizer treatments, turfgrass species and seed densities 1) on pollutant (phosphate, nitrate, TSS, total phosphorus and total nitrogen) concentrations and total export during small to moderate precipitation events by conducting rain simulations 2) on pollutant (phosphate, nitrate, TSS, total phosphorus and total nitrogen) concentrations and total exports of already polluted water during storm events by conducting storm simulations and 3) on pollutant (phosphate, nitrate, and TSS) concentrations and total export of winter runoff during snowmelt. Based on the stated objectives, the hypotheses tested were:

1. Fertilizer treatment and grass species would not significantly affect the nutrient concentrations from turfgrass during rain simulations, but seed density would demonstrate a significant influence on TSS concentrations.

2. Fertilizer treatment would significantly affect the nutrient concentrations during storm simulations and seed density would impact TSS, but grass species would not demonstrate significance.

3. Fertilizer treatment would significantly influence phosphate concentrations during the winter months, but grass species and seed density would not significantly impact nutrient loss. In addition, nutrient concentrations during snowmelt would be higher than those experienced during the summer and fall.

Chapter 2. Materials and Methods

2.1. Rationale, 2015

Several experimental mesocosms were established to replicate what is typical of the rough portion of golf courses in terms of maintenance practices (Table 2). Specifically, the grass species, grass cut height, fertilizer formula and fertilizer application rates were controlled and adjusted to match the realistic conditions. The roughs were simulated because they often account for the largest portion of the golf course land area and are acting as a buffer zone along the water bodies. To determine what is considered to be typical of golf courses in the Simcoe County area, superintendents or head greens keepers of various courses were contacted and asked to participate in interviews. Those that participated in the interviews were Hawk Ridge Golf and Country Club (Orillia, ON.), Settlers' Ghost Golf Club (Barrie, ON.), Big Bay Point Golf and Country Club (Innisfil, ON.), Lake St. George Golf and Country Club (Washago, ON.) and The Briars Golf Club (Sutton, ON.).

Turfgrass Maintenance	Hawk Ridge	Settlers Ghost (SG) /Big Bay Point (BBP)	Lake St. George Country Club	Briars Golf Club
Grass Species	Rough: Kentucky bluegrass	SG: Creeping bentgrass BBP: Mostly Kentucky bluegrass	Rough: Kentucky bluegrass	Rough: Kentucky bluegrass
Distribution of course	Rough>Fairway>tee	SG: Rough (30-35 acres) BBP: Rough (15acres)	Rough>Fairway	Rough (50 acres)
Mowing Height	Rough: 2 inches	Rough: 2 inches	Rough: 1 inches	Rough: 2-3 inches
Clippings Practice	Rough: left on	Rough: left on	Rough: left on	Rough: left on
Fertilizer Type	Fall: fast release Summer: slow release	Nitrogen slow-release	Nitrogen slow-release	Slow release 46 : 0 : 0
Fertilizer Frequency	Once/year; released based on temperature	Twice/year (spring and fall)	Once or twice/year	Once or twice/year

Table 2. Summary of interviewees' responses when asked what were typical grass species, mowing height, fertilizer formula and fertilizer application rates for rough portion of golf courses.

2.1.1. Rationale, 2016

The major objective of the study in the summer of 2016 was to confirm the trends observed for both nitrate and phosphate during the 2015 rain simulation experiments as a function of time and fertilizer treatment. It was decided that two ("No fertilizer" and

“Typical”) of the five fertilizer treatments employed in 2015 would be repeated on Kentucky bluegrass (Table 5) to reduce the variables influencing nutrient measurements.

2.2. Location

For the establishment of mesocosms and subsequent sample collection, a property in southern Ontario located in the community of Gamebridge (44°29'47.8"N 79°09'11.9"W) was utilized during the growing season of 2015 and 2016. This area is part of the Great Lakes/St. Lawrence climate zone or hardiness zone 5b for plants (Natural Resources Canada, 2017). It is typical for this region to experience a humid continental climate with an average summer temperature that can be above 30ºC (86ºF) and the warmest month of the year being July (Ontario Tourism Marketing Partnership [OTMP], 2016). During the winter months temperature can drop to below -13ºC (9ºF), with the coldest month of the year being January (OTMP, 2016).

2. 3. Construction of experimental mesocosms, 2015

Mesocosms were made by using 70.2 cm x 40.0 cm x 16.5 cm (27.6” x 15.8” x 6.5”) polypropylene storage tubs (Fig. 2). The bottom containers were used for collecting infiltrate and had pieces of pipe placed in them which were cut to 15.24 cm (6”) in length to aid in supporting the weight of the top bin when it was filled with soil. Before soil was added to the top container it had a layer of coarse gravel, plastic mesh with 1 cm^2 openings and landscape tarp. This layering was done to allow the passage of infiltrate along the bottom of the top container without further removal or addition of nutrients happening from the soil before it entered the collection bin. To allow the movement of water from the top container to the collection bin a slit was cut in the bottom of the top bin and the lid of the collection bin (Fig. 2). Landscape tarp was used to prevent gravel from falling through the slits. For the collection of surface runoff that was generated by these mesocosms the top containers were modified so that collection troughs could be inserted at one end of the container and adjusted to match the ground level. These collection troughs were made to have an average slope of 8% but the bottom cut of each trough was made to match the level of the soil.

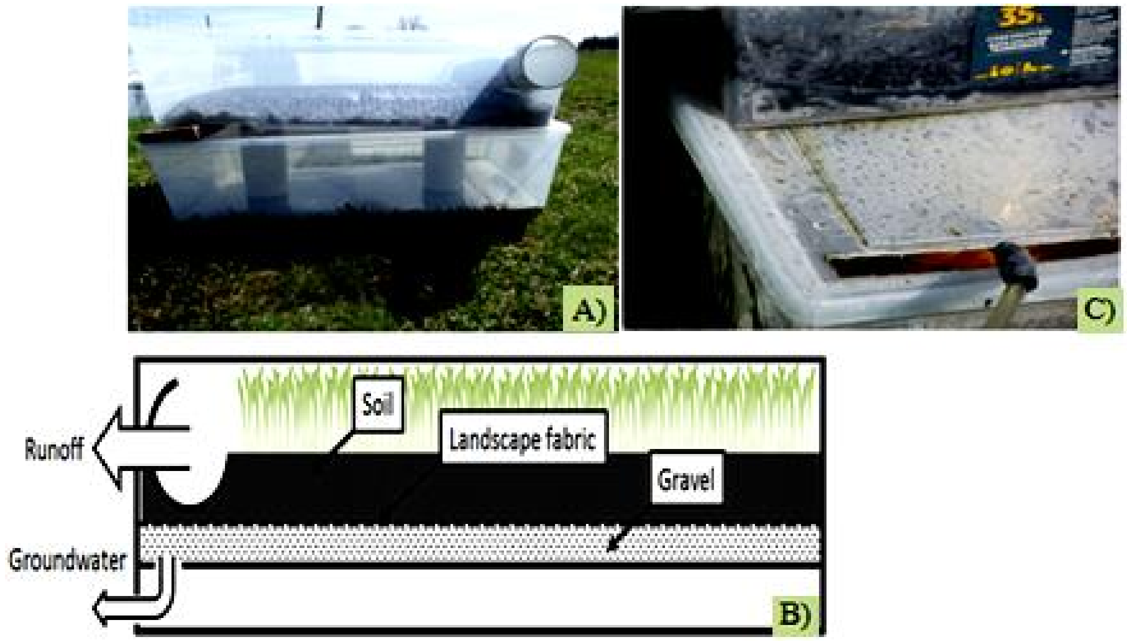

Figure 2. Display of the layering described in the bottom of the mesocosms A) and B) with dimensions 70.2 cm x 40.0 cm x 16.5 cm. Shown in C) is the type of cut that has been made to allow infiltrate to be collected.

The bins were filled with soil by adding Premier top soil (12.7 cm (5") and Scotts turf builder (a nutrient-enhanced topsoil/mulch mix for turfgrass establishment) on top (2.54 cm (1"), then lightly mixed together using a small gardening rake. Once the bins were filled with soil, gaps were noticed between the soil and plastic bin edge which created conditions favourable for preferential water flow to occur as mentioned by Barton and Colmer (2006); where water and nutrients move unevenly through the soil minimising the opportunity for plant roots to utilise applied water and nutrients. To prevent preferential water flow from taking place, stakes were placed on either side of each mesocosm. A total of 26 mesocosms (Fig. 3) were established all of which were adjusted to have a 5% slope and were seeded with either Barrister Kentucky bluegrass or Shark Creeping bentgrass provided by Quality Seeds Limited (Vaughn, Ontario). After the seeds were germinated the grass was maintained at a height of 2” in accordance with the interview responses and clippings were not removed.

Figure 3. Photograph showing the experimental mesocosms used in 2015 study

2. 3.1. Construction of experimental mesocosms, 2016

For this set of experiments new mesocosms were made in boxes made of plywood with the same dimensions for length and width as 2015. This was done to avoid the problems associated with the flexibility of the plastic containers. To solve the problem of water absorption by the wood, the interior of the boxes were made impermeable to water by applying a liquid rubber seal. Mesocosms were built with a slit in the bottom similar to the 2015 plastic tubs. In these boxes the same layering of materials (gravel, plastic mesh and landscape tarp) was used. The second container for infiltrate collection was eliminated by attaching a pipe that had been cut in half on a slope in one direction on the bottom of the box with silicone for directing infiltrate into a collection container. In order to allow the placement of a measuring cup under the new collection pipe the mesocosms were elevated off the ground by placing them on wood skids (Fig. 4). These new mesocosms were not as deep as the plastic ones used in 2015; therefore 7.62 cm (3") of Premier top soil was used to fill the boxes with an additional 2.54 cm (1") of Scotts turf builder as the top layer. The mesocosms were seeded on June 2, 2016, using 3.7g (3 lb of seed/1000 $ft.^2$) of Barrister Kentucky bluegrass seed provided by Quality Seeds Limited. All of the mesocosms were adjusted to have an average slope of 5% once established and maintained at height of 2" after germination.

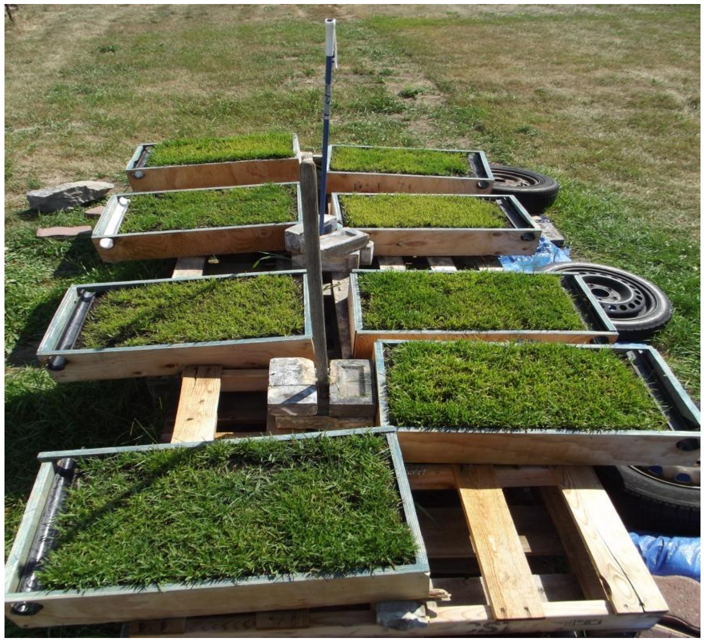

Figure 4. Photograph showing the experimental mesocosm used in 2016 study (Eight new mesocosms in 2016 on growing day 63).

2.4. Fertilizer treatments

Each mesocosm was labeled alphabetically from letters A to W. The mesocosms were seeded on May 29, 2015 but mesocosms W1 and W2 contained soil only and received no seed or fertilizer throughout this study so that "background" measurements of nutrient contributions from soil could be made. One fertilizer treatment that contained phosphorus was included in the 2015 study, while three other treatments were zero phosphorus formulas and the fifth treatment received no fertilizer. These consisted of "With phosphorous" (WP) (1 lb N, 1 lb P/1000 $ft.^2$), "Typical" (T) (1 lb N/1000 $ft.^2$), "Lower than typical" (LT) (0.5 lb N/1000 $ft.^2$), "Higher than typical" (HT) (2 lb N/1000 $ft.^2$) and "No fertilizer" (NF) which was the control treatment in this study. In 2016, the "Typical" (T) and "No fertilizer" (NF) treatments were repeated.

In 2015 four mesocosms per treatment were prepared. Of these, two were seeded with Kentucky bluegrass and two with Creeping bentgrass (Table 3). During the 2016 study each treatment (T and NF) was replicated 4 times with Kentucky bluegrass (Table 4). In 2015 two fertilization events were performed; one in August to simulate summer applications and one in October to simulate fall applications whereas only a summer fertilization in July was applied in the 2016 study period. The summer application of fertilizer used a slow release urea source with a polymer coating called XCU 46-0-0 (Alliance Agri-Turf). During the summer application of fertilizer mesocosms that were assigned to the T were given 2.4 g of XCU fertilizer as this was calculated to be the appropriate amount for the mesocosm size being used (2,808 cm^2). Corresponding amounts of fertilizer for the LT and HT treatments consisted of 1.2 g and 4.8 g of XCU per mesocosm, respectively. The WP mesocosms receiving phosphorous and otherwise "Typical" amounts of nitrogen had 2.4 g of monoammonium phosphate (MAP) 11-52-0

(Alliance Agri-Turf) of fertilizer added and an additional 1.8 g of XCU (an amount reduced from that of other "T" treatments to account for the nitrogen content of the MAP fertilizer) (Table 5).

Fall fertilization was accomplished by using a quick release 34-0-0 (Alliance Agri-Turf) ammonia sulphate fertilizer. 3.2 g each was applied to each mesocosm for T and WP treatments. The WP treatment also had 2.4 g of MAP, but a reduction to the ammonium source was not calculated so the WP treatment received a slightly higher nitrogen amount than the other T. LT and HT received 1.6 g and 6.4 g of the 34-0-0 formula, respectively (Table 6). Mesocosms U1-V2 were treated with fertilizer similar to the other four mesocosms receiving the T. Over the timeframe when fertilizer events occurred, best management practices identified by Shuman (2004) were followed such as light watering at the time of fertilization and rain simulations were carried out at least three days after fertilizer application.

Mesocosm	Fertilizer Treatment	Grass Type	Seed Amount
A	T	Kentucky bluegrass	3lbs/1000 $ft.^2$
B	T	Creeping bentgrass	3lbs/1000 $ft.^2$
C	T	Kentucky bluegrass	3lbs/1000 $ft.^2$
D	T	Creeping bentgrass	3lbs/1000 $ft.^2$
E	WP	Kentucky bluegrass	3lbs/1000 $ft.^2$
F	WP	Creeping bentgrass	3lbs/1000 $ft.^2$
G	WP	Kentucky bluegrass	3lbs/1000 $ft.^2$
H	WP	Creeping bentgrass	3lbs/1000 $ft.^2$
I	LT	Kentucky bluegrass	3lbs/1000 $ft.^2$
J	LT	Creeping bentgrass	3lbs/1000 $ft.^2$
K	LT	Creeping bentgrass	3lbs/1000 $ft.^2$
L	LT	Kentucky bluegrass	3lbs/1000 $ft.^2$
M	HT	Creeping bentgrass	3lbs/1000 $ft.^2$
N	HT	Kentucky bluegrass	3lbs/1000 $ft.^2$
O	HT	Creeping bentgrass	3lbs/1000 $ft.^2$
P	HT	Kentucky bluegrass	3lbs/1000 $ft.^2$
Q	NF	Creeping bentgrass	3lbs/1000 $ft.^2$
R	NF	Kentucky bluegrass	3lbs/1000 $ft.^2$
S	NF	Creeping bentgrass	3lbs/1000 $ft.^2$
T	NF	Kentucky bluegrass	3lbs/1000 $ft.^2$
U1	T	Kentucky bluegrass	1lbs/1000 $ft.^2$
U2	T	Kentucky bluegrass	1lbs/1000 $ft.^2$
V1	T	Kentucky bluegrass	5lbs/1000 $ft.^2$
V2	T	Kentucky bluegrass	5lbs/1000 $ft.^2$
W1	NA*	NA*	NA*
W2	NA*	NA*	NA*

Table 3. Summary of the 2015 fertilizer assignment, seed amount and grass species of each mesocosm.

NA* - Not Applicable because no seed or fertilizer applied.

Mesocosm	Fertilizer Treatment	Grass Type	Seed Amount
A	NF	Kentucky bluegrass	3 lb/1000ft.^{2}
B	T	Kentucky bluegrass	3 lb/1000ft.^{2}
C	NF	Kentucky bluegrass	3 lb/1000ft.^{2}
D	T	Kentucky bluegrass	3 lb/1000ft.^{2}
E	NF	Kentucky bluegrass	3 lb/1000ft.^{2}
F	T	Kentucky bluegrass	3 lb/1000ft.^{2}
G	NF	Kentucky bluegrass	3 lb/1000ft.^{2}
H	T	Kentucky bluegrass	3 lb/1000ft.^{2}

Table 4. Summary of 2016 fertilizer assignment, seed amount and grass species of each mesocosm.

Fertilizer Treatment	Amount of Fertilizer	Fertilizer Formula
NF	0 g	NA*
LT	1.2 g	46-0-0
T	2.4 g	46-0-0
HT	4.8 g	46-0-0
WP	1.8 g & 2.4 g	46-0-0 & 11-52-0

Table 5. Summary of 2015 summer fertilizer formulas and amounts for each treatment. The same amounts and formulas were used in 2016 for the repeated NF and T treatments.

NA*- formula not applicable to NF treatment because none applied

Fertilizer Treatment	Amount of Fertilizer	Fertilizer Formula
NF	0 g	NA*
LT	1.6 g	34-0-0
T	3.2 g	34-0-0
HT	6.4 g	34-0-0
WP	3.2 g & 2.4 g	34-0-0 & 11-52-0

Table 6. Summary of 2015 fall fertilizer formulas and amounts.
NA*- formula not applicable to NF treatment because none applied

2.5. Rain simulation

Construction of the rain simulator included a porous drip soaker hose arranged so that the pattern of drips was uniform over the surface of each mesocosm (Fig. 5). On the days when simulations were planned to occur the lower collection container was pre-rinsed with tap water. In addition, the slope was checked and adjusted as needed to maintain an average slope of 5% and soil moisture was measured with an average of 7.4 (39%) being aimed for and frequently achieved by watering the mesocosms the day before.

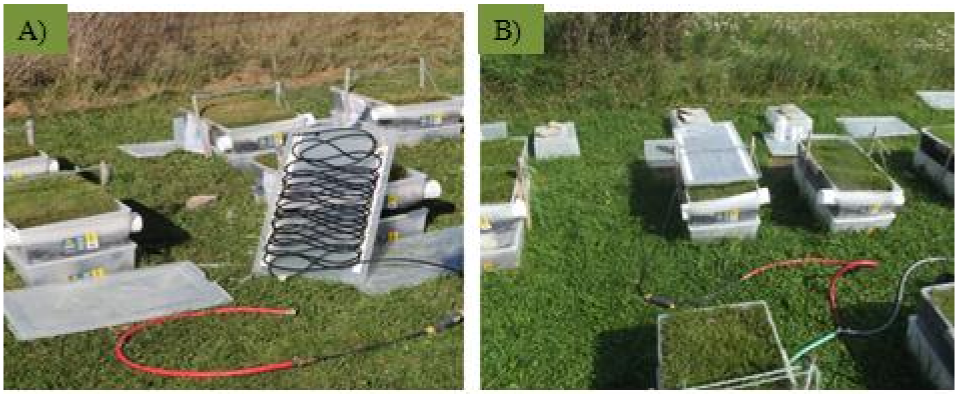

Figure 5. The rain simulator used for the experiment (A) and in use (B).

During 2015 four rounds of rain simulations were performed between June 30, 2015 (one month after seeding) and October 30, 2015. The purpose of the rain simulator was to achieve a flow rate comparable to a real precipitation event with a maximum flow rate of 8.4 ml/second; which was determined to be based on previous work conducted by Carmi and Berliner (2008), Shuman (2004) and King et al. (2001) that reported flow rates of 41 mm/hr., 27 mm/hr. and 152 mm/hr. respectively. When these flow rates were converted to ml/sec. based on the mesocosm size the corresponding flow rates were 2.3 ml/sec. (Carmi and Berliner, 2008), 1.5 ml/sec. (Shuman, 2003) and 8.4 ml/sec. (King et al., 2001). During the first day of rain simulations there were difficulties in achieving the desired flow rate as the lowest one that was measured was 13 ml/sec and the simulations were carried out by using this flow rate for mesocosms D, F, H, J, Q and S, all mesocosms containing Creeping bentgrass. For July 15 Kentucky bluegrass mesocosms N, P, R, T, V2 and U2, a lower flow rate was recorded of 9 ml/sec. For the remainder Kentucky bluegrass mesocosms (E, G, I, U1 and V1) sampled on July 16, an even lower flow rate of 6 ml/sec was recorded.

It was after these initial rain simulations that a flow rate of 8 ml/sec was achieved and utilized for all the subsequent simulations that took place. When rain simulations were completed for each mesocosm, the top bin was pulled back so that the modified hose of a shop vacuum could collect the generated infiltrate as demonstrated by the right hand image in Fig. 2; over the course of all rain simulations no surface runoff was produced. After the collection of water samples they were transferred into three different bottles with a measuring cup. The samples were later analyzed for nitrate, phosphate and TSS. The total volume of water collected was also recorded. In between each simulation the shop vacuum and measuring cups were rinsed with tap water three times.

In 2016 the same rain simulator was used and the same procedure was followed with slight modifications to sample collection. Due to the differences in construction of the mesocosms the shop vacuum was no longer needed and buckets were placed on the ground below the pipe channelling water towards them. Once the simulation began to generate infiltrate a measuring cup was used to collect water (Fig. 6) for phosphate, nitrate, total suspended solids (TSS), total phosphorus and total nitrogen samples; the total volume of water collected was recorded.

Figure 6. A 2016 rain simulation with raised mesocosms in progress (left) and collection of infiltrate with measuring cup (right).

2. 6. Storm simulation

Storm simulations were conducted to examine the capacity of the mesocosms to retain nutrients during storm events. This simulation created flow rates that could result from precipitation building up on impervious surfaces in urban areas such as a parking lot before running through turfgrass. The methodology for storm simulations was based on the information provided by a New Jersey manual on standard design of storm events (Blick, Kelly, & Skupien, 2004). In an attempt to match the variable influent flow rate during a design storm (Fig.7), three different flow rates were used in the creation of storm simulations.

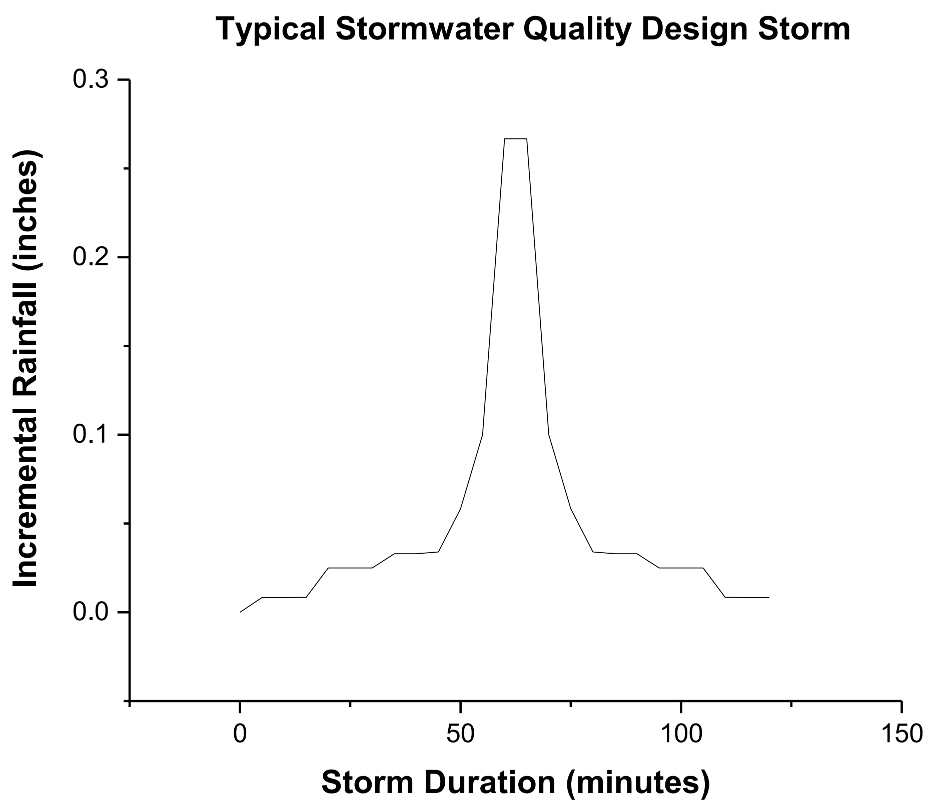

Figure 7. Stormwater Quality Design Storm based on precipitation rate of 1.25”/2 hrs. (Blick et al., 2004)

The design storm used flow rates and times for the first round of storm simulations as follows: 10 minutes at 165 ml/min, followed by 10 minutes at 500 ml/min, followed by 20 minutes at 1,500 ml/min, then 10 minutes at 500 ml/min and finally 10 minutes at 165 ml/min. However, the full duration of storm simulations (1 hour) was not always accomplished (average 41 minutes) because of the fixed volume capacity (35 L) for water collection. These flow rates were characteristic of what would be expected from a storm landing on an impervious surface as described by Blick et al. (2004). Runoff was generated for only 17 of the 26 mesocosms after the first round of storm simulations, possibly due to low soil compaction. Consequently, in the planning of the second round of storm simulations the highest intensity was increased to 3,000 ml/min while the other two flow rates remained the same in the hopes that the majority of the mesocosms would produce runoff. Problems with generating runoff are not uncommon because infiltration rates of turfgrass are generally much greater than normal precipitation rates and runoff generation is more a function of rainfall amount than intensity (Shuman, 2004; Easton & Petrovic, 2005; Dunne & Black, 1970). Controlling the lower flow rates for both rounds of simulations was done by using plastic cups that had holes punctured in the bottom. Water within the cups was maintained at an appropriate level to achieve the desired flow rate by inserting plastic pipe for an overflow outlet (Fig. 8).

