Sveriges lantbruksuniversitet
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Water movement and phosphorus leaching from managed and unmanaged grass buffer strips

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Master's Thesis in Soil Science

Examensarbeten, Institutionen för mark och miljö, SLU 2016:06

Uppsala 2016

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Credits: 30 ECTS
Level: Second cycle, A2E
Course title: Independent project/degree project in Soil Science - Master´s thesis
Course code: EX0430

Place of publication: Uppsala
Year of publication: 2016
Title of series: Examensarbeten / Institutionen för mark och miljö, SLU
Number of part of series: 2016:06
Online publication: http://stud.epsilon.slu.se

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Abstract

This study was undertaken in a long term field trial investigating the effect of grass buffer strips on nutrient leaching. The treatments included were harvested grass (HG), non-harvested permanent grass (PG) and cultivated soil without a buffer strip (control). Infiltration rate (IR), field saturated hydraulic conductivity (Kfs) and saturated hydraulic conductivity (Ks) were determined. In addition, rainfall simulation in the laboratory was conducted on soil cores (20 cm diameter, 20cm high) after these had been exposed to repeated freezing and thawing in the field for one month. Large spatial variability of hydraulic properties was demonstrated when comparing different treatments. Both PG and HG were shown to have a higher IR during the summer compared to the control. Levels of turbidity in the water were significantly higher ($p \leq 0.05$) in the control plots than PG plots. Water concentration of dissolved reactive phosphorus (DRP) was significantly ($p \leq 0.05$) higher in both PG and HG treatments than in the control. This was proposed to be due to mobilizing of P from vegetation during the freezing and thawing cycles. A linear regression model of TOC on DRP gave a coefficient of determination R^2=0.45.

Keywords: dissolved reactive phosphorus, hydraulic conductivity, organic matter, rainfall simulation, turbidity

Popular Science Summary

Crop and pasture yields have been increased dramatically since the mid 20th century due in part to the application of phosphorus fertilizers. The unintended side effect of this has been increased eutrophication of waterways which has decreased water quality and depleted aquatic ecosystems. In Nordic countries phosphorus is most often the limiting nutrient for algal growth in waterways. Elevated levels of phosphorus, often arising from upstream agricultural run-off, has often led to increases in eutrophication.

There has been a large effort to reduce phosphorus discharge to waterways, particularly from agricultural sources. One of the most commonly implemented, often through government subsidies, has been the creation of a grass 'buffer strips' between agricultural fields and waterways. These consist of a narrow strip (i.e. 6m in width) usually sown with grass and left uncultivated for a number of years. These are primarily intended to reduce nutrient discharge from overland flow, whereby water flowing above the soils surface transports nutrients to waterways. Particularly on sloping surfaces grass buffer strips have shown to be largely effective at reducing phosphorus loads to waterways.

Less studied however has been their effectiveness on land with flat topography with water transport primarily occurring beneath the soils surface. Also less studied is how effective different forms of phosphorus, particularly the dissolved reactive phosphorus, behave in grass buffer strips or whether the dynamics can change due to weather events such as freezing and thawing of vegetation during the autumn. Cutting and removing grass from a buffer strip has been one management practice suggested to reduce leaching of dissolved reactive phosphorus.

Recent research suggests there are large variances in grass buffer strip effectiveness depending on one or a combination of topographic and seasonal factors. In some instances, particularly on flat topography, they have been shown to be completely ineffective.

This thesis aimed to determine how cutting and removing grass affected both water movement and leaching of phosphorus, both particulate phosphorus and dissolved reactive phosphorus. It was found that water movement generally decreased at the soils surface due to the impacts of cultivation but only during the summer. With regards to leaching of phosphorus both permanent and cut grass leached less particulate phosphorus but concentrations of dissolved reactive phosphorus was >10X higher in both grass treatments than when the field was left uncultivated. As the grass underwent freezing and thawing cycles the elevated concentrations of dissolved phosphorus was attributed to the breakdown of organic matter which liberated large amounts of phosphorus in the dissolved form.

It is clear from this research that an area which needs to be investigated more in the research of management of grass buffer strips is reducing the liberation of dissolved phosphorus. One suggestion from the results of this study could be cutting and removing the grass in the summer to reduce levels of decaying organic matter later releasing dissolved reactive phosphorus later in the autumn. However, alternative measures (i.e. chemical amendments to grassed plots) are necessary to abate eutrophication.

Table of Contents.

1. Introduction.

Vegetated buffer strips have been shown to be effective in reducing the total phosphorus contribution from upstream agricultural land to downstream waterways, albeit mainly in short term studies emphasizing overland flow (Dorioz et al. 2006; Hoffmann et al. 2009; Kronvang et al. 2007; Roberts et al. 2011). Predictions of P retention in vegetated buffer strips are extremely challenging because of the complex interactions and feedback between hydrology and biogeochemical transformations of P, and therefore their effectiveness can vary greatly (Hoffmann et al. 2009; Noij et al. 2013).

The effectiveness of vegetated buffer strips has been demonstrated on sites with slopes 12-18% (Uusi-Kämppä & Jauhiainen 2010), 11-16% (Dillaha et al. 1988) to slopes as flat as 2% (Noij et al. 2013). However, the study by Noij et al. (2013) demonstrated ineffectiveness of vegetated grass buffer strips in fields with four other hydrogeological classes in the Netherlands, which had <2% slope and deep groundwater flow. Thus it was concluded that the occurrence of surface run-off or a shallow flow groundwater is a pre-condition for vegetated buffer strips to be effective in mitigating phosphorus pollution.

The management practices of the field upstream from the vegetated buffer strip will have an impact on the effectiveness in mitigating phosphorus pollution. Uusi-Kämppä and Jauiainen (2010) undertook a study investigating both management practices of an adjacent field in conjunction with a vegetative buffer strip. This study found that vegetative buffer strips were significantly more effective in reducing total (TP) and particulate (PP) loads to the environment compared to no buffer strip, when the upstream field was being conventionally tilled compared to being direct drilled or being in a pasture phase. This can be attributable to soil tillage contributing more particulate phosphorus (PP) to the vegetated buffer strip to begin with, thus more could be removed. This study, also demonstrated a higher total volume of water and total solids in surface run-off both when the (adjacent) upstream field was cultivated and when there was no vegetative buffer strip. When conventional tillage reduces porosity and water infiltration (Azooz & Arshad 1996), it is likely that the greater volume of water is carried in surface run-off, transporting with it more PP from the cultivated area.

Since dissolved reactive P (DRP) behaves very differently to PP in its interactions with the soil and water the effectiveness of vegetative buffer strips may be totally different for retention of these two P forms (Dillaha et al. 1988; Uusi_Kämppä & Jauhiainen 2010; Uusi-Kämppa et al. 2011). Permanently vegetated buffer strips are ineffective in reducing losses in DRP and losses often increase on field margins when left uncultivated (Muukkonen et al. 2009; Uusi-Kämppä & Jauhiainen 2010). One likely reason for this is that phosphorus can be biologically converted to more labile P forms within the vegetated buffer strip and be delivered as DRP (Roberts et al. 2011). Mowing and removing the vegetation from the buffer strip can reduce DRP in spring surface run-off (Uusi-Kämppä & Jauhiainen 2010).Elevated levels of DRP is of concern as DRP is available for immediate algal uptake in surface waters, thus higher levels will enhance eutrophication.

