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A Soil Additive to Reduce Runoff and Pollutants from Dairy Heavy Use Areas

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A Soil Additive to Reduce Runoff and Pollutants from Dairy Heavy Use Areas

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INTRODUCTION

Agricultural pollution from farming and animal feeding operations is the primary source of stream pollution in the United States. Nutrient, sediment and pathogen pollution are three of the top seven causes of stream impairments (USEPA, 2009). All of these pollutants are associated with animal feeding operations. Different treatment methods can be used to reduce pollutants in runoff from heavy use areas such as constructed wetlands, vegetative filters and soil amendments. Possible soil amendments for heavy use areas include coal ash, polyacrylamide (PAM) and polysaccharides (PSD). Coal ash is the combustion by-product of burning coal (Graber et al., 2006) but can cause heavy metal contamination of surface waters above drinking and surface water quality limits when used as a soil additive (Carlson and Adriano, 1993; NDDH and EERC, 2003). Polyacrylamides have been shown to reduce suspended solids, nutrients and pathogen pollution in irrigation runoff but can contain residual acrylamide molecules which are a known neurotoxin (Seybold, 1994). Due to the potential release of heavy metals and toxins from coal ash and PAMs, these soil additives are not safe to use in the presence of dairy cows.

Polysaccharide soil additives are an alternative to apply in heavy use areas when dairy cows are present. Several PSD soil additives have been tested in agricultural settings but not in heavy use areas. These additives include microalgal, cellulose/starch xanthate, guar, and psyllium. Due to the prevalence of stream impairments and the variety of PSD soil additives available for treatment of heavy use areas, the objective of this paper was to review PSD soil additives for potential use on heavy use areas and their effects on runoff water quality. This review will also determine the research needs associated with soil additives for heavy use areas.

CONFINED ANIMAL FEEDING OPERATIONS

An animal feeding operation (AFO) is a facility where the following two conditions are met (USEPA, 2012):

(i) "Animals (non-aquatic) have been, are, or will be stabled or confined and fed or maintained for a total of 45 days or more in a 12 month period"

(ii) "Crops, vegetation, forage growth, or post-harvest residues are not sustained in the normal growing season over any portion of the facility"

Part (i) of the definition states that an animal needs to be kept on the lot for a minimum for 45 non-consecutive days in any 12 month timespan. If an animal is confined for any part of a day, no matter how long, the animal meets 1 day of the 45 day requirement. Part (ii) of the definition is used to distinguish between grazing/pasture land and confined areas (USEPA, 2012).

A large confined animal feeding operation (CAFO) has a dairy cow population greater than 700. An AFO is considered a medium CAFO when there are 200-699 mature dairy cows and one of the following two conditions are met (USEPA, 2012):

A. "Pollutants are discharged into waters of the United States through a manmade ditch, flushing system, or other similar manmade devices"
B. "Pollutants are discharged directly into waters of the United States which originate outside the facility and pass over, across or through the facility or otherwise come into direct contact with the animals confined in the operation"

A small AFO can be designated a CAFO when it is determined that the AFO contributes pollutants to the surface water via the methods described for medium CAFOs (A and B above) (USEPA, 2012).

AFOs designated as CAFOs are required to meet a no-discharge standard. The CAFO may not release pollutants into the environment except when precipitation greater than the 25-year, 24-hour storm causes an overflow. Typically CAFOs meet water quality criteria using the best practicable control technology (BPT) currently available. However, the National Pollution Discharge Elimination System (NPDES) permit allows for CAFOs to request for voluntary performance standards. The voluntary performance standard is a second way for the CAFO to meet or exceed the no-discharge and water quality standards through implementing site specific technologies other than the BPT (Federal Register, 2012).

HEAVY USE AREAS

A heavy use area is agricultural land that is frequently used by animals and machinery. According to NRCS, heavy use area protection is defined as the stabilization of heavy use areas by using vegetative cover, surface materials and/or installing needed structures (NRCS, 2010). Heavy use areas, which are also called loafing areas or sacrifice lots, can contribute significant pollutants to stream through runoff due to high concentrations of manure and urine on exposed soil (Younos et al., 1998). Examples of heavy use areas include feedlots, confinement houses, cowyards and animal walkways.

2012 CAFO RULE

In 2012 the USEPA implemented two changes to the CAFO rule. The first change was to remove the parts of the law which were vacated by the 5th Circuit Court of Appeal. The USEPA removed the requirement for a CAFO which "proposes to discharge" to get a permit under the NPDES (Federal Register, 2012a). A CAFO is designated "proposed to discharge" when an assessment determines that the CAFO is "designed, constructed, operated or maintained such that a discharge may occur" (USEPA, 2010). In order for a CAFO to discharge into waters of the United States a NPDES permit is required (USEPA, 2012). A NPDES permit includes effluent

limitations and standards, monitoring and reporting requirements, record keeping requirements, special conditions and standard conditions (USEPA, 2012).

The second amendment made to the CAFO Rule in 2012 was to withdraw a proposed rule change in how information is reported. CAFOs do not have to directly submit information to the USEPA. Instead the USEPA will continue to collect information via other sources such as NPDES programs at the federal and state levels to make sure CAFOs are meeting the water quality standards (Federal Register, 2012b).

CHARACTERISTICS OF POLLUTANTS ASSOCIATED WITH CONFINED ANIMAL FEEDING OPERATIONS

Typical pollutants found in animal waste runoff include pathogens (e.g. *Escherichia coli)*, sediment, nitrogen (N) and phosphorus (P) (USDA, 2009; USEPA, 2001). The pollutants and their relation to CAFOs are described below.

Escherichia coli

Microbial water quality is tested primarily using indicator bacteria such as fecal coliforms, total coliforms, and fecal streptococci (Baxter-Potter and Gilliland, 1988). *Escherichia coli* (*E. coli*) is the most common bacteria found in the intestines of warm-blooded animals and can occur with a density as high as $1x10^9$ organisms per gram of wet weight animal feces (Mawdsley et al., 1995).

Fecal pollution can cause sickness to humans from pathogens. The use of *E. coli* as indicator bacteria only suggests the possible presence of fecal pathogens. Even though *E. coli* only indicates the possibility of fecal contamination it is still used because direct tests for fecal pathogens are too expensive and time-consuming (USEPA, 2001). The significance of finding fecal coliforms, specifically *E. coli*, in surface water is that it indicates possible fecal

contamination by a warm blooded animal. In one study, 99.9% of the total coliforms found in fecal matter were *E.coli*, and the remaining 0.1% were *Enterobacter/Citrobacter* (Dufour, 1977).

Cattle manure contains between 4.4 and 6.8 x10^7 enterobacteria per gram of dry weight, of which 90% can be *E. coli* (Hrubant et al.1972). Fecal coliform pollution has been linked to the presence of grazing cattle. A study monitoring a stream adjacent to a pasture found that fecal coliforms increased soon after the introduction of cattle to a pasture. If the pasture was not managed well for grazing habits the fecal coliform count remained high in the stream up to three months after removal of the cattle (Baxter-Potter and Gilliland, 1988). During storm events fecal pollution in a partially grazed watershed rose to at least 50 times the background level in an adjacent stream (Kunkle, 1970).

Escherichia coli Transport

Escherichia coli can enter runoff from soil pores and water films as a free moving microorganism, by being attached to waste or a soil particle, or by detaching from the soil surface and transported as free moving microorganisms (Tyrrel and Quinton, 2003). *Escherichia coli* is most likely to be attached to organic or clay soils because microbes are attracted to negatively charged particles (Mawdsley et al., 1995). Muirhead et al. (2006) found approximately 32% of all the *E. coli* cells were attached to soil particles in runoff, of which the majority were attached to soil particles less than 2µm. When *E.coli* was attached to soil particles greater than 45 µm, transport of *E. coli* in overland flow was reduced. Survival of *E. coli* in water is dependent on the temperature of the water, nutrient levels, light, predation and competition with other bacteria (Mawdsley et al., 1995). Survival of *E.coli* increases as temperature is lowered between 4 and 37º C (Flint, 1987). Freezing of *E. coli* at -15℃ in a freezer for 2 days did not inactivate all the *E. coli* and some viable bacteria still remained in the

sample (Gao et al., 2007). Solar radiation can be fatal to exposed *E. coli* (Sundaravadivel and Vigneswaran, 2001). Excess nutrients can increase *E. coli* survival time in surface water (Lim and Flint, 1989).

Nitrogen

Nitrogen contamination in water can be linked to dairy cattle CAFOs because of the release of N from manure (Sudduth and Loveless, 2004). Hutson et al. (1998) found 20% of the total nitrogen (TN) in manure from a lactating dairy cow came from urea, which was equal to the amount of ammonia nitrogen content. The remaining 60% of the TN content in manure was organic nitrogen. High N concentrations in surface waters can help lead to eutrophication, but is not as important to eutrophication as P in freshwater (Driscoll et al., 2003).

Phosphorus

Phosphorus contamination in runoff can be directly linked to the presence of cattle and can be found in inorganic, organic, dissolved or solid forms (Hart et al., 2004; USEPA, 2001). Greater than 70% of the P found in manure is in the organic form (Spellman and Whiting, 2010). Sharpley and Syers (1976) found runoff concentrations of dissolved P increased immediately after cattle grazing and decreased to background levels after 20 days. Cattle can disturb and compact soil by walking on it, which in turn decreases infiltration and increases soil particles and particulate P in runoff (Hart et al., 2004). Phosphorus concentrations in water increase with a rise in sediment because P has the ability to adsorb onto soil particles (Bosch et al., 2006). Areas such as cow paths and barnyards can contribute to P pollution (Hively et al., 2005).

Suspended Solids

Suspended solids are inorganic or organic material (Inoue et al., 2009) held in suspension through the movement of water (Bilotta and Brazier, 2008). Suspended solids concentration

(SSC) in runoff is mostly determined by soil texture, infiltration rate, rainfall intensity, slope and ground cover (Hively et al., 2005). An increase in SSC can cause changes in the physical, chemical and biological properties of water. Chemical changes occur when nutrients and other pollutants attach to adsorption sites on sediment in runoff (Bilotta and Brazier, 2008; Inoue et al., 2009). Similarly, Inoue et al. (2009) found that adsorption onto suspended sediment was effective at transporting P and pathogens in runoff.

Runoff is considered the primary method for SSC transport from land to surface water. There are three stages for sediment movement in runoff: detachment, transport and deposition. Suspended solids in runoff negatively affect surface water bodies through increased sedimentation and turbidity (Tyrrel and Quinton, 2003).

WATER QUALITY CHARACTERISTICS OF CONFINED ANIMAL FEEDING OPERATIONS

Studies in the United States and Canada have shown several pollutants are found in feedlot runoff and their concentrations have ranged widely from site to site (Table 1). Various experimental designs were used in the studies reported in Table 1. For example, Edwards et al (1972) reported concentrations from a single barnlot watershed. Miller et al. (2004) completed a study on a research feedlot which contained 32 identical pens with approximately 15 cattle in each pen. And Miner et al. (1966) constructed two feedlots of identical size which were either surfaced with concrete or left as soil. Both Kreis (1972) and Manges (1975) completed studies on commercial feedlots which totaled 12,000 and 33,000 cattle, respectively. Even with the variety of study designs, all reported maximum pollutant concentrations higher than untreated domestic wastewater.

Table 1: Range of Cattle Feedlot Runoff Concentrations in the USA and Canada

Type	Feedlots/ Pens/ Plots[8](#)	Location	NO_3-N (mg/L)	NH_3-N (mg/L)	TKN (mg/L)	TN (mg/L)	PO_4 (mg/L)	TP (mg/L)	TSS or SSC (mg/L)	Total Solids (mg/L)	Fecal Coliform (10^3CFU/100 ml)	Source
Barnlot watershed	1	OH	1.1-2.1	-	-	21.6-38.7	-	3.2-7.2	-	-	-	Edwards et al. (1972)[1]
Surfaced clay feedlot	32	AB	<0.04 23.5	0.7-85.9	-	19.2-173	-	2.1-61.2	-	-	0.1-100000	Miller et al. (2004)[2]
Unsurfaced feedlot	1	KS	0.1-6.0	1-62	50-540	-	15-45[3]	-	1500-10500	-	3300-542000	Miner et al. (1966)
Concrete feedlot	1	KS	0.1-11	1.3-139	94-1000	-	20- 80[2]	-	1400-12000	-	3300-790000	Miner et al. (1966)
Feedlot	>6	NE	0.0-217	-	-	11-8590	-	4-5200	-	-	-	Gilbertson (1975)
Dirt feedlot	2	TX	0.0-163	2-85	8-279	-	-	-	-	-	-	Wells et al. (1970)[4]
Concrete feedlot	4	TX	0.0-1270	33-774	68-1315	-	-	-	-	-	-	Wells et al. (1970)[4]
Surfaced feedlot	6	TX	0.05-2.3	4-173	-	-	21-223	-	745-17202	3110-28882	200-17000	Kreis (1972)
Feedlot	26	KS	0.0-48	-	-	85-1580	-	9-482	-	214-19252	-	Manges et al. (1975)
Feedlot	8	TX, CO, NE, KS, SD	-	-	-	3000-17500	-	47-300	-	2986-17500	-	Clark et al. (1975)[5]
Feedlot	1	KS	-	-	-	165-1580	-	9-242	-	0.84-1.92*	-	Manges et al. (1971)
Feedlot	1	SD	-	-	225.64	-	62.8	-	-	-	-	Madden and Dornbush (1971)[6]
Feedlot	6	NE	0.0-17	-	-	65-555	-	13.9-46.6	-	0.24-1.74*	-	Gilbertson et al. (1971)
Unsurfaced lot	12	WI	0.1-5.6	-	-	0.8-86.3	-	0.2-39.6	-	-		Vadas and Powell (2013)[7]
Sacrifice lot: dairy	1	VA	0.88-8.08	-	3.50-44.08	3.52-46.15	0.13-3.27	0.94-23.00	40-8390	-	-	Younos et al. (1998)

*Percent total solids
1: Values represent range of mean annual concentrations over 3 year period
2: *E. coli* concentrations
3: 70% Limits
4: Combined concentration range from two different feeds for the cattle from average values per rainstorm; TKN calculated as the sum of organic N and ammonia N per storm
5. Review article, reported minimum/maximum of average concentrations from various feedlots throughout United States
6: Snowmelt and runoff; mean concentrations calculated in lbs pollutant/in runoff, concentrations reported from site #15 in study
7: Concentrations are the yearly storm average over 4 year period
8: Number reported is the greatest number of feedlots, pens or runoff plots used in the study design

Nitrate (NO_3) concentrations have ranged from 0 to 1270 mg L^{-1}, while TN and total Kjeldahl nitrogen (TKN) have ranged from 4 to 8,590 mg L^{-1} and 8 to 1,315 mg L^{-1}, respectively. TKN concentrations in feedlot runoff follow COD concentrations, where an increase in COD (organic matter) would result in an increase in TKN (Miner et al., 1966). Concentrations of TKN and NO_3 were higher in runoff from concrete or surfaced feedlots than unsurfaced ones. Nitrate concentrations were lower than 100 mg L^{-1} in all studies except for two. A study by Gilbertson et al. (1975) reported a maximum NO_3 concentration in runoff of 217 mg L^{-1} but the mean concentration of NO_3 was only 10 mg L^{-1}. And Wells et al. (1970) reported NO_3 concentrations as high as 1,270 and 163 mg L^{-1} in runoff from surfaced and unsurfaced feedlots, respectively. In general, NO_3 concentrations from feedlots are less than 10 mg L^{-1} due to anaerobic conditions typically found in feedlots. However under aerobic conditions, NO_3 concentrations will increase due to nitrification (Miner et al. 1966).