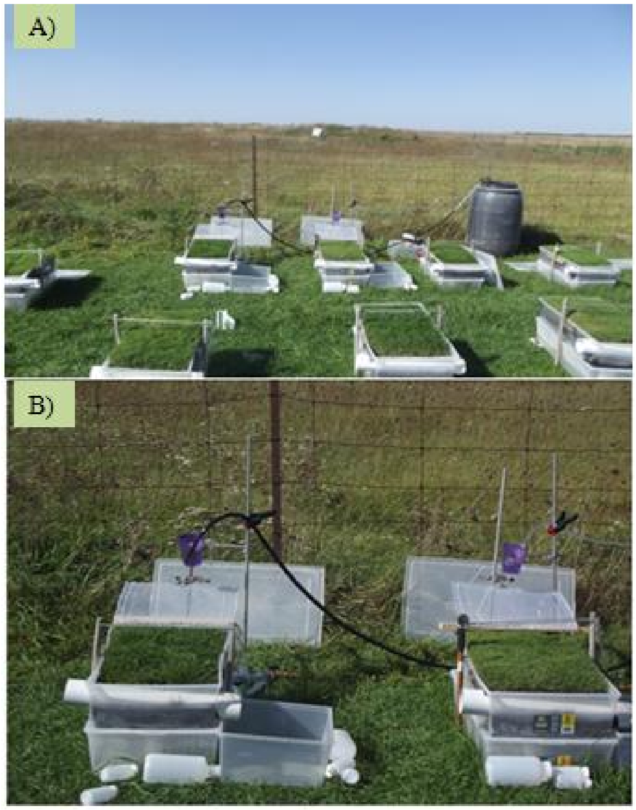

Figure 8. Image of storm simulations A) showing the whole set up with rain barrel, B) with a close up view of cups used for measuring flow rates.

In addition to subjecting the mesocosms to flows corresponding to a storm event running off of a parking lot, the concentration of pollutants in the inlet water used for these simulations was adjusted to simulate typical storm water. A 50 gallon water barrel was filled and had 4.5 g of NK 21-0-21 (Alliance Agri-Turf), 1.0 g of MAP 11-52-0 fertilizer added and 18.9 g of sil-co-sil 106 (U.S. silica, 2016) fine silica sediment was added. The main nitrogen source for NK 21-0-21 fertilizer is ammonia nitrogen instead of ammonium nitrate and is highly soluble (The Agromart Group, 2017). Silica sediment was used instead of natural sediment to prevent adsorption from occurring which could alter the nutrient concentrations of the inlet water. However, it is reasonable to assume that the data collected can be anticipated for natural sediment. Moreover, the inclusion of sediment during storm simulations is of interest because of its role in the transport of dissolved nutrients as discussed (Wang et al., 2015; Balogh et al., 1992). A submersible bilge pump was used to keep water circulating and to promote homogeneous distribution of sediment and nutrients in the barrel. The bilge pump was turned on one minute prior to initiating simulations. The preparation of the mesocosms in advance of storm simulations was the same as for rain simulations, and consisted of pre-rinsing the bottom collection containers, measuring the soil moisture of each mesocosm that was planned to be sampled that day, checking the slope and adjustments needed so that all mesocosms maintained an average slope of 5%. Storm simulations were not carried out in the 2016 season.

2.7. Winter snowmelt

To explore the effect of seasonality on phosphate and nitrate concentration in runoff within the Canadian climate, sampling of snowmelt was performed. The 26 field mesocosms were left exposed to accumulate snow for the months of December 2015 to March 2016 (Fig. 9). When weather was forecasted to be warm enough for snow melt to occur containers were positioned below the runoff troughs to collect the water generated (Fig. 10). In order to prevent these containers from blowing away they were safely fastened with the help of supporting objects such as rocks. Nutrient testing took place the following day after the collection of samples. This experiment was not repeated in 2016.

Figure 9. All 26 mesocosms covered in snow on March 3,2016

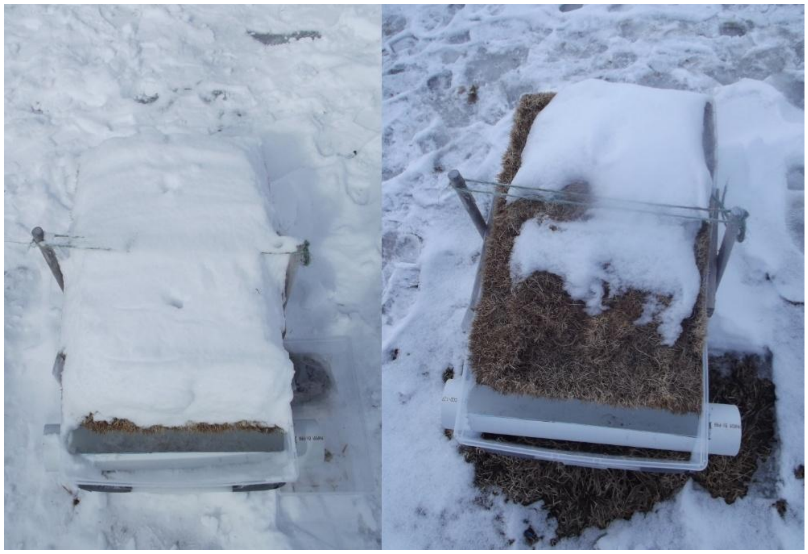

Figure 10. Mesocosm O on March 3, 2016 (left) and on March 6, 2016 after snowmelt.

2.8. Nutrients testing

Nutrient testing of the water samples collected from the mesocosms was carried out by following the standard analytical procedures. The nitrate content was analyzed by following the HACH method 8192, for low range nitrate (HACH, 2015). Phosphate concentrations were measured by following HACH method 8048 for reactive phosphorus, orthophosphates (HACH, 2014). Optical density measurement for nitrate analysis was carried out in a HACH DR 900 colorimeter while the phosphate concentrations were determined using either the HACH DR 900 colorimeter or Thermo Scientific Genesys 10UV spectrophotometer.

2.8.1. Phosphate

Processing of water samples collected for phosphate analysis was done using a modified version of the HACH method 8048 with ascorbic acid powder pillows and a determination of orthophosphates (PO_4^{3-}) (HACH, 2014). Each sample was shaken before 10 ml were poured into a glass vial, which had been previously rinsed with a 1:1 hydrochloric acid solution and allowed to dry. Then the contents of a PhosVer 3 pillow packet (HACH, 2014) was added to the sample and shaken for 30 seconds. After mixing the contents, samples were given a 10 minute reaction time before optical density was measured in either a HACH DR 900 colorimeter or a Thermo Scientific Genesys 10UV spectrophotometer at 880 nm. Final concentrations were calculated from a line of best fit obtained for a range of 0.02 mg/L (minimum detection limit) and 2.50 mg/L (maximum detection limit).

2.8.2. Nitrate

Water samples that were collected for nitrate analysis, if refrigerated, were left for a minimum of one hour to get to room temperature before being processed using the cadmium reduction method (HACH, 2015). Samples were shaken well before transferring 15 ml into a test tube along with the contents of a NitraVer 6 pillow packet. This mixture was then shaken vigorously for three minutes. 10 ml of this mixture was transferred to a glass vial and the contents of a NitraVer 3 pillow packet was added and shaken gently for 30 seconds. After a final reaction time of 15 minutes the glass vial was placed in the DR 900 colorimeter for reading the optical density measurement. The concentration was finally expressed in mg/L of nitrate (NO_3). Deionized water was used as a blank and treated identically. The detection range of this method was 0.01 mg/L to 0.5 mg/L.

2.8.3. Total suspended solids

The total suspended solids (TSS) analysis was carried out according to the procedure outlined by Environmental Protection Agency (EPA) U.S.A. (EPA, 1999). In this study, a larger volume of water (averaging 816 ml) instead of 700 ml as recommended by EPA (1999) was filtered and a lower drying temperature of 85 ˚C for 24 hours was used. After the completion of a rain or storm simulation, the samples were stored in a refrigerator until analysis (usually within a couple of days after collection). The water samples were filtered through Whatman glass microfiber filters (1.0 µm pore size) with a 42.5 mm diameter and the volume of water filtered was recorded. TSS was calculated by weighing the dried filters (before and after the filtration) in a microbalance. The following equation was used to calculate the total suspended solids:

$$\text{Total Suspended Solid:} \quad \text{mg/L} = \frac{(A-B) \times 1{,}000}{C}$$

where A is the weight of the filter with the residue in mg, B is the weight of the filter alone in mg and C is the volume of sample filtered in ml.

2.8.4. Total Export

As mentioned above, while rain, storm and snowmelt sampling took place the total volume collected was measured to allow for total export calculations to be carried out. This mass measurement was employed to provide a means by which high amounts of phosphate or nitrate due to high volume or high concentrations (or both) could be determined. Moreover, if changes in volume occurred this measurement would allow for a more accurate comparison with time. As such the total export for all sampling intervals was calculated to determine the export in mass (mg) using the following equation:

Total export: = A x B

where A is the measured concentration (mg/L) and B is the collected volume (L).

2.8.5. Total nitrogen and Total phosphorus

When storm simulations were carried out in 2015 additional samples were collected for total nitrogen and total phosphorus analysis. During the 2016 study period samples for total nitrogen and total phosphorus testing were collected during rain simulations only. These samples were placed in a freezer immediately after collection and transported to the Environmental Laboratory at the Lakehead University Centre for Analytical Services in Thunder Bay, Ontario. Total Nitrogen was performed using the Skalar autoanalyzer system. Sample was mixed with a potassium peroxodisulfate/sodium hydroxide solution and heated to 90 °C. The solution was then mixed with a borax buffer and all nitrogen species were converted by UV radiation to nitrate and colorimetric determination followed WNOX. This method accounts for nitrogen in the form of azide, azine, azo, hydrazone, nitrate, nitrite, nitro, nitroso, oxime and semi-carbazone, as opposed to the Total Kjeldahl Nitrogen method (SKALAR Methods for Total Nitrogen in Water Catrn# 475-426) (J.

Roarco, personal communication, December 5, 2016). The minimum detection limit was 0.05 mg/L.

Total Phosphorus was also carried out by following a UV digestible method in a Skalar autoanalyzer system where polyphosphate and some organophosphorus compounds are determined by converting to ortho-phosphorus. This was performed online with the Skalar autoanalyzer system by adding sulphuric acid to the sample stream and heating at 97°C. Following hydrolyzation, the sample underwent further digestion with peroxodisulfate under UV radiation generating the orthophosphate ion. The ortho-phosphate ions in a sample reacted online in an acidic solution containing molybdate and antimony ions to form a phospho-molybdic acid, which was reduced by ascorbic acid. For the determination of dissolved total phosphorus, the sample is filtered through a 0.45 mm filter and colorimetric determination followed WPO4 (SKALAR Methods for Total Phosphorus in Water Catrn# 503-010) (J. Roarco, personal communication, December 5, 2016). When conducting the total phosphorus the minimum detection limit was 0.005 mg/L.

2.9. Statistical analysis

All forms of nutrients or TSS examined in this study are graphed to show the changes in concentration over time. Time is represented by sampling intervals one to four, which describe growing days 10 to 39, 72 to 84, 122 to 133 and 138 to 151 respectively in the 2015 study period (Table 7). For storm simulations sampling interval one corresponds to growing days 101 to 112 and sampling interval two corresponds to growing days 162 to 172. Winter snowmelt sampling took place on growing days 234, 275 and 303 for sampling interval one, two and three respectively. In 2016, sampling intervals one to five occurred on growing day 32, 46, 54, 67 and 76 respectively.

Study Period	Sample Method	Sample Interval	Growing Days
2015	Rain simulation	1	10 to 39
2015	Rain simulation	2	72 to 84
2015	Rain simulation	3	122 to 133
2015	Rain simulation	4	138 to 151
2015	Storm simulation	1	101 to 112
2015	Storm simulation	2	162 to 172
2015	Winter snowmelt	1	234
2015	Winter snowmelt	2	275
2015	Winter snowmelt	3	303
2016	Rain simulation	1	32
2016	Rain simulation	2	46
2016	Rain simulation	3	54
2016	Rain simulation	4	67
2016	Rain simulation	5	76

Table 7. Summary of sample methods used for 2015 and 2016 with the growing days they took place on.

Variations in water nutrient concentrations and export values (dependent variable) due to fertilizer treatment, seed density and grass species (independent variables) were studied by performing one-way ANOVA analysis using IBM SPSS Statistics 20 software. Multiple factors were not analyzed at the same time (example treatment, species, time etc.) using Multivariate analysis in SPSS because not enough data was collected. A comparison of water nutrient concentrations for the 2015 and 2016 study period was done by conducting an independent t-test. In addition, the independent t-test was used for testing the difference between studied nutrient concentrations and export amounts of infiltrate and runoff, infiltrate and inlet water and runoff and inlet water, in the 2015 storm simulations. Sample size during storm simulations were not always the same because of the difficulties with consistently producing sufficient runoff volume. While performing one-way ANOVA analysis, if Levene's test for homogeneity of variance was violated, the Welch ANOVA was applied (Lund & Lund, 2013). The outliers were not removed because of the limited number of data points. After performing the one-way ANOVA statistical analysis, when significant variations were detected, a Tukey's post hoc analysis was carried out to study pair-wise variations.

Chapter 3. Rain Simulation Study

3.1. Phosphate and Total phosphorus

3.1.1. Impact of fertilizer treatment

Overall, the volume of infiltrate decreased with each sample interval (Fig. 11). For performing statistical analysis, phosphate concentrations that were below the detection limit were assigned a value of 0.01 mg/L (half the minimum detection limit). When all the treatments were considered, the phosphate concentration varied from below the detection limit to a maximum of 1.51 mg/L (Fig. 12). Type of treatment was not found to have a significant influence on the infiltrate phosphate concentrations (one-way ANOVA $F_{(4,14.98)}= 2.85$, $p= 0.06$). Soil only mesocosms had phosphate concentrations (average 0.49 mg/L) that could not be utilized as a “background” measurement because they were not consistently higher or lower than measurements collected from the other treatments. Infiltrate phosphate export was calculated using the total volume measurements of the infiltrated water. As such, phosphate export was not significantly affected by the fertilizer treatments (one-way ANOVA $F_{4,35}= 1.91$, $p= 0.13$). The total export for all treatments varied between below the minimum detection limit to 12.4 mg (Fig. 13).

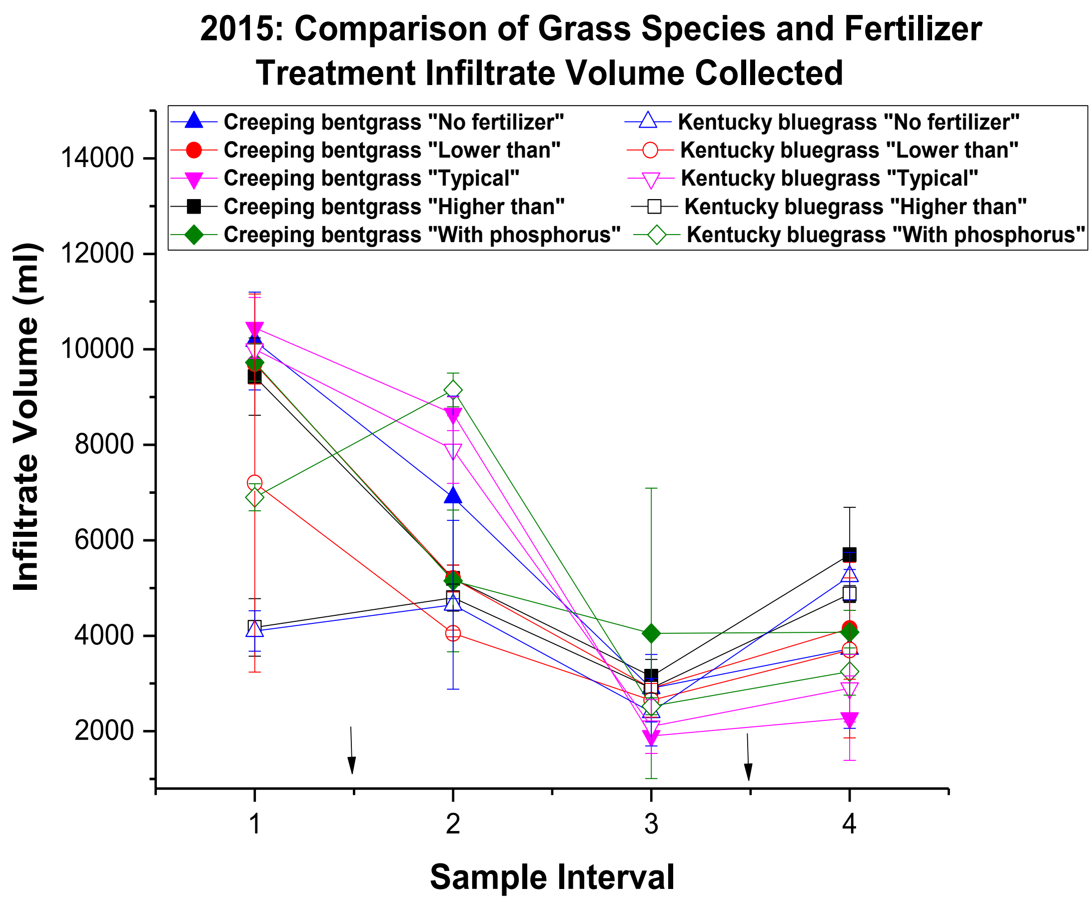

Figure 11. Changes in infiltrate volume collected with sample interval. Average volume of 11,746 ml applied for interval 1 and 9,960 ml for the remainder. Arrows indicate when fertilizer was applied and symbols show averaged volume collected. Error bars represent standard deviation.

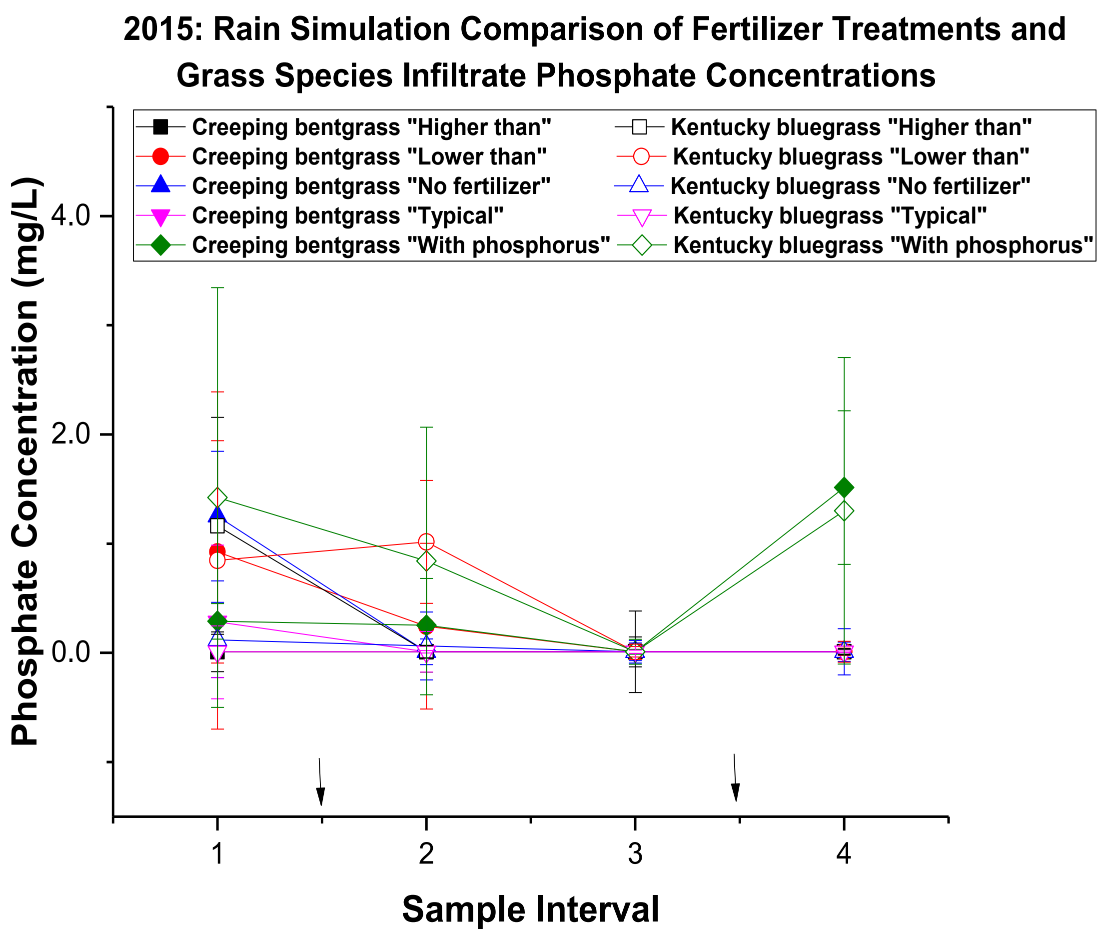

Figure 12. Changes in infiltrate phosphate concentrations with sample interval. Arrows indicate when fertilizer was applied and symbols show averaged concentration. Error bars represent standard deviation.

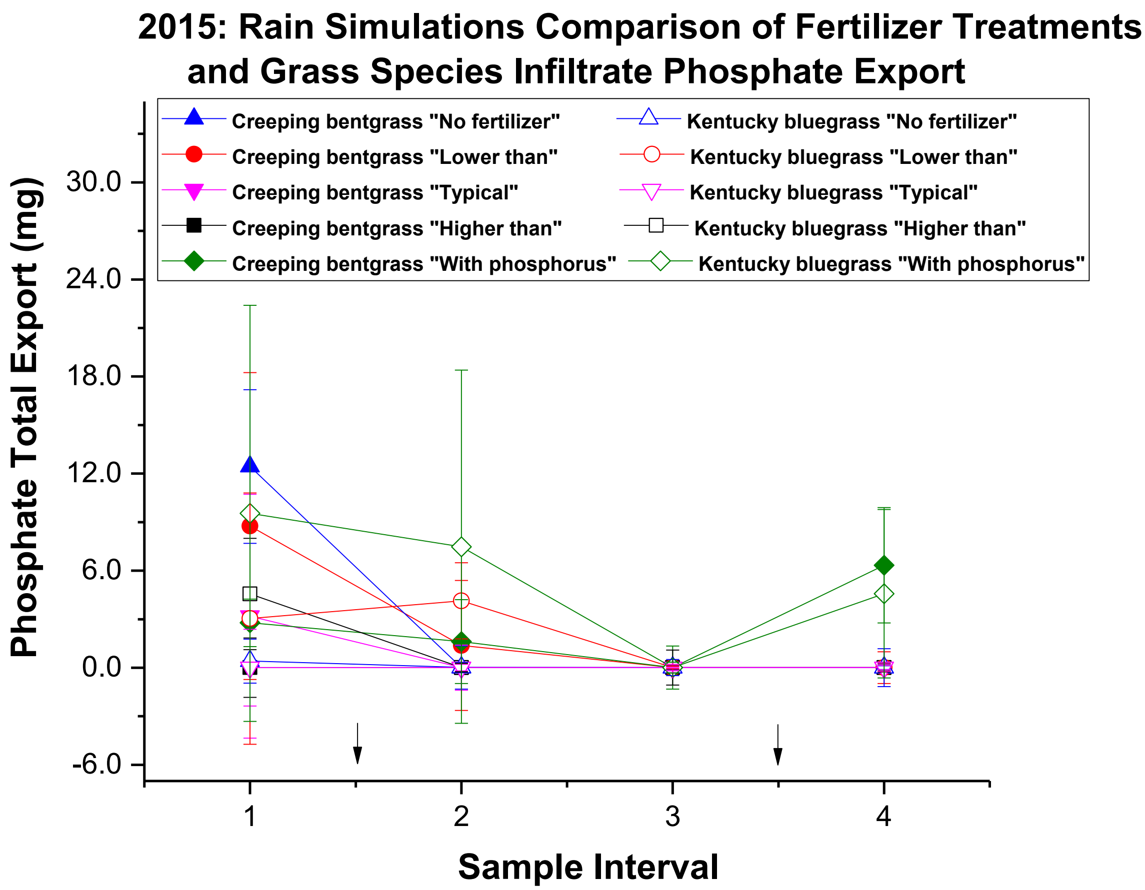

Figure 13. Changes in infiltrate phosphate total export with sample interval. Arrows indicate when fertilizer was applied and symbols show averaged export. Error bars represent standard deviation.

In 2016 the total infiltrate volume collected was consistent among all sampling intervals except for interval 2 (Fig. 14). During the majority of the sample intervals the flow rate was frequently measured to be 10 ml/second despite adjustments, rather than the target of 8 ml/second. It was only in sample interval two the flow rate of 8 ml/second was reached. Phosphate concentrations ranged from below the detection limit to a maximum of 1.14 mg/L (Fig 15). Variation in fertilizer treatments studied in 2016 did not significantly affect phosphate concentrations in infiltrate (one-way ANOVA $F_{1,8} = 0.06$, $p=0.82$), nor phosphate export ($F_{1,8}= 0.07$, $p= 0.79$) (Fig. 16). In addition, results of one-way ANOVA on fertilizer treatment in 2016 showed no significant variation in infiltrate total phosphorus concentrations ($F_{1,4}= 4.291$, $p= 0.10$) (Fig. 17).