Particulate P may also be lost through sub-surface drainage through macropores although this has been far less studied than overland flow losses (Roberts et al. 2011). Ulén and Persson (1999) measured phosphorus losses in a Swedish clay soil via sub surface drains and found that total P was largely lost in rapid episodes. On average nearly half of the total P transport occurred during 140 hours in six year period. Similar to overland flow, PP comprised of the total P losses (63%). Macropores, mainly cracks, attributed to shrink/swell and freeze/thaw cycles were the main causes for the leaching. This study showed that sub surface processes can have an important effect on particulate P leaching. Therefore P leaching mitigation strategies need to consider reducing P losses through the soil and not just over it.

The objective of this experiment is to investigate the effect of vertical flow on P leaching. It is hypothesized that PP leaching will be more pronounced in soil columns from conventionally cultivated plots than in those from grassed plots due to a decrease in aggregate stability.

2. Materials and Methods

2.1 Experimental design

The experimental buffer strips are positioned on the southern side of a field located approximately 15 km south of Uppsala in eastern Sweden (59°43’60’’N; 17°41’21’’E). The field trial was established in 2010 for investigating the effectiveness of grassed buffer strips in reducing nutrient and pesticide leaching from a conventionally managed arable field. The field has a very slight slope with a gradient of about 1% in the north south direction and the soil is a clay loam (32.3% clay, 33.1% silt, 34.6% sand) with an organic carbon content of 13 g Kg^{-1} (Larsbo et al. 2015). The 12 experimental plots are 6 m by 6 m in size and all are drained at 1m depth by a 6 m long central drain which collects sub surface run-off. On the lowest side of each plot surface run-off water is collected. Both drain water and surface water runs to a measuring station where water volume is continuously logged by a datalogger. Water sampling takes place flow-proportional guided by the logger and the composite samples are collected regularly (at most weekly) depending on the water flow.

The treatments are arranged in a randomized block design with 4 blocks with 3 treatments as follows;

Control- conventionally cultivated. Autumn ploughed to a depth of 20 cm and harrowed to a depth of 6 cm prior to sowing. In spring 2015 these plots were sown with Canola.
PG- Permanent Grass. A mixture of timothy (*Phleum pretense* L.) and red fescue (*Festuca rubra* L.) was sown in 2010. Timothy was the dominant species at the time of the measurements.
HG- Harvested Grass. As above but the grass was harvested once a year in the autumn then raked off the buffer zone area.

2.2 Infiltration measurements in summer and autumn

Infiltration measurements took place in two periods; 11-14 August 2015 and 26-27 September 2015, respectively. A twin ring infiltrometer with inner ring diameter of 30 cm was used in the north east corner of every plot after a patch of surface vegetation was removed by cutting. The device was then inserted by hammering with a heavy wooden ram to a depth of 10 cm.

Water was then added to a depth of 13 cm inside the inner ring and filled to the top of the outer ring. The rate of fall of water was then measured initially at 30 second time intervals. If water movement was slow this could be changed to 1 or 2 minute time intervals. Measurements were recorded until a steady state was reached. This consisted of at least five successive measurements that had equal difference from the previous measurement. A steady state was generally achieved between 25 minutes to 1 hour in the summer and 15-40 minutes during the autumn. The summer values typically needed 1-12 replenishments whilst the autumn measurements typically needed 0-1 replenishment to achieve a steady state. The latter was due to continuous rain prior to the measurements in the autumn which decreased infiltration rate.Sorptivity (S) and infiltration rate (IR) were calculated from the data. S was calculated by first plotting cumulative infiltration against the square root of time ($\sqrt{t}$), for small t (first 4 measurements). This because t increases the gravitational gradient predominates thus diminishing the effect of sorptivity (Hillel 2004). Then a linear equation was fitted to these points and S was derived from the coefficient of this linear equation. This approach is a graphical method of what was discussed in Hillel (2004) from the theory originally composed by Philp (1957a). IR was calculated according to equation 1 (Reynolds et al. 2002; Bagarello et al. 2012). This was derived from plotting all the values of t which were at a steady state against cumulative infiltration and fitting a linear equation in Microsoft Excel. qs Was then derived from the coefficient of this equation and fitted to equation 1.

(1) IR = qs (where qs is the quasi steady state of infiltration)

This calculation method may be prone to overestimate IR as it neglects the effect of the pressure head caused by the ponded water (Reynolds et al. 2002) or by the lateral infiltration at larger t (Thony et al. 1991).

2.3 Field saturated hydraulic conductivity (Kfs) at different depths

For each plot Kfs was determined during two seasonal sampling periods; in summer (August 17 – September 4) and in autumn (September 28 – October 14). In order to determine Kfs two adjacent holes, 10 cm in diameter where successively augered to 25, 45 and 55 cm depth and measured at two times, initial and three days after the first measurement. Both holes where augered within close proximity (~0.5 m) to where the surface infiltration measurements were taken to minimize destruction to the plot, these plots being part of a long term experiment established in 2011. In the summer plots were replenished with water 3 times prior to the measurements commencing for the 10-25 cm depth, 2 times for 25-45cm depth and not pre-wetted at the 45-55cm depths assuming that the earlier replenishments would suffice. At the shallower depths during summer more pre-wetting was required as in most plots water movement was too rapid to measure and unlikely to reflect Kfs. At the summer 45-55 cm depths and during the autumn measurements, no pre-wetting was required as water velocity from the auger hole was much slower and measurements were assumed to reflect Kfs at much shorter times.

After the pre-wetting period (if required) the rate of fall of the head of water in the auger hole was measured using the same time intervals as the infiltration measurements with the twin ring infiltrometer. We also recorded the number of water replenishments in each auger hole. Measurements where stopped once the measurements where adjudged to be at a steady state which was usually a minimum of 5 measurements with no change in the rate of water movement. Kfs was calculated according to the Porchet solution, equation 2 (van Hoorn 1979);

$$(2)\quad Kfs = 1.15*r(\log(ho + r/2) - \log(ht + r/2))/t$$

Where r is radius of auger hole = 10 cm, t= time, ho is the height (cm) of the water above the bottom of the hole at t=0, ht is the height (cm) of the water above the base of the auger hole at the time of the last measurement.

The Porchet solution neglects the effects of the pressure head caused by the water column above the base of the auger hole and the negative pressure head in the surrounding unsaturated soil (Messing & Jarvis 1990). This method also largely neglects the structural changes in the soil profile, assuming an Isotropic, homogeneous and rigid porous soil (Messing & Jarvis 1990). Similar to the double ring method the height of the head (h0) of the water above the base of the hole should be kept as small as possible and more accurate measurements are thus achieved with a larger diameter. When measurements are expected to exhibit large seasonal variations (i.e. in soils where water flow is dominated by cracks in the soil during the summer, yet may be largely impermeable when fully swollen) it is better to use a simple method to obtain Kfs and measurements should thus be taken comparatively (Messing and Jarvis 1990). Due to the comparative nature of the treatment effects this experiment opted for a more simple method to determine Kfs rather than a method such as tension disc infiltrometer.