Total phosphorus (TP) concentrations in cattle feedlot runoff have ranged from 4 to 5,200 mg L^{-1}. The maximum concentration of TP in runoff reported by Gilbertson et al. (1975) was 5,200 mg L^{-1} but the mean was 300 mg L^{-1}. Phosphate concentrations in feedlot runoff also have increased with COD concentrations and PO_4 concentrations were found to be higher in runoff from concrete feedlots compared to unsurfaced feedlots (Miner et al. 1966).

Suspended solids concentration or TSS in feedlot runoff ranged from 745 to 12,000 mg L^{-1}. Miner et al. (1966) reported that concentrations of SSC in runoff were greater from the surfaced feedlot than the unsurfaced feedlot unless samples collected were preceded by a long period of light rain. Concentrations of SSC in runoff observed by Kreis (1972) were the highest reported (Table 1) (mean = 5,900 mg L^{-1}), and were measured from 6 feedlot pens with 125 cattle each constructed on bedrock and covered with gravel. Feedlot pens studied by Kreis (1972) contained

several inches of manure cover throughout the study period. Water retention characteristics of manure on the plots may have reduced the runoff amount. However when manure becomes moist it becomes more easily transported by runoff (Miner et al., 1966).

Bacterial concentrations in runoff ranged from 100 to 790 million CFU 100 ml^{-1} (Table 1). The highest concentrations of bacteria were measured in the summer months, partly due to manure being more soluble at warmer temperatures. Bacteria concentrations in runoff also increased with an increase in organic matter (Miner et al. 1966).

Younos et al. (1998) was the only study to report concentrations of pollutants in runoff from feedlots with dairy cows and not beef cattle. The range in observed concentrations were lower for dairy cows than beef cattle. Clark et al. (1975) reported concentrations from eight different beef cattle feedlots and found that the differences in concentrations for nutrients and SSC could be attributed to rainfall intensity, stocking rate and time between runoff events.

Maximum concentrations of TN, TP, and SSC summarized in Table 1 exceeded pollutant levels found in untreated domestic wastewater. TN, TP and SSC concentrations ranged from 10.7 to 36.8, 4.8 to 15, and 52 to 420 mg L^{-1}, respectively (Thomas et al., 1974). Maximum *E. coli* concentrations exceeded an average concentration of 1 million CFU 100 mL^{-1} in untreated municipal wastewater (Kern et al., 2000).

Concentrations of sediment and pollutants adsorbed onto sediment particles in runoff are affected by rainfall characteristics, topography, soil erodibility, soil cover, and land use (Younos et al., 1998). Excess rainfall and erodible soil increases SSC in runoff therefore increasing pollutant concentrations as well. Topography with steep slopes can also increase soil erodability due to an increase in surface runoff velocity. An increase in vegetative cover can increase soil stability and reduce runoff.

EFFECTS OF CONFINED ANIMAL FEEDING OPERATIONS ON NUTRIENT MASS BALANCES

The accumulation of nutrients in CAFOs is a leading factor in the potential pollution risk to water resources from feedlot runoff (Koelsch and Lesoing, 1999). Importing animal feed adds excess nutrients into the nutrient cycle (Mallin and Cahoon, 2003; Zhang et al. 2006).

Compared to pasture farming, CAFOs create a smaller environment where waste can accumulate (Jongbloed and Lenis, 1998). Release of pollutants in manure and urine from dairy cattle is exacerbated by being in a confined space. Cows produce 86 lbs of fecal material per 1,000 lbs animal unit (AU) per day whereas humans were found to have a median rate of 0.23 lbs $person^{-1}$ day^{-1} (Cummings et al., 1992). Thus a feedlot of 700 1000 lb AUs is equivalent to a city of 261,740 people. Associated with that fecal material are several pollutants (Table 2). A Great Lakes area study found that beef cattle and dairy operations had the greatest impact on water quality compared to all other livestock operations (Sonzogni et al.1980).

Klausner (1993) analyzed N and P mass balances for dairy farms in New York. Mass balances for N were conducted on four farms with cow populations of 45, 85,120 and 1,300. Phosphorus mass balances were conducted on three farms with populations of 45, 85 and 120 cows. Nutrient inputs included purchased feed, purchased fertilizer, legume N fixation and rainfall. Outputs and losses included the export of milk, meat or crops. The annual N mass balance (Table 3) for the dairy farm that had the largest population (1,300 cows) indicated 68% of the N inputs were unaccounted for in the outputs. The dairy farms with populations of 45, 85 and 120 cows had 64, 64 and 76%, respectively, of nitrogen unaccounted for in the mass balance. The P nutrient balance for the dairy farm with 120 cows (Table 4) indicated approximately 81%

Table 2: Characteristics of manure from dairy cattle per 1,000 lb. live dairy cow mass per day (ASAE, 2003).

Pollutant	Unit	Mean	Standard Deviation
Total Manure	lbs.	86	17
Urine	lbs.	26	4.3
Total Kjeldahl Nitrogen	lbs.	0.45	0.096
Ammonia Nitrogen	lbs.	0.079	0.083
Total Phosphorus	lbs.	0.094	0.024
Total Solids	lbs.	12	2.7
Fecal Coliform Bacteria	Colonies ft^{-3}	$5.19x10^{10}$	$9.05\ x10^{10}$

Table 3: The mass balance of N on one dairy farm with 1,300 cows (Klausner, 1993).

Inputs	Nitrogen (tons $year^{-1}$)	Percent of Total Input
Purchased fertilizer	9.8	4
Purchased feed	205.0	87
Nitrogen Fixation by Legumes	21.3	9
Sum	236.1	100
Outputs		
Milk	72.8	31
Meat	3.2	1.5
Crops Sold	0.0	0
Sum	76	32.5
Residual	160.1	67.8

Table 4: The mass balance of P on one dairy farm with 120 cows (Klausner, 1993).

Inputs	Phosphorus (tons $year^{-1}$)	Percent of Total Input
Purchased fertilizer	1.3	19
Purchased feed	5.4	81
Sum	6.7	100
Outputs		
Milk	1.1	16
Meat	0.2	3
Crops Sold	0.0	0
Sum	1.3	19
Residual	5.4	80.6

of imported P was unaccounted for in the milk, crops sold and meat which left the dairy farm. Dairy farms with populations of 45 and 85 cows each had 79 and 68% excess P, respectively. The study by Klausner (1993) did not take into account the N and P losses in leaching, runoff, soil storage, erosion and N losses in ammonia volatilization and denitrification when calculating the mass balances.

Koelsch and Lesoing (1999) analyzed nutrient mass balances for 33 livestock (16 cattle and 17 swine) operations and found 25 of the operations had 50% more N and P input than output. Imported feed constituted 33-77% of the N input and 62 to 71% of the P input in the farms studied. For farms smaller than 2,500 animals, the primary source of N input was imported fertilizer. In a later study Klausner et al. (1998) found that N and P inputs were 72 and 59%, respectively, greater than the exports on one dairy farm in New York. Purchased feed in Klausner et al. (1998) constituted 60-80% of the N and P imbalance. Frink (1969) studied nutrient mass balances on dairy farms in Connecticut and found that 85% of P input was unaccounted for in milk and meat. Approximately 53% of N was unaccounted in the exports of milk, meat and volatilization.

RUNOFF TREATMENT METHODS

Methods developed by the NRCS to treat runoff from dairy heavy use areas and reduce its associated pollutant loads in the environment discussed in this review include constructed wetlands, vegetative filter strips and soil additives (NRCS, 2010; NRCS, 2010; NRCS, 2011; NRCS, 2013). Both constructed wetlands and vegetative filter strips use filtration, sedimentation, volatilization, biochemical conversions, accretion, adsorption and nutrient uptake to mitigate pollutants in runoff (ASWCC et al., 1997; Dickey and Vanderholm, 1981; Dickey and Vanderholm, 1989; Dillaha et al., 1988). Soil additives are either inorganic or organic. Soil

additives can be applied directly to the soil or through irrigation. Examples of soil amendments include PAM, coal fly ash, and PSDs. Each of these treatment methods are further described.

Constructed Wetlands

Constructed wetlands, as defined in NRCS (2010), are designed to mimic natural wetlands and use plants, soils and microorganisms to remove contaminants from water (USEPA, 1997). Studies have shown concentration removal rates between 59-94% of TSS, 54-89% of NH_3-N, 37-86% of TKN, 44-83% of TP and 89-99% of coliform bacteria (Hunt and Poach 2001). Cronk (1996) summarized studies of constructed wetlands used to reduce pollution from dairy farm runoff and found TP, TKN and TSS mass was reduced 54-93%, 57-96%, and 57-99%, respectively. Since CAFO pollution is high in nutrients and solids, constructed wetlands should not be used alone in the treatment of wastes (Hunt and Poach, 2001; USEPA, 1997). Examples of recommended pretreatment methods are settling basins or lagoons before the wastewater enters the wetland (Cronk, 1996; Hunt and Poach, 2001).

Vegetative Filters

Vegetative filter strips/treatment areas, as defined in NRCS (2011) and NRCS (2008), are areas of vegetation used to reduce pollutants such as sediment and nutrients in runoff from entering surface water (Fajardo et al., 2001). Vegetative filters are designed with 10 year life spans following NRCS guidelines (NRCS, 2011). Filter strips have been found to reduce TP between 78 to 83%, TN by 84%, TKN by 80%, SSC between 73 to 95%, fecal coliform by 69%, NH_4 by 63% and PO_4 by 74% (Young et al., 1980; Dillaha et al., 1988, Dickey and Vanderholm 1989).

Soil Additives

Soil amendments (NRCS, 2013) have been used to improve the quality of runoff and soil infiltration properties. They can be added either in a dry form or through an amendment-liquid mixture (*e.g.* irrigation). Additives can be natural, synthetic, inorganic or organic (Brady and Weil, 2008). Common soil amendments include coal ash, PAM and PSD. A variety of PSDs are available for treatment of runoff in agriculture, which include microalgal, cellulose and starch xanthate, guar, polyvinyl alcohol, chitosan and psyllium.

Coal Ash

Coal ash has been used to stabilize soils in feedlots (Gilley et al., 2009; NDDH and EERC, 2003). The chemical properties of fly ash are dependent on the coal used and burning conditions. However, fly ash made from the burning of coal contains mutagenic and carcinogenic compounds (Graber et al., 2006). Ash also contains gypsum, which has been shown to act a as soil stabilizer (Graber et al., 2006). The addition of coal fly ash to soil can increase pH and can cause heavy metal contamination above drinking and surface water quality limits (Carlson and Adriano, 1993; NDDH and EERC, 2003). Other negative effects of fly ash include high concentrations of salts, boron and other toxic elements, increased cementation of soil (Graber et al., 2006), and reduction in available nitrogen and phosphorus. Positive effects of fly ash on soils include increasing water holding capacity, improved soil texture and increased pH in acidic soil (Carlson and Adriano, 1993).

Pond ash is a type of fly ash that was placed in evaporative ponds and dewatered. Gilley et al. (2009) studied the effect of pond ash on manure related pollutants. *Escherichia coli* concentrations in runoff were not affected by the addition of the ash to plots constructed in cattle feedlots. Total phosphorus mass was found to be significantly ($p= 0.05$) lower in runoff from the

plots containing the ash (1.28 kg ha^{-1}) compared to the control plots (2.56 kg ha^{-1}). Mass export of NH_4 in pond ash amended plots was higher (1.16 kg ha^{-1}) when compared to the control plots (0.50 kg ha^{-1}). The pond ash amended surface did not have a significant effect on TN or NO_3. Chang et al. (1977) found that adding fly ash increased the binding strength of soil, decreased bulk density and increased the water holding capacity of soil. Coal ash was also found to initially increase the hydraulic conductivity of soil. However a maximum hydraulic conductivity was reached and additional fly ash added to the soil decreased the hydraulic conductivity (Carlson and Adriano, 1993; Chang et al., 1977). Cox et al. (2001) found coal ash to improve the infiltration rate in soil by approximately 72%.

Polyacrylamides

Polyacrylamide is a synthetic water-soluble organic polymer that is used to stabilize soil aggregates and flocculate suspended particles. Polyacrylamides are made by joining acrylamide molecules (Barvenik, 1994) derived from natural gas (Flanagan et al., 2003; Sojka and Lentz, 1996). An acrylamide molecule has the chemical formula of C_3H_5NO and is the backbone molecule of PAM. Polyacrylamides used to treat erosion are often long chain copolymers with a high molecular weight, in which there are greater than 150,000 acrylamide monomers per molecule. In order for the PAM to stabilize soil, a functional group is changed in the acrylamide chain (Sojka and Lentz, 1996). Polyacrylamides can be formulated to have specific charges, molecular weights and joining acrylamides with certain monomers may improve erosion control (Green and Stott, 1999).