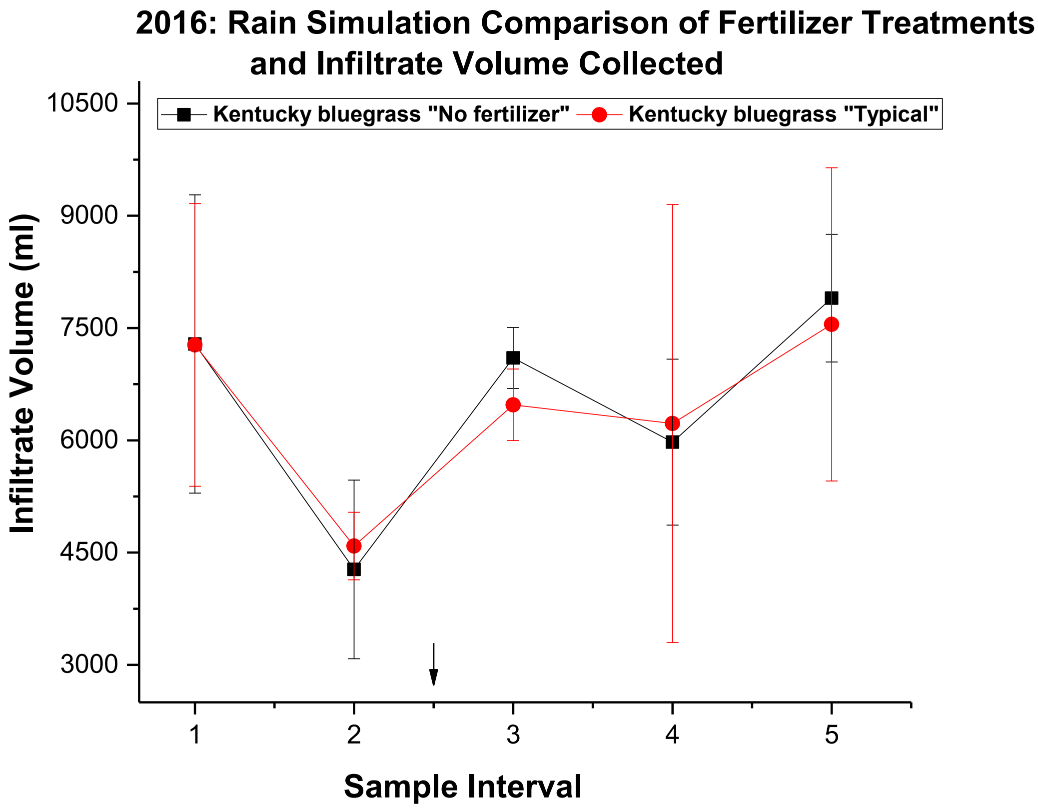

Figure 14. Changes in infiltrate volume collected with sample interval an average of 11,242 ml applied for interval 1, 3, 4 and 5. Interval 2 had an average of 9,480 ml applied. Arrow indicates when fertilizer was applied and symbols show averaged volume collected with error bars representing standard deviation.

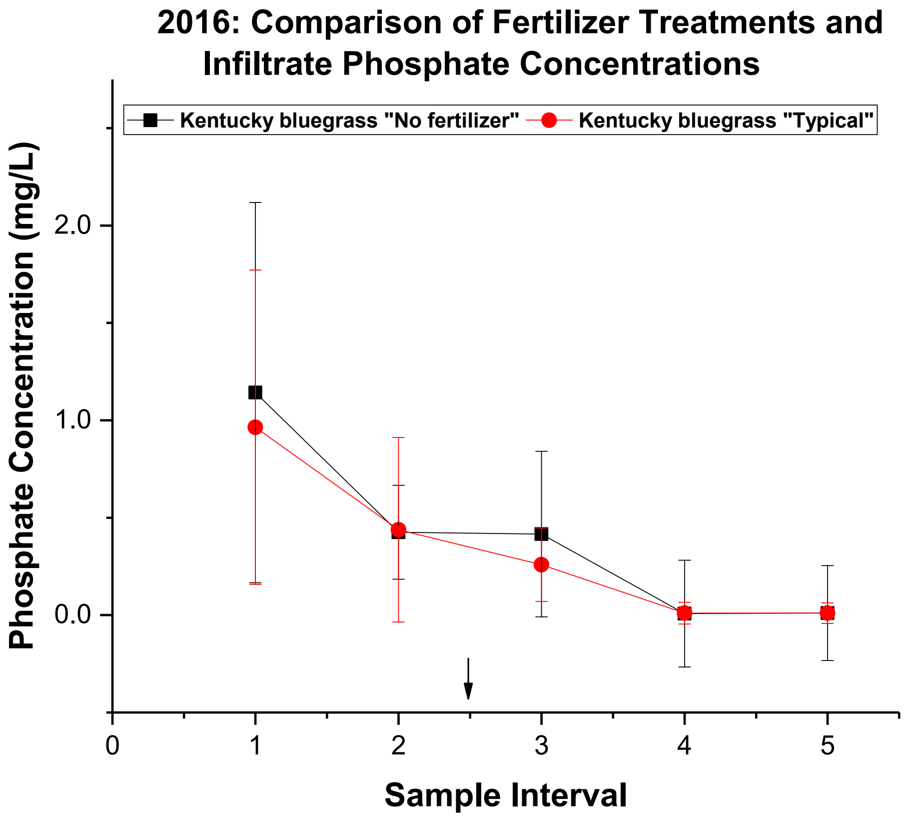

Figure 15. Changes in infiltrate phosphate concentrations with sample interval. Arrow indicates when fertilizer was applied and symbols show averaged concentration with error bars representing standard deviation.

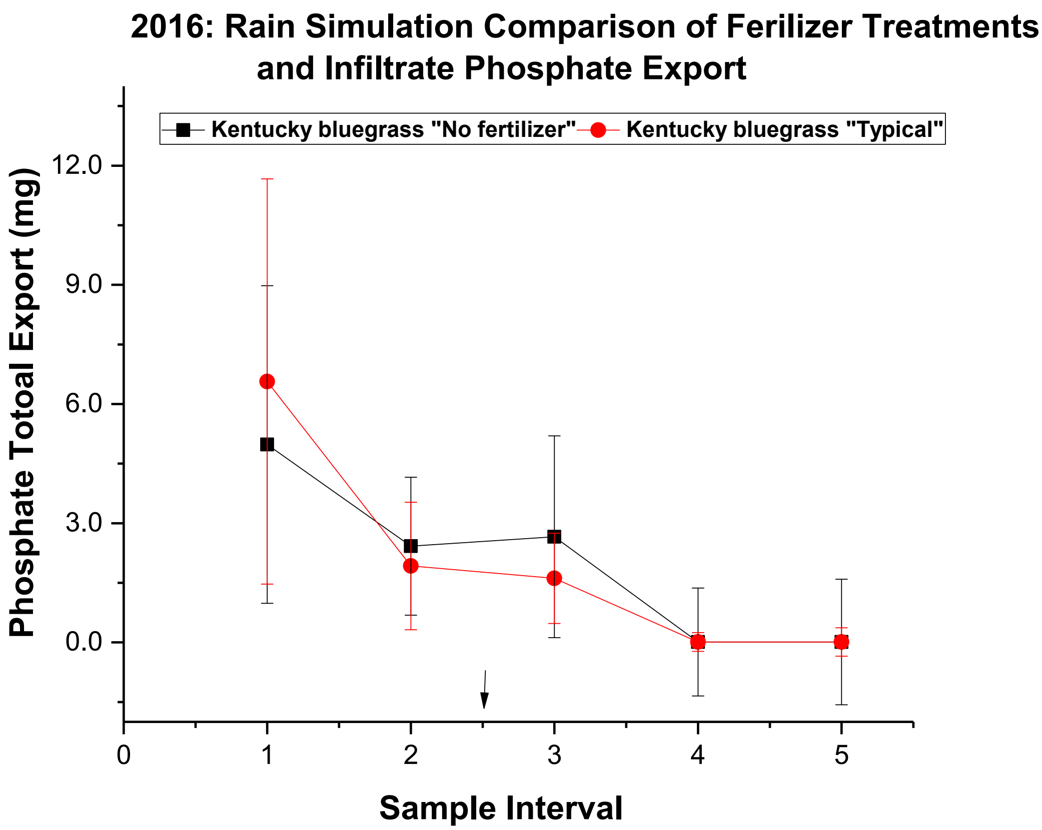

Figure 16. Changes in infiltrate phosphate total export with sample interval. Arrow indicates when fertilizer was applied and symbols show averaged export with error bars representing standard deviation.

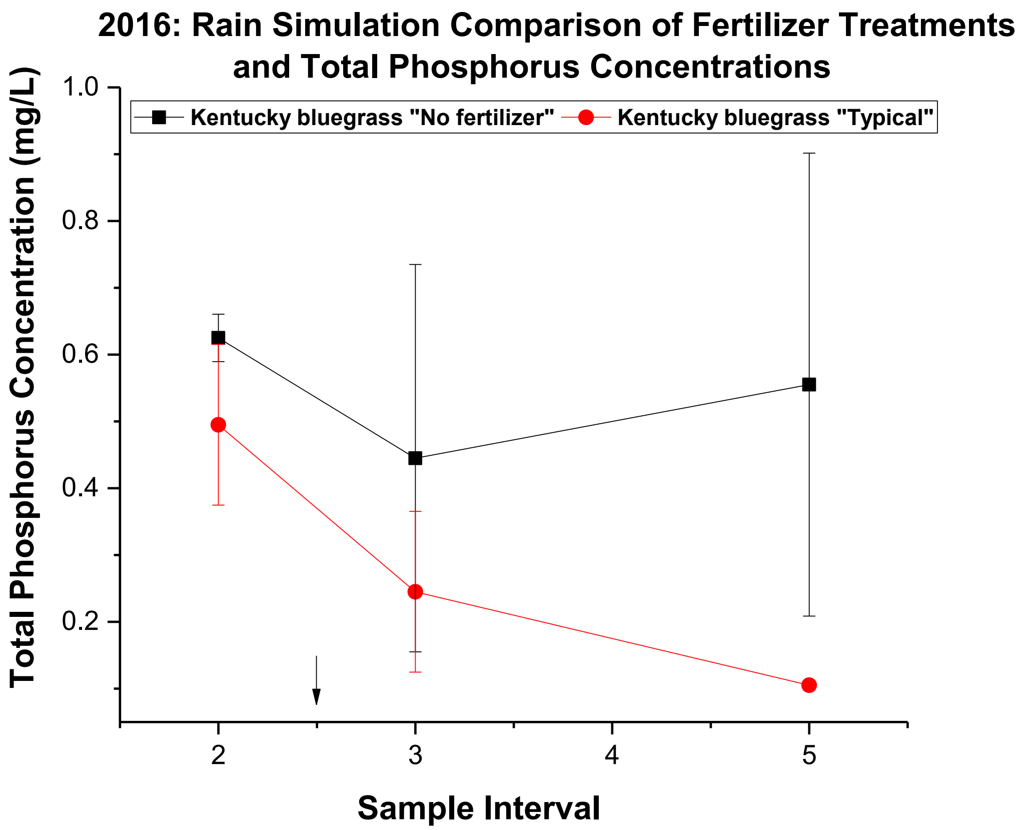

Figure 17. Changes in infiltrate total phosphorus concentrations with sample interval. Arrow indicates when fertilizer was applied and symbols show averaged concentration for sample interval 2, 3 and 5 with error bars representing standard deviation.

3.1.2. Impact of grass species

In the 2015 study period phosphate concentrations were not significantly influence by grass species (one-way ANOVA $F_{1,38} = 0.43$, $p= 0.52$). The minimum export of phosphate for both grass species was below the detection limit and a maximum export was 12.4 mg and 9.5 mg for Creeping bentgrass and Kentucky bluegrass, respectively. An analysis of total phosphate export in the infiltrate of each grass species showed no significant difference (one-way ANOVA $F_{1,38}= 0.02$, $p= 0.89$). Statistical analysis also revealed there was no significant difference in infiltrate phosphate concentrations for Kentucky bluegrass between the two study periods (t-test, $t= -0.125$, $p= 0.90$), nor in the total export (t-test, $t= -0.313$, $p= 0.76$).

3.1.3. Impact of seed density

The effect of seed density variation for Kentucky bluegrass was studied only in 2015 for the T treatment mesocosms. The seed densities examined in the study were 1 lb/1000 ft.^{2}, 3 lb/1000 ft.^{2} and 5 lb/1000 ft.$^{2.}$ After counting the shoot densities for these mesocosms it was determined that the seed density did not alter shoot density (Fig. 18).

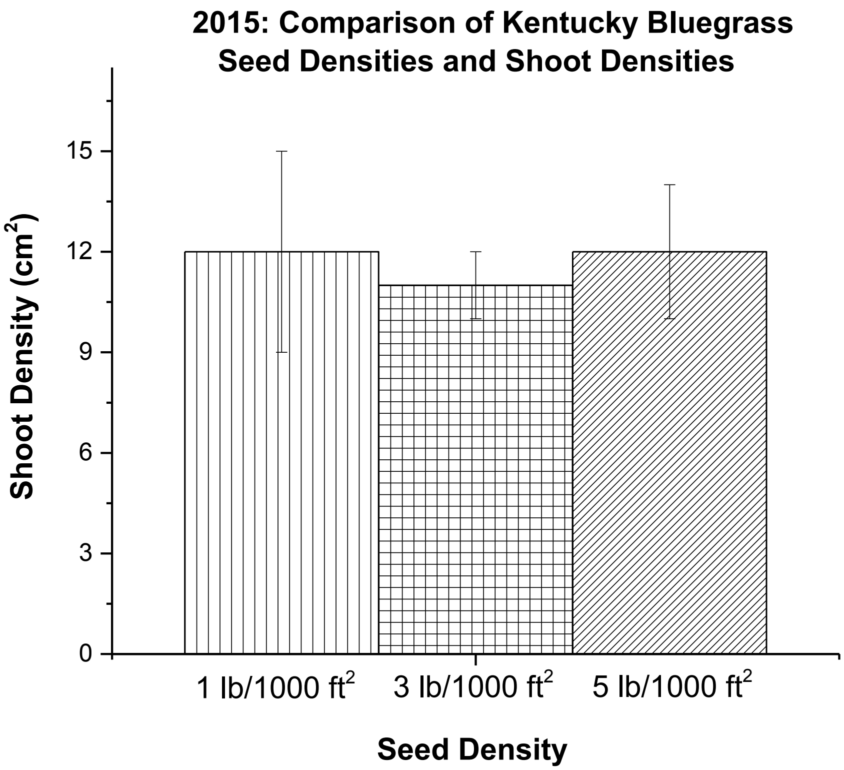

Figure 18. Shoot density of Kentucky bluegrass mesocosms with altered seed densities and treated with the “Typical” treatment. Bars show average density/cm^2 with error bars representing standard deviation.

Infiltrate phosphate concentrations for different seed densities (Fig. 19) showed no significant difference (one-way ANOVA $F_{(2,4.0)} = 1.13$, $p= 0.41$). The phosphate export was found to be below detection limit for 3 lb/1000 ft.^{2} seed density while the export varied between below the detection limit to 4.85 mg/L and 3.85 mg/L for 1 lb/1000 ft.^{2} and

5 lb/1000 ft.^{2} seed densities, respectively. Seed density did not significantly influence the export of phosphate in the infiltrate (one-way ANOVA $F_{2,9}$= 0.66, p= 0.54).

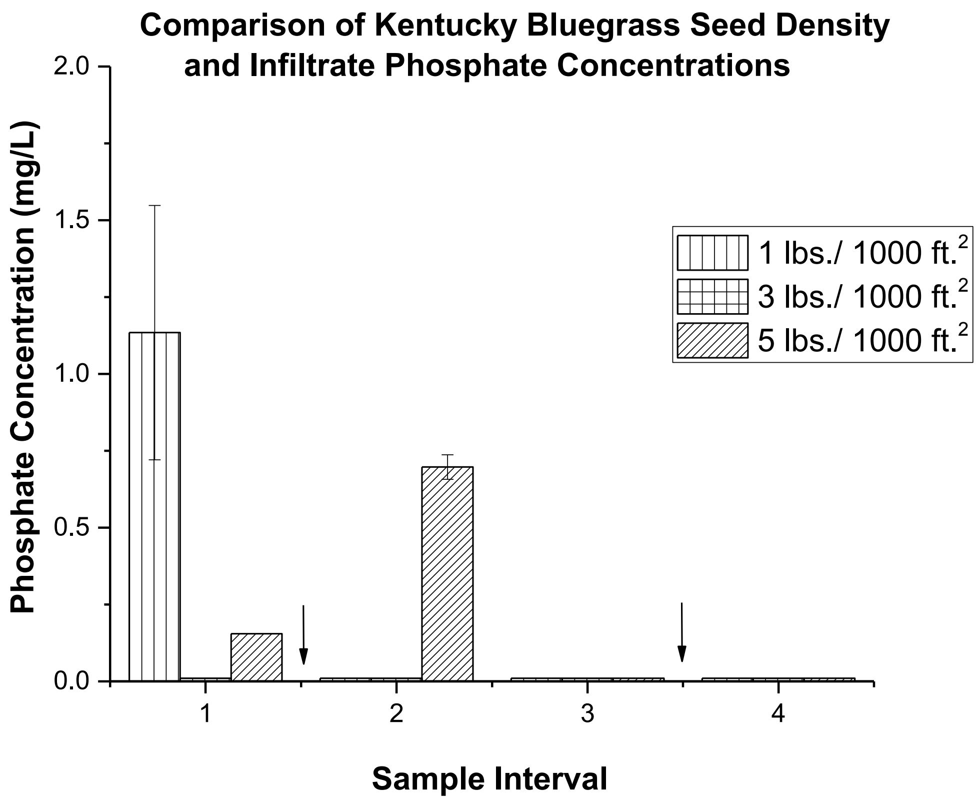

Figure 19. Average infiltrate phosphate concentrations for each seed density in 2015. Arrows indicate when fertilizer was applied and error bars represent standard deviation.

3.1.4. Discussion: phosphate and Total phosphorus

The phosphate concentration for the rain simulation studies demonstrated a general decrease with time during both the study periods. This trend of decreasing phosphate concentrations and export with sample interval was expected for all treatments not receiving phosphorus addition. In 2015 phosphate concentrations and infiltrate volume decreased with sample interval; whereas in 2016 phosphate concentrations and export declined while infiltrate volume collected remained fairly consistent (except for the decrease at interval 2); as such a correlation of phosphate on the amount of volume collected was not observed. Infiltrate volume collected in 2016 did not display a change with sample interval likely because all of the samples collected were within the same time frame as sample interval 1 and 2 for 2015 where a distinguishable decrease was also not experienced. Thus, 2016 infiltrate phosphate concentrations and export followed the same decreasing trend as shown in 2015 for all treatments not receiving phosphorus. Overall infiltrate phosphate concentrations exceeded the EPA guideline (0.10 mg/L) multiple times in the 2015 and 2016 study period.

Fertilizer treatment was not anticipated to have a significant effect on infiltrate phosphate concentrations because of efficient utilization during the growing season. However, the effect of the WP treatment on the concentration of phosphate in the infiltrate was noticeable (as expected) as is shown in Fig. 13 but not significant. In the 2016 study period fertilizer treatment was not found to significantly affect the infiltrate phosphate or total phosphorus concentrations.

When sampling began in the early phase of turfgrass development, it is reasonable to suggest that phosphate concentrations in infiltrate were higher than the remainder of the sampling because the root systems had not developed sufficiently to effectively utilize the available phosphorus within the soil (Wong, Chan, & Cheung, 1998). However, the noticed decrease in infiltrate phosphate concentrations for all treatments by sample interval 3 could be due to the root system being more developed as time progressed. In addition, the concentration gradient created by higher concentrations within the root cells than the soil requires energy to be expended for phosphate uptake to occur because phosphate cannot move freely into root cells (Hull, 1997). Over time phosphate uptake momentarily increases as a function of this mechanism (Hull, 1997).

Research carried out by Shuman (2002) also reported the highest concentration of phosphorus resulted from the first simulated rainfall event with a dramatic decrease in subsequent events (Shuman, 2002). This study showed grass species and density had no significant impact on the phosphate concentration of the infiltrate which is in agreement with Stier and Kussow (2006).

3.2. Nitrate and Total nitrogen

3.2.1. Impact of fertilizer treatment

The infiltrate nitrate concentrations in 2015 had a maximum of 5.70 mg/L, (Fig. 20). Fertilizer treatment was not found to significantly affect the infiltrate nitrate concentrations (one-way ANOVA $F_{(4,12.09)}$= 2.60, p= 0.09), nor export (one-way ANOVA $F_{(4,16.57)}$= 1.45, p= 0.26) (Fig. 21). Average nitrate concentration for the soil only mesocosms (1.81 mg/L) could not be utilized as a "background" measurement because it was not determined to be consistently higher or lower than all other measurements.

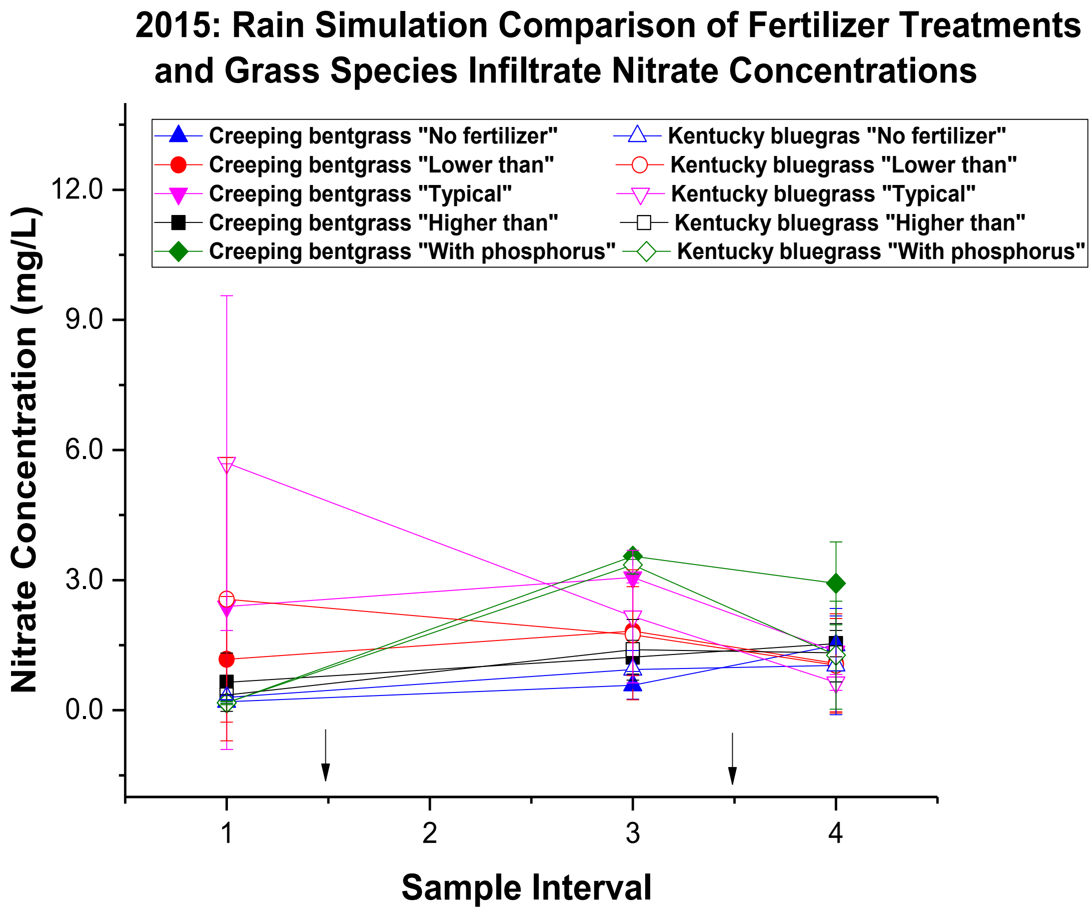

Figure 20. Changes in infiltrate nitrate concentrations with sample interval. Arrows indicate when fertilizer was applied and symbols show averaged concentration with error bars representing standard deviation.