2.4 Particulate (PP) and dissolved reactive phosphorus (DRP) leaching from soil columns

The rainfall simulator consisted of a fixed boom spray mounted on a steel bar (1.6m long). This was positioned 1 m above a stainless steel bench. The stainless steel bench had a slight slope so excess water would drain into a gutter which ran along the length of the center of the bench. The boom had 4 nozzles 15cm apart which were adjusted to a height of 70cm above the bench. The boom would spray the length of the bar then back at 60 second time intervals to simulate rainfall of 11-13 mm/hr. All the apparatus was contained within sliding doors of plexiglass to prevent wind turbulence and spray drift. The water that was used during the rainfall simulation was prepared in the lab at SLU to resemble the chemical composition of the rainfall in the area.

A rainfall simulation was conducted to determine P leaching with percolated water. 12 soil columns, one soil column for each plot (20 cm in diameter 20 cm high) were collected on 15th October 2015. This was done by pushing the columns into the soil with a tractor mounted hydraulic ram. Once samples were collected they were left on the ground in the field for one month to undergo natural freezing and thawing cycles since the temperature was often beneath 0°C in the nights. To avoid the base and the sides of the soil columns undergoing more freezing than if they remained in place in the topsoil, they were wrapped in several layers of housing insulation and then with a layer of plastic, with only the top soil surface of the column exposed to open air. One day before the beginning of the rainfall simulation the samples were taken to the laboratory. Then vegetation was removed from the HG soil columns by cutting with secateurs and the columns were left at room temperature overnight. The next day excess soil was removed from the bottom of the column by picking away at loose aggregates with a knife rather than cutting a straight edge across the base of the soil column. This was done to avoid the destruction/clogging of macropores.

The rainfall simulation was completed for plots 1-6 between November 13 - 15 and November 15-17 for plots 7-12. Two successive rainfall simulations were performed for each plot at 24 hr intervals during the aforementioned dates. Once the bottom of each column was prepared, a wire mesh was placed under the bottom to prevent large soil particles leaching. Then they were placed inside a funnel base to collect the percolating suspension. This base was drained by a tube, which ran through the gutter of the stainless steel bench to be collected in 2 L PVC containers positioned on the floor. A schematic diagram of this method of collection is given in Liu et al. (2012). The irrigations ran for 3 hrs and sample collection was ended 24 hrs after the end of the irrigation.

The samples were collected after 24 hrs, and then the volume of each bottle was recorded. After this they were measured for turbidity in 250 ml sub samples, with a Hach 2100 Turbidimeter. First measurements were done after manual shaking, and the second measurement was done after the sample had been left standing for 4.5 hours at room temperature (t=4.5hr) for sedimentation of particles/aggregates larger than clay size. Measurements determining concentration of total P, DRP and total organic carbon (TOC) was performed at the laboratory at the department for Water and Environment, SLU laboratory according to EU standards. Dissolved reactive P was analyzed after filtration with filters with pore diameter 0.2µm. All chemical analysis was performed from samples of the same plot mixed for the 2 irrigation events in equal ratios (50:50), because collected water volumes did not vary much between the irrigations. Particulate P was estimated by subtracting DRP from total P.

2.5 Hydraulic conductivity and macroporosity in small soil cores

The way in which water is transmitted or retained in a structured clay soil is related directly to the make-up of pores (Messing & Jarvis 1990; Messing & Jarvis 1993; Azooz and Arshad 1996). Pores are usually arbitrarily assigned as either macropores, mesopores or micropores. Typically the interaggregate spaces are collectively termed macropores which serve as the principal avenues for infiltration and drainage of water as well as aeration. In contrast the intra-aggregate spaces are termed as the micropores which usually are involved in retention of water and solutes (Hillel 2004). A clay soil with flat platy aggregates will have more surface area and therefore more capacity to retain water (and/or associated solutes potentially) due to a larger capacity for capillary binding in micropores, than a soil with large spherical aggregates (Grip and Bishop 2000). The graphical relation between pore size distribution in a soil and water potential is termed soil moisture characteristic or pF-curve (Grip and Bishop 2000). In other words, it is expected that a soil which has a higher proportion of macropores also displays higher rates of water transmission. The pF curve is a functional relationship between the soil matric potential (suction) and volumetric water content. Soil pF is determined as follows; Soil samples are drained at increasing suctions (negative pressures) and the soil sample is weighed to determine residual water content for a given suction. From this relationship the pore size distribution is determined (Grip and Bishop 2000). The higher the proportion of micropores the more tightly bound water is, as a greater suction is required to drain these micropores. This relationship is of vital importance when modeling how water and solutes are transported or retained in soil (Hewelke et al. 2015). Applications of the pF curve are extensive and it can be important in determining irrigation rate in agriculture and in modeling environmental water and solute movement (Hewelke et al. 2015; Bardhan et al. 2015).

Between October 16-19 small soil cores (7.2 cm diameter, and 5 cm high) were collected from each plot at 7 separate depths (cm); 0-5, 5-10, 10-15, 20-25, 25-30, 40-45, 50-55. Since large amount of earthworms were observed in the plots when sampling, the cores were wrapped in many layers of plastic wrap after weighing and before being placed in an oven at 55°C in order to kill the worms and prevent further bioturbation. Then the samples were progressively saturated from the bottom up at approximately 1 cm intervals over a one week period in a large plastic box. It was evident that the oven methods had been successful in killing any remnant earthworms in the samples due to the foul odor omitted whilst performing the saturation.

Then Ks parameters were measured according to the constant head soil core tank method with a pressure head of 10 cm and Ks was calculated according to Darcy’s law (Reynolds et al. 2002).

(3) $Ks = (V * Tcorr * dx) / (t * A * dH)$

Where V = volume of water, T corr = adjusted temperature, dx= length, t = time, A = area, dH = height of water head. (There was an excel sheet already pre-made by laboratory staff at SLU which allowed for simple computation of the Ks according to formula (3).

After the Ks experiment was done the soils were then drained at increasing suctions to determine macroporosity. These suction heads were; 0.05 m, 0.5 m, 1m, 10m. After the 10m suction was performed the soil cores were dried at 105°C to determine water content at sampling and to calculate dry bulk density. Using a default value of 2.66 g/cm3 for particle density, soil porosity was calculated for each step of soil level.
A regression analysis was performed for Ks and macroporosity (% of pores>30µm diameter) for the 0-5cm cores. Whilst there are several definitions of a macropore (Beven and German 1982), it was decided to analyze the portion of macropores >30 µm diameter as several other studies have defined a macropore as this size (Beven and German 1982) which corresponds to draining all pores at 1 meter suction.

2.6 Statistical Analysis

For statistical analysis of all data the computer software package SAS version 9.4 was used (SAS institute, 2012). ANOVA was performed according to a complete randomized block design for the response variables of the following data sets; IR, Kfs (summer and autumn analyzed separately) Ks, turbidity, PP and DRP.
Means comparison for treatments and blocks either used Tukeys or lsd means separation procedures, depending on whether results looked more likely to encounter a type 1 or type 2 error. For comparison of interactions the Tukey-Kramer method was used in the *lsmeans* (least square means) statement for unbalanced data.

An ANCOVA was performed to investigate relationship between the variables volume of percolated water and treatment according to a complete randomized block design also in the *proc glm* procedure (SAS institute, 2012). The co-variate with treatment was the volume of water percolated.

Where regression analysis was performed, for example between TOC and DRP the *proc reg* procedure was used in SAS.

3. **Results and Discussion**.