Polyacrylamides are sold either in solution, inverse emulsion or as a powder. Inverse emulsion PAM is aqueous droplets that are stabilized by surfactants in another liquid, typically a product of petroleum distillation. When PAM is sold in solution, the PAM concentration in high

molecular weight molecules is limited because the solution will become too viscous to be used in a practical manner. A high molecular weight for PAM ranges from 1-5 x 10^6 grams per mole whereas a low molecular weight PAM is anything less than 10^5 grams per mole (Barvenik, 1994).

How Polyacrylamides Work

Properties of soil which effect PAM effectiveness include types of clay, pH, the ionic strength of the soil solution and the species of ions in the soil solution (Flanagan et al., 2003). Species of ions influence PAM binding by either acting as an ion bridge for bonding or blocking adsorption sites on soil particles (Laird, 1997). Organic matter in soils was found to reduce sorption sites on the inorganic material therefore reducing PAM effectiveness in laboratory studies (Lu et al., 2002).

Polyacrylamide stabilizes the soil through adsorption onto the clay particles in the soil aggregate (Seybold, 1994) by acting as a bond between the individual particles (Laird, 1997). By keeping the soil aggregate bound together, fewer particles can detach and enter runoff as suspended sediment. Improving soil aggregate stability prevents particles from breaking off and clogging soil pores which can create a surface seal. Binding of the soil aggregates increases soil infiltration and decreases runoff and erosion (Flanagan et al., 2003; Green and Stott, 1999). If the aggregate is broken, the inner particles may not have been treated with PAM and can enter the soil pores causing surface crust formation (Levy and Miller, 1999). Important properties of PAM are the surface charge, configuration, molecular weight and size. Polyacrylamides bind to soil aggregate in a variety of ways depending on the charge of the ion (Flanagan et al., 2003).

Desorption of PAM from a soil particle is nearly impossible due to the long chain of the polymer. In order for a PAM to desorb from a soil particle all bonds need to be broken at the

same time and remain detached long enough for PAM and the soil particle to separate. If the PAM and soil particles are subjected to drying, the bonding between the two particles becomes irreversible (Seybold, 1994).

Polyacrylamide Bonding

Polyacrylamides come in three charges, nonionic, cationic and anionic. While properties of soil can influence the efficacy of PAM, the method of bonding is determined by the charge of the molecule. Each of the different bonding methods for different PAM charges are described below.

Nonionic Polyacrylamides

Nonionic PAM is used to settle out particles in wastewater treatment. Unlike anionic or cationic PAM, nonionic PAM is rarely used in soil remediation (Barvenik, 1994). The ineffectiveness of nonionic PAM in stabilizing soil could be due to the PAM coiling when in the presence of water. Coiling of the nonionic PAM reduces its effectiveness because the polymer is not as long. The extra length on the ionic PAMs is due to repulsion between like charges along the polymer chain allowing PAM to attach to more adsorption sites on the soil aggregate (Laird, 1997). When nonionic PAM is used as a soil amendment, the polymer adsorbs to soil via van der Waals forces (Seybold, 1994).

Cationic Polyacrylamide Bonding

Cationic PAM use in soil is limited (Green and Stott, 1999). When used in soils, research has shown cationic PAM bind directly to soil particles (Seybold, 1994). Laird (1997) showed that a cationic PAM had a high adsorption rate on different clay minerals after being treated with a calcium salt solution and being rinsed to remove extra salts. The clays treated with cationic

PAM released Ca^{+} ions indicating PAM removed the exchangeable ions that were previously adsorbed onto the soil surface and bonded directly to the soil aggregate.

Anionic Polyacrylamide Bonding

Polyacrylamides with a negative charge bond to soil aggregates through cationic bridges and are dependent on the amount of exchangeable ions, clay, pH and PAM size (Seybold, 1994). Laird (1997) showed that cation bridging was vital for anionic PAM bonding to clay surfaces. Cation bridging uses a divalent ion because monovalent ions can only adsorb onto either the clay or the PAM whereas a divalent cation can adsorb to both. Laird (1997) found that the type of clay and ion available altered the way anionic PAM bonded to the soil structure and caused the anionic PAM to bind through hydrogen bonding and hydrophobic bonding.

Green and Stott (1999) conducted an experiment to compare the flocculation rates of PAM with and without the presence of a divalent cation. The experiment showed PAM was less effective when a divalent cation was not present to act as a bridge. Shainberg et al. (1990) tested the effect of adding a cation source to an anionic PAM while treating soil in simulated rainfall. Phosphogypsum (PG) was added to the PAM treatment which has a dry composition of 97% $CaSO_4$. The PG released a Ca^{2+} cation when reacting with PAM to form a cation bridge between the PAM and the soil. It was found that when PAM was added at rates of 20 kg ha^{-1} with PG the infiltration rate was increased by ten times. The addition of PG to PAM decreased the runoff from an 80 mm storm to 18.8% of the rainfall.

Lu et al. (2002) showed PAM's effectiveness in adsorbing to soil particles increased 28 times when in the presence of divalent cations compared to monovalent cations. Polyacrylamide was not as effective when tested on soil organic matter due to a decrease in available adsorption sites. Polyacrylamide adsorbs more effectively onto clay minerals than fine sands and when a

cation was added the difference in PAM adsorption to clays increased by a minimum of ten times.

Polyacrylamide Toxicity

Polyacrylamides are susceptible to mechanical degradation, chemical alteration, sunlight, wetting/drying cycles and soil cultivation. Polyacrylamides have been shown to be non-toxic but acrylamides are a known neurotoxin (Seybold, 1994). While PAM does not degrade into acrylamides, residual acrylamide molecules from the production of PAM can enter the environment through PAM usage. Acrylamides do not dissolve in water and adsorb very little onto sediments. Acrylamides are biodegradable but the half-life is increased in anaerobic conditions (Barvenik, 1994; Seybold, 1994).

Polyacrylamide Effectiveness

Polyacrylamides have been used to reduce pollutant loads in runoff associated with agriculture. Studies primarily show the effectiveness of PAM to reduce pollution in furrow irrigation with or without manure applied as fertilizer. Polyacrylamide has been found to reduce TP concentrations in runoff by as much as 92%, and suspended sediment by 57-97% (Lentz and Sojka, 1994; Lentz et al., 1992; Lentz et al., 2001). It also has increased infiltration rates of soil compared to control furrows with no PAM application (Lentz and Sojka, 1994; Lentz et al., 1992). Polyacrylamide efficacy at removing NO_3 from water ranged from no effect to 83% (Lentz and Sojka, 1994; Lentz et al., 2001). Polyacrylamide had no significant effect on NO_3 because the results were influenced by an unexpectedly high concentration in a single furrow. Krauth et al. (2008) concluded that the application of anionic PAM to a cotton farm was able to reduce P in runoff an average of 0.436 mg/L and TSS in runoff an average of 148.67 mg/L in rain events (n=3).

Entry and Sojka (2000) studied the effect of PAM, PAM + CaO and PAM + $Al(SO_4)_3$ on the removal of nutrients in runoff. The additional compounds added to the PAM treatment reduced TP in runoff up to an additional 75% compared to PAM only treatments. The experiment also found that the PAM treatment with and without additives had removed 99% of fecal coliforms after 1 meter in the furrow compared to the fecal coliform count of the source material. Spackman et al. (2003) tested the effect of PAM on fecal coliform in runoff from furrows which received manure fertilization. The study showed that fecal coliform decreased by 99% after 28 days whether or not the furrow was treated with PAM. Sojka and Entry (2000) used furrow irrigation and different flow rates to test the effect of PAM on microbial transport in runoff and found that after one meter at the lowest flow rate, the total bacteria mass reduced by 90.2% compared to the control at the lowest flow rate.

Polysaccharides

Polysaccharides are polymers of monosaccharides (Potter, 1986) and are a commonly used alternative to PAM as a soil conditioner. Polysaccharides are typically cationic with small to medium molecular weight and work similarly to PAM when stabilizing soil aggregates (Graber et al., 2006). Microalgal, chitosan, cellulose and starch xanthate, guar, polyvinyl-alcohol (PVA) and psyllium are all PSDs which have been tested as a soil conditioner.

<u>Microalgal</u>

Microbes which create PSDs can be used as a soil stabilizing agent. The effectiveness of the microbe depends on the strain, amount present and the environmental conditions. Microbial amendments physically bind with the soil or produce PSDs which bind with soil aggregates (Martens and Frankenberger Jr, 1992). Multiple studies have shown increased aggregate stability after using microbial soil amendments. Martens and Frankenberger Jr (1992) treated

soils with 13 different amendments, incubated the soils at 25ºC and found aggregate stability increased by 17-78% with an average of 32% over eight weeks. *Nostoc muscorum* treated soils were found to increase soil carbohydrate content by 2.96-3.49 times over 300 days, which increased aggregate stability 18%. The increase in aggregate stability was significantly correlated with the increase in carbohydrate content (Rogers and Burns, 1994). *Nostoc muscorum* increased stability of aggregates larger than 250 micrometers by 12 times after 180 days and 66% after 365 days compared to the control (de Caire et al., 1997).

Chlamydomonas mexicana and *Chlamydomonas sajao* were able to increase PSD content in the top 2 mm of soil 20-129% in a greenhouse experiment compared to the non-treated soils, which would suggest the ability to increase the aggregate stability of soil. Only 25% of the samples saw an increase in PSD content between 3-8 mm because 99% of the algae stayed within the top 2 mm of the soil. Algae was only able to grow and produce PSDs when moisture levels were at 100% field capacity (Barclay and Lewin, 1985). A second study using *C. mexicana* concluded that 1.5-3.3% more water was retained and carbohydrates increased 132-164% in the first 300 mm of soil depth compared to non-treated soil. The study also compared treated and non-treated soil aggregate stability by slaking in water and dry sieving. *Chlamydomonas mexicana* increased dry and wet soil aggregate stability by as much as 17.9 and 21.7%, respectively (Table 5) (Metting and Rayburn, 1983). Metting (1987) applied a high rate (7.8 kg dry weight ha^{-1}) and a low rate (0.8 kg dry weight ha^{-1}) of microalgal additive to soils and found no significant difference between low rate and the control after three years. The high treatment rate was found to improve soil aggregate stability.

Table 5: Change in aggregate stability of *Chlamydomonas mexicana* treated soils (Metting and Rayburn, 1983)

Soil	Wet Stability (% aggregation)	Dry Stability (% aggregate)
Quincy Loam Fine Sand	21.7	10.1
Quincy Loam Sand	18.9	16.3
Warden Silt Loam	2.0	17.9
Ritzville Silt Loam	-0.4	1.8

Cellulose/Starch Xanthate

Cellulose and starch xanthates are synthetic soil amendments made from paper, wood pulp or cotton gin waste (Wood and Oster, 1985). They are attracted to soil aggregates because of a charged disulfide group and potentially can have a higher molecular weight than PAM (Orts et al., 1999). Similar to PAMs, xanthates bond to the clays in soil (Wood and Oster, 1985). Xanthate has been shown to reduce soil loss in runoff by 80% in plots if applied at rates of 80 mg L^{-1}. At this application rate Xanthate was eight times less effective than PAM, which reduced soil loss in furrow runoff by up to 97% at concentrations of 10 and 5 mg L^{-1} (Orts et al., 1999). Wood and Oster (1985) tested xanthates on three different soils with varying exchangeable sodium levels. Xanthate removed almost all solids in runoff (Table 6) and reduced crust strength in soils by 30-60% (Table 7). Xanthate also increased the hydraulic conductivity of soil with nonsaline solution between 50 and 340%.

Guar

Guar is a PSD and soil conditioner derived from the bean *Cyamopsis tetragonoloba* bean (Wallace, 1986). Ben-Hur and Letey (1989) tested anionic, nonionic and cationic guars as a soil amendment in a lab setting. Soil irrigated with two cationic guars produced infiltration rates of 19 and 14.2 mm h^{-1}. Soil irrigated with nonionic guar produced an infiltration rate of 5.2 mm h^{-1} while anionic guar produced a rate of 4 mm h^{-1}. The untreated control had an infiltration rate of 4 mm h^{-1}. The higher infiltration rate and increased aggregate stability in the cationic guar treated soil suggests that the positively charged guar binds with the negatively charged clay particles. A soil which was covered with a material to protect from rainfall impact was found to have a similar infiltration rate as the cationic guar amended soil. The similar infiltration rates show that the cationic guar was able to prevent soil aggregates from breaking apart and creating a surface

Table 6: Erosion (mg of solid/ml runoff) during sprinkler irrigation with nonsaline water (Wood and Oster, 1985)

	Untreated			Xanthate			Polyvinyl Alcohol		
Exchangeable Na	0.0	0.05	0.10	0.0	0.05	0.10	0.0	0.05	0.10
Soil Type									
Arlington	15	25	12	0	0	1	0	0	0
Fallbrook	6	3	4	0	1	0	0	2	0
Pachappa	11	9	8	2	0	0	2	0	8

Table 7: Penetration pressure (Gg m^{-2}) between soil depths of 0 to 10 mm to measure crust strength (Wood and Oster, 1985)

	Untreated			Xanthate			Polyvinyl alcohol		
Exchangeable Na	0.0	0.05	0.10	0.0	0.05	0.10	0.0	0.05	0.10
Soil type									
Arlington	1.3	1.6	1.8	0.9	0.7	0.7	0.9	1.5	1.0
Fallbrook	2.1	2.2	2.3	1.5	1.0	1.7	2.1	1.9	2.5
Pachappa	1.7	1.5	2.2	0.9	0.6	1.4	2.0	1.2	2.2

seal. Cationic guar irrigated soil was able to limit clay dispersion which prevented a full surface seal from forming.