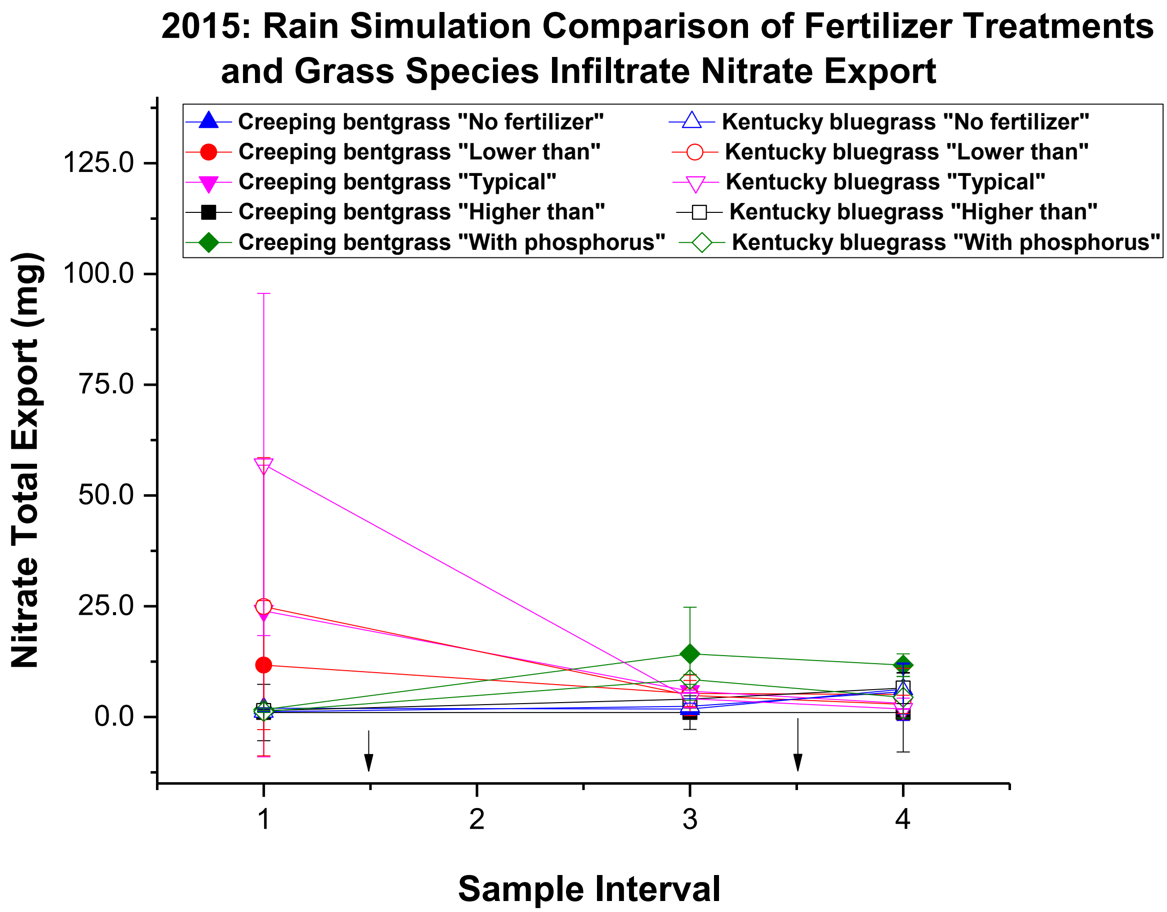

Figure 21. Changes in infiltrate nitrate total export with sample interval. Arrows indicate when fertilizer was applied and symbols show averaged export with error bars representing standard deviation.

In 2016 NF and T treatments did not influence the infiltrate nitrate concentrations (one-way ANOVA $F_{1,8} = 1.98$, $p = 0.19$) (Fig. 22). Similarly, one-way ANOVA showed no significant variation to nitrate export between various fertilizer treatments ($F_{1,8} = 1.55$, $p = 0.25$) (Fig. 23). During 2016 rain simulations, samples were collected for total nitrogen analysis on sampling interval two, three and five. The range of concentrations for the NF and T treatment also indicated fertilizer treatment did not significantly affect the infiltrate total nitrogen concentrations in 2016 (one-way ANOVA $F_{(1, 5.58)} = 0.14$, $p = 0.73$) (Fig. 24).

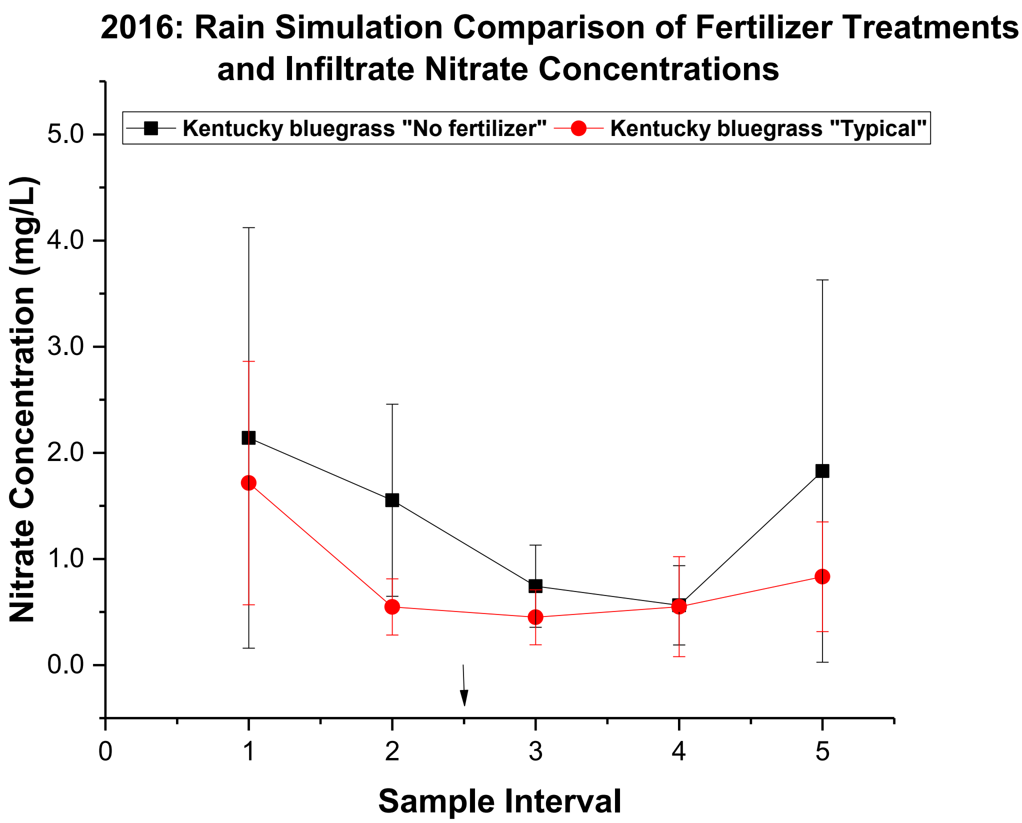

Figure 22. Changes in infiltrate nitrate concentrations with sample interval. Arrow indicates when fertilizer was applied and symbols show averaged export. Error bars represent standard deviation.

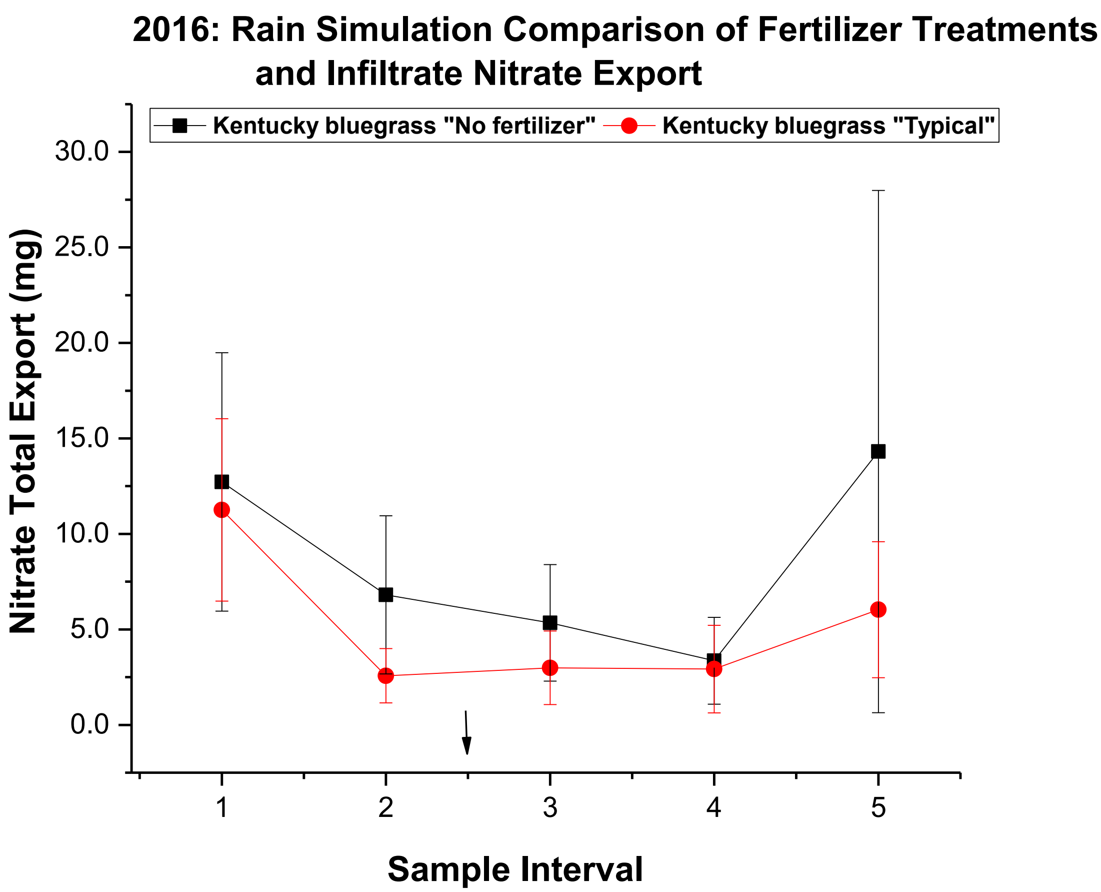

Figure 23. Changes in infiltrate nitrate total export with sample interval. Arrow indicates when fertilizer was applied and symbols show averaged export. Error bars represent standard deviation.

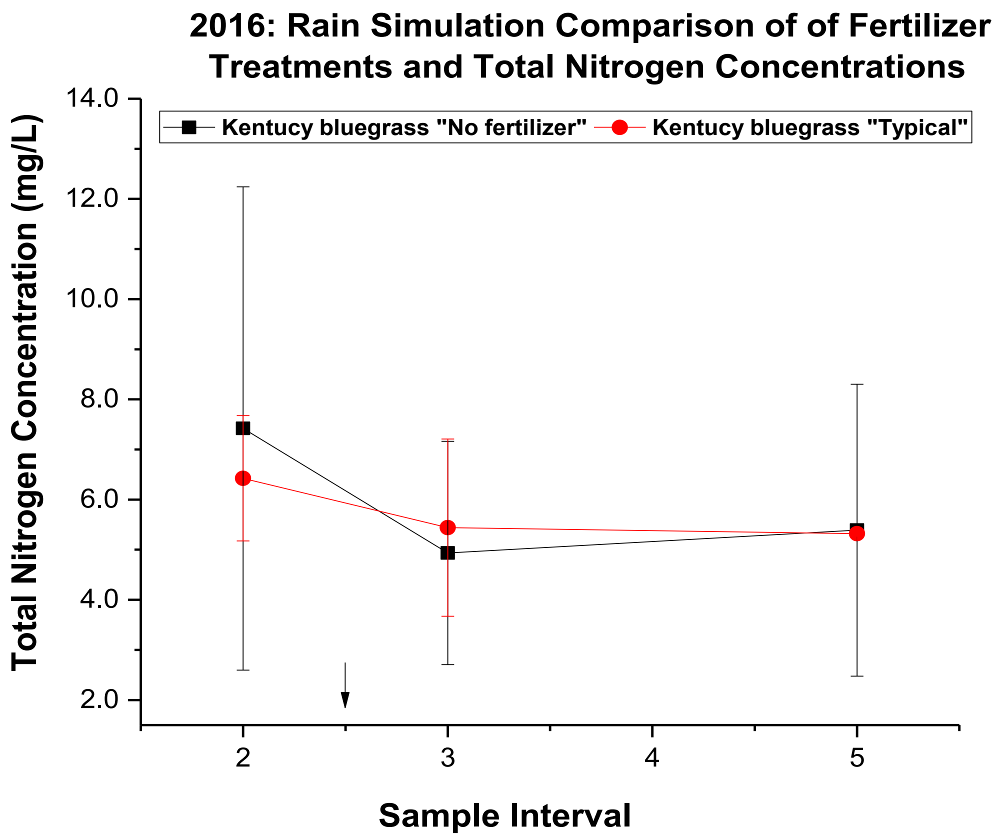

Figure 24. Changes in infiltrate total nitrogen concentrations with sample interval. Arrow indicates when fertilizer was applied and symbols show averaged concentrations for sample interval 2, 3 and 5. Error bars represent standard deviation.

3.2.2. Impact of grass species

During the 2015 study period, Kentucky bluegrass had the highest infiltrate nitrate concentration recorded (5.70 mg/L). The results of one-way ANOVA on infiltrate nitrate concentrations between the two grass species was not found to be significant ($F_{1,28} = 0.01$, $p = 0.91$). Similarly, grass species was also determined to have no significant impact on nitrate export ($F_{1,38}= 0.01$, $p= 0.93$). In 2016 Kentucky bluegrass infiltrate nitrate concentrations had a maximum of 2.14 mg/L. When the Kentucky bluegrass nitrate concentrations for 2015 and 2016 were compared no significant difference was found (t-test, $t = 0.469$, $p = 0.65$) with a mean of 1.38 mg/L ± 1.86 mg/L in 2015 and 1.10 mg/L ± 0.64 mg/L in 2016. Infiltrate nitrate exports of the two study periods did not show a significant difference (t-test, $t= 0.507$, $p= 0.62$).

3.2.3. Impact of seed density

When the seed density of Kentucky bluegrass was altered in 2015 the infiltrate nitrate concentrations were not found to vary significantly (one-way ANOVA $F_{2,6} = 1.42$, $p= 0.31$) (Fig. 25). Ranges in infiltrate nitrate export were also not significantly different (one-way ANOVA $F_{(2,3.43)}= 0.52$, $p= 0.64$).

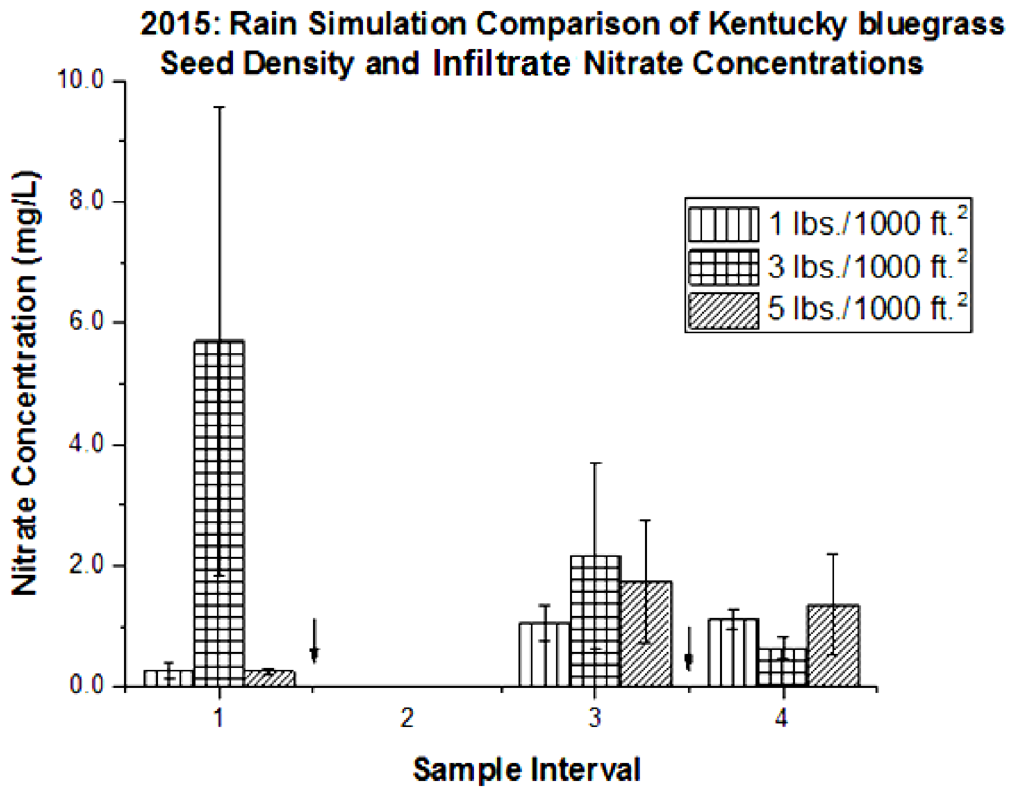

Figure 25. Average infiltrate nitrate concentrations for each seed density in 2015. Arrows indicate when fertilizer was applied and error bars represent standard deviation.

3.2.4. Discussion: nitrate and Total nitrogen

During both the study periods the infiltrate nitrate concentrations and export exhibited the same trend of a general decrease with sampling interval. Unlike the phosphate results; nitrate concentrations appeared to correlate to infiltrate volume in 2015 and 2016. There were no instances of infiltrate nitrate exceeding the Canadian Council of Ministries of the Environment (CCME) limit of 13 mg/L for freshwater systems. The higher concentrations at the beginning of the sampling periods are possibly related to immature roots and shoots not efficiently using available nutrients. Results from sample interval 2 were disregarded due to expired reagent packets being used during analysis procedures; as such interval 2 measurements were not included in the presentation of results or statistical analysis. Slow- release fertilizers operate using the fundamental mechanism; where water finds its way through the coating to the fertilizer particle, solubilize the fertilizer inside the granule and the fertilizer solution makes its way through the coating into the environment (Varadachari & Goertz, 2010). As such the slow-release XCU fertilizer can be released over a period of weeks (Varadachari & Goertz, 2010) resulting in a lag in nitrate availability and increases at interval 3 in 2015 and interval 5 in 2016. In addition, the percentage of urea released from the XCU fertilizer employed is not limited by the occurrence of “lock off” (only a portion of the available urea being released because the sulfur coating is too thick) (Agrium, 2017).

The results of this study showed no variation in infiltrate nitrate concentrations due to grass species or seed density. These results are similar to Kussow (2008) and Rice and Horgan (2010) who reported similar nitrogen retention for Kentucky bluegrass (98%) and Creeping bentgrass (88%). It is reasonable to assume that seed density was not a significant factor because shoot densities were not affected by the variations to seed density. The infiltrate nitrate concentrations found in this study are similar to what has been published following other studies. For instance, King et al. (2007) reported a maximum concentration of 3.52 mg/L and the results from 2015 and 2016 had maximum concentrations of 3.55 mg/L and 3.85 mg/L, respectively.

3.3. Total suspended solids

3.3.1. Impact of fertilizer treatment

The infiltrate TSS concentrations for the 2015 study period generally decreased with sampling interval for all the treatments. The TSS concentrations during 2015 study ranged from 0.7 mg/L to 16.2 mg/L (Fig. 26). Variation to fertilizer treatment was not found to have a significant impact on infiltrate TSS concentrations (one-way ANOVA $F_{4,34}$= 1.11, p= 0.37). Analysis of infiltrate TSS export did not result in significant variation (one-way ANOVA $F_{4,34}$= 0.32, p= 0.87) (Fig. 27). Soil only mesocosms had an average TSS concentration of 11.6 mg/L which could not be utilized as a "background" measurement because it was within the range of the other TSS concentrations reported.

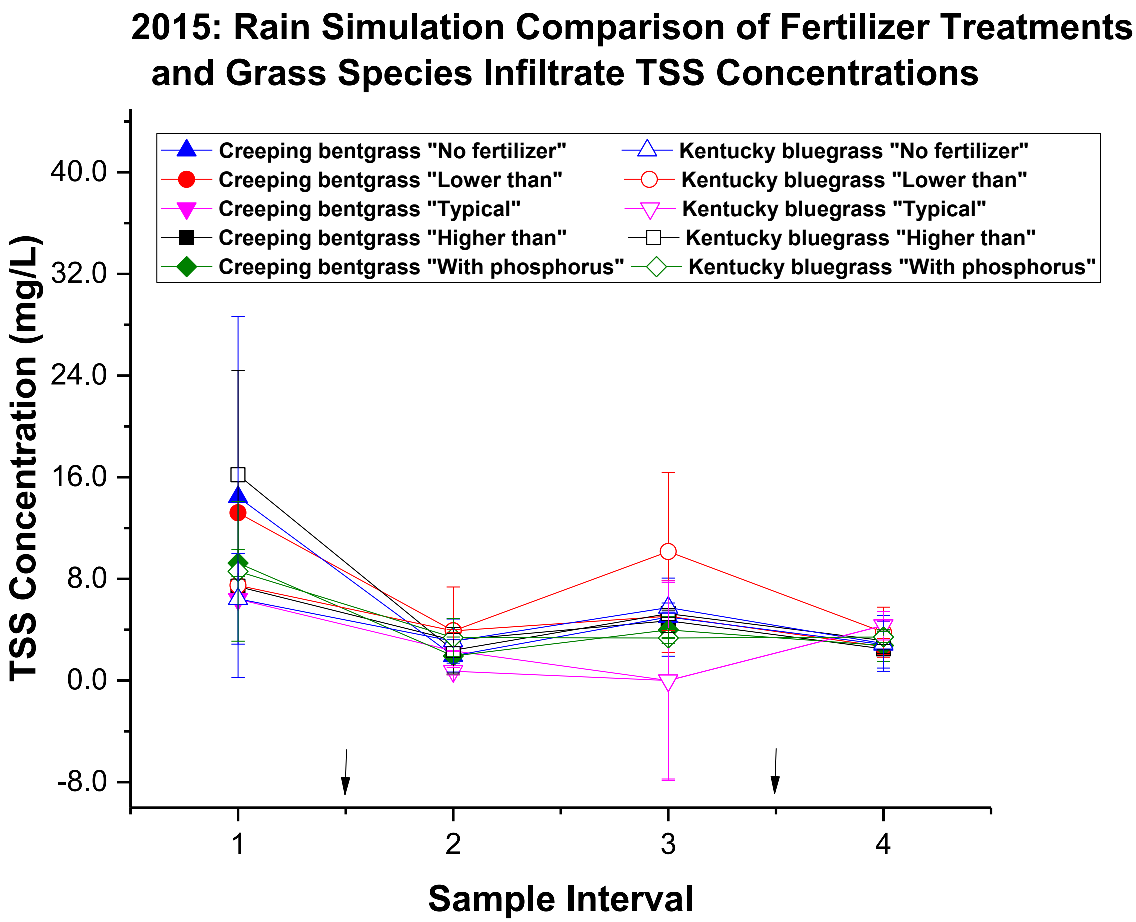

Figure 26. Changes in infiltrate TSS concentrations with sample interval. Arrows indicate when fertilizer was applied and symbols show averaged concentration with error bars representing standard deviation.

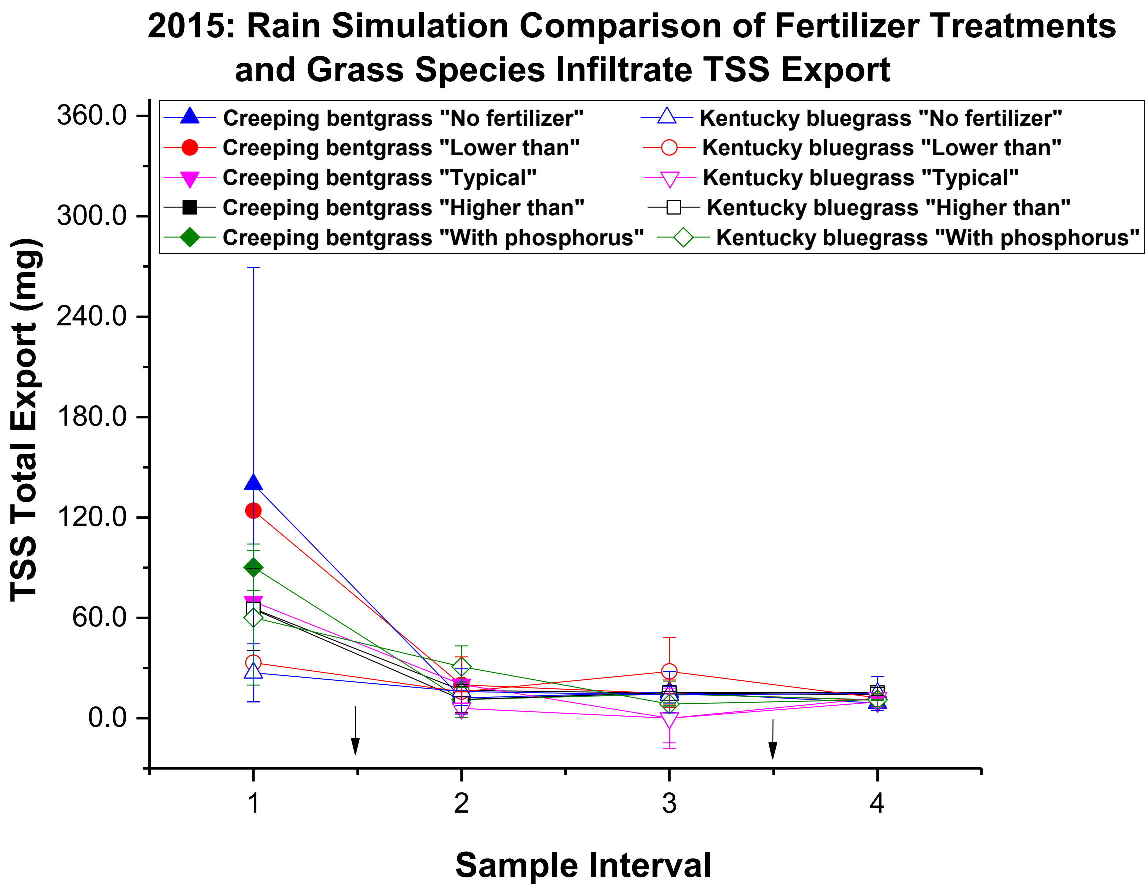

Figure 27. Changes in infiltrate TSS total export with sample interval. Arrows indicate when fertilizer was applied and symbols show averaged concentration with error bars representing standard deviation.