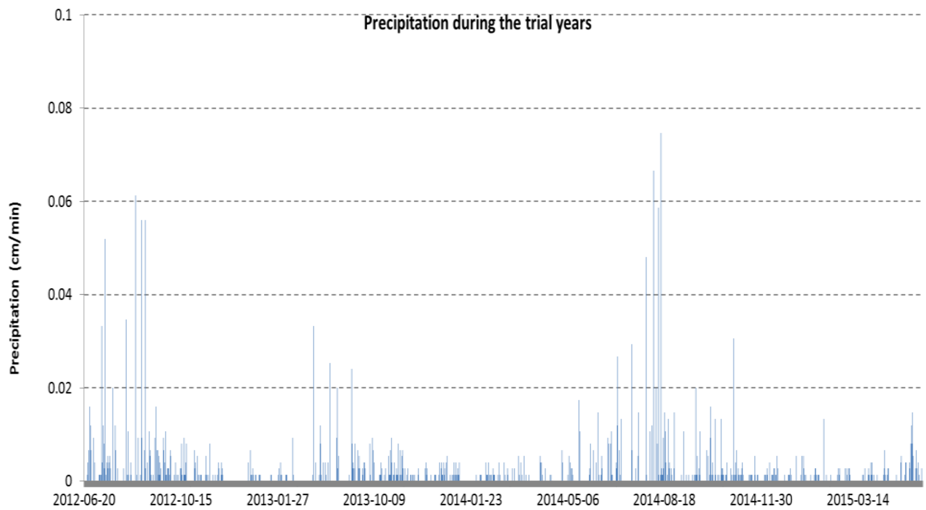

Fig.1. *Amount of rainfall expressed in cm/min (to synchronize with Kfs measurements) from June 20 2012 until May 15 2015 at the site of the experiment. This is included to compare rainfall to infiltrations rate (IR). Rainfall never exceeded even the lowest recorded values of IR so water will be portioned to infiltration.*

3.1 Infiltration

Typically the rate of infiltration will be initially large and decrease with time until reaching a quasi-steady state (qs) which reflects IR at the soils surface (Reynolds et al. 1991; Thony et al. 1991; Reynolds et al. 2002). The intention of the double ring is to provide a buffer which physically prevents divergent lateral flow under the measuring cylinder, which would otherwise occur with a single ring. The theory being, that as infiltration under the buffered area absorbs lateral water flow, only vertical flow directly under the measuring cylinder will remain (Reynolds et al. 2002).
It has been found however, that the buffered ring system does not entirely eliminate lateral flow (Thony et al. 1991; Reynolds et al. 2002) and therefore is prone to overestimate infiltration rate measurements. The lateral (horizontal) flow described can be attributable to sorptivity (S). S is directly influenced by the water holding capacity and initial water content of the soil (Philip 1957a). In a dry soil it could be expected to be an initially

large lateral flow (high S value), particularly at small time (t) (Philip 1957b). Then as time progresses, Ks rather than S, will become the more dominant driving mechanism for water movement.

The differences in infiltration rate (IR) were attributable to block (Pr > F= 0.0035) season (Pr > F = <.0001) and treatment (Pr > F = <.0001) in addition there were significant interactions between block and treatment (Pr > F = <.0001), treatment and season (Pr > F = <.0001) (Fig.2.) as well. Large spatial variability, as indicated by the significance between blocks, is consistent with previous studies investigating IR or Kfs (Messing and Jarvis 1990; Messing and Jarvis 1993; Sobieraj et al. 2004; Alleto and Coquet 2009; Kellor et al. 2012). Essentially these studies found IR (or Kfs) can vary considerably even over short distances. Whilst it is important to note that there will be differences arising in Kfs due to spatial variability it is not very informative for this experiment to discuss in detail but rather acknowledge it will diminish the chances of finding statistically significant differences between treatment effects in both the Kfs and Ks values.

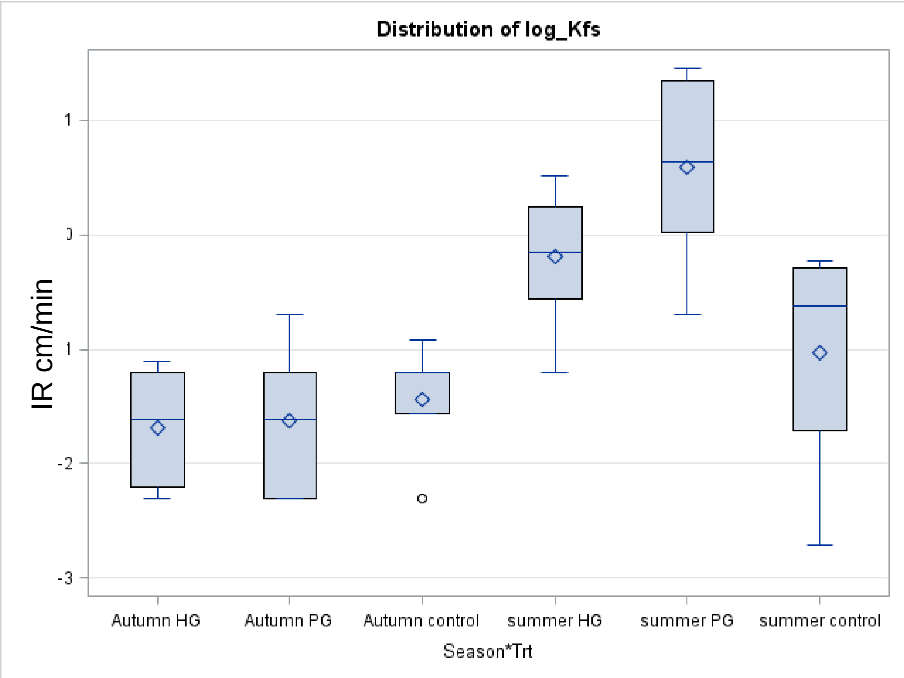

Fig.2. *Box plot showing the effects of treatment and season on IR, cm/min (log transformed). Diamonds indicate means, middle horizontal line represents median values, shaded area of boxes show 25th and 75th percentiles, blue lines represent minimum and maximum values and circles show outliers*.

Fig.2. shows IR (log-transformed) in HG, PG and control treatments in summer and autumn measurements. There were significant differences between the seasons for PG and HG (Pr > |t|= <.0001). In contrast no significant differences were observed between the control plots between summer and autumn sampling periods (Pr > |t| = 0.9722).Fig.2 illustrates that both PG and HG treatments conduct more water during the summer compared to control plots, but no significant difference was found between the two grass treatments. Nevertheless, IR was generally high for the whole field due to cracks in the summer and to the abundant earthworms in the autumn throughout the upper soil profile. Thus, surface run-off is unlikely to be an issue here as the lowest IR value recorded was 0.066 cm/min which is only exceeded 3 times in 3 years with available rainfall data (Fig.1.).

Being located on such flat topography (1% slope) surface run-off is expected to be negligible in comparison to infiltration when comparing water partitioning at the soil surface. Where water infiltration is uneven its movement is more likely to be dominated by rapid fluxes through macropores as found by Ulén and Persson (1999). Given these results it could be expected that water flow probably was dominated more by preferential flow in control plots during the summer than the PG and HG treatments. In the summer large cracks were observed in the control plots and appeared to penetrate the plough layer, while no such cracks were observed in the PG or HG plots.