Page and Quick (1979) found guar gum to be highly viscous in solution, therefore hard to use in the field, and when stored at room temperature the product degraded quickly. Degradation of guar would decrease effectiveness and cause a need for reapplication of the soil amendment. Guar was also found to have almost no effect on soil crust formation. Wallace (1986) conducted experiments in a laboratory setting using guar in suspension at different rates (0, 600 and 1200 kg ha^{-1}) in acidic solution and found the effect of the additive disappeared after three weeks. Weaver (1984) used wet sieving to test cationic starches, PVA, guar and PAM on aggregate stability. The amendments were added then the soil was sieved. The soil remaining on the sieve was calculated as the % retained (Weaver et al., 1984). Polyvinyl alcohol retained the highest percentage of soil after wet sieving, followed by guar and PAM (Table 8). Contrary to Weaver (1984), Helalia and Letey (1988) concluded PAM was more effective than guar. Helalia and Letey (1988) measured flocculation rates in solution of anionic, nonionic and cationic guar at varying soil adsorption ratios and concentrations using optical transmittance. PAM products with the same charge as the guar were found to more successful at promoting flocculation, which is likely due to the PAM's larger molecular weight. Cationic guar was the best at promoting flocculation with a transmittance rate close to 100%, while anionic guar had transmittance close to 60%.

Polyvinyl alcohol

Polyvinyl alcohol is a synthetic organic stabilizing agent that comes in a powder form and has been found to increase infiltration and water movement in soil high in clay content. In a laboratory study, Wood and Oster (1985) used PVA that was added in solution slowly to let the

Table 8: % soil retained after wet sieve when 0.1 grams of material applied to 100 grams of soil (Weaver, 1984)

Material	Material type	% Soil Retained after Wet Sieve
Control	-	10
Cato 8	Cationic Starch	27
Cato 14	Cationic Starch	25
Cato 15	Cationic Starch	23
t-CS-1	Tertiary cationic starch	35
t-CS-2	Tertiary cationic starch	43
t-CS-3	Tertiary cationic starch	47
q-CS-1	Quaternary cationic starch	42
q-CS-2	Quaternary cationic starch	44
q-CS-3	Quaternary cationic starch	42
Galactasol 210	Guar gum	88
Galactasol 811	Cationic guar gum	88
Galactasol 813	Cationic guar gum	86
Galactasol 817	Cationic guar gum	86
PAM	Polyacrylamide	74
PVA	Polyvinyl Alcohol	97

PVA infiltrate, then dried the soil sample prior to simulating rainfall to test the amendments efficacy at stabilizing soil. The application of PVA as a soil conditioner under simulated rainfall conditions reduced the soil crust strength by 30% (Table 7) and reduced almost all sediment in runoff (Table 6). A second study found PVA increased the hydraulic conductivity between 50 and 420%, and was able to reduce soil surface crust strength by 21% compared to the control soil (Page and Quick, 1979).

Chitosan

Chitosan is a PSD and is used as a flocculating agent. However it is not used as much as PAM because it costs twice as much. Lab experiments using mini-furrows indicated that chitosan has the potential to be just as effective as PAM at a concentration of 20 mg/L. However lab results were not duplicated in field experiments. Chitosan was applied at rates of 10 mg/L in a field and found that solids in runoff were reduced by 49%. Even though the flocculation of solids was not as high as expected based on laboratory experiments, chitosan was still an effective amendment because it flocculated algae and fine sediment in the field. Chitosan did not achieve similar results to lab experiments because it readily attached to sediment particles and was flocculated out of solution at the beginning of the furrow before the SSC samples were taken (Orts et al., 1999).

Psyllium

Psyllium is a plant derived soil additive made from ground seed husks. As a soil additive, psyllium has had little research, but in one study it increased the maximum water holding capacity of soil 242%. Psyllium also increased porosity, TN content and bacteria in the soil in that study (Patil et al., 2011). In another study by CALTRANS (2011), wood mulch, paper mulch or straw were mixed with psyllium and the mixtures were applied on soil for erosion control.

Psyllium mixtures were found to increase runoff compared to bare soils but decreased sediment in the runoff. The wood mulch psyllium mixture increased average TKN and P in runoff by 0.54 and 0.56 mg L^{-1}, respectively. Paper mulch and psyllium did not increase average TKN but increased average P in runoff by 0.28 mg L^{-1} (CALTRANS, 2011).

Conclusions

There are three primary ways to treat runoff pollution from animal waste: constructed wetlands, vegetative filters and soil additives. Coal ash as a soil additive has little research available and causes heavy metal pollution. Polyacrylamide has been extensively researched and is effective at reducing pollutants through soil stabilization and increased infiltration. However PAM is difficult to dissolve in water and apply to the field due to high viscosity. The degradation of PAM makes it necessary for multiple applications in one season (Green and Stott, 1999). Polyacrylamides also release residual acrylamide molecules, which is a known neurotoxin (Seybold, 1994). An inexpensive, nontoxic, soil additive which is effective at reducing pollutants in CAFO runoff needs to be found.

Stabilizer® is an organic product made from psyllium seed husks and applied in a similar manner as PAM. Stabilizer® is a PSD which has not been studied as a soil amendment in dairy heavy use areas. Psyllium seed husk has little research available but has been shown to increase the maximum water holding capacity and porosity of soil (Patil et al., 2011) and decrease the suspended sediment in runoff (CALTRANS, 2011). Limitations of PSD treatments such as guar, PVA, microalgal and chitosan include: increasing the viscosity of the application solution, large quantities to achieve results similar to PAM, a short shelf life, and being twice the cost of PAM (Orts et al., 1999; Page and Quick, 1979; Wallace, 1986).

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A SOIL ADDITIVE TO REDUCE RUNOFF AND POLLUTANTS FROM DAIRY HEAVY USE AREAS

ABSTRACT

Dairy farm heavy use areas are a known contributor of runoff pollution. A paired watershed study evaluated the soil additive Stabilizer® and its effect on pollutants in runoff from a dairy heavy use area in Connecticut. During the calibration period when no treatment was applied, a statistically significant ($p=0.05$) relationship between paired observations from the control and treatment watersheds was found for discharge, suspended solids (SSC), total phosphorus (TP), total nitrogen (TN), and *Escherichia coli* (*E. coli*) mass exports in runoff. During the treatment period Stabilizer® was applied to the treatment watershed at a rate of 0.08 kg m^{-2} and was disked 3 cm into the soil. An analysis of covariance (ANCOVA) determined there was a significant change in the y-intercept of the regression equations before and after the application Stabilizer® for discharge, runoff depth, runoff coefficient, mass exports of TP and SSC, but not the export of TN and *E. coli*. Stabilizer® reduced discharge by 79%, TP mass export by 95% and mass SSC by 84%. TN concentration increased after the application of psyllium by 87%. Psyllium should be considered as a possible treatment method for dairy heavy use area runoff but more research is needed to determine the effect of psyllium on TN in runoff.

INTRODUCTION

Agricultural pollution is the main cause of stream impairment in the United States. Nutrient, sediment and pathogen pollution are three examples of agricultural pollution (USEPA, 2009), all of which are associated with animal feeding operations. Pollution from animal waste is worsened in confinement areas because they create a smaller environment where animal waste accumulates (Jongbloed and Lenis, 1998). Due to the excess bacterial and nutrient pollution associated with dairy confined animal feeding operations (CAFOs), all runoff and wastewater needs to be treated using the best practicable control technology (BPT) unless it is an overflow caused by a 25 year, 24 hour storm or larger. However, National Pollution Discharge Elimination System (NPDES) permits allow for site specific technologies called voluntary performance standards to meet the no discharge requirement (USEPA, 2012).

Constructed wetlands (Cronk, 1996; Hunt and Poach, 2001), vegetative filters (Dillaha et al., 1988; Koelsch et al., 2006), and soil additives such as polyacrylamides (PAM) (Barvenik, 1994; Seybold, 1994), and polysaccharides (PSD) (Orts et al., 1999; Page and Quick, 1979; Weaver, 1984) have been studied for their effects on soil stabilization and runoff water quality. Coal ash has been used to treat feedlots (Gilley et al., 2009) but can cause an increase in hazardous compounds in runoff which may exceed surface water and drinking water quality standards (Carlson and Adriano, 1993; NDDH and EERC, 2003). Polyacrylamides have been shown to decrease SSC, TP, TN and bacterial concentrations in furrow irrigation (Lentz and Sojka, 1994; Lentz et al., 2001; Spackman et al., 2003). However, PAM may contain residual acrylamide molecules which are known neurotoxins (Seybold, 1994). Polysaccharide treatments have been studied as soil additives, including microalgal (Martens and Frankenberger Jr, 1992), cellulose and starch xanthate (Wood and Oster, 1985), polyvinyl alcohol (PVA) (Wood and Oster, 1985), chitosan (Orts et al., 1999) and guar (Ben-Hur and Letey, 1989; Helalia and Letey,

1988). Issues with the PSD treatments studied include increasing the viscosity of the application solution, large quantities to achieve results, a short shelf life and not being economically feasible (Orts et al., 1999; Page and Quick, 1979; Wallace, 1986).

Another potential soil PSD additive is psyllium. Psyllium powder was found to increase the maximum water holding capacity, porosity, TN and biological activity in the soil in a laboratory setting while testing the effect biopolymer additives have on seed germination (Patil et al., 2011). The powder was also tested for erosion control in a mixture with wood mulch, paper mulch or straw. Psyllium mixtures were found to increase runoff compared to bare soils but decrease sediment erosion. The wood mulch psyllium mixture increased average TKN and P in runoff by 0.54 and 0.56 mg L^{-1}, respectively. Paper mulch and psyllium did not increase average TKN but increased average P in runoff by 0.28 mg L^{-1} (CALTRANS, 2011). Psyllium has primarily been used on athletic fields, horse race tracks, driveways and walkways and has not been used in the treatment of agricultural heavy use areas. Some advantages of psyllium are that it is organic, non-toxic and it produces no hazardous waste. Due to the limitations of other soil additives, the purpose of this study was to test the effectiveness of psyllium at reducing pollutants in runoff from dairy heavy use areas. The product Stabilizer® was used in this study.

MATERIALS AND METHODS

Study Site

The heavy use area was located at a dairy farm in Connecticut. Control and treatment watersheds (Figure 1) were monitored at their outlets both before and after the application of the soil amendment for volume and water quality characteristics. The control watershed was 3.12 ha in area, with 2.56 ha designated as in heavy use. The treatment watershed was 0.49 ha with 0.40 ha designated as in heavy use. It had exposed soil with little to no vegetation due to a constant

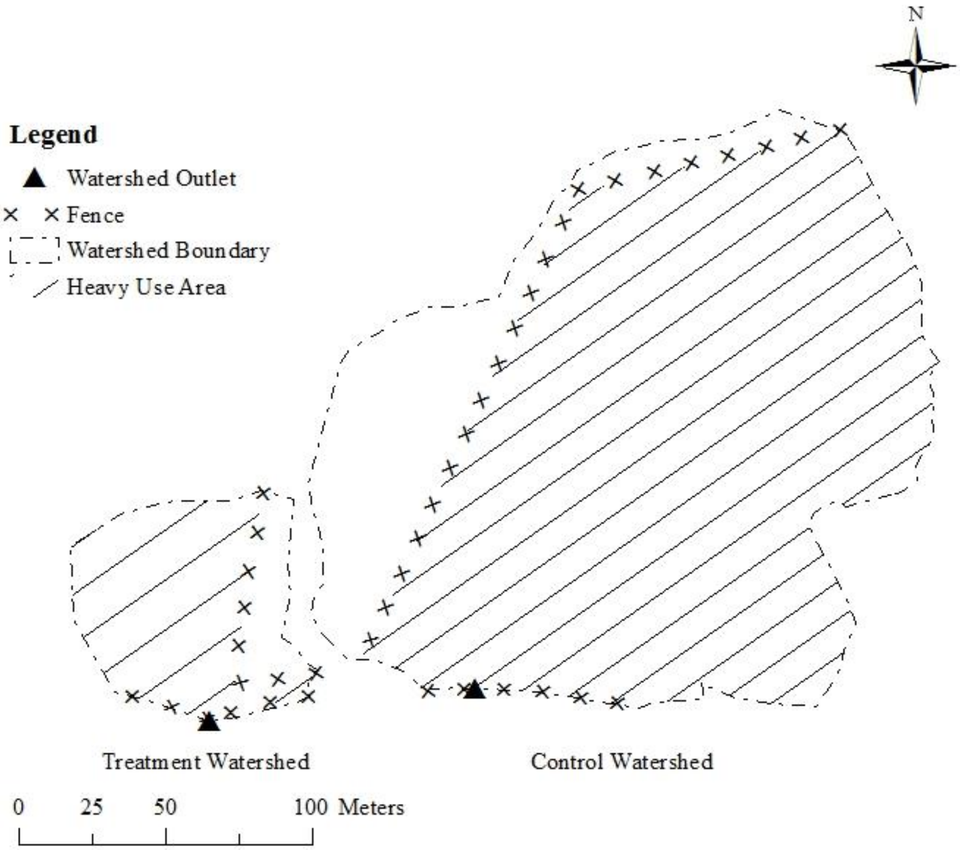

Figure 1: Field site map

presence of dairy cows. The control watershed also had areas of exposed soil due to the dairy cows, but part of the eastern half of the control watershed was covered in vegetation during various times throughout the experiment.

Study Design

The paired watershed technique (Clausen and Spooner, 1993) was used for this study. This approach uses a control and treatment watershed and calibration and treatment time periods. During the calibration period both watersheds were monitored in their untreated state. Discharge observations and water quality characteristics were measured when both the control and treatment watersheds produced a sample. After determining a significant relationship between paired observations for the control and treatment watersheds for the characteristics measured using analysis of variance (ANOVA), the treatment was added to the randomly chosen treatment watershed. Paired observations continued using the same methods applied in the calibration period. Analysis of covariance was used to determine if the slopes and intercepts were different between the treatment and control watershed observations before and after the soil amendment was added to the treatment watershed.

Site Design

A 0.46 m H-flume with wing walls was used to measure discharge. A bubbler flow meter (Model 4230, ISCO®) and sampler (Models 3710 and 2900, ISCO®) were used to record discharge and take samples. Flow-weighted samples were collected in a 2L bottle and were replaced after every storm. Samples were collected within 24 h of the end of the storm. Precipitation was measured using manual and recording rain gages.