In 2016 the infiltrate TSS concentrations had a maximum value of 25.2 mg/L (Fig. 28). It was found that fertilizer treatment did not have a significant effect on infiltrate TSS concentrations or export (Fig. 29) (one-way ANOVA $F_{1,8} = 0.20$, p=0.66 and $F_{1,8} = 0.19$, p=0.67) respectively.

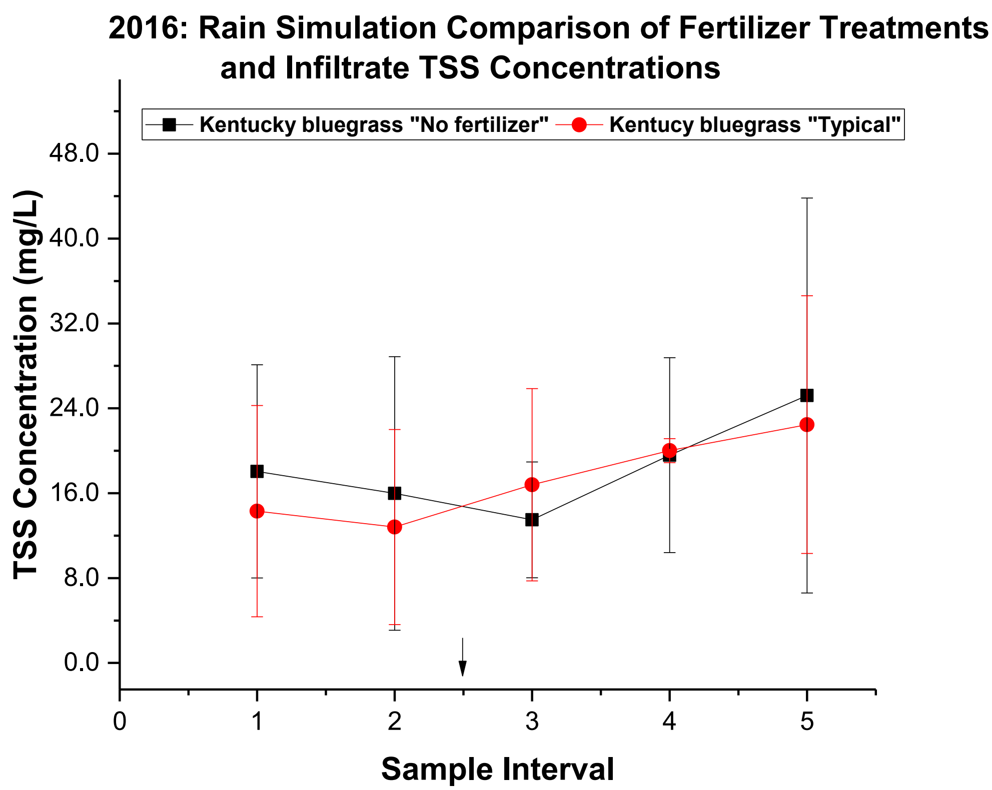

Figure 28. Changes in infiltrate TSS concentrations with sample interval. Arrow indicates when fertilizer was applied and symbols show averaged concentration with error bars representing standard deviation.

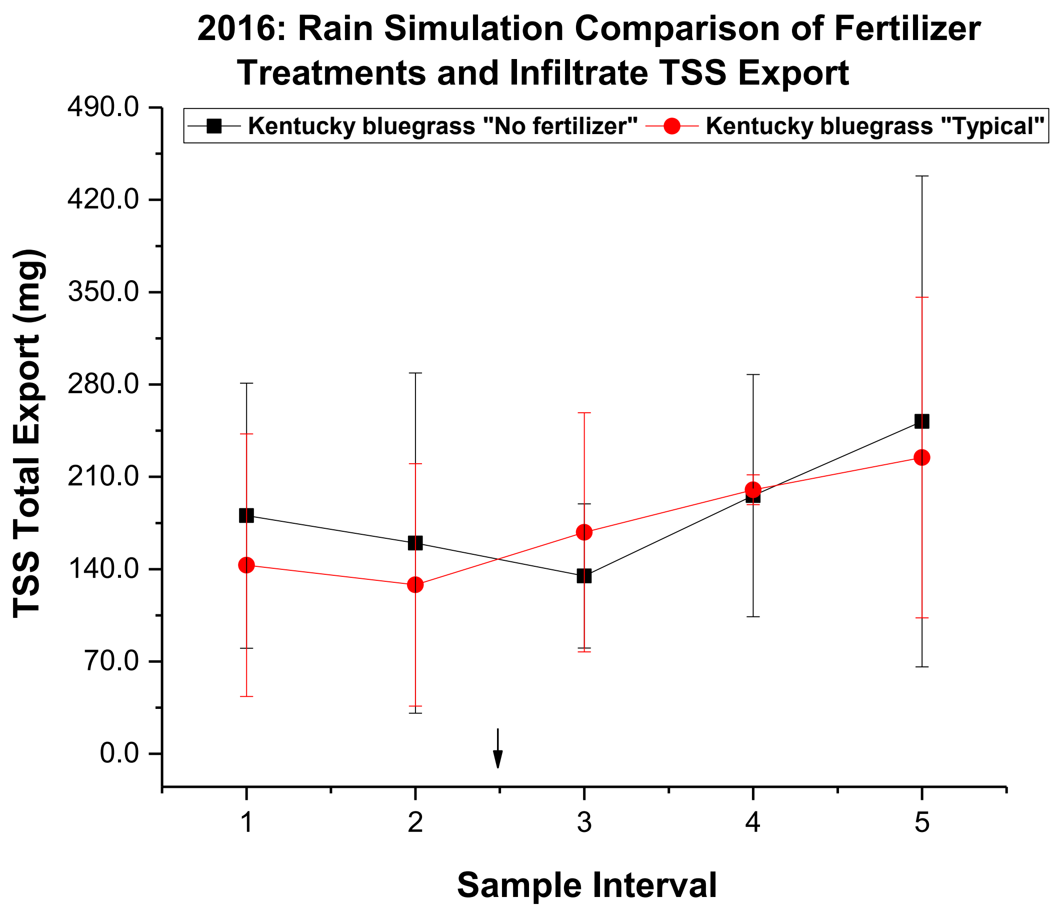

Figure 29 Changes in infiltrate TSS total export with sample interval. Arrow indicates when fertilizer was applied and symbols show averaged concentration with error bars representing standard deviation.

3.3.2. Impact of grass species

During 2015 study period Creeping bentgrass and Kentucky bluegrass had a maximum infiltrate TSS concentrations of 14.5 mg/L and 16.2 mg/L, respectively. These ranges in TSS concentrations for both grass species were not significantly different (one-way ANOVA $F_{1,37} = 0.003$, $p=0.95$). Also the infiltrate export of TSS for both grass species was not significantly different ($F(_{1,25.6}) = 1.80$, $p = 0.19$). The range of infiltrate TSS concentrations and export in 2016 for Kentucky bluegrass was significantly different from those measured in 2015 (t-test, $t = -8.694$, $p<0.0005$ and $t = -11.897$, $p<0.0005$ for TSS concentrations and total export respectively).

3.3.3. Impact of seed density

In 2015 seed density did not significantly influence the ranges of infiltrate TSS concentrations (one-way ANOVA $F_{2,8} = 1.51$, $p = 0.28$) (Fig. 30). In addition seed density did not impact TSS export (one-way ANOVA, $F_{2,8} = 4.03$, $p = 0.06$).

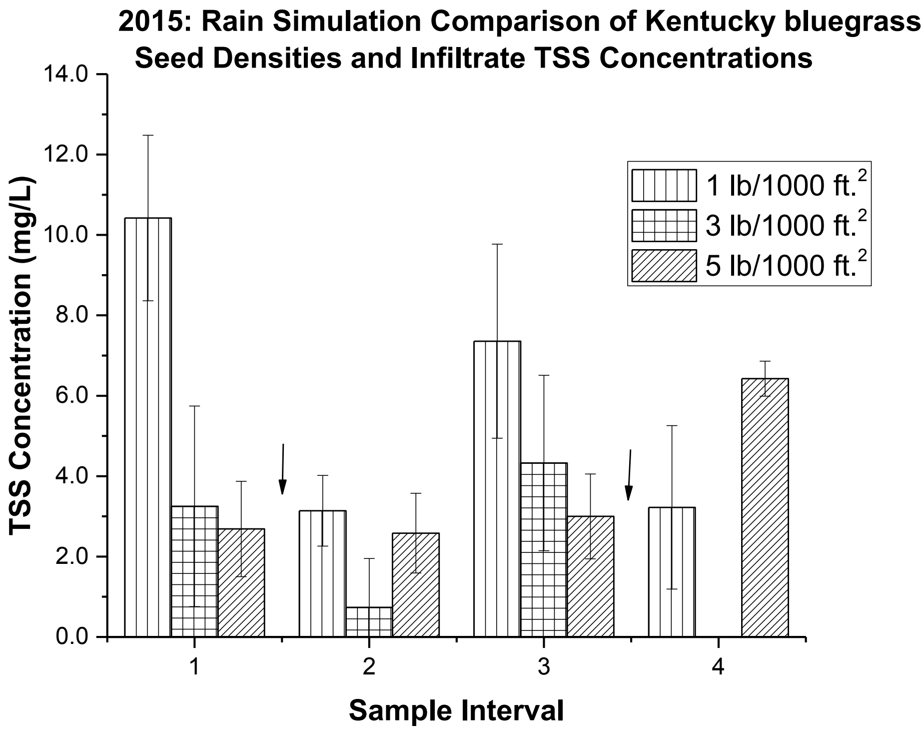

Figure 30. Average infiltrate TSS concentrations for each seed density in 2015. Arrows indicate when fertilizer was applied and error bars represent standard deviation.

3.3.4. Discussion: Total suspended solids

Overall, the TSS in the 2015 study period showed a decrease with time. This trend of decreasing concentration with time was expected because as turfgrass matures a higher amount of soil coverage is achieved. With less soil exposure soil erosion reduces thereby lowering TSS concentrations (Easton & Petrovic, 2004). Moreover, sediment that gets displaced can be deposited again when water encounters a barrier that is able to slow down the water flow and reduce its energy and consequently the carrying capacity (Moss et al., 2006). Results reported in this research show TSS decreasing with time from 16.2 mg/L to 2.5 mg/L and are similar to those presented by Borst (2011) where total solids also decreased with time. TSS concentrations in 2016 did not follow the same decreasing trend with sample interval as displayed in 2015. When standard deviation is considered it was determined to be higher in 2016 and the overlap of measurements does not indicate the occurrence of a trend. This agrees with the collected volume measurements also being within range of one another and not displaying a trend with sample interval. It is believed that TSS concentrations in 2016 lack a trend with sample interval because all of the samples were collected within the same amount of time as interval 1 and 2 in 2015. In addition, the 2016 study period had higher flow rates, less soil depth and gravel that was dirtier (coated with clay like sediment) than the gravel used in 2015 which may have contributed to TSS concentrations being higher than in 2015.

It was not anticipated that grass species would have a significant effect on TSS concentrations as Creeping bentgrass and Kentucky bluegrass are both cool-season species and have similar growth patterns including shoot densities (Kussow, 2008) and the results support this assumption. No difference in sediment loading with different species of turfgrass was also reported by Kauffman & Watschke (2007). However, it was expected that differences to seed density would alter the shoot density which would result in significant variation of TSS concentrations. This prediction was not supported by the data because shoot density was not reduced or increased based on the alterations made to seed density. Fertilizer treatment was not expected to significantly influence the infiltrate TSS concentrations or export because the root zone of turfgrass provides limited potential for elements to pass through (Beard & Green, 1994). This expectation was supported by the results.

3.4. Conclusion

Phosphate was frequently measured at levels high enough to contribute to algal blooms as previously stated by King et al. (2006), Rice and Horgan (2010) and King et al. (2012). However fertilizer treatment, grass species and seed density were not found to significantly influence infiltrate phosphate concentrations or export. Although phosphate exceeded the EPA guideline multiple times in 2015, 53% occurred during the first sampling interval and 46% were due to the WP treatment. In 2016 the phosphate concentrations exceeded the EPA guideline 60% of the time during the early sampling intervals, but decreased to below the detection limit by sampling interval four and five.

This study supports the conclusion proposed by other studies that nitrate is frequently not of a concern as there were no instances of infiltrate nitrate exceeding the CCME limit of 13 mg/L (Baris et al., 2010; Davis & Lydy, 2002; Kussow, 2008). Higher nitrate and TSS concentrations from the first sample interval are similar to what other researchers reported (Bowman et al., 2002; Borst, 2011) and the variables examined did not result in any significant differences being found for infiltrate nitrate or TSS concentrations or export. Therefore, results from both the study periods indicate that turfgrass age might be a factor in reducing nutrient loss. It was initially hypothesised for 2015 rain simulations that fertilizer treatment and grass species would not significantly affect the nutrient concentrations from turfgrass during rain simulations, but seed density would demonstrate a significant influence on TSS concentrations. Similarly, 2016 rain simulations would not indicate fertilizer treatment to be a significant influence on the infiltrate nutrient concentrations or total export. Statistical analysis of the nutrient measurements from 2015 and 2016 support the hypothesis presented; except for the hypothesis of seed density influencing TSS in 2015. As mentioned, it is believed that seed density was not a significant factor on the concentrations of the nutrients because shoot density was not altered. Thus, the results from this study provide useful information about how as turfgrass matures concerns related to phosphate, nitrate and TSS concentrations and exports can be mitigated.

Chapter 4. Storm Simulation Study

4.1. Phosphate and Total phosphorus

4.1.1. Impact of fertilizer treatment

In 2015 the overall infiltrate volume collected increased from sample interval one to interval two, but the time elapsed to exceed the fixed holding capacity for infiltrate decreased (Fig. 31) because the maximum flow rate increased. After performing statistical analysis the infiltrate phosphate concentrations (Fig. 32) were found to be significantly different based on fertilizer treatment (one-way ANOVA $F_{4,15} = 5.93$, $p = 0.005$) and a Tukey's post hoc revealed the WP treatment was significantly different from the HT ($p= 0.01$), LT ($p= 0.05$) and NF ($p=0.004$) treatments. While storm simulations were conducted samples were collected for total phosphorus analysis and the concentrations (Fig. 33) were not found to be significantly different (one-way ANOVA $F_{(1,4)}= 4.41$, $p= 0.10$). However, the phosphate concentrations for the spiked inlet water were significantly higher than the infiltrate concentrations (t-test, $t= -4.842$, $p= 0.001$).

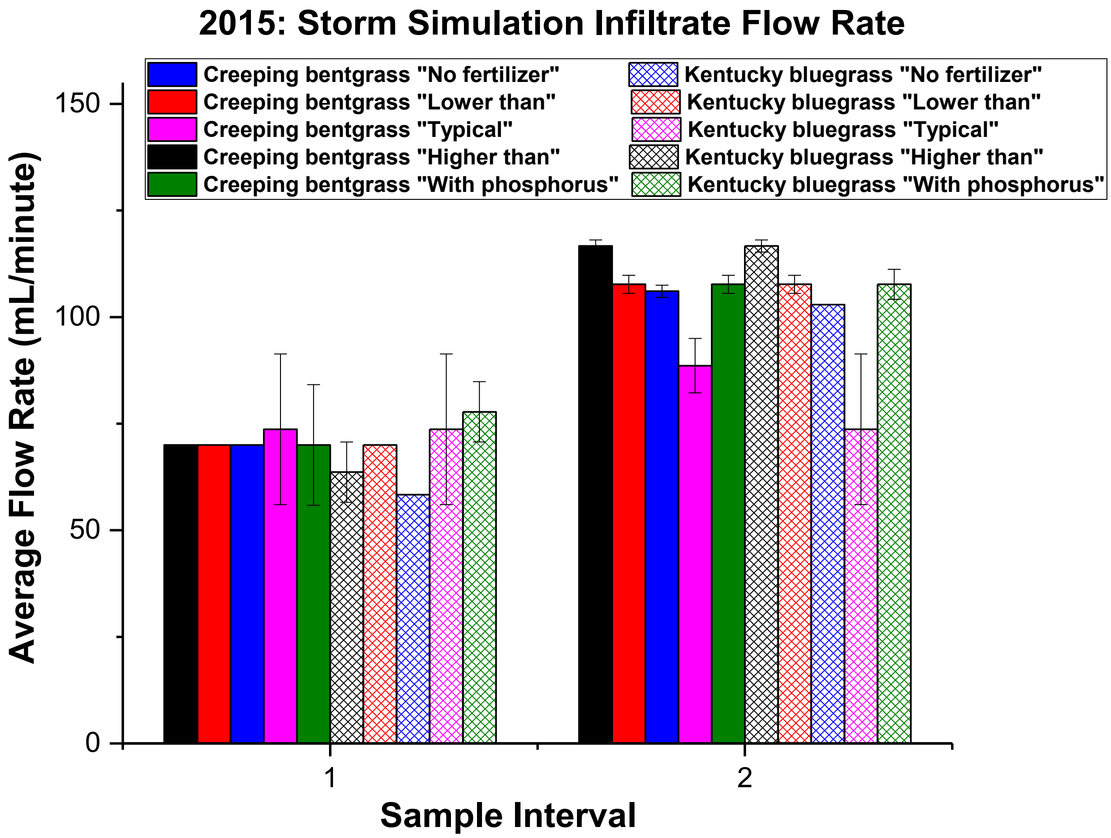

Figure 31. Changes in average infiltrate flow rate with sample interval. Average volume of 39,980 ml applied for interval 1 and 45,260 ml applied during interval 2. Bars show averaged flow rates with error bars representing standard deviation.

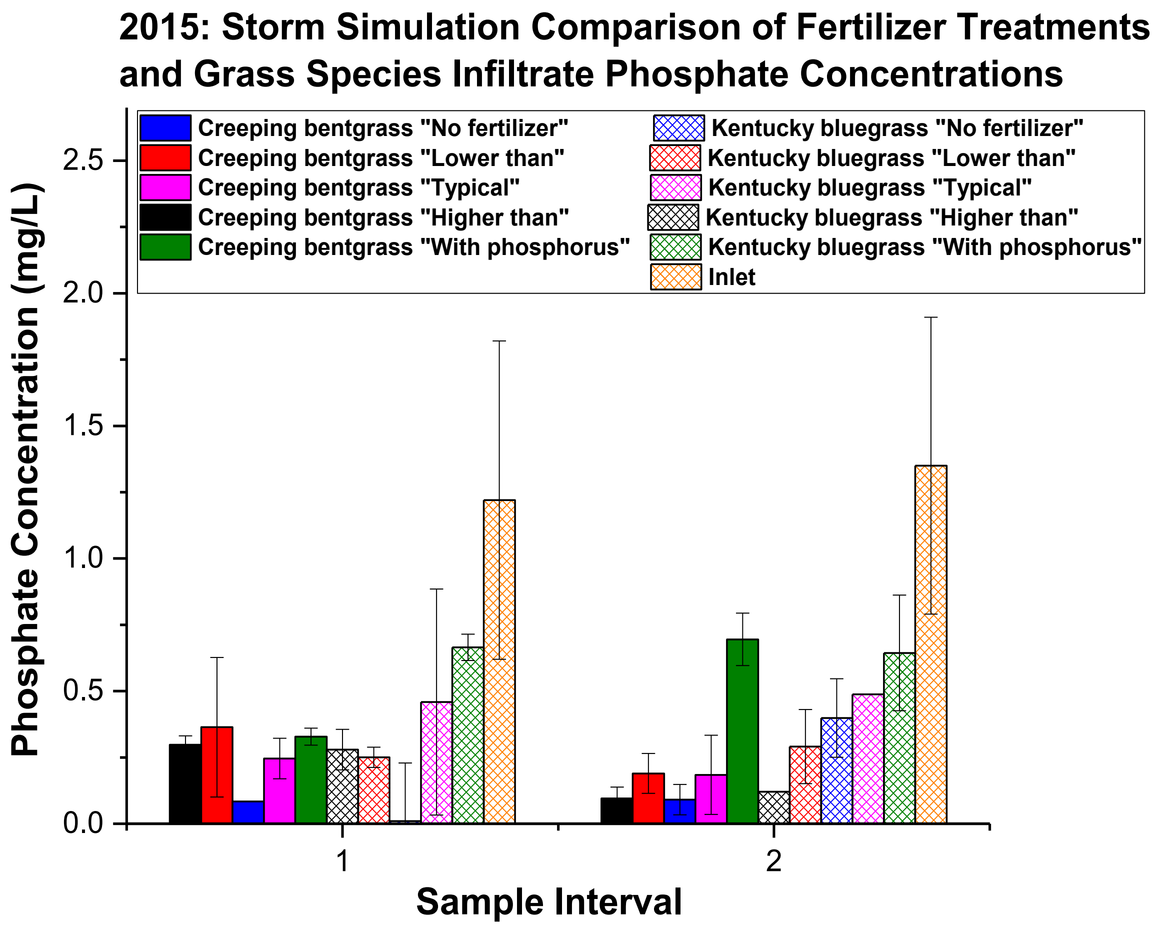

Figure 32. Changes in infiltrate phosphate concentrations with sample interval. Bars show averaged concentration and error bars represent standard deviation.

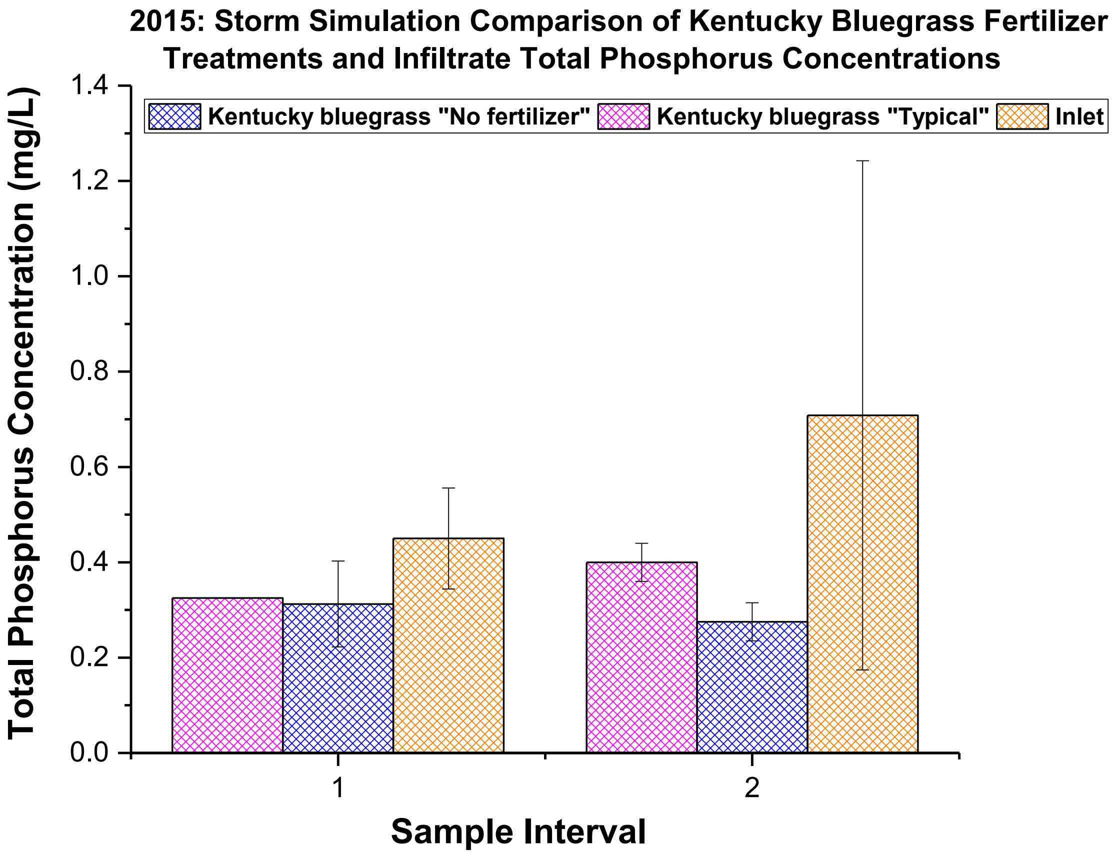

Figure 33. Changes in infiltrate total phosphorus concentrations with sample interval. Bars show averaged concentration with error bars representing standard deviation.

Runoff volume also increased or remained similar from sample interval one to interval two (Fig. 34). It was determined that fertilizer treatment did not significantly impact the phosphate (Fig. 35) or total phosphorus (Fig. 36) concentrations of runoff (one-way ANOVA $F_{4,12}$= 1.74, p= 0.21 and $F_{1,4}$= 2.65, p= 0.18). Similarly, the ranges of phosphate export (Table 8) in the runoff water was not significantly influenced by fertilizer treatment (one-way ANOVA $F_{4,12}$= 1.39, p= 0.29). A significant difference between the infiltrate and runoff phosphate concentrations was observed (t-test, t= -5.379, p< 0.0005). Total phosphorus concentrations of the inlet water were not significantly different from the infiltrate (t-test, t= 1.336, p= 0.22) and runoff water (t-test, t= 1.056, p= 0.33). Significance was also not determined between the infiltrate and runoff total phosphorus (t-test, t= -0.495, p= 0.64). In addition, the phosphate concentrations in the inlet water were not significantly higher than the runoff water (t-test, t= -1.694, p= 0.10).