3.2 Field saturated hydraulic conductivity (Kfs) with inversed auger hole method

The results of the inversed auger hole measurements where highly variable and largely uninformative with regards to any treatment effects. All values needed to be log transformed to meet the assumption of equal variance. Significant differences were attributable to block ($P \leq F = 0.0085$), depth ($P \leq F = <0.0001$) as well as interactions between both time replicate (+/- 3 days) and depth ($P \leq F = <0.0001$) and treatment and depth ($P \leq F = 0.0132$). The 10-25cm depth is where most treatment effects are expected to take place due to the action of cultivation in the control plots. At this depth however there were no significant differences between the treatments (control and HG $Pr > |t| = 0.9936$; control and PG HG $Pr > |t| = 0.9134$ using Tukey Kramer adjustment). Yet at the 10-25cm depth differences arising due to time replicate were highly significant (initial and + 3 days $Pr > |t| = <0.0001$) with the mean log Kfs value being of the + 3 day measurements being higher (+ 3 days = -0.47, initial = -1.78) . This indicates that the effect of pre-wetting has a larger effect on Kfs than treatments. As the initial measurements will saturate the soil in the immediate vicinity of the auger hole, the measurements at + 3 days are being taken from soils with a higher initial moisture content.

3.3. Turbidity after rainfall simulation on soil columns

Fig.3 which shows the distribution of turbidity between the treatments, highlighted greater turbidity values in the control percolates. However lsd means separation (Table.1) shows how only the PG and control were statistically different at the $p \leq 0.05$ level. Since the mean value of control was high, it can also be assumed that control had generally higher turbidity than HG also (1048.6 NTU control, 397.1 NTU HG). In future experiments more samples could help alleviate the problem of statistically insignificant results particularly when there is such a large numerical difference in the means of HG and control.

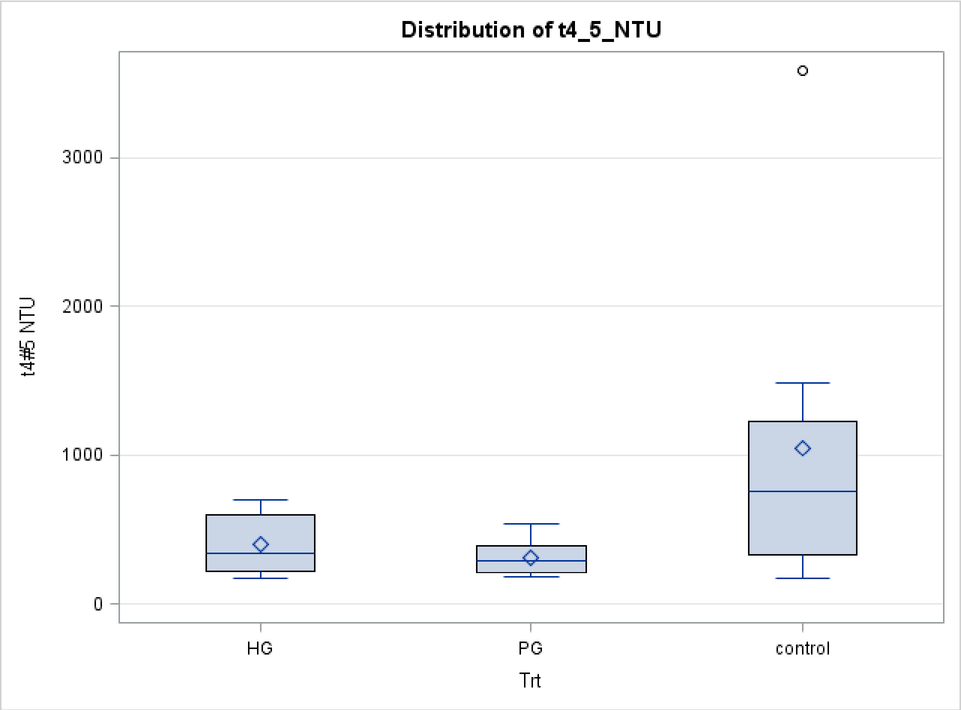

Fig.3. *Box plot represents treatment effects on turbidity (NTU) after being allowed to settle for 4.5 hours. Diamonds represent means, horizontal lines medians, shaded areas 25th and 75th percentiles, blue lines representing minimum and maximum values and circles show outliers*.

Means with the same letter are not significantly different.				
t Grouping		Mean	N	Trt
	A	1048.6	8	control
	A			
B	A	397.1	8	HG
B				
B		311.9	8	PG

Table.1. *Highlights that the control and the PG treatments for turbidity of water percolate are significantly different with the lsd means separation method, whilst HG and control treatments are most likely different, but it couldn't be shown statistically*.

Turbidity has been shown to be a good indicator of total suspended solids (TSS) (Ayoub et al. 2014; Wu et al. 2014) and suspended clay particles (Etana et al. 2009) in water samples. Previous studies have shown that increasing intensity and depth of cultivation increases turbidity in percolated water samples (Etana et al. 2009; Muukkonen et al. 2009a; Muukkonen et al. 2009b). The results presented in Fig.4 and Fig.3 are therefore consistent with those previous studies, if both HG and PG are taken as the equivalent to no-till.

Based on the rainfall simulation experiment a relationship between volume and treatment was indicated. Control cylinders that were highly impermeable appeared to have very clear percolated water collected in the PVC container. In comparison, control cylinders that seemed to be conducting water rapidly appeared to have large volumes and murky water. This appeared to indicate a relationship between the treatment effects and the soils ability to conduct water. Since turbidity seemed to be influenced by both volume of permeated water and treatment, an ANCOVA was performed between these two parameters.

There was a good fit for turbidity being a result of both the volume of water permeated and treatment at both irrigation events, R_2= 0.6587 and R_2 = 0.6538 respectively. This relationship is shown in Fig.4.

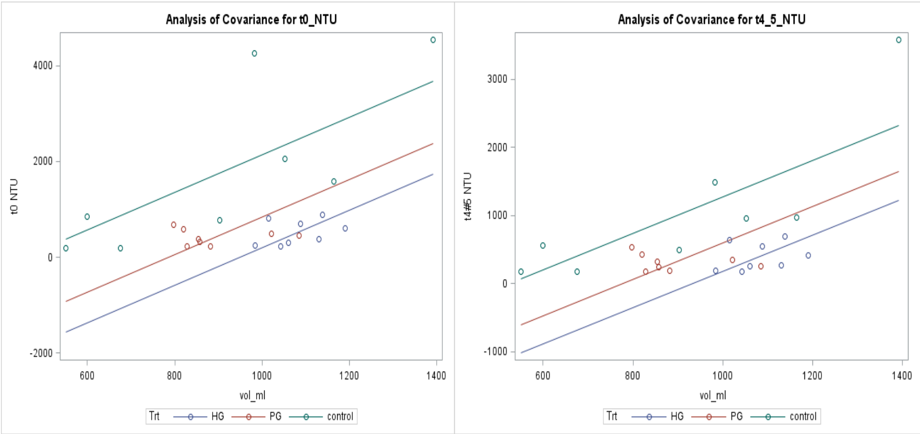

Fig. 5. *Turbidity (NTU) as a function of volume of water permeated between different treatments was shown by an ANCOVA for both turbidity measurements, t=0 (left) and t=4.5 hrs (right)*.

Fig.4 highlights how turbidity was dependent on the volume of percolated water (ml) and treatment. Where control plots were highly impermeable, as indicated by a small volume, their turbidity was also reduced greatly in the percolated water and vice versa. The most likely explanation for the good fit (R^2= 0.65 at t = 4.5 hrs) is that more vertical erosion with the downward movement of water leading to more TSS in the permeated water. In contrast, where control plots have been highly impermeable, little water passed through few continuous pores without carrying soil particles.