Sample Analysis

Samples were transported from the field to the lab in a cooler with ice packs and divided using a churn splitter. TN and TP were analyzed by the Center for Environmental Science and Engineering at the University of Connecticut using USEPA methods 353.2 and 365.4, respectively (USEPA, 1974; USEPA, 1978). Within 6 h of collection, *E. coli* analysis was performed using 3M Petrifilm and AOAC official method 991.14 (3M, 2001). Suspended solid concentrations were determined using Standard Method SSC 2540D (APHA et al., 1998). Soil samples were taken using a trowel at random locations in each watershed and mixed in a bucket. A subsample was sent to the Soil Testing Laboratory at Kansas State University for analysis of pH, TP, organic matter, TN, total carbon (TC), texture and Mehlich P.

Statistical Methods

The normality of the observations was analyzed using the Shapiro Wilk test and data were log transformed for all water characteristics measured. Analysis of variance was used to determine the relationship between observations in the control and treatment watersheds. Following the treatment period, ANCOVA was used to determine if the slopes and y-intercepts of the control and treatment period regression equations were statistically different. SAS version 9.4 was used for all analysis (SAS Institute, 2012). Mass export (kg ha^{-1} yr^{-1}) was calculated by multiplying the discharge and concentration for each sample and dividing by the watershed area and time period. Results of the paired watershed study include the means, predicted mean values, and percent change during the treatment period. Predicted treatment watershed values were calculated using the regression equation from the calibration period. Percent change was calculated as the difference between the observed and predicted geometric means using Eq. [1]:

$$\%change = [\frac{(observed - predicted)}{predicted}] * 100 \qquad [1]$$

RESULTS AND DISCUSSION

Precipitation

A National Oceanic and Atmospheric Administration (NOAA) observation station 17.7 km from the study site at Bradley International Airport (GHCND: USW00014740) was used to compare observed to normal precipitation (Table 9). Observed rainfall was 36.2% and 10.7% less than normal in 2013 and 2014, respectively. In April 2015, there was -8.5% difference in rainfall compared to the climate normal.

Discharge

During the calibration period 14 runoff events occurred, of which two had equipment malfunctions and no samples. Following the Stabilizer ® treatment, six discharge events occurred. Discharge, runoff coefficients and runoff depth were higher for the control watershed than the treatment watershed for all paired observations in the study. Weekly mean discharge decreased due to the treatment by 79% ($p < 0.001$) based on a comparison of discharge predicted by the calibration equation and observed discharge (Table 10). Weekly mean runoff depth and weekly mean runoff coefficients were similarly reduced by 79% ($p < 0.001$). Since these terms of discharge are highly related, they would be expected to show similar results. Discharge decreased from the treatment watershed across the full range of observed values (Figure 2). Psyllium is known to increase the water holding capacity of the soil (Patil et al., 2011), which may have reduced runoff from the watershed in this case.

Runoff coefficients from both the control and treatment watersheds were low, which was not expected because the soil appeared compacted from dairy cows. The geometric mean event runoff coefficient observed on the control watershed decreased from 0.34 to 0.04 from the calibration to treatment period. Similarly, the treatment watershed mean event runoff coefficient decreased from 0.08 during the calibration period to 0.003 during the treatment period. Without

Table 9: Precipitation (mm) from April to November in 2013, 2014, and 2015 at Bradley International Airport (GHCND: USW00014740) in Connecticut and percent departure from normal precipitation (in parentheses)

Month	Normal	2013	2014	2015
April	94.5		138.7 (46.8)	86.4 (-8.5)
May	110.5		113.3 (2.5)	
June	110.5		43.4 (-60.7)	
July	106.2	106.4 (0.24)	114.0 (7.4)	
August	99.8	174.8 (75.1)	95.5 (-4.3)	
September	98.6	90.9 (-7.7)	41.1 (-58.2)	
October	111.0	55.6 (-49.9)	99.7 (-10.3)	
November	98.8	101.9 (3.1)	95.8 (-10.65)	
TOTAL	829.8	529.6 (-36.2)	741.4 (-10.7)	

Table 10: Mean predicted and observed values and percent change from the control and treatment watersheds during the calibration and treatment periods consistent

	Calibration Period			Treatment Period						ANCOVA	
						Treatment Watershed					
Characteristic	Sample Size (n)	Control Watershed	Treatment Watershed	Sample Size (n)	Control Watershed	Observed	Predicted	Calibration Equation	% Change	*F*	*P*
Discharge (m^3/week)	14	133.7	4.8	6	23.36	0.27	1.32	$y = 0.2559x^{0.664^{***}}$	-79.3^{***}	22.55	<0.001
Depth (cm/week)	14	0.43	0.10	6	0.075	0.006	0.027	$y = 0.2341x^{0.664^{***}}$	-79.3^{***}	22.57	<0.001
Runoff coefficient	14	0.34	0.08	6	0.133	0.010	0.047	$y = 0.137x^{0.532^{***}}$	-79.0^{***}	13.98	<0.001
TN (mg L^{-1})	12	24.3	44.1	5	33.61	99.26	53.15	$y = 6.97x^{0.578^{N.S.}}$	86.8^{*}	7.56	0.003
TP (mg L^{-1})	12	13.9	30.2	5	12.75	27.22	29.10	$y = 10.895x^{0.386^{N.S.}}$	$-6.5^{N.S.}$	1.47	0.268
SSC (mg L^{-1})	12	834.3	1661.9	5	493.16	187.11	1305.03	$y = 73.01x^{0.465^{N.S.}}$	$-85.7^{N.S.}$	3.84	0.036
E. coli (CFU 100 mL^{-1})	12	6.5E+06	1.2E+07	5	2,000	20,000	900,000	$y = 78932.4x^{0.323^{N.S.}}$	$-97.6^{N.S}$	3.2	0.059
TN (kg $ha^{-1}yr^{-1}$)	12	54.4	21.0	5	8.65	2.77	34.06	$y = 0.437x^{0.890^{**}}$	$-91.9^{N.S.}$	10.94	<0.001
TP (kg $ha^{-1}yr^{-1}$)	12	31.2	14.3	5	3.32	0.77	15.86	$y = 0.498^{0.855^{***}}$	-95.2^{*}	21.9	<0.001
SSC (kg $ha^{-1}yr^{-1}$)	12	2794.1	789.0	5	126.27	14.37	89.27	$y = 1.22x^{0.773^{***}}$	-83.9^{*}	21.52	<0.001
E. coli (kg $ha^{-1}yr^{-1}$)	12	1.46E+07	5.91E+06	5	2,228.23	2,586.15	447.6	$y = 0.129x^{1.084^{*}}$	$477.7^{N.S.}$	9.42	0.001

*p < 0.05; **p < 0.01; ***p < 0.001; N.S. non-significant

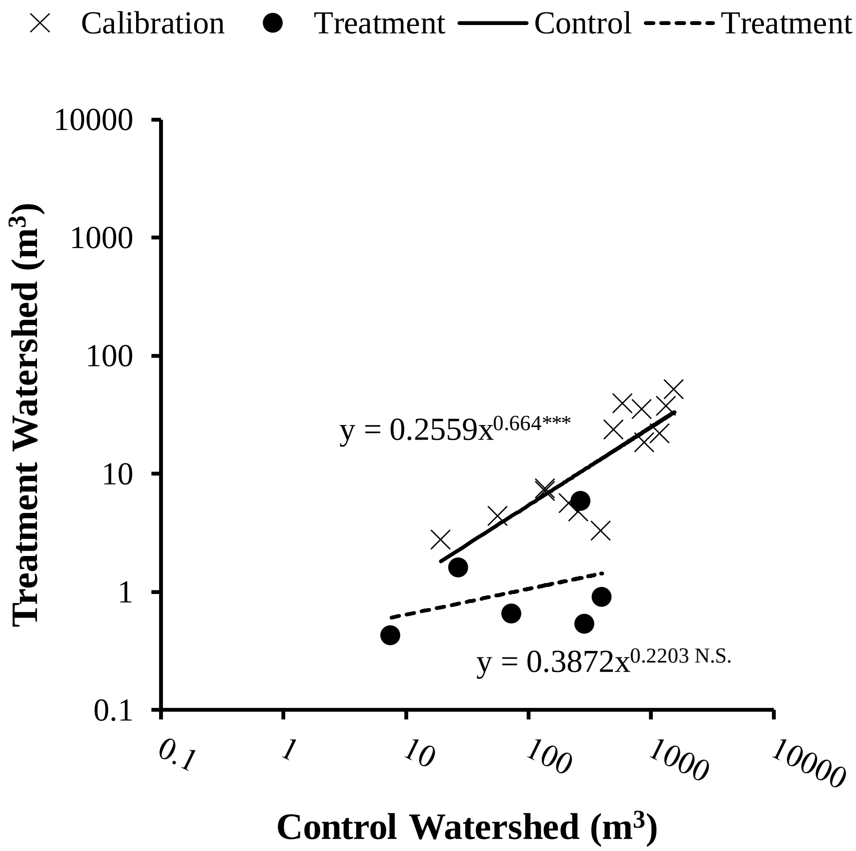

Figure 2: Paired runoff volume from the control and treatment watersheds during the calibration and treatment periods for a dairy heavy use area in Connecticut

considering compaction, a cultivated, sandy soil with less than 1% slope would be expected to have an estimated runoff coefficient of 0.2 (Novotny, 2003). Changes in the topography of the treatment and control watersheds may have occurred due to scraping both before and after the Stabilizer® treatment. Scraping may have increased detention storage in both watersheds.

In the only other study of the use of psyllium to reduce erosion, CALTRANS (2000) used psyllium in combination with wood mulch or paper mulch as a soil additive. The laboratory study used rainfall simulation and runoff plots. Psyllium in combination with either wood or paper mulch caused an increase in discharge by 18.5 and 27%, respectively, compared to bare soil (CALTRANS, 2000). They did not explain any reasons for the increase in discharge after the psyllium treatment in this study.

Total Nitrogen

During the calibration period 12 TN samples were collected and a significant regression relationship was determined (R^2= 0.66, p=0.001) between paired observations of mass export from the control and treatment watersheds, but not for TN concentrations (Table 10). Following the application of Stabilizer ®, five TN samples were obtained. Based on ANCOVA, there was no significant difference in the y-intercepts (p = 0.27) and slopes (p=0.07) between the regression equations for the calibration and treatment periods for TN mass export. However, after the application of psyllium, TN concentration increased by 86.8% (p = 0.048) when compared to the predicted TN concentration using the calibration equation and observed mean concentration. While psyllium had no effect on TN mass export in this study, in the CALTRANS (2000) plot study, TKN increased in runoff from one of the two psyllium treatments. Psyllium has also been shown to increase TN concentrations in soils due to an increased soil microbial count (Patil et al., 2011). Total nitrogen concentrations could have also increased due to the

application of the psyllium. The TN concentration of psyllium was 17,400 ppm, which was high compared to the 5,900 ppm reported for dairy manure in the Agricultural Waste Management Handbook (USDA, 2009).

Total nitrogen concentrations in this experiment in runoff from both watersheds were similar to those observed in other studies (Figure 3). Concentrations of TN in runoff have ranged widely between feedlots studied. Total nitrogen concentrations in feedlot runoff can even be higher than concentrations in untreated domestic wastewater (Figure 3).

Total Phosphorus

The treatment of Stabilizer® reduced TP mass export (Figure 4) by 95% ($p < 0.001$) when compared to the predicted means using the calibration equation (Table 10). There was no significant relationship between paired observations based on the concentrations of TP measured in runoff from the control and treatment watersheds during the calibration period. The reduction in TP mass export is likely linked to the decrease in discharge from the watersheds. In the CALTRANS (2000) study psyllium combined with paper or wood mulch increased TP concentrations in runoff by 133% and 257%, respectively, when compared to bare soil (CALTRANS, 2000).

Total phosphorus concentrations in runoff from this study were similar to observations from other studies of beef and dairy cow feedlot runoff (Figure 5). Runoff from both watersheds in this study had a higher maximum and a similar minimum concentration of TP as untreated domestic wastewater.

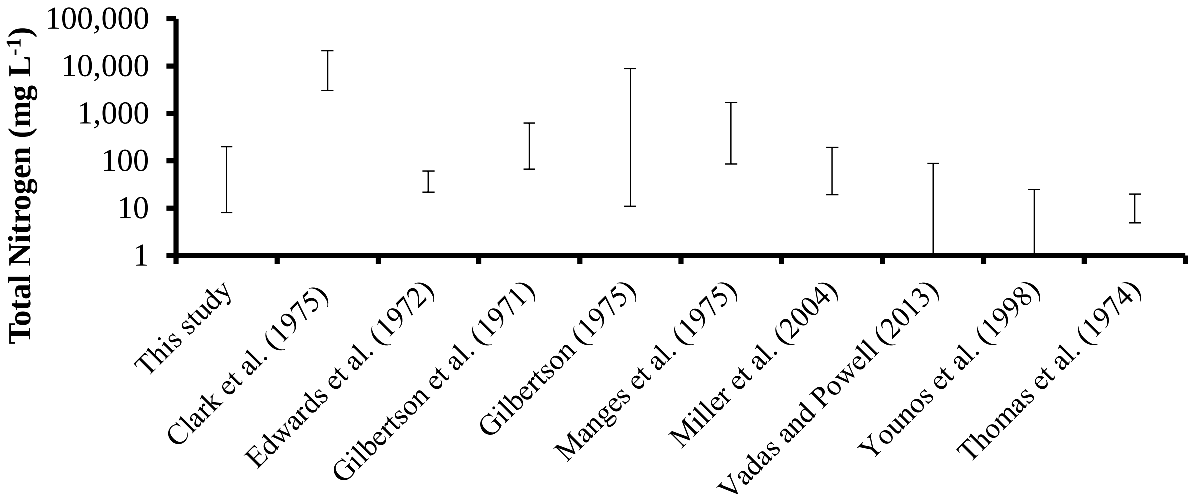

Figure 3: Total nitrogen concentrations in runoff from feedlots. Thomas et al. (1974) is for untreated domestic wastewater.