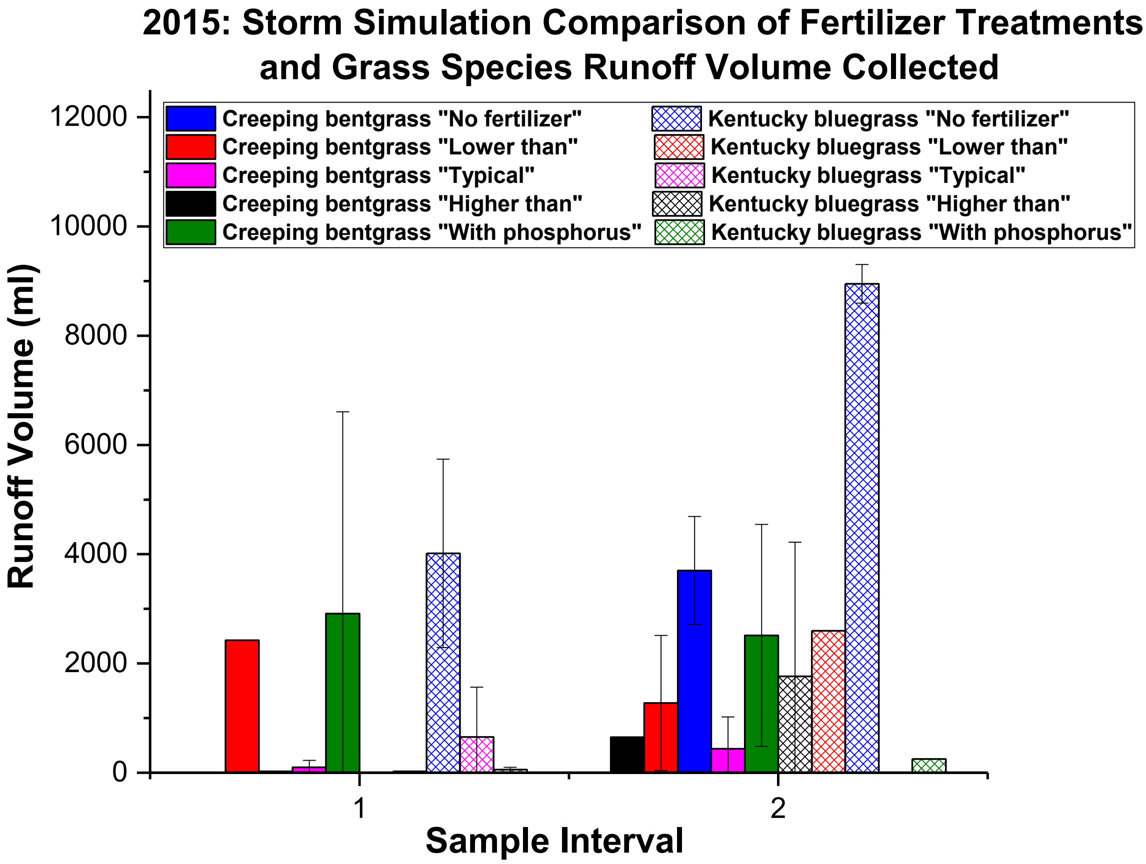

Figure 34. Changes in runoff volume collected with sample interval and an average volume of 39,980 ml applied for interval 1 and 45,260 ml applied during interval 2. Bars show averaged volume collected with error bars representing standard deviation.

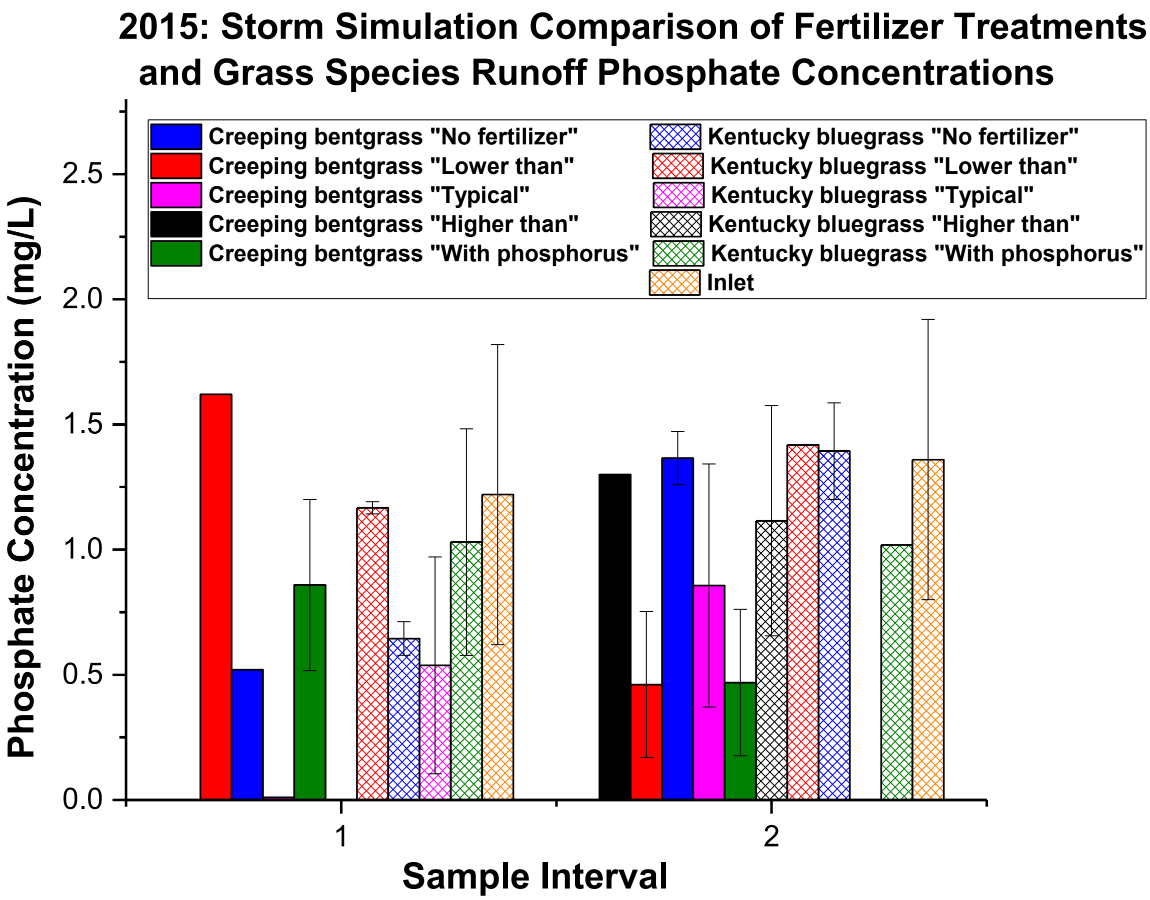

Figure 35. Changes in runoff phosphate concentrations with sample interval. Bars show averaged concentration and error bars represent standard deviation.

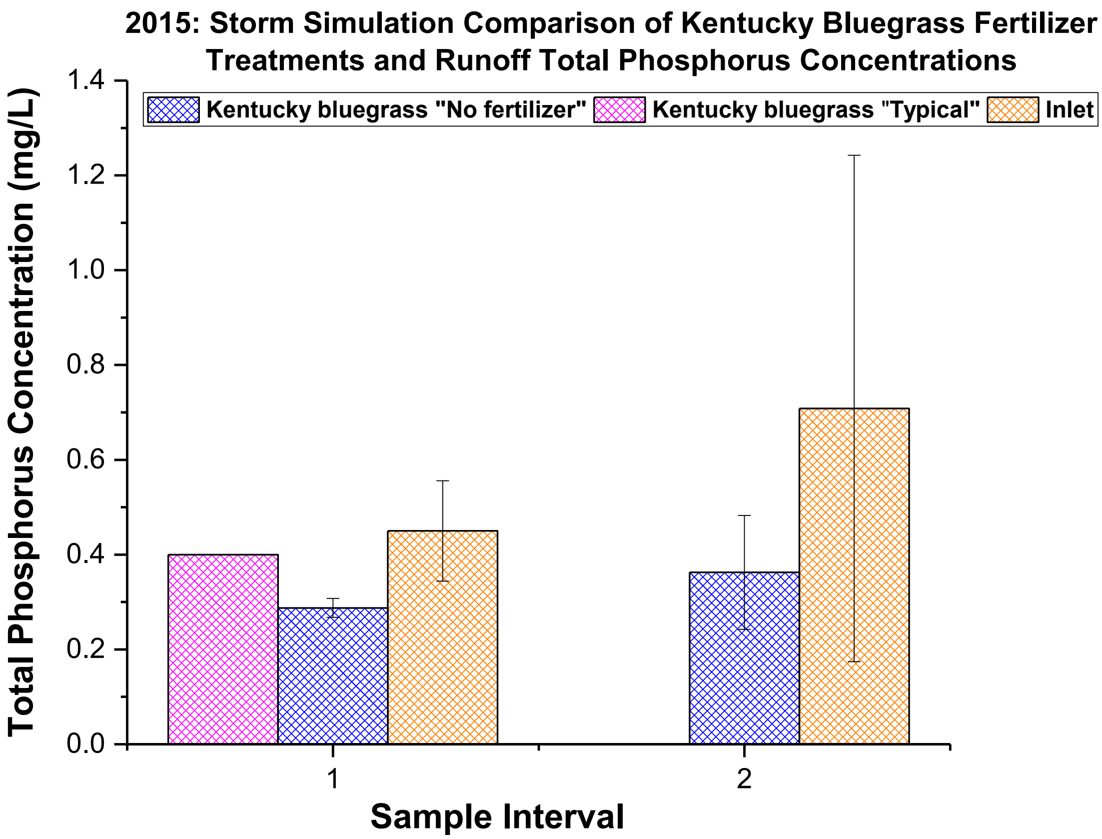

Figure 36. Changes in runoff total phosphorus concentrations with sample interval. Bars show averaged concentration and error bars representing standard deviation.

Fertilizer Treatment	Range of Runoff Phosphate Export
No fertilizer	[a]0.0 mg - 12.5 mg
Lower than	[a]0.0 mg - 3.9 mg
Typical	[a]0.0 mg - 0.5 mg
Higher than	0.8 mg - 2.5 mg
With phosphorus	0.1 mg - 3.1 mg

Table 8. Summary of the average runoff phosphate export values from 2015 with an average standard deviation of 1.5 mg. Inlet import values were 52.1 mg and 57.8 mg for sample interval 1 & 2 respectively.
[a] - value of 0.0 provided because measurement was below the detection limit

4.1.2. Impact of grass species

When storm simulations were carried out in the 2015 study period the infiltrate generated for Creeping bentgrass and Kentucky bluegrass had phosphate concentrations that were not determined to be significantly different (one-way ANOVA $F_{1,18}$= 1.34, p = 0.26). During storm simulations in 2015 runoff was also studied and the ranges in phosphate concentration for both grass species was not significantly different (one-way ANOVA $F_{1,15}$ = 1.10, p= 0.31). Moreover, phosphate exports for Creeping bentgrass and

Kentucky bluegrass runoff was not noticed as being significantly different (one-way ANOVA $F_{1,15}$= 0.30, p= 0.59).

4.1.3. Impact of seed density

During the storm simulations not enough data was collected from the mesocosms with altered seed densities to allow for statistical analysis using one-way ANOVA to be performed. However, the concentrations and export for phosphate presented in Table 9 do not display wide variation or dependence on seed density.

	Seed Density		
	1 lb/1000 ft.^{2}	3 lb/1000 ft.^{2}	5 lb/1000 ft.^{2}
Infiltrate Phosphate Concentrations	0.11 mg/L & 0.21 mg/L	0.46 mg/L & 0.49 mg/L	0.46 mg/L & 0.49 mg/L
Runoff Phosphate Concentrations	0.46 mg/L & 0.49 mg/L	[b]0.54 mg/L	0.29 mg/L & 0.61 mg/L
Runoff Phosphate Total Export	0.0 mg & 0.6 mg	[b]0.5 mg	0.2 mg & 0.6 mg

Table 9. Summary of the infiltrate and runoff phosphate concentrations and runoff export for each seed density from 2015 storm simulations.
[a]- value of 0.0 represents a measurement that was below the detection limit
[b]- only one value provided because relevant mesocosms only generated enough runoff during one of the two simulations

4.1.4. Discussion: phosphate and Total phosphorus

Phosphate concentrations from NF and T treatment are higher than total phosphorus but the corresponding phosphorus from phosphate is lower (a factor of approximately 3.1) than total phosphorus, as one would expect. Results indicate that phosphate concentrations were higher in runoff water than in infiltrate, consistent with results presented by Beard and Green (1994). Lower phosphate concentrations in the infiltrate were expected during storm simulations because the process of infiltration allows for more effective nutrient retention by means of biological processes in both the turfgrass roots and soil, whereas surface runoff only has the physical barrier of turfgrass shoots. Grass species and seed density were not expected to affect the phosphate concentrations because both grass species involved in the study were cool-season species and may have similar capacities of nutrient uptake and growth patterns (Kussow, 2008; Rice & Horgan, 2010; Barton & Colmer, 2006). The assumption for grass species was supported by the results, but conclusive statements cannot be made in regards to seed density because statistical analysis was not performed. However, the measurements presented in Table 9 suggest no dependence of phosphate concentrations or export on seed density which could be due to the occurrence of similarity in shoot densities.

Fertilizer treatment was not anticipated to significantly influence phosphate concentrations or export in the runoff water because the phosphate from fertilizer applications would be within the soil below the turfgrass surface. However, phosphate concentrations would be higher in the runoff than the infiltrate because only physical mechanisms performed by turfgrass shoots would provide nutrient retention (Moss et al., 2006); whereas infiltrate would have nutrients removed by the turfgrass roots and soil. Despite the lower phosphate concentrations in the infiltrate, the WP fertilizer treatment would significantly affect the phosphate concentrations of the infiltrate because of the cumulative effects of phosphorus addition from fertilizer application and polluted inlet water. The present study results supported this. Although Lehman et al. (2009) did not focus on infiltrate and their results are similar to this study because they reported phosphorus levels decreased in water samples when phosphorus was not supplied (zero phosphorus formulas were used). During the storm simulation portion of this research infiltrate phosphate concentration exceeded the EPA limit of 0.10 mg/L 75% of the time and runoff exceeded it 94 % of the time.

4.2. Nitrate and Total nitrogen

4.2.1. Impact of fertilizer treatment

When focusing on the range of infiltrate nitrate concentrations for each treatment (Fig. 37) one-way ANOVA analysis found that fertilizer treatment did not significantly affect nitrate concentrations ($F_{4,15} = 1.17$, $p = 0.36$). Total nitrogen concentrations (Fig. 38) were also not significantly different between fertilizer treatments (one-way ANOVA $F_{1,5} = 2.21$, $p = 0.20$). During the study the inlet water nitrate concentrations were not found to be significantly different from those of the infiltrate samples (t-test $t = 0.232$, $p = 0.82$).

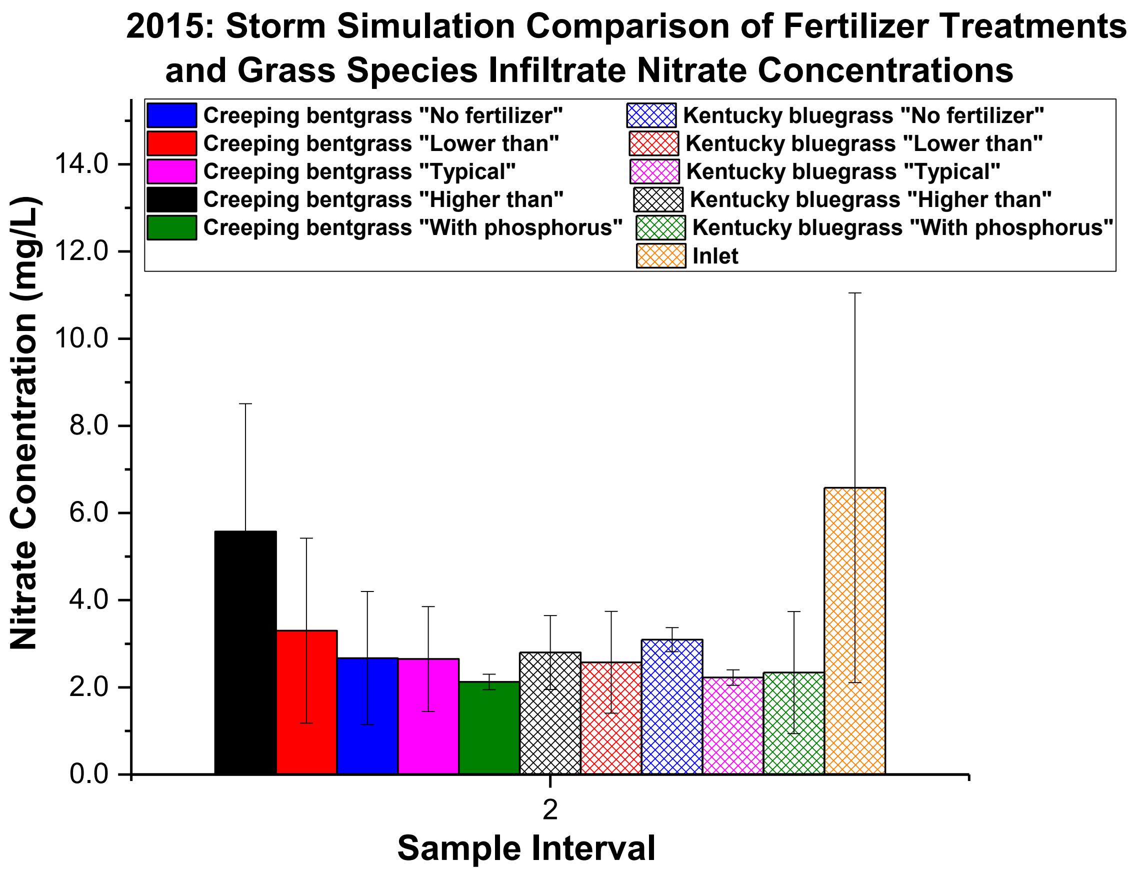

Figure 37. Changes in infiltrate nitrate concentrations with sample interval. Bars show averaged concentration and error bars represent standard deviation.

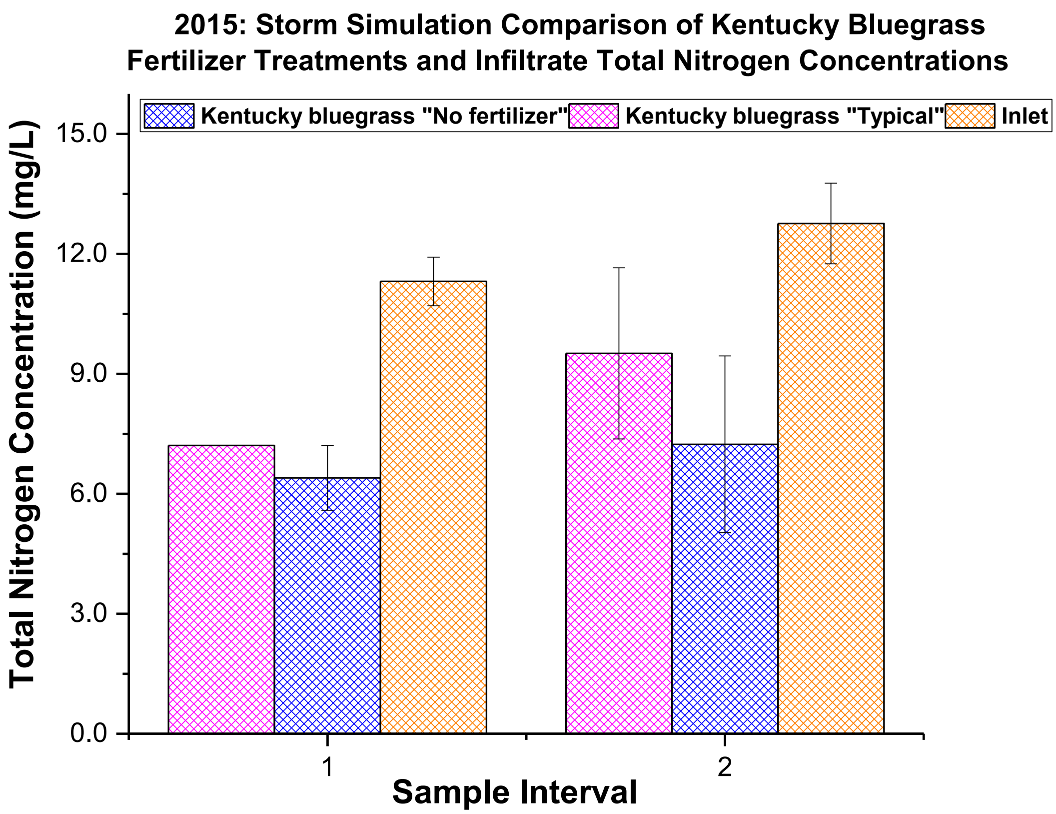

Figure 38. Changes in infiltrate total nitrogen concentrations with sample interval. Bars show averaged concentration with error bars representing standard deviation.

The nitrate concentrations in the runoff water were not determined to be influenced by fertilizer treatment (one-way ANOVA $F_{(4,5.39)} = 1.13$, $p= 0.43$) (Fig. 39). Moreover, fertilizer treatment did not have a significant effect on runoff nitrate export (Table 10) (one-way ANOVA $F_{(4,4.86)}= 2.47$, $p= 0.17$). Total nitrogen concentrations measured for the runoff of NF and T treatments ranged from 9.84 mg/L to 10.65 mg/L and 12.72 mg/L respectively (Fig. 40), but limited data did not allow a statistical comparison. An examination comparing the infiltrate and runoff nitrate concentrations also indicated a lack of significant difference (t-test, $t= 1.337$, $p = 0.19$). Also the total nitrogen and nitrate concentrations of runoff and inlet water were not significantly different (t-test, $t= -1.226$, $p= 0.29$; t-test, $t= -1.191$, $p= 0.35$), respectively.

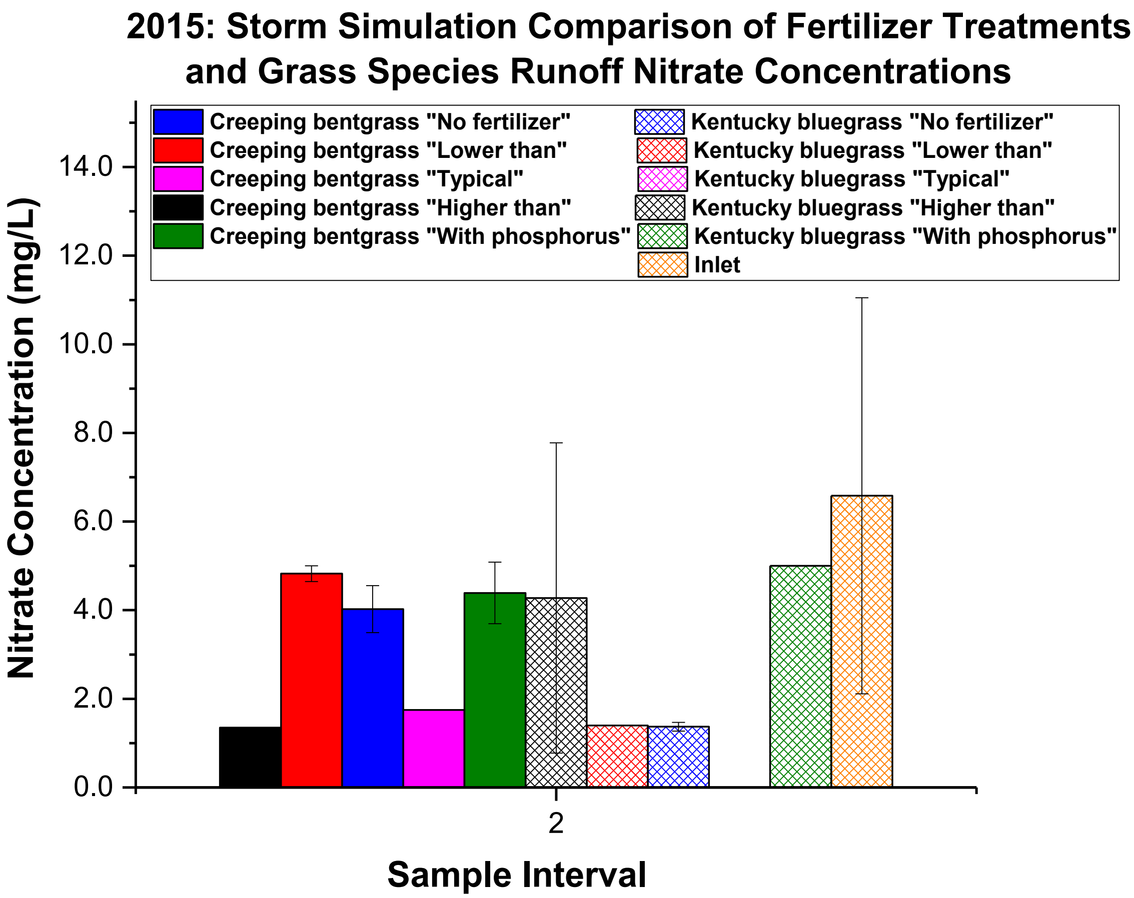

Figure 39. Changes in runoff nitrate concentrations with sample interval. Bars show averaged concentration with error bars representing standard deviation.