3.4 Particulate & dissolved reactive phosphorus in water percolate

Differences in total P between all treatments were found to be insignificant ($P \leq F = 0.42$). So instead the concentrations of particulate (PP) and dissolved reactive (DRP) in the water percolates were investigated. The cultivated plots (control) had a trend to for a higher PP than both the grassed treatments. However statistical significance was probably not found due to the wide range of values for the control samples (0.324 mg/l - 2.023 mg/l PP). The results for PP are consistent with measurements of turbidity (Fig.3.) where the control samples had a wide range of turbidity (NTU) values.

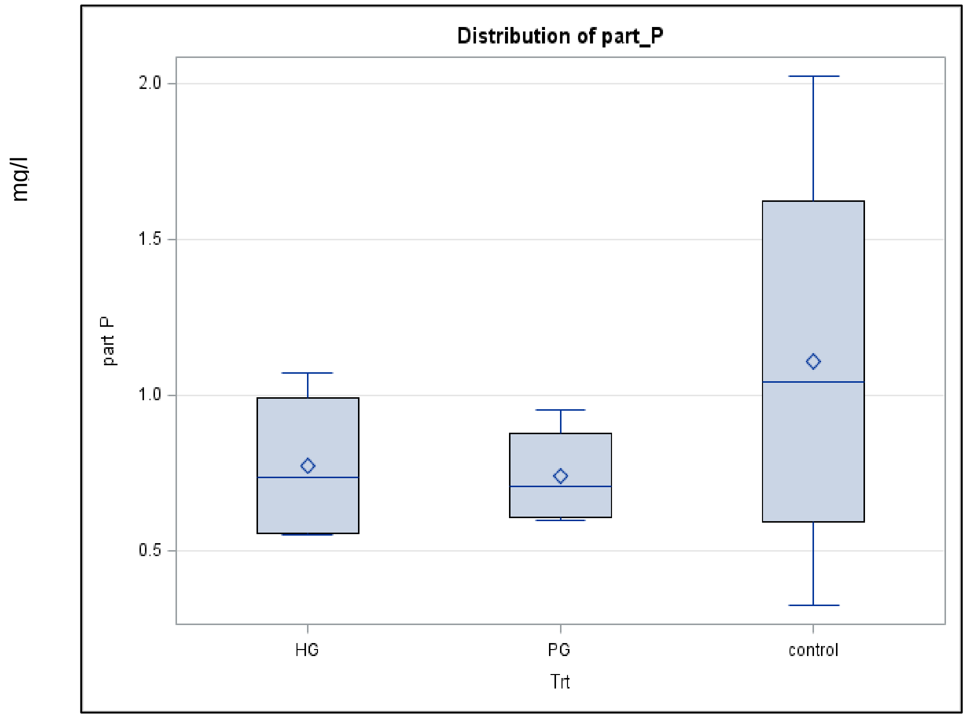

Fig.5. *Box plot representing concentration of PP (mg/l) in the permeated water for the three treatments. Diamonds represent the means, horizontal lines the medians, shaded boxes the 25th and 75th percentiles and blue lines the minimum and maximum values. Whilst the control plot has a higher mean it also has a wide spread of values as shown by both the 25th and 75th percentiles in addition to the blue lines. Due to this the differences between the treatments were found to be insignificant (Pr> F =0.4065)*.

Where overland flow is measured it can be expected that PP values will be higher in the absence of a grassed buffer zone from an arable field (Dillaha et al. 1988; Vought et al. 1994; Syversen 2005; Borin et al. 2005; Harper et al. 2009; Uusi-Kämppä & Jauhiainen 2010). In the present study also, cultivated areas demonstrated a higher PP value in the percolated water. This would indicate that the absence of a grassed buffer strip would deliver more PP to the groundwater or lose more to water bodies via drainage tiles. However at this scale a rainfall simulation experiment can only evaluate deliverance values of PP to the tile drains from the upper soil layer (0-20 cm) within the buffer zones. To simulate the filtering efficiency of buffer strips with regard to cultivated soil upstream we would need a more complex study, for example by applying loose soil material to the soil columns before rainfall simulation.

For PP leaching through the top-soil comparisons have been made between agronomic practices such as direct drilling, depth of tillage and growing annual catch crops (Ulén et al. 2010; Liu et al. 2014;), rather than a comparison of perennial grass and conventional tillage. The grassed treatments are taken as the equivalent of the treatments which receive the least mechanical disturbance in this study. Autumn ploughing disturbs the soil more, and then the loose soil is subjected to large quantities of water during snow melt periods, which are common between autumn and the spring. This increases PP leaching during these periods. Uusi-Kämppä & Jauhiainen (2010) found an adjacent field under conventional tillage contributed more PP to surface run-off than if the field was under pasture or no-till. It is therefore reasonable to assume that where infiltration capacity is high, cultivation will also contribute more PP to the tile drain.

Previous studies have found turbidity to be a reliable indicator of PP in percolated water samples (Etana et al. 2009; Marttila & Kløve 2012). This has also been extended to give a good correlation in water samples between PP and NTU (Etana et al. 2009; Marttila & Kløve 2012). Etana et al. 2009 found good correlations between turbidity and clay content as well as between turbidity and PP after investigating 5 Swedish clay soils. When investigating nutrient losses from Finnish peatland forest Marttila & Kløve (2012) also found good correlations between total P, total N, PP and turbidity, providing large fractions of these nutrients occurred in the particulate form. However in the same study, the reliability of turbidity measurements to indicate nutrient concentrations decreased markedly as the portion of dissolved nutrients increased, such as during snowmelt periods. Pavanelli & Bigi (2005) also found turbidity to be a good indicator of TSS measured by direct measurements, however this reliability decreased with higher concentrations of TSS and an increased number of dilutions. Pavanelli & Bigi (2005) concluded that higher component of coarser solids in suspension likely decreases the reliability of turbiditity measurements.

As Fig.3 shows long-term measurements in the current field trial showed that PP was highly correlated to turbidity in sub surface drainage (Etana et al. 2015). However this was not the case in this study with rain simulation where regression analysis between PP and NTU at t= 4.5 hrs had a poor coefficient of determination (0.0071). The poor relationship could be due to the contribution of particulate organic matter which contained a high portion of PP (Fig.7). The correlation between TOC and turbidity was also poor but better than the relationship between turbidity and PP (R_2= 0.119). Freezing and thawing cycles appear to create a unique scenario where phosphorus is in the form of particulate organic matter, or large portions of P are in the form of DRP (Fig.7.).