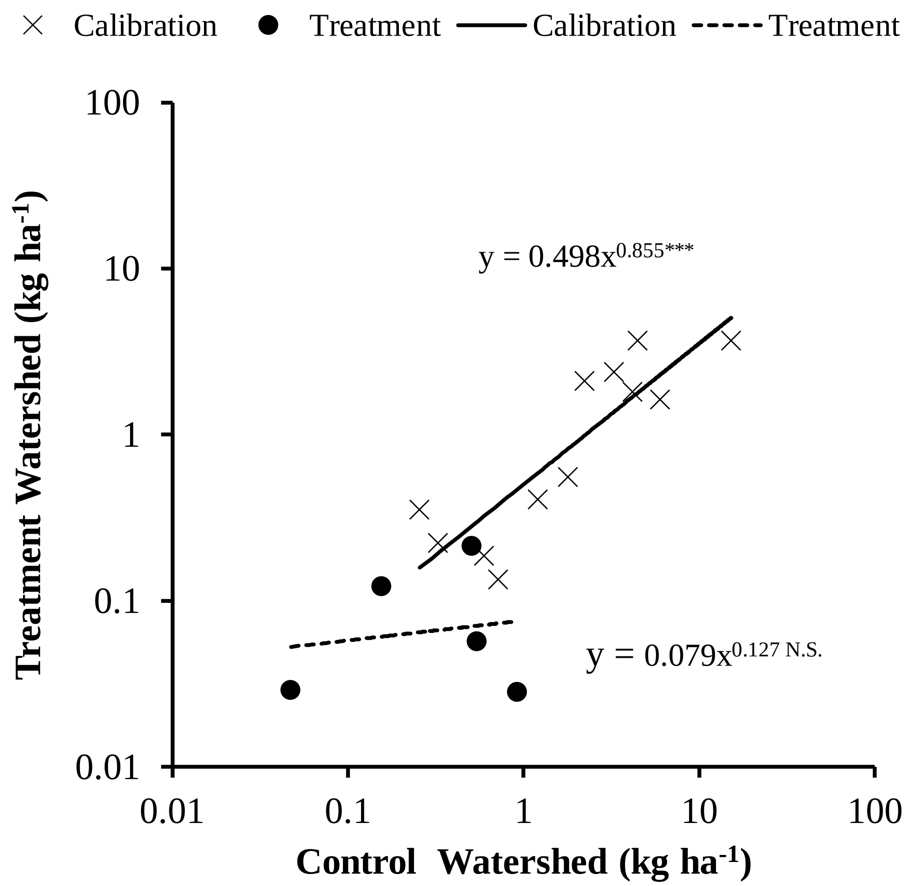

Figure 4: Paired mass TP export from the control and treatment watersheds during the calibration and treatment periods for a dairy heavy use area in Connecticut

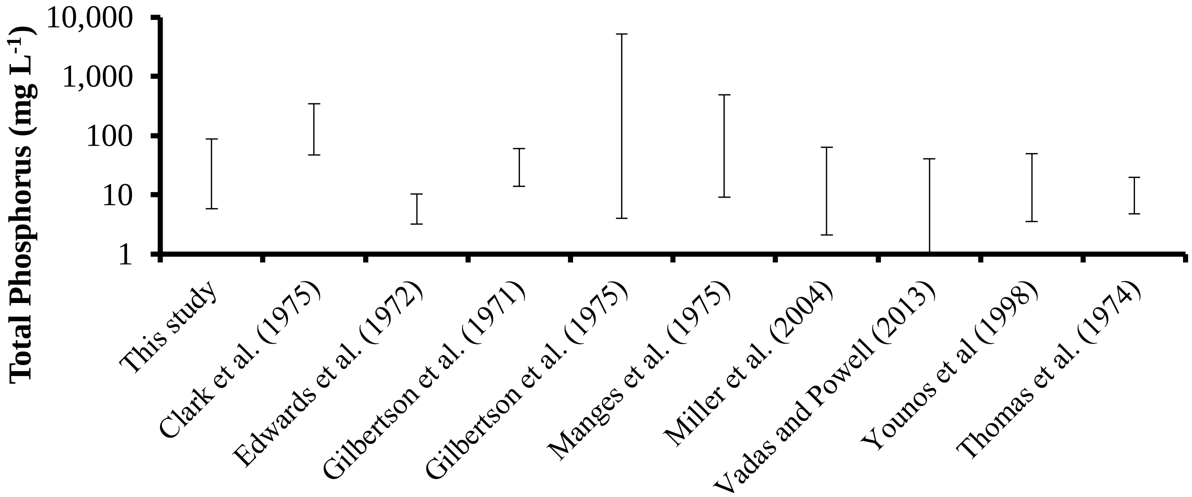

Figure 5: Total phosphorus concentrations in runoff from feedlots. Thomas et al. (1974) is for untreated domestic wastewater.

Escherichia coli

A statistically significant regression relationship (R^2 = 0.397 , p = 0.028) was found during the calibration period for the paired observations of *E. coli* mass export for the control and treatment watersheds, but not for *E. coli* concentrations. However, the relationship between the observations was not strong based on the low R^2 value. Based on ANCOVA, there was no significant difference in the y-intercepts (p = 0.093) and slopes (p=0.383) between the regression equations for the calibration and treatment periods for *E. coli* mass export. The addition of psyllium to soil has been previously shown to increase microbial counts in soil (Patil et al., 2011).

Bacterial concentrations varied seven orders of magnitude in runoff from these watersheds. Mean concentrations of *E. coli* in runoff from the control and treatment watersheds were greater than the average untreated municipal wastewater concentration for fecal coliform of 1 million CFU 100 mL^{-1} during the calibration period (Kern et al., 2000). However, concentrations of *E. coli* in runoff from both watersheds during the treatment period were lower than the average untreated municipal wastewater concentration, perhaps because temperatures were cooler when samples were taken. The *E. coli* criteria for contact recreation in water is a geometric mean of 126 CFU 100 mL^{-1}or a sample maximum of 235 CFU 100 mL^{-1} (CT DEEP, 2012). The geometric mean *E. coli* concentrations in runoff were higher than both the allowable geometric mean and single sample maximum criteria set by the CT DEEP (2012). *Escherichia coli* concentrations in runoff ranged from 0 to 40,000 CFU 100 mL^{-1} when minimum temperatures were less than 3º C during the spring. There was only one study of *E. coli* in runoff from heavy use areas even though feedlots have been known to be a major source of stream

pollution. This study reported concentrations of *E. coli* between 100 and 100 million CFU $100mL^{-1}$ (Miller et al., 2004).

Suspended Solids

The treatment reduced SSC mass export by 84% ($p < 0.001$) when compared to the predicted mean using the calibration equation (Figure 6) (Table 10). This finding was consistent with results from the CALTRANS (2000) study, which observed SSC concentration reductions of 61 and 87% for the psyllium plus paper or wood mulch treatments, respectively. A reduction in SSC is likely due to psyllium binding to soil and stabilizing soil aggregates in a manner similar to PAM (CALTRANS, 2003).

Suspended solids concentrations varied widely in this study but concentrations were similar to those found in feedlot studies. Suspended solids concentrations found in runoff from this study were typically greater than concentrations found in untreated domestic wastewater reported by Thomas et al. (1974). Miner et al. (1966) concluded that concrete feedlots had higher SSC in runoff than unsurfaced feedlots and that SSC increased with warmer weather, lower rainfall rate or increased moisture conditions.

Soil Analysis

Soil samples were taken before and after the application of Stabilizer® in both control and treatment watersheds. Soil texture for both watersheds was similar to the expected values for a Manchester sandy loam soil during the calibration period (NRCS, 2015). Following the treatment, the sand content was greater than and the silt content was less than the expected values for a Manchester soil (Table 11). Soil sampled during the treatment period in the control watershed changed from a sandy loam to a loamy fine sand, but the soil in the treatment watershed did not change texture. Both the control and treatment watersheds had a higher pH

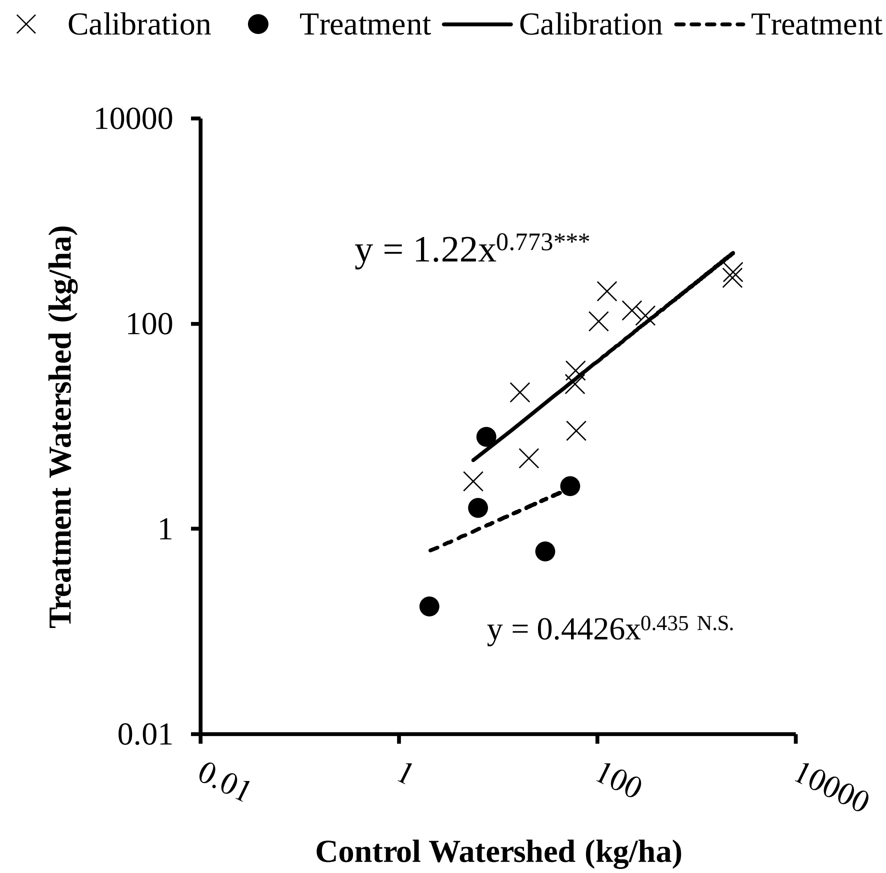

Figure 6: Paired mass SSC export from the control and treatment watersheds during the calibration and treatment periods for a dairy heavy use area in Connecticut

Table 11: Soil testing results for the control and treatment watersheds in a Connecticut dairy heavy use area before and after the application of Stabilizer®

Watershed	Period	pH	Mehlich P (ppm)	Organic Matter (%)	Total N (ppm)	Total C (%)	Total P (ppm)	Texture % Sand		
								Sand	Silt	Clay
Control	Calibration	7.5	461	5.26	2800	3.57	1038	60	36	4
Treatment	Calibration	8.06	522	7.02	2200	3.00	981	60	36	4
Control	Treatment	7.45	445	4.80	2000	2.71	992	78	18	4
Treatment	Treatment	7.97	466	4.88	1900	2.61	891	74	22	4

than expected (pH 4.5-6) during both the treatment and calibration periods. The organic matter content also was higher in both watersheds than reported for this soil type (3.5%). Phosphorus concentrations measured in our soils were high compared to the expected average range of 500 to 1000 ppm (Troeh and Thompson, 2005). Concentrations of TN in soils were in the range of expected values for A horizons of 200 to 5,000 ppm but were higher than expected values for cultivated soils (1,500 ppm) (Brady and Weil, 2008).

Conclusions

Runoff from heavy use areas is predictably high in SSC, TN, TP and *E.coli*. In some cases, concentrations of these water quality characteristics can be higher than those observed in untreated domestic watewater. The concentrations measured in runoff from the two watersheds ranged widely from storm to storm which was consistent with other studies. Stabilizer® reduced discharge, mass export of TP and mass export of SSC. There was no statistical change in the mass export of *E. coli* and TN, but an increase in TN concentrations did occur following the treatment. Scraping of surface materials may have had a confounding effect on our results by temporarily increasing surface detention within both watersheds. Based on the results of the experiment, psyllium should be considered as a possible soil amendment for use on dairy heavy use areas but additional research is needed on psyllium’s effect on nitrogen concentration. The length of time the treatment may remain effective is unknown.