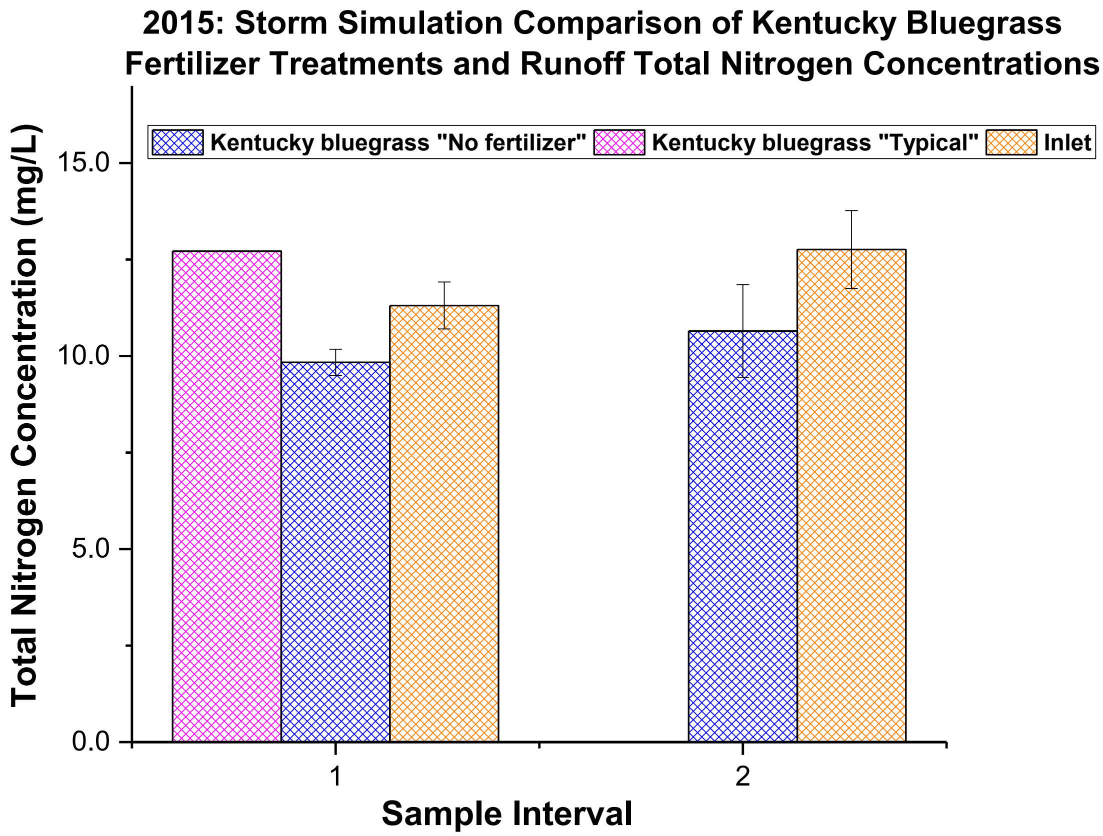

Figure 40. Changes in runoff total nitrogen concentrations with sample interval. Bars show averaged concentration and error bars represent standard deviation.

Fertilizer Treatment	Range of Runoff Nitrate Export
No fertilizer	1.4 mg - 4.0 mg
Lower than	1.4 mg - 4.8 mg
Typical	1.5 mg
Higher than	1.4 mg - 4.3 mg
With phosphorus	4.4 mg - 5.0 mg

Table 10. Summary of the average runoff nitrate export values from 2015.

4.2.2. Impact of grass species

There was not a significant difference between the infiltrate nitrate concentrations of the two grass species (one-way ANOVA $F_{1,18} = 1.06$, $p = 0.32$). One-way ANOVA analysis determined that grass species did not significantly impact nitrate concentrations or export in the runoff ($F_{1,15} = 0.11$, $p = 0.74$; one-way ANOVA $F_{1,15} = 0.08$, $p = 0.76$), respectively.

4.2.3. Impact of seed density

During the storm simulations not enough data was collected from the mesocosms with altered seed densities to allow for statistical analysis using one-way ANOVA to be performed. However, the concentrations and export measurements for nitrate are presented in Table 11 and do not appear to express a dependency on seed density.

	Seed Density		
	1 lb/1000 ft.^{2}	3 lb/1000 ft.^{2}	5 lb/1000 ft.^{2}
Infiltrate Nitrate Concentrations	2.13 mg/L	4.93 mg/L	2.34 mg/L
Runoff Nitrate Concentrations	4.95 mg/L	*	1.80 mg/L
Infiltrate Nitrate Total Export	48.7 mg	70.1 mg	74.9 mg
Runoff Nitrate Total Export	5.0 mg	*	1.8 mg

Table 11. Summary of the infiltrate and runoff nitrate concentrations and runoff export for each seed density from 2015 storm simulations.
*- no value reported because relevant mesocosms did not generate runoff

4.2.4. Discussion: nitrate and Total nitrogen

Total nitrogen concentrations for the inlet water remained consistent with only a slight increase from an average of 11.31 mg/L to 12.76 mg/L. Nitrate concentrations measured during the first sample interval were disregarded because they were suspected of being inaccurate due to expired reagent packets being used during analysis procedures. Thus, statistical analysis and the results presented were for the second sample interval data only.

Fertilizer treatment and seed density were not anticipated to affect nitrate or total nitrogen concentrations. This was concluded based on reviewing literature that indicated nitrate is frequently not of concern (Cohen, Svrjcek, Durborow, & Barnes, 1999; Hindahl et al., 2009; King et al., 2007) because of high uptake efficiency (Shuman, 2003; Al-Rawashdeh & Abdel-Ghani, 2008). Nitrate and total nitrogen concentrations in the infiltrate and runoff did not vary significantly due to fertilizer treatment, supporting the prediction. Results in Table 11 show that the 1 lb/1000 ft^2 density had the highest runoff nitrate concentration and export. An absence of significance for fertilizer treatment could be due to the rainfall intensity and soil moisture (7.4 or 39%) prior to rainfall that was employed in this study because other research has proposed the transport of nitrate is directly related to these factors (Shuman, 2004).

Nitrate concentrations measured in the runoff during the storm simulations had a maximum of 5.00 mg/L and is higher than the maximum concentration of 3.50 mg/L reported by King et al. (2007). The difference between the maximum concentrations could be due to higher dilution during sample collection for King et al. (2007) because water samples were collected from a stream. When examining nitrate concentrations the CCME guideline for freshwater (13 mg/L) was never exceeded in runoff or infiltrate (CCME, 2012). Similarly total nitrogen concentrations were not recorded higher than the Canadian Environmental Sustainability Indicators (CESI) trigger value of 650 mg/L for mesotrophic freshwater systems (CESI, 2008).

4.3. Total suspended solids

4.3.1. Impact of fertilizer treatment

In 2015 the maximum infiltrate TSS concentration was 14.7 mg/ L (Fig. 41). After performing statistical analysis it was found that fertilizer treatment did not have a significant impact on infiltrate TSS concentrations (one-way ANOVA $F_{4,15} = 1.17$, $p = 0.36$). However, the infiltrate TSS concentrations were found to be significantly lower than the inlet (t-test, $t = -5.608$, $p = 0.001$) with a decrease from a mean of 78.6 mg/L ± 34.3 mg/L for influent water to a mean of 5.8 mg/L ± 3.8 mg/L for the infiltrate.

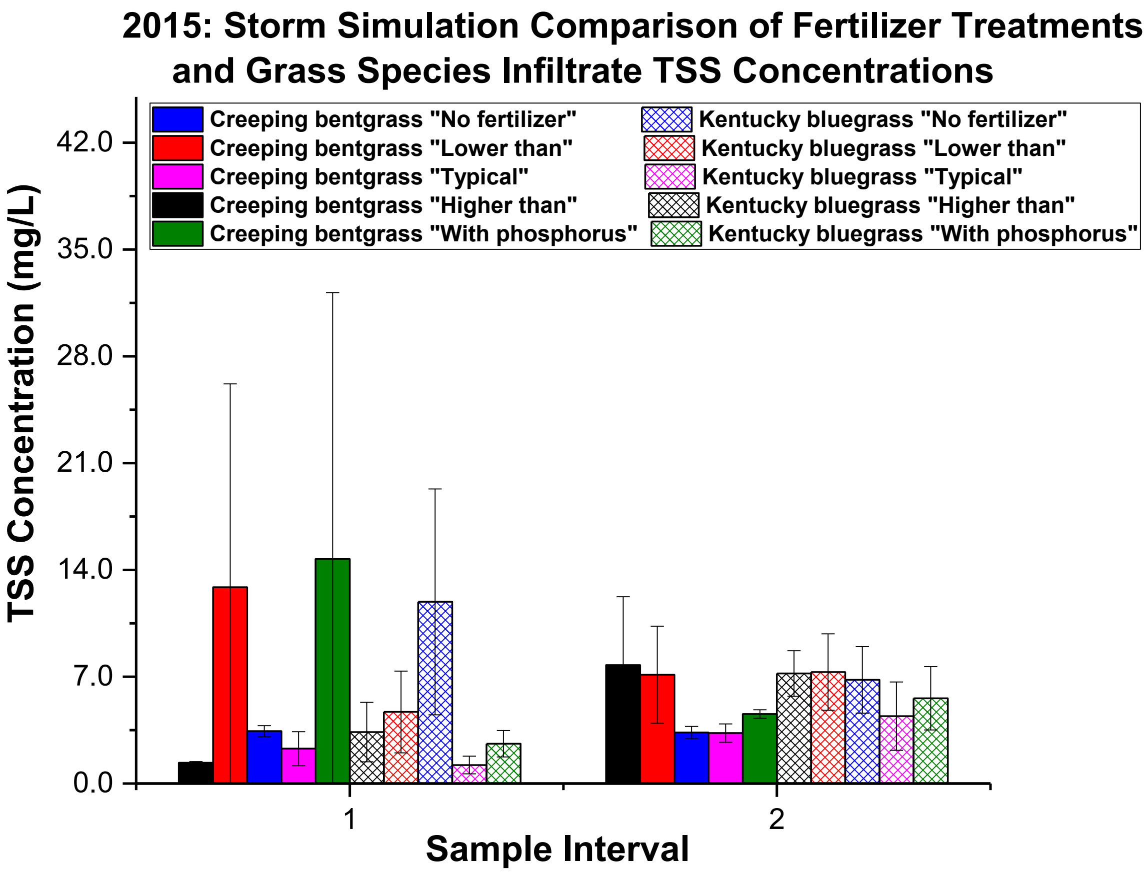

Figure 41. Changes in infiltrate TSS concentrations with sample interval. Bars show averaged concentration with error bars representing standard deviation. Inlet TSS concentrations were 60.8 mg/L and 102.4 mg/L for sample interval 1 and 2 respectively.

The TSS concentrations and export in the runoff were not determined to be significantly influenced by fertilizer treatment (one-way ANOVA $F_{4,8}$= 0.76, p= 0.58; $F(_{4,3.20})$= 1.73, p= 0.33), respectively (Fig. 42 & Fig. 43). A significant difference between the infiltrate and runoff TSS concentrations was identified (t-test, t= -6.296, p< 0.0005) with an increase from a mean of 5.8 mg/L ± 3.8 mg/L for infiltrate to 16.7 mg/ L ± 6.2 mg/L for runoff. In addition, the TSS concentrations of the runoff water were significantly lower (t-test, t= -4.721, p= 0.005) than the inlet water.

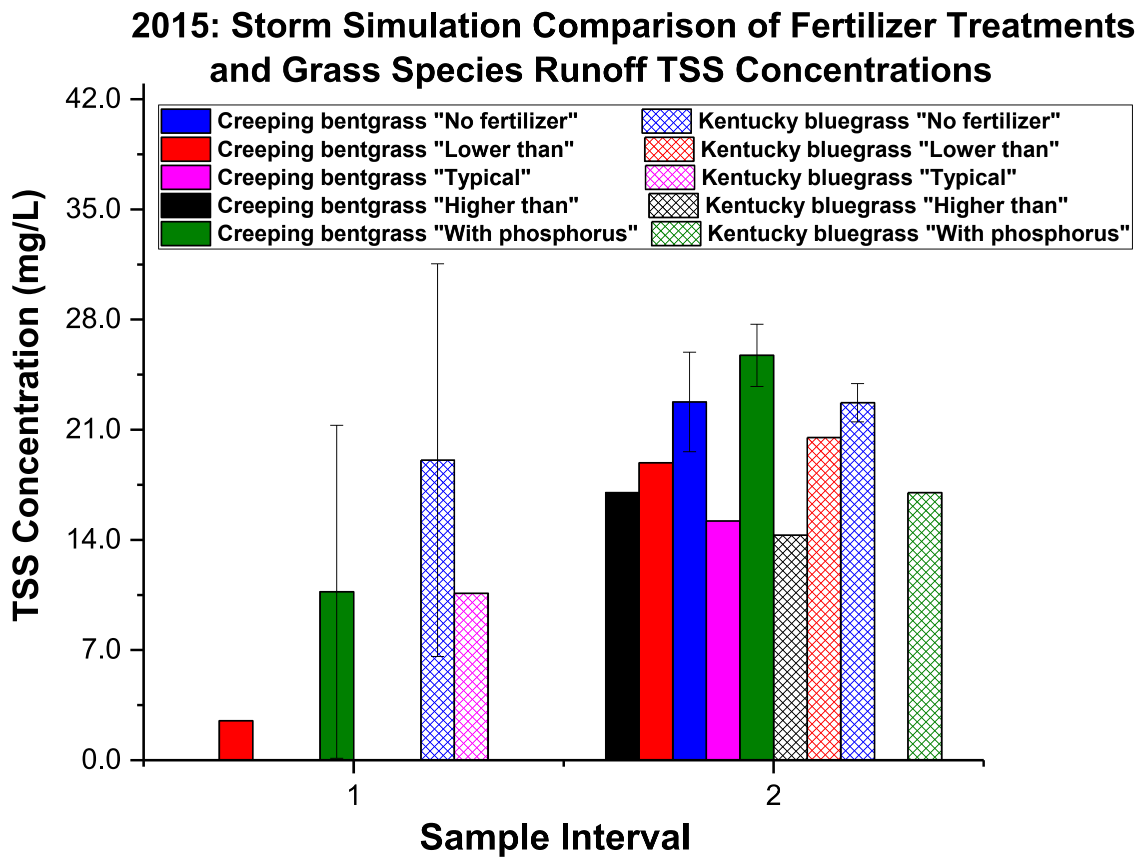

Figure 42. Changes in runoff TSS concentrations with sample interval. Bars show averaged concentration and error bars represent standard deviation. Inlet TSS concentrations were 60.8 mg/L and 102.4 mg/L for sample interval 1 and 2 respectively.

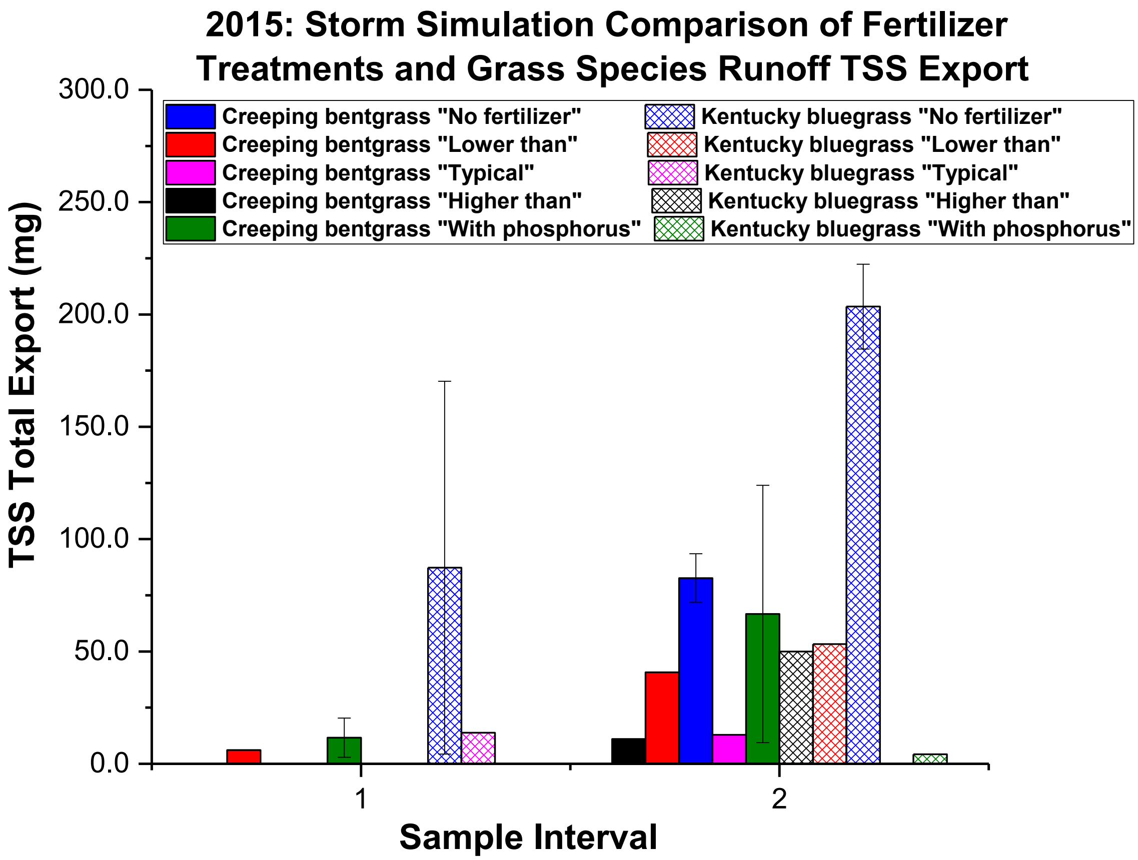

Figure 43. Changes in runoff TSS total export with sample interval. Bars show averaged concentration with error bars representing standard deviation. Inlet import values were 2,591.3 mg and 4,364.3 mg for sample interval 1 and 2 respectively.

4.3.2. Impact of grass species

When storm simulations were carried out in the 2015 study period the infiltrate generated for Creeping bentgrass and Kentucky bluegrass had TSS concentrations that were not identified as being significantly different (one-way ANOVA $F_{1,18} = 0.11$, $p= 0.75$). Runoff TSS concentrations and export were also not significantly different based on grass species (one-way ANOVA $F_{1,11} = 0.12$, $p= 0.73$; $F_{1,11}= 1.41$, $p= 0.26$), respectively.

4.3.3. Impact of seed density

During the storm simulations not enough data was collected from the mesocosms with altered seed densities to allow for statistical analysis using one-way ANOVA to be performed. However, the concentrations and export measurements for TSS are presented in Table 12. When the reported measurements are observed the lowest seed density experienced the highest TSS concentration in the infiltrate.

	Seed Density		
	1 lb/1000 ft.^{2}	3 lb/1000 ft.^{2}	5 lb/1000 ft.^{2}
Infiltrate TSS Concentrations	[a]0.0 mg/L & 7.4 mg/L	1.2 mg/L & 4.4 mg/L	[a]0.0 mg/L & 5.3 mg/L
Runoff TSS Concentrations	[b]18.5 mg/L	[b]10.6 mg/L	12.1 mg/L & 24.3 mg/L
Runoff TSS Total Export	[b]40.1 mg	[b]13.8 mg	12.1 mg & 17.6 mg

Table 12. Summary of the infiltrate and runoff TSS concentrations and export for each seed density from 2015 storm simulations.
[a]- value of 0.0 represents a measurement that was below the detection limit
[b]- only one value provided because relevant mesocosms only generated enough runoff during one of the two simulations

4.3.4. Discussion: Total suspended solids

An inspection of the inlet water TSS concentrations indicated that the sil-co-sil sediment may not have been evenly distributed in the inlet water; specifically during the second round of storm simulations. It is suspected that the higher TSS concentrations that occurred resulted because the sil-co-sil sediment was too coarse for the bilge pump to adequately suspend the sediment. The inlet water may have experienced higher TSS concentrations than the runoff because of the mechanisms (sediment capture and physical barrier) proposed by Moss et al. (2006). Moreover, infiltrate concentrations were determined to be lower than both the inlet and runoff because of the soil's physical barrier, slowing the flow of water even further allowing for more sediment removal.

It was anticipated that the fertilizer treatment would not significantly affect the runoff TSS concentrations, which was supported by the statistical analysis. This expectation was based on literature that has reported substantial reductions in stormwater TSS by turfgrass filters (Soldat & Petrovic, 2008). Seed density results were not examined with statistical methods (due to lack of sufficient number of samples). However, it is suspected that the seed densities employed did not vary the shoot density enough to make a difference in the turfgrass' ability to remove sediment (Kussow, 2008) (Fig. 18).

4.4. Conclusion

When exploring the impact that fertilizer use can have on water quality it has been reported that nutrient loads to water bodies can increase with fertilization (King et al., 2001; King et al., 2007). Results of the storm simulation study only determined the WP treatment to have significantly higher phosphate concentrations in the infiltrate compared to all other treatments. Infiltrate phosphate and TSS concentrations were significantly lower than those in the inlet water, but nitrate concentrations of the infiltrate, runoff or inlet had no significant difference. Runoff phosphate and TSS concentrations were significantly higher than the infiltrate for all treatments and only TSS runoff concentrations were significantly lower than the inlet water. Inlet water, infiltrate and runoff total phosphorus and total nitrogen concentrations for the NF and T treatment were not determined to be significantly different. Moreover, grass species displayed no significant effect on any of the nutrients in the infiltrate or runoff. Initially it was hypothesised that fertilizer treatment would significantly affect the nutrient concentrations during storm simulations and seed density would impact TSS, but grass species would not demonstrate a difference. The results reported support the hypothesis; except for the effect of seed density on TSS. Furthermore the results discussed support the findings of other literature reporting that turfgrass is a viable option for buffer areas (Moss et al., 2006; Stier & Kussow, 2006); specifically for reducing total phosphate and TSS.

Chapter 5. Winter Snowmelt Study

5.1. Phosphate

5.1.1. Impact of fertilizer treatment

Overall the volume collected from snowmelt decreased with sample interval (Fig. 44). The examination of phosphate concentrations based on fertilizer treatment did not show any significant variation (one-way ANOVA $F_{4,20} = 2.26$, $p= 0.09$) (Fig. 45). Moreover, fertilizer treatment did not significantly impact the export of phosphate during winter snowmelt (one-way ANOVA $F(_{4,6.67}) = 1.69$, $p= 0.26$) (Fig. 46).

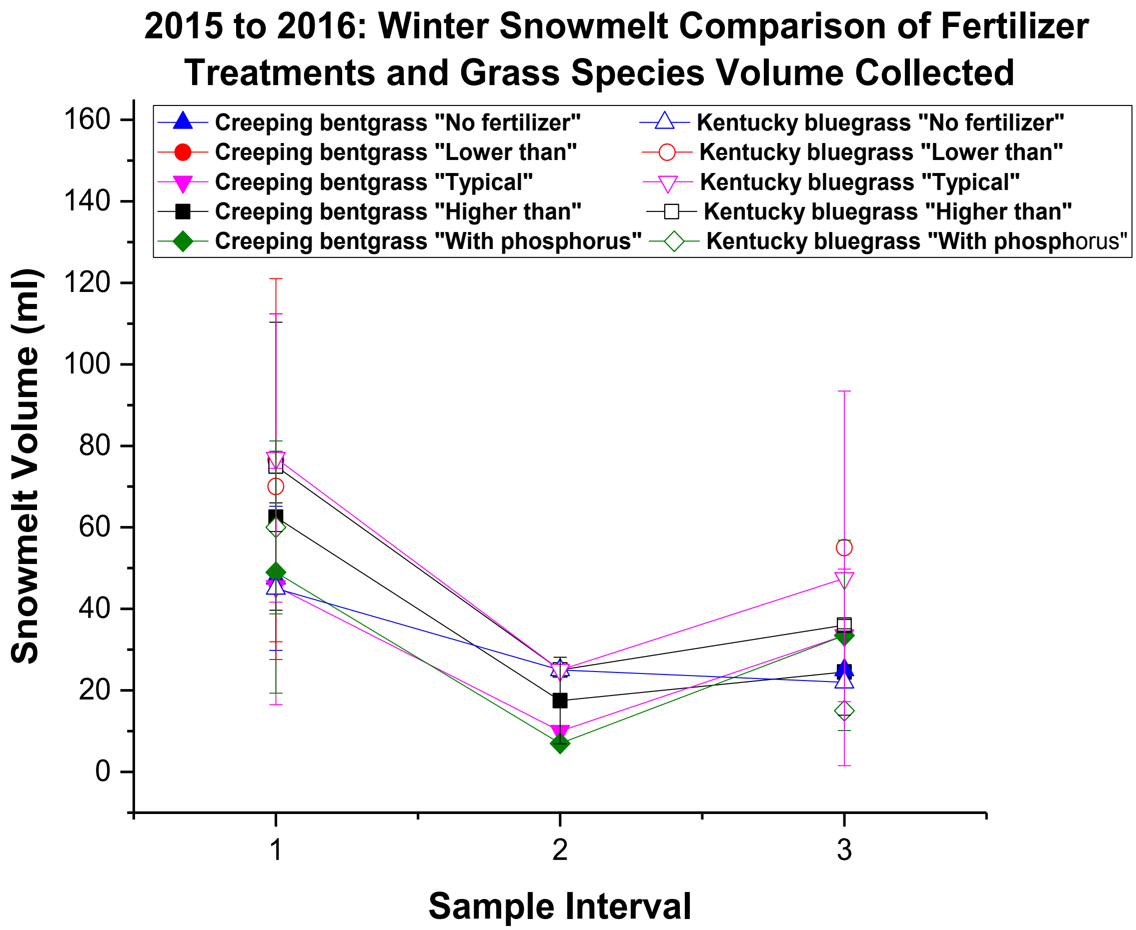

Figure 44. Changes in volume collected with sample interval. Symbols show averaged volume collected and error bars represent standard deviation.