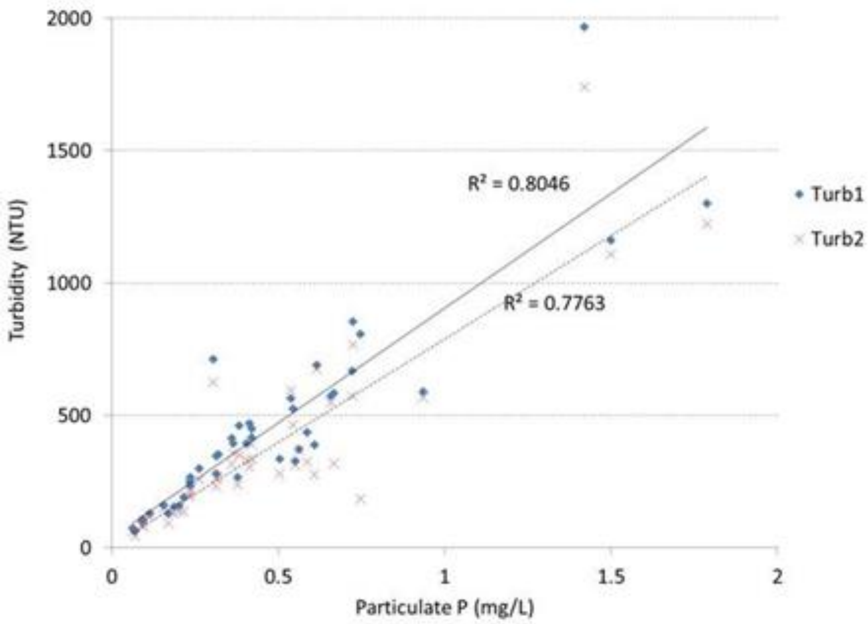

Fig.6. The diagram *from Etana et al. (2015), is based on long-term measurements in the current field trial. It demonstrates that PP was highly correlated to turbidity in water collected from sub-surface drainage.*

Grassed buffer strips tend to deliver more DRP both in surface and tile drains compared to the same site without a buffer strip (Dillaha et al. 1988; Dorioz et al. 2006; Hoffman et al. 2009; Uusi-Kämppä & Jauhiainen 2010; Uusi-Kämppä et al. 2011;). Fig.7. highlights a significantly ($p \leq 0.05$) more amount of DRP in both PG and HG treatments compared to the control treatment. Mean concentrations for both the grassed buffer treatments are >10 orders of magnitude higher than for the control after the rainfall simulation experiment. This runs quite contrary to the aim of grassed buffer strips which aim to reduce P loads to surface water, especially as DRP is in a far more biologically active form in waterways (Hoffmann et al. 2009).

Leaching of DRP is also usually highest during springtime run-off after snowmelt in cold climate (Uusi-Kämppä et al. 2011), which precedes optimal algal growing conditions within waterways in Nordic conditions (Inkala et al. 1997). It is certainly one of the most pressing concerns for constructing grassed buffer strips for nutrient removal.

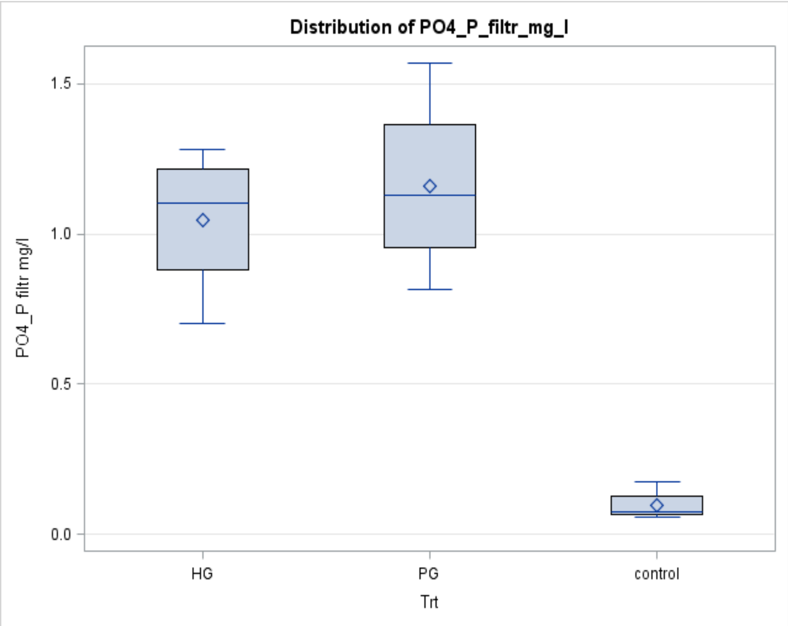

Fig.7 *Box plot highlighting differences in concentrations of dissolved reactive phosphorus DRP (PO-4 mg/l) in the water permeate after the simulated rainfall experiment. Diamonds represent means, horizontal lines medians, shaded area 25th and 75th percentiles and blue lines minimum and maximum values.*

Means with the same letter are not significantly different.			
t Grouping	Mean	N	Trt
A	1.161	4	PG
A			
A	1.0478	4	HG
B	0.0955	4	control

Table.2. *shows means for both HG and PG treatments are >10 X higher compared to control treatments, and that the mean value for the control is significantly different at the $p \leq 0.05$ level compared to the two grassed treatments.*

The elevated levels of DRP in the water from vegetative grassed buffer strips compared to the control was could be due to the rapid degradation of above ground biomass during repeated freezing and thawing cycles.Fig.8 confirms this assertion showing a significantly elevated TOC in PG treatments compared to control. Higher amounts of decaying vegetation from freezing and thawing cycles in the PG plots lead to increased TOC, which contained both particulate and dissolved reactive P. Fig.9. shows a linear regression (R_2= 0.448) indicating that as TOC increases so does DRP. Whilst the mean value of TOC in PG was higher than HG, the difference was not statistically significant. The little difference there was however lends support to the theory that cutting and removing grass in a grassed buffer strip in the autumn prior to breakdown by freezing and thawing could be a good way to reduce leaching of DRP from grassed buffer strips (Uusi-Kämppä and Jauhiainen 2010).

Previous studies have concluded that repeated freezing and thawing can release considerable amounts of DRP from decaying plant matter in vegetated buffer strips (Uusi-Kämppä et al. 2011), various common Swedish catch crops (Liu et al. 2014) and annual ryegrass catch crops (Bechmann et al. 2005). In addition organic matter breakdown by mechanical disturbance can also liberate increased amounts of DRP in no-till soils with stubble remaining when compared to the cultivated ones (Muukkonen et al. 2009a; Muukkonen et al. 2009b).The results of this experiment are therefore consistent with these previous findings that where there is a large amount of organic matter on the soils surface there is also highly likely to be an increased leaching of DRP.

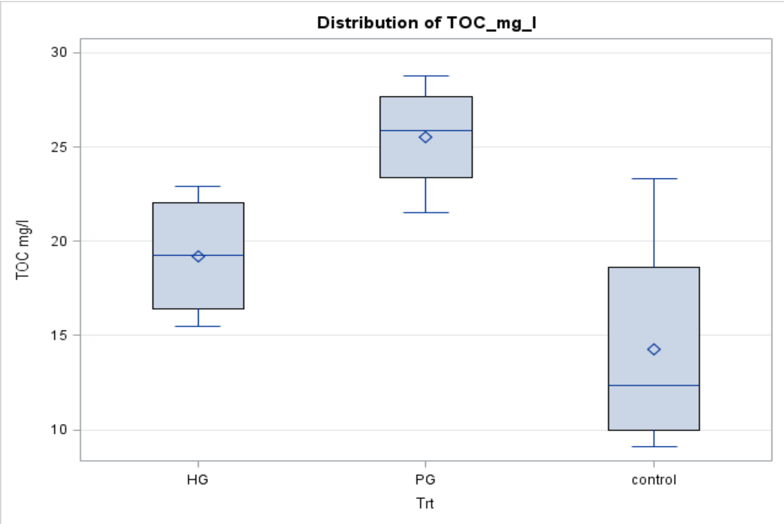

Fig.8. *Box plot which shows differences in TOC concentrations (mg/l) between the different treatments. Diamonds represent means, horizontal lines medians, shaded areas 25th and 75th percentiles and blue lines minimum and maximum values*.