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APPENDIX A: Data for the calibration and treatment periods for the Stabilizer® experiment

Table A1: Calibration period data for the control and treatment watersheds from a dairy heavy use area in Connecticut

Date	Station	Manual Rain Gage (cm)	Recording Rain Gage (cm)	Discharge (m^3)	Depth (cm)	Runoff Coefficient	SSC (mg L^{-1})	*E. coli* (CFU 100 ml^{-1})	TN (mg L^{-1})	TP (mg L^{-1})	Mass SSC (kg ha^{-1})	Mass N (kg ha^{-1})	Mass P (kg ha^{-1})	Mass *E. coli* (CFU ha^{-1})
6/19/13	Control	1.32	0.00	2.6	0.008	0.006	.	.	.	.	.	.	.	.
6/28/13	Control	3.18	2.57	116.4	0.373	0.118	.	.	.	.	.	.	.	.
7/9/13	Control	1.40	0.10	1.8	0.006	0.004	.	.	.	.	.	.	.	.
7/10/13	Control	5.94	5.03	1560.7	5.008	0.843	4645.83	8000000	45.44	30.40	2326.70	22.76	15.22	4006506
7/13/13	Control	1.85	1.75	216.3	0.694	0.374	860.00	6000000	75.72	17.50	59.70	5.26	1.21	416487
8/3/13	Control	4.14	2.16	501.2	1.609	0.389	14360.00	4000000	43.52	37.44	2309.76	7.00	6.02	643386
8/9/13	Control	4.19	3.86	593.4	1.904	0.454	1606.67	27000000	31.40	17.32	305.92	5.98	3.30	5140965
8/27/13	Control	3.40	2.13	138.5	0.444	0.131	127.12	5000000	12.16	7.36	5.65	0.54	0.33	222174
9/3/13	Control	2.24	2.24	19.5	0.063	0.028	.	.	.	.	.	.	.	.
9/13/13	Control	2.69	2.11	260.3	0.835	0.310	741.18	15000000	15.28	8.60	61.90	1.28	0.72	1252760
9/23/13	Control	2.29	1.93	139.5	0.448	0.196	462.50	7000000	18.21	13.40	20.71	0.82	0.60	313397
9/24/13	Control	0.05	0.00	1.1	0.004	0.072	.	.	.	.	.	.	.	.
10/8/13	Control	0.79	0.79	56.9	0.183	0.232	914.29	11000000	22.20	14.04	16.69	0.41	0.26	200811
11/27/13	Control	4.83	3.99	1357.6	4.357	0.903	515.96	2000000	15.43	9.64	224.77	6.72	4.20	871285
4/16/14	Control	3.63	2.59	394.5	1.266	0.349	482.35	8000000	26.17	14.20	61.07	3.31	1.80	10128587
4/27/14	Control	2.11	1.96	11.4	0.036	0.017	.	.	.	.	.	.	.	.
5/2/14	Control	5.77	4.50	1197.2	3.842	0.666	328.36	3000000	22.26	11.70	126.15	8.55	4.49	1152550
5/17/14	Control	3.86	3.07	857.6	2.752	0.713	378.95	4000000	13.20	8.10	104.29	3.63	2.23	1100855
5/23/14	Control	4.47	3.33	901.1	2.892	0.647	.	.	.	.	.	.	.	.
6/19/13	Treatment	1.32	0.00	0.0	0.000	0.000	.	.	.	.	.	.	.	.
6/28/13	Treatment	3.18	2.57	0.0	0.000	0.000	.	.	.	.	.	.	.	.
7/9/13	Treatment	1.40	0.10	0.0	0.000	0.000	.	.	.	.	.	.	.	.
7/10/13	Treatment	5.94	5.03	52.4	1.062	0.179	2961.54	26000000	67.20	34.64	314.56	7.14	3.68	2761590
7/13/13	Treatment	1.85	1.75	5.7	0.115	0.062	2216.67	2000000	74.20	35.20	25.55	0.86	0.41	23055
8/3/13	Treatment	4.14	2.16	24.0	0.485	0.117	5700.00	3000000	41.92	33.60	276.56	2.03	1.63	145558
8/9/13	Treatment	4.19	3.86	39.7	0.804	0.192	1472.73	28000000	50.44	29.60	118.33	4.05	2.38	2249785
8/27/13	Treatment	3.40	2.13	7.5	0.153	0.045	188.89	6000000	18.96	14.64	2.88	0.29	0.22	91533

Table A1: Calibration period data for the control and treatment watersheds from a dairy heavy use area in Connecticut (continued)

Date	Station	Manual Rain Gage (cm)	Recording Rain Gage (cm)	Discharge (m^3)	Depth (cm)	Runoff Coefficient	SSC (mg L^{-1})	*E. coli* (CFU 100 ml^{-1})	TN (mg L^{-1})	TP (mg L^{-1})	Mass SSC (kg ha^{-1})	Mass N (kg ha^{-1})	Mass P (kg ha^{-1})	Mass *E. coli* (CFU ha^{-1})
9/3/13	Treatment	2.24	2.24	2.8	0.057	0.025	.	.	.	.	.	.	.	.
9/13/13	Treatment	2.69	2.11	4.8	0.098	0.036	911.11	5000000	17.16	13.72	8.94	0.17	0.13	49036
9/23/13	Treatment	2.29	1.93	7.2	0.146	0.064	333.33	5000000	16.22	12.76	4.86	0.24	0.19	72837
9/24/13	Treatment	0.05	0.00	0.0	0.000	0.000	.	.	.	.	.	.	.	.
10/8/13	Treatment	0.79	0.79	4.4	0.090	0.114	2360.00	42000000	66.32	39.16	21.25	0.60	0.35	378176
11/27/13	Treatment	4.83	3.99	38.0	0.769	0.159	1742.86	4000000	45.84	23.48	134.04	3.53	1.81	307633
4/16/14	Treatment	3.63	2.59	3.3	0.068	0.019	5083.33	40000000	53.47	82.20	34.40	0.36	0.56	270699
4/27/14	Treatment	2.11	1.96	0.0	0.000	0.000	.	.	.	.	.	.	.	.
5/2/14	Treatment	5.77	4.50	22.1	0.447	0.078	4600.00	60000000	128.35	82.40	205.78	5.74	3.69	2684050
5/17/14	Treatment	3.86	3.07	35.5	0.719	0.186	1460.00	52000000	46.48	29.14	105.00	3.34	2.10	3739776
5/23/14	Treatment	4.47	3.33	18.5	0.375	0.084	.	.	.	.	.	.	.	.

TableA2: Treatment period data for the control and treatment watersheds from a dairy heavy use area in Connecticut

Date	Station	Manual Rain Gage (cm)	Recording Rain Gage (cm)	Discharge (m^3)	Depth (cm)	Runoff Coefficient	SSC (mg L^{-1})	*E. coli* (CFU 100 ml^{-1})	TN (mg L^{-1})	TP (mg L^{-1})	Mass SSC (kg ha^{-1})	Mass N (kg ha^{-1})	Mass P (kg ha^{-1})	Mass *E. coli* (CFU ha^{-1})
7/4/14	Control	1.12	1.05	404.704	1.299	0.457	413.33	24000000	14.77	7.14	53.68	1.92	0.93	3116866.30
7/27/14	Control	0.77	0.72	23.956	0.077	0.039	.	.	.	.	.	.	.	.
8/14/14	Control	1.84	.	273.059	0.876	0.187	88.00	0	7.89	5.82	7.71	0.69	0.51	0.00
11/17/14	Control	1.45	1.15	292.173	0.938	0.255	.	.	.	.	.	.	.	.
11/24/14	Control	0.34	0.27	0.651	0.002	0.002	.	.	.	.	.	.	.	.
4/6/15	Control	0.39	0.3	27.026	0.087	0.088	736.84	0	.	.	6.39	.	.	0.00
4/9/15	Control	0.84	0.52	73.482	0.236	0.111	1288.89	40000	.	.	30.39	.	.	943.22
4/13/15	Control	0.36	0.24	7.617	0.024	0.027	844.44	40000	.	.	2.06	.	.	97.77
7/4/14	Treatment	1.12	1.05	0.906	0.018	0.006	1400.00	59000000	58.40	15.30	2.57	0.11	0.03	108279.64
7/27/14	Treatment	0.77	0.72	0.000	0.000	0.000	.	.	.	.	.	.	.	.
8/14/14	Treatment	1.84	.	5.918	0.120	0.026	647.62	1000000	34.20	17.82	7.76	0.41	0.21	11986.46
11/17/14	Treatment	1.45	1.15	0.538	0.011	0.003	.	.	.	.	.	.	.	.
11/24/14	Treatment	0.34	0.27	0.000	0.000	0.000	.	.	.	.	.	.	.	.
4/6/15	Treatment	0.39	0.3	1.586	0.032	0.032	493.33	10000	.	.	1.58	.	.	32.12
4/9/15	Treatment	0.84	0.52	0.651	0.013	0.006	450.00	10000	.	.	0.59	.	.	13.19
4/13/15	Treatment	0.36	0.24	0.425	0.009	0.009	200.00	0	.	.	0.17	.	.	0.00

APPENDIX B: Shapiro Wilk Normality Test

Table B1: Shapiro Wilk normality test data for observations of runoff quantity and quality from a paired watershed study on a dairy heavy use area in Connecticut

	Non Transformed		Log_{10} Transformed	
Characteristic	W	P Value < W	W	P Value < W
Discharge (m^3)	0.70	<0.001	0.94	0.086
Depth (cm)	0.75	<0.001	0.95	0.22
Runoff Coefficient	0.84	0.001	0.96	0.43
SSC (mg L^{-1})	0.65	<0.001	0.98	0.96
Mass SSC (kg ha^{-1})	0.45	<0.001	0.97	0.63
TP (mg L^{-1})	0.76	<0.001	0.94	0.18
Mass P (kg ha^{-1})	0.66	<0.001	0.96	0.50
TN (mg L^{-1})	0.85	0.002	0.94	0.19
Mass N (kg ha^{-1})	0.70	<0.001	0.93	0.14
E. coli (CFU 100 mL^{-1})	0.76	<0.001	0.92	0.058
Mass *E. coli* (CFU ha^{-1})	0.78	<0.001	0.97	0.66

APPENDIX C: ANOVA Regression Analysis

Table C1: ANOVA of regression results from a study on a dairy heavy use area in Connecticut for discharge

Analysis of Variance				
Source	DF	Mean Square	F Value	Pr>F
Model	1	2559.36657	30.05	<0.001
Error	12	85.09214		
Corrected Total	13			

Table C2: ANOVA of regression results from a study on a dairy heavy use area in Connecticut for log transformed discharge

Analysis of Variance				
Source	DF	Mean Square	F Value	Pr>F
Model	1	1.77278	26.01	<0.001
Error	12	0.06816		
Corrected Total	13			

Table C3: ANOVA of regression results from a study on a dairy heavy use area in Connecticut for runoff depth

Analysis of Variance				
Source	DF	Mean Square	F Value	Pr>F
Model	1	1.04885	30.02	<0.001
Error	12	0.03494		
Corrected Total	13			

Table C4: ANOVA of regression results from a study on a dairy heavy use area in Connecticut for log transformed runoff depth

Analysis of Variance				
Source	DF	Mean Square	F Value	Pr>F
Model	1	1.76977	26.02	<0.001
Error	12	0.06802		
Corrected Total	13			

Table C5: ANOVA of regression results from a study on a dairy heavy use area in Connecticut for SSC concentration

Analysis of Variance				
Source	DF	Mean Square	F Value	Pr>F
Model	1	12196718	4.95	0.0502
Error	10	2463184		
Corrected Total	11			

Table C6: ANOVA of regression results from a study on a dairy heavy use area in Connecticut for log transformed SSC concentration

Analysis of Variance				
Source	DF	Mean Square	F Value	Pr>F
Model	1	0.70879	4.53	0.0591
Error	10	0.15640		
Corrected Total	11			

Table C7: ANOVA of regression results from a study on a dairy heavy use area in Connecticut for mass SSC

Analysis of Variance				
Source	DF	Mean Square	F Value	Pr>F
Model	1	95863	26.15	<0.001
Error	10	3665.23091		
Corrected Total	11			

Table C8: ANOVA of regression results from a study on a dairy heavy use area in Connecticut for log transformed mass SSC

Analysis of Variance				
Source	DF	Mean Square	F Value	Pr>F
Model	1	4.10216	33.15	<0.001
Error	10	0.12374		
Corrected Total	11			

Table C9: ANOVA of regression results from a study on a dairy heavy use area in Connecticut for TN concentration

Analysis of Variance				
Source	DF	Mean Square	F Value	Pr>F
Model	1	1179.56680	1.26	0.2873
Error	10	933.84445		
Corrected Total	11			

Table C10: ANOVA of regression results from a study on a dairy heavy use area in Connecticut for log transformed TN concentration

Analysis of Variance				
Source	DF	Mean Square	F Value	Pr>F
Model	1	0.21868	3.55	0.0888
Error	10	0.06153		
Corrected Total	11			

Table C11: ANOVA of regression results from a study on a dairy heavy use area in Connecticut for mass TN

Analysis of Variance				
Source	DF	Mean Square	F Value	Pr>F
Model	1	45.04158	25.81	<0.001
Error	10	1.74484		
Corrected Total	11			

Table C12: ANOVA of regression results from a study on a dairy heavy use area in Connecticut for log transformed mass TN

Analysis of Variance				
Source	DF	Mean Square	F Value	Pr>F
Model	1	2.49512	19.35	0.0013
Error	10	0.12895		
Corrected Total	11			

Table C13: ANOVA of regression results from a study on a dairy heavy use area in Connecticut for TP concentration

Analysis of Variance				
Source	DF	Mean Square	F Value	Pr>F
Model	1	32.30966	0.05	0.8213
Error	10	601.06635		
Corrected Total	11			

Table C14: ANOVA of regression results from a study on a dairy heavy use area in Connecticut for log transformed TP concentration

Analysis of Variance				
Source	DF	Mean Square	F Value	Pr>F
Model	1	0.078551	1.13	0.3119
Error	10	0.06920		
Corrected Total	11			

Table C15: ANOVA of regression results from a study on a dairy heavy use area in Connecticut for mass TP

Analysis of Variance				
Source	DF	Mean Square	F Value	Pr>F
Model	1	11.38999	14.29	0.0036
Error	10	0.79713		
Corrected Total	11			

Table C16: ANOVA of regression results from a study on a dairy heavy use area in Connecticut for log transformed mass TP

Analysis of Variance				
Source	DF	Mean Square	F Value	Pr>F
Model	1	2.31926	31.13	<0.001
Error	10	0.07451		
Corrected Total	11			

Table C17: ANOVA of regression results from a study on a dairy heavy use area in Connecticut for *E. coli* concentration

Analysis of Variance				
Source	DF	Mean Square	F Value	Pr>F
Model	1	1.31639E12	0.00	0.9531
Error	10	5.030418E14		
Corrected Total	11			

Table C18: ANOVA of regression results from a study on a dairy heavy use area in Connecticut for log transformed *E. coli* concentration

Analysis of Variance				
Source	DF	Mean Square	F Value	Pr>F
Model	1	0.11046	0.35	0.5683
Error	10	0.31737		
Corrected Total	11			

Table C19: ANOVA of regression results from a study on a dairy heavy use area in Connecticut for mass *E. coli*

Analysis of Variance				
Source	DF	Mean Square	F Value	Pr>F
Model	1	6.551283E12	4.65	0.0564
Error	10	1.407808E12		
Corrected Total	11			

Table C20: ANOVA of regression results from a study on a dairy heavy use area in Connecticut for log transformed mass *E. coli*

Analysis of Variance				
Source	DF	Mean Square	F Value	Pr>F
Model	1	2.53548	6.59	0.0281
Error	10	0.38502		
Corrected Total	11			