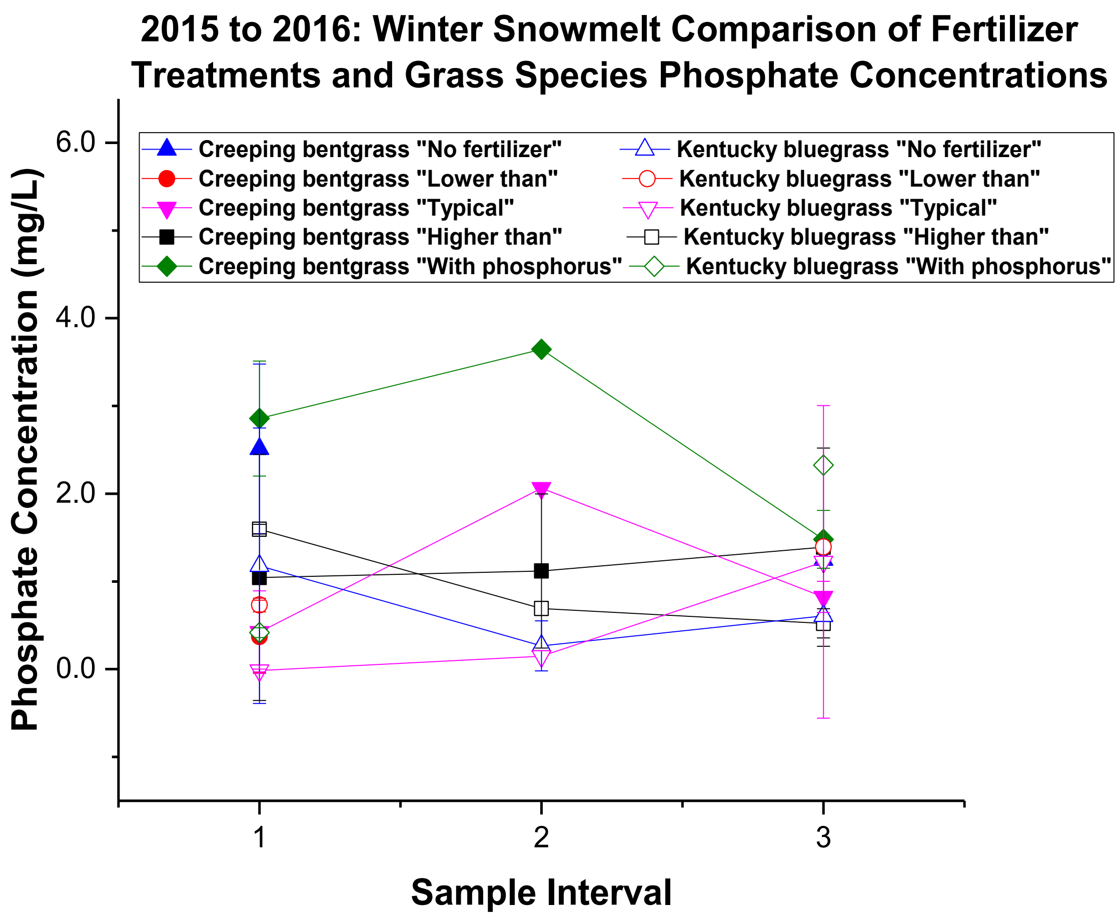

Figure 45. Changes in phosphate concentrations with sample interval. Symbols show averaged concentration with error bars representing standard deviation.

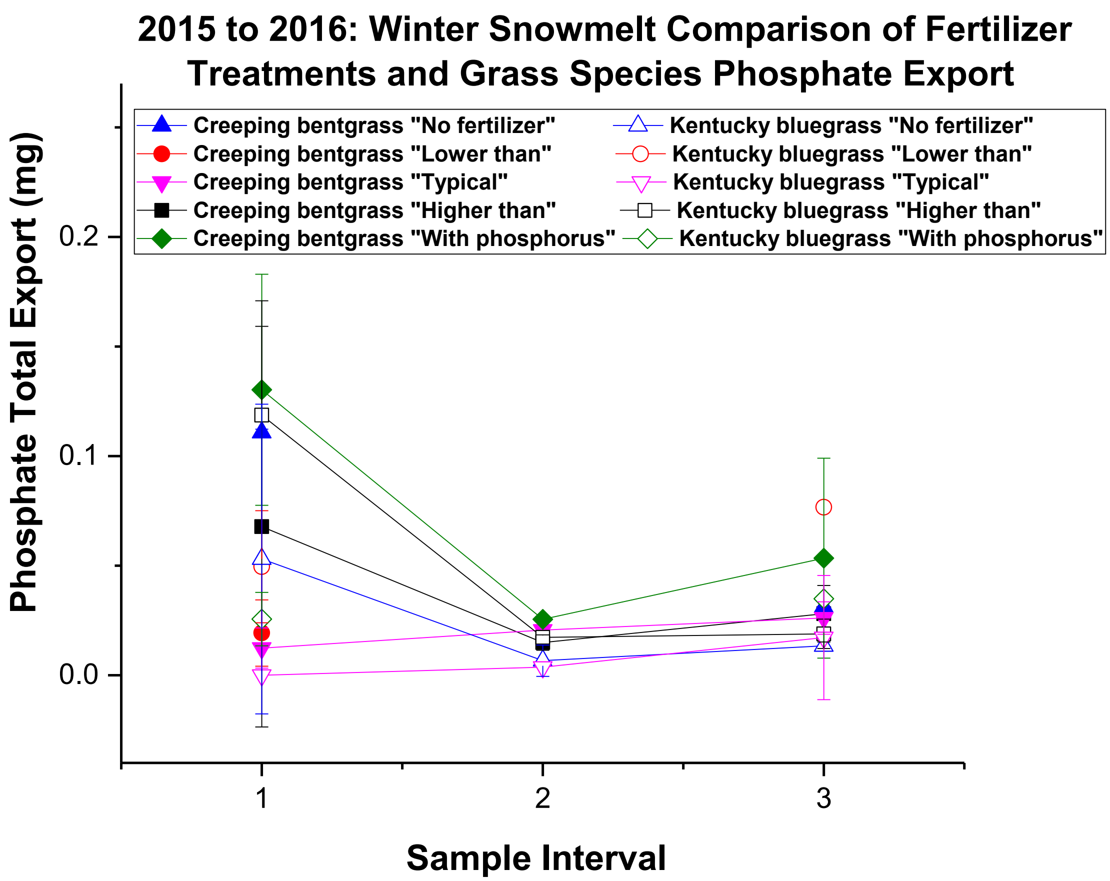

Figure 46. Changes in phosphate total export with sample interval. Symbols show averaged concentration and error bars represent standard deviation.

5.1.2. Impact of grass species

In the 2015 to 2016 winter snowmelt study period phosphate concentrations between grass species varied significantly (one-way ANOVA $F_{1,23} = 4.66$, p= 0.04). However, the phosphate export during snowmelt for both grass species did not vary significantly (one-way ANOVA $F_{1,23}= 0.30$, p= 0.59).

5.1.3. Impact of seed density

During this study, the snowmelt phosphate concentrations and the total phosphate export did not vary significantly with varying seed densities (one-way ANOVA $F_{2,6} = 0.37$, p= 0.71 and$F_{2,6}= 0.19$, p= 0.83), respectively.

5.1.4. Discussion: phosphate

Grass species and seed density were not anticipated to have a significant effect on snowmelt phosphate concentrations and export, but the WP fertilizer treatment was anticipated to impact phosphate concentrations. The WP fertilizer treatment was predicted to have a significant influence on phosphate concentration because of the phosphorus addition and its higher leaching potential compared to nitrate (Wong et al., 1998; Shuman, 2003). The results did not support the expectation that fertilizer treatment would significantly affect phosphate loss, eventhough the WP treatment consistently had higher concentrations as shown in Fig. 45.

Seed density was not identified as a significant factor influencing phosphate loss, thereby supporting the prediction. It is proposed that seed density was not a significant factor on phosphate concentrations or export because alterations to seed density did not result in differences among shoot densities. However, the phosphate concentrations were significantly different between the grass species which was not the predicted outcome. An examination of the overall mean for each species suggested that Creeping bentgrass with a mean of 1.58 mg/L ± 1.00 mg/L experienced higher phosphate losses during the winter months than Kentucky bluegrass 0.85 mg/L ± 0.65 mg/L. Although, limited data was obtained for Kentucky bluegrass mesocosms receiving the WP treatment. Thus, it is suspected that the snowmelt phosphate concentrations reported for the Creeping bentgrass mesocosms receiving the WP treatment are the source of difference between grass species.

Winter snowmelt phosphate concentrations exceeded the EPA discharge limit of 0.10 mg/L 96% of the time. When the results of this research are compared to what others have reported the phosphate loss appears similar. For instance, Bierman et al. (2010)

reported a mean runoff concentration of 1.01 mg/L during frozen soil conditions and this study has a mean of 1.12 mg/L. Maximum phosphate concentrations during snowmelt were higher (3.65 mg/L) than all other samples, including storm simulation runoff concentrations which had a maximum value of 1.62 mg/L. These findings confirm what other research has stated about the highest phosphate loss occurring when soil is frozen (Bierman et al., 2010; Kussow, 2008; Easton & Petrovic 2004).

5.2. Nitrate

5.2.1. Impact of fertilizer treatment

Variations to the type of fertilizer treatment was not found to be a significant factor influencing snowmelt nitrate concentrations (one-way ANOVA $F_{4,17} = 0.29$, $p= 0.87$) (Fig. 47). It was also found that fertilizer treatment did not have a significant impact on the snowmelt nitrate export (one-way ANOVA $F_{4,16}= 0.22$, $p= 0.93$). The maximum nitrate export for each treatment was 0.6 mg, 0.3 mg, 0.2 mg, 0.3 mg and 0.3 mg for the HT, LT, NF, T and WP treatments respectively (Fig. 48).

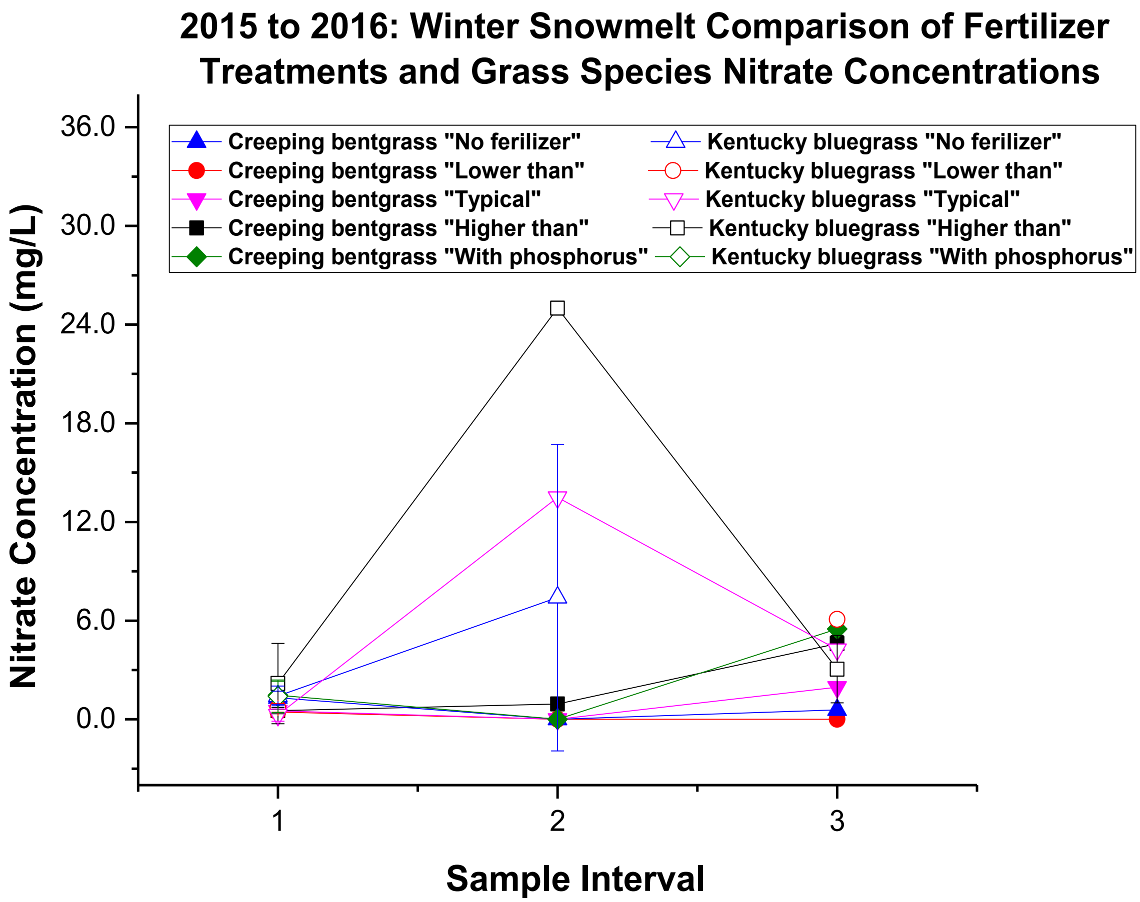

Figure 47. Changes in nitrate concentrations with sample interval. Symbols show averaged concentration with error bars representing standard deviation.

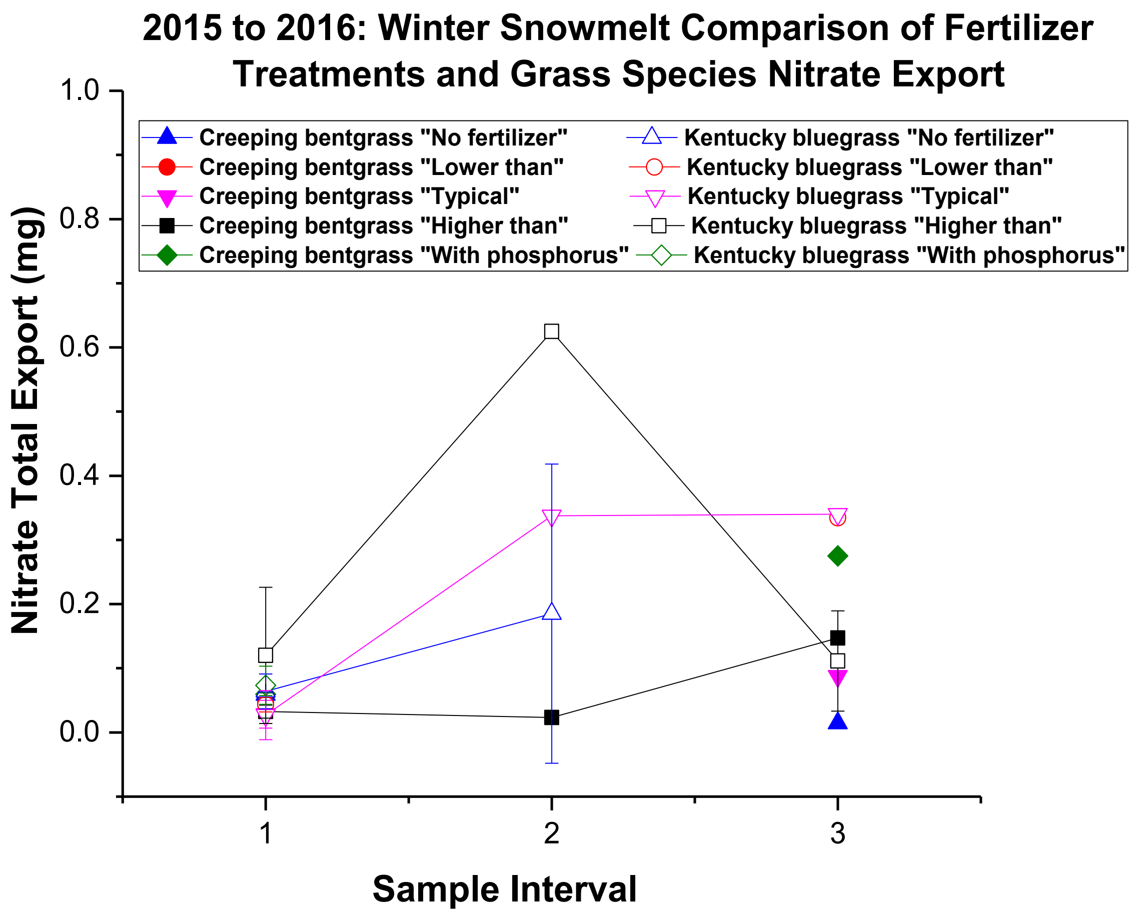

Figure 48. Changes in nitrate total export with sample interval. Symbols show averaged concentration with error bars representing standard deviation.

5.2.2. Impact of grass species

Nitrate concentrations in the snowmelt water did not vary significantly between grass species (one-way ANOVA $F(_{1,11.10}) = 3.43$, $p= 0.09$), although Kentucky bluegrass had the highest nitrate loss with a concentration of 25.00 mg/L. One-way ANOVA analysis determined that grass species did not significantly impact nitrate exports ($F(_{1,13.99})= 4.26$, $p= 0.06$).

5.2.3. Impact of seed density

The winter snowmelt nitrate concentrations and export measurements for each seed density are presented in Table 13. The summarised results indicate that seed density was not a significant factor that effected snowmelt nitrate concentration or export.

Seed Density	**Snowmelt Nitrate Concentrations**	**Snowmelt Nitrate Total Export**
1 lb/1000 ft.^{2}	[b]0.75 mg/L	[ab]0.0 mg
3 lb/1000 ft.^{2}	4.25 mg/L - 13.50 mg/L	[a]0.0 mg - 0.3 mg
5 lb/1000 ft.^{2}	[b]11.48 mg/L	[b]0.5 mg

Table 13. Summary of the nitrate concentrations and export measured during the 2015 to 2016 winter snowmelt sampling period.
[a]- value of 0.0 represents a measurement that was below the detection limit
[b]- only one value provided because relevant mesocosms only generated enough sample volume during one of the three collection days

5.2.4. Discussion: nitrate

Expectations for nitrate loss when performing winter snowmelt sampling were that fertilizer treatment, grass species and seed density would not have a significant effect on the nitrate concentrations. This was expected because regardless of grass species or seed density the turfgrass would be in a dormant state and mechanisms of actively growing turfgrass that influence nutrients would no longer be occurring. Fertilizer treatment was not anticipated to be a significant factor on snowmelt nitrate concentrations or export because a quick release fertilizer was used for the fall application and the turfgrass would utilize available nitrate before the reduced microbial activity and plant uptake (King et al., 2006).

The results presented support these predictions, and the results presented in Table 13 offer support to the hypothesis that seed density is not a significant factor. Although fertilizer treatment was not determined to have a significant effect on nitrate loss, the highest nitrate concentration of 25.00 mg/L was observed from the HT treatment. However, during this study an unusually cold winter for the area was experienced which did not allow for the anticipated higher volumes of snowfall to take place before melting. Rather, three phases of mild snowfall followed by warmer temperatures generated the data collected. This resulted in several missing data points because not enough volume was produced for both nitrate and phosphate analysis. Consequently, averaging of nitrate results was not always possible if only one of the replicates generated enough volume for nitrate processing. Moreover, the 13.50 mg/L measurement observed for the T treatment and the 25.00 mg/L recorded for the HT treatment could be the outliers since they both lacked a replicate. The average nitrate concentration for this study during snowmelt was 3.52 mg/L which is comparable to 2.45 mg/L as reported by Kussow (2008). When the 13.50 mg/L and 25.00 mg/L measurements are considered the CCME limit of 13 mg/L was exceeded twice over the course of the winter sampling period.

5.3. Total suspended solids

Over the duration of the sampling period from December 2015 to April 2016, the volume of sample collected was not enough to perform TSS analysis and will not be mentioned in the following sections.

5.4. Conclusion

During winter snowmelt sampling from December 2015 to April 2016, phosphate and nitrate concentrations peaked in March. Fertilizer treatment and seed density were not found to be factors influencing phosphate loss. However, statistical analysis did reveal significance between Creeping bentgrass and Kentucky bluegrass phosphate concentrations. Fertilizer treatment and grass species was not determined to significantly alter nitrate loss during snowmelt. Statistical analysis comparing the snowmelt nitrate concentrations and export from the different seed densities could not be performed, but the results summarised in Table 13 indicate that nitrate concentrations and export did not have a dependence on seed density. An examination of the TSS concentrations and export during winter snowmelt could not be carried out because of low sample volume. Initially it was hypothesized that fertilizer treatment would significantly influence phosphate concentrations during the winter months, but grass species and seed density would not significantly impact nutrient loss. In addition, nutrient loss during snowmelt would be higher than those experienced during the summer and fall.

The results presented did not support the hypothesis in regards to phosphate because fertilizer treatment was not determined to significantly vary phosphate concentrations. Furthermore, phosphate concentrations were identified as being significantly different between grass species. The revealed difference in phosphate concentrations between the grass species is suspected to be due to missing data for the Kentucky bluegrass WP mesocosms. Although statistical analysis could not be carried out on seed density and nitrate concentrations, it can be inferred from the summarised results that seed density was not a significant influence on nitrate during winter snowmelt; which supports the original hypothesis. In addition, the hypothesis that nitrate and phosphate concentrations would be higher over the winter months was supported with a maximum concentration of 3.65 mg/L for phosphate and 25.00 mg/L for nitrate during the winter months. Thus, the results from this study suggest that the use of zero phosphorus formulas can noticeably reduce phosphate loss (Fig. 45) and grass species may be a significant factor in limiting nutrient inputs during winter months.

Chapter 6 Conclusions

Nutrient pollution is an issue of growing concern as the degradation of limited freshwater resources continues (Karr, 1991). When water quality is impaired the implications are not only ecological, but financial as well (Dodds et al., 2008). Strategies employed to limit nutrient exports to inland water bodies include reducing the frequency and amount of runoff that occurs during precipitation events. The primary method for achieving runoff reductions is to establish vegetative buffers between anthropogenic landscapes and water bodies (Moss et al., 2006) because bare soil can markedly reduce water infiltration (Carmi & Berliner, 2008) and therefore increase the amount of runoff. Hence, turfgrass within the urban landscape has the potential to mitigate nutrient exports and therefore protect water quality (King et al., 2001). Using data collected from turfgrass mesocosms, it was established that turfgrass can reduce nutrient concentrations and exports to water bodies.

In order to determine the impact of turfgrass on infiltrate nutrient concentrations the effect of fertilizer treatments, turfgrass species and seed densities was examined by performing rain simulations. The objective of studying the listed management practices was to determine and compare the efficacies of best management strategies reported by other researchers to reduce nutrient input from golf courses. During rain simulations it was hypothesized that fertilizer treatment and grass species would not significantly affect the infiltrate nutrient concentrations from turfgrass, but seed density would demonstrate a significant influence on TSS concentrations. The results of this research supported these predictions except for the effect of seed density on TSS because shoot density was increased or decreased by altering seed density as anticipated.

Another objective of this research was to explore the impact of fertilizer treatments, turfgrass species and seed densities on nutrient concentrations and total exports of already polluted water during storm events. Results from the storm simulations indicated that turfgrass responds predictably to already polluted water and has a capacity to improve water quality. Nutrient concentrations measured were the highest in the inlet water followed by runoff and infiltrate. Conversely, the export of phosphate, nitrate and TSS was consistently higher in infiltrate than in runoff because of differences in the amount of water volume being collected. When the storm simulations were carried out runoff was not consistently generated. This could have resulted from differences in soil compaction; a factor not quantified in this study. The hypothesis tested was that fertilizer treatment would significantly affect the nutrient concentrations during storm simulations and seed density would impact TSS, but grass species would not demonstrate significant variation. Fertilizer treatments resulted in a significant difference in phosphate concentrations in infiltrate but not in nitrate or TSS. Variations with respect to grass species and seed density were not determined. Therefore, the hypothesis was supported except in regard to seed density.

The winter snowmelt portion of this research generated phosphate and nitrate concentrations that were higher than the other sampling periods. Objectives of this research were to determine the influence of turfgrass on nutrient concentrations during winter snowmelt, again with respect to fertilizer treatment, turfgrass species and seed density. However, due to the nature of winter sample collection and the abnormal winter experienced from December 2015 to April 2016, these results should be considered a qualitative attempt at examining the effect of seasonality on nutrient exports to inland water bodies. The initial hypothesis tested was that fertilizer treatment would significantly influence phosphate concentrations during the winter months, but grass species and seed density would not significantly impact nutrient loss. In addition, nutrient loss during snowmelt would be higher than those experienced during summer and fall. In relation to the prediction of fertilizer treatment and grass species, the hypothesis was not supported when phosphate concentrations were analyzed. Moreover, a higher nutrient loss during the winter snowmelt than summer and fall was not experienced.

Future research focusing on different fertilizer treatments, grass species, seed density and the seasonal effect of winter on nutrient loss would benefit from long-term studies with higher replication. More replication in 2015 when only duplicates were established would have provided more data and addressed the problem of missing runoff measurements during storm simulations and winter snowmelt sample collection. Additional data for seed density analysis also would have allowed for statistical analysis to be carried out on results from storm simulations and winter snowmelt nitrate. Furthermore, the inclusion of nutrient and TSS analysis from soil only conditions would be valuable in future research and this was short term research carried out on young turfgrass that was grown from seed at the beginning of each study period. Thus, it is suggested that a more comprehensive picture of turfgrass behaviour and functional benefits could be gathered from a long-term study that incorporates soil only measurements.

This research was carried out by following some of the practices that have already been identified as best management practices. They include applying fertilizer when rainfall is not expected within 48 hours, using controlled-release products and using a zero phosphorus fertilizer formula except for the WP treatment (King & Balogh, 2013). The research performed combined field conditions with laboratory control over specific variables to allow for the collection of data from small to moderate precipitation events.

The results of this study demonstrated that nutrient pollution is an important issue, especially in regards to phosphorus originating from golf courses. This study also supported the claims that when phosphorus is removed from fertilizer formulas the amount transported to inland water bodies can be reduced (Lehman et al., 2009; Davis & Lydy, 2002). The land development activities taking place result in deteriorating water quality and therefore it is imperative that research continues to discover methods to mitigate nutrient inputs to freshwater systems.

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