Means with the same letter are not significantly different.				
t Grouping		Mean	N	Trt
	A	25.525	4	PG
	A			
B	A	19.225	4	HG
B				
B		14.275	4	control

Table.3. *Means separation using lsd grouping showing that there is a significant difference between the PG and control treatments*.

Table.3. shows that by cutting and removing grass significantly reduces the amount of TOC in the water percolate and will therefore also reduce the amount of DRP as shown by the relationship in Fig.9

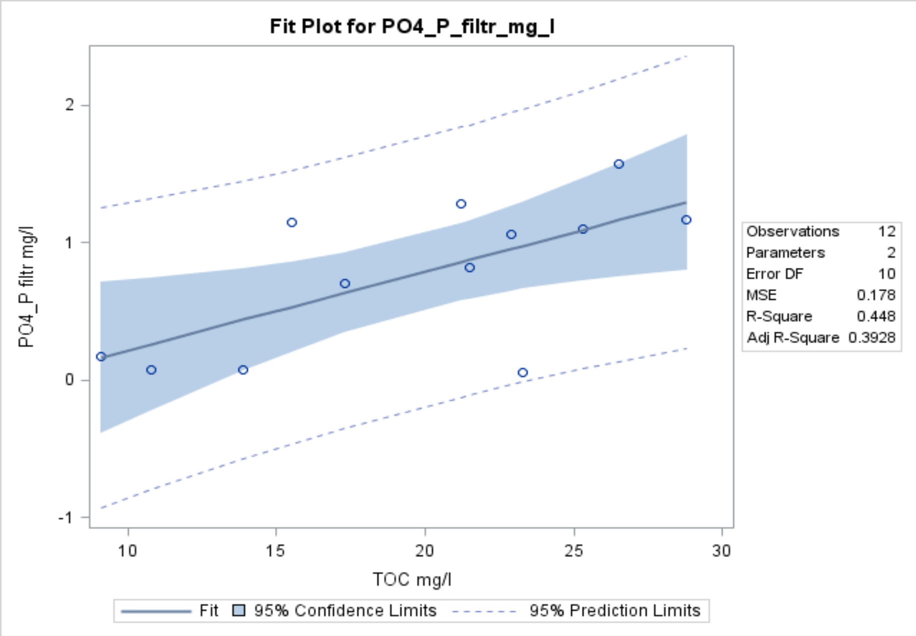

Fig.9. *Relationship between TOC (mg/l) and DRP (PO-4 mg/l) after simulated rainfall experiment, taking DRP as the response variable. Given the number of observations a R_2 value of 0.448 gave a good fit for the model*.

3.5 Saturated hydraulic conductivity and porosity in small soil cores

The dominant reason for differences in the laboratory Ks experiment arose due to block (Pr > F = 0.006) indicating there was wide spatial variability as indicated in Fig.12. Another possible reason for non significant differences between treatments could be the large number of large earthworms throughout the trial site which may have altered macroporosity. It was stated by Messing and Jarvis (1990) that earthworm hole channels can act as macropores which can dominate water movement through sections of the soil profile. Spatial variability in earthworm abundance has been observed previously in tile drained clay soils (Alakukku et al. 2010) and in combination with an inherent spatial variability in the soil has contributed to a wide variation in this study.

The results of the laboratory measurements highlight the large variation that can be encountered when measuring Ks. It can't be known exactly what has led to such a wide variance in Ks values (earthworms, inherent spatial variability) but it does highlight that more replicates should be taken from a particular depth if statistical significance is likely to be obtained.

Fig.10.illustrates that as the portion (%) of macropores >30µm increases so does Ks at the 0-5cm measurements. Due to restrictions of time and lab space (for the drainage boxes) only 6 samples at this depth could be analyzed, so treatment effects were not analyzed. Despite this a reasonable fit (R_2 = 0.5778) could be fitted to a simple linear regression model (Fig.10.). This would suggest that in the top 5 cm of the soil profile, management practices affecting macroporosity also affect Ks. Messing and Jarvis (1990) found a similar relationship on an Ultuna clay soil where field saturated hydraulic conductivity (Kfs) could be expressed as a function of macroporosity.

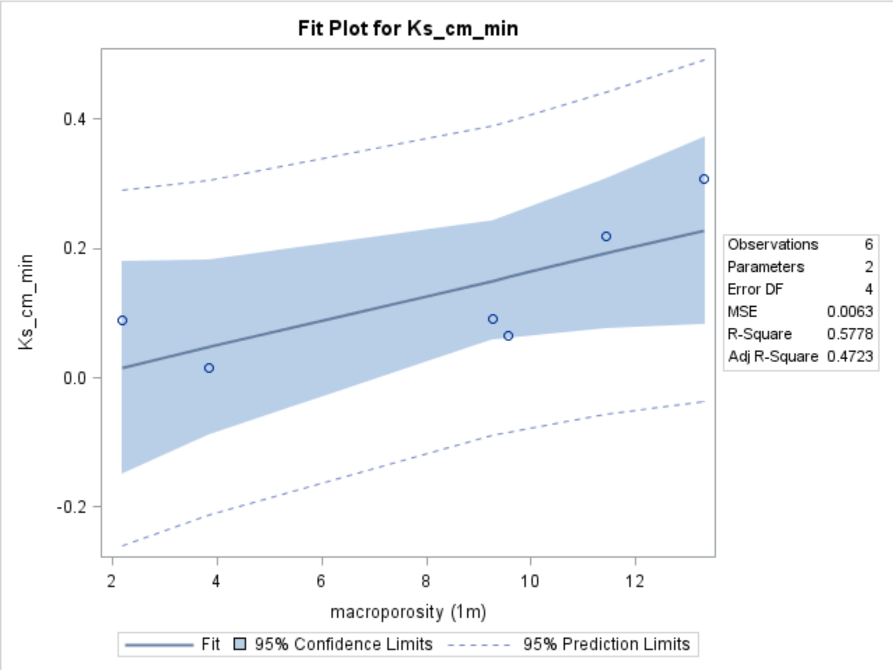

Fig.10. *Ks (cm/min) as a function of macroporosity (% of pores >30µm) at 1 m suction*.

4. **Conclusion**.

Cutting and removing vegetation from grass buffer strips may reduce leaching of dissolved reactive phosphorus from above ground biomass. However the levels of dissolved reactive phosphorus in harvested grass was still greater than that in the control so other complementary or alternative measures (i.e. chemical amendments to grassed plots) are necessary to abate eutrophication. It is likely that leaching of solids may occur in rapid fluxed through macropores. This was indicated by the relationship between turbidity and the volume of water percolate collected after the rainfall simulation.
Focusing more on the top 15 centimeters of the soil profile, where most of the pronounced changes infiltration where shown to happen, and then

linking this to particulate or dissolved reactive phosphorus leaching patterns would be recommended for future studies.

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6. Acknowledgements

I would sincerely like to thank Ararso Etana for his supervision throughout the entirety of the project.

In addition I would like to thank Christina Öhman, Anna Maria Mingot Soriano and Liselott Evasdotter for their assistance in the lab as well as Richard Childs for help in the field.