Table C21: ANOVA of regression results from a study on a dairy heavy use area in Connecticut for runoff coefficient

Analysis of Variance				
Source	DF	Mean Square	F Value	Pr>F
Model	1	0.02885	10.59	0.0069
Error	12	0.00216		
Corrected Total	13			

Table C22: ANOVA of regression results from a study on a dairy heavy use area in Connecticut for log transformed runoff coefficient

Analysis of Variance				
Source	DF	Mean Square	F Value	Pr>F
Model	1	0.58146	8.65	0.0123
Error	12	0.06720		
Corrected Total	13			

APPENDIX D: ANCOVA Regression Analysis

Table D1: ANCOVA results from a study on a dairy heavy use area in Connecticut for log transformed discharge

Analysis of Variance				
Source	DF	Mean Square	F Value	Pr>F
Model	3	2.23478729	22.55	<0.001
Error	16			
Corrected Total	19			
Intercept			21.01	<0.001
Slope			2.94	0.1056

Table D2: ANCOVA results from a study on a dairy heavy use area in Connecticut for log transformed depth

Analysis of Variance				
Source	DF	Mean Square	F Value	Pr>F
Model	3	2.23458033	22.57	<0.001
Error	16	0.09900611		
Corrected Total	19			
Intercept			21.04	<0.001
Depth			2.94	0.1055

Table D3: ANCOVA results from a study on a dairy heavy use area in Connecticut for log transformed runoff coefficient

Analysis of Variance				
Source	DF	Mean Square	F Value	Pr>F
Model	3	1.33475539	13.98	<0.001
Error	16			
Corrected Total	19			
Intercept			21.58	<0.001
Slope			4.27	0.0554

Table D4: ANCOVA results from a study on a dairy heavy use area in Connecticut for log transformed SSC concentration

Analysis of Variance				
Source	DF	Mean Square	F Value	Pr>F
Model	3	0.55159863	3.84	0.0362
Error	13	0.14382310		
Corrected Total	16			
Intercept			4.30	0.0586
Slope			2.55	0.1346

Table D5: ANCOVA results from a study on a dairy heavy use area in Connecticut for log transformed mass SSC

Analysis of Variance				
Source	DF	Mean Square	F Value	Pr>F
Model	3	4.28768179	21.52	<0.001
Error	13	0.19922820		
Corrected Total	16			
Intercept			8.91	0.0105
Slope			0.62	0.4453

Table D6: ANCOVA results from a study on a dairy heavy use area in Connecticut for log transformed *E. coli* concentration

Analysis of Variance				
Source	DF	Mean Square	F Value	Pr>F
Model	3	8.97371984	3.20	0.0590
Error	13	2.80498276		
Corrected Total	16			
Intercept			3.68	0.0772
Slope			0.02	0.8874

Table D7: ANCOVA results from a study on a dairy heavy use area in Connecticut for log transformed mass *E. coli*

Analysis of Variance				
Source	DF	Mean Square	F Value	Pr>F
Model	3	13.63401195	9.42	0.0014
Error	13	1.4467573		
Corrected Total	16			
Intercept			3.25	0.0945
Slope			0.82	0.3827

Table D8: ANCOVA results from a study on a dairy heavy use area in Connecticut for log transformed TP concentration

Analysis of Variance				
Source	DF	Mean Square	F Value	Pr>F
Model	3	0.07973945	1.47	0.2676
Error	13	0.05410632		
Corrected Total	16			
Intercept			0.04	0.8440
Slope			0.38	0.5491

Table D9: ANCOVA results from a study on a dairy heavy use area in Connecticut for log transformed mass TP

Analysis of Variance				
Source	DF	Mean Square	F Value	Pr>F
Model	3	2.16381421	21.90	<0.001
Error	13	0.09878812		
Corrected Total	16			
Intercept			7.21	0.0187
Slope			4.21	0.0609

Table D10: ANCOVA results from a study on a dairy heavy use area in Connecticut for log transformed TN concentration

Analysis of Variance				
Source	DF	Mean Square	F Value	Pr>F
Model	3	0.36136049	7.56	0.0035
Error	13	0.04776956		
Corrected Total	16			
Intercept			4.78	0.0476
Slope			0.13	0.7223

Table D11: ANCOVA results from a study on a dairy heavy use area in Connecticut for log transformed mass TN

Analysis of Variance				
Source	DF	Mean Square	F Value	Pr>F
Model	3	1.40103380	10.94	<0.001
Error	13	0.12804544		
Corrected Total	16			
Intercept			1.32	0.2715
Slope			3.77	0.0741

APPENDIX E: Figures

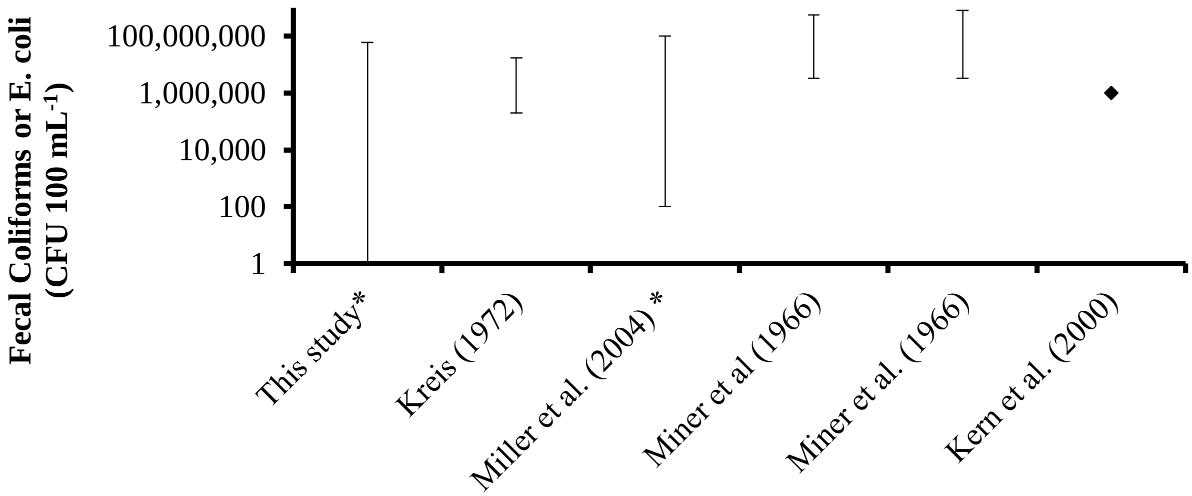

Figure E1: Fecal Coliform Bacteria (**E. coli)* concentrations for feedlots. Kern et al. (2000) is for average municipal.

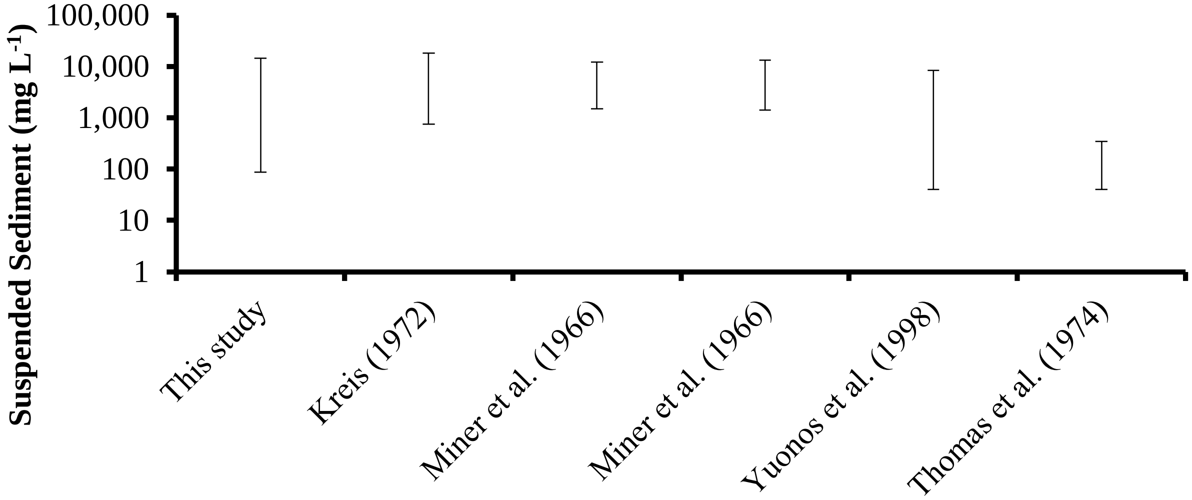

Figure E2: Suspended solids concentrations for feedlots. Thomas et al. (1974) is for untreated domestic wastewater.

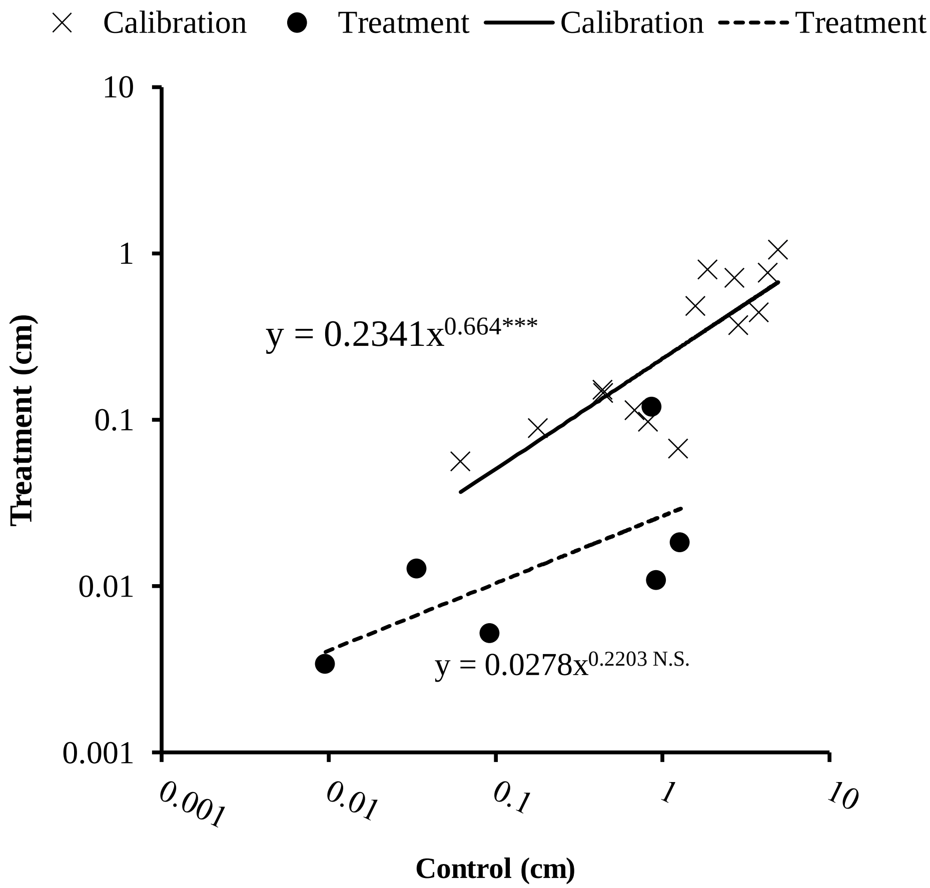

Figure E3: Paired runoff depth from the control and treatment watersheds during the calibration and treatment periods for a dairy heavy use area in Connecticut

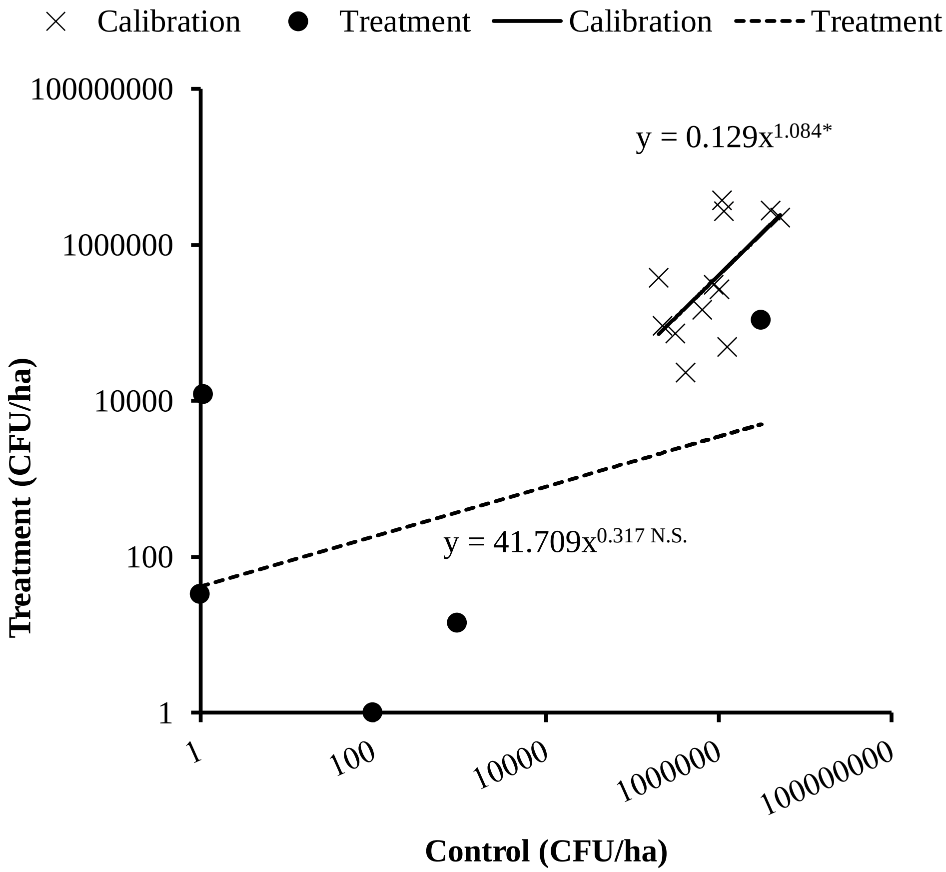

Figure E4: Paired mass *E. coli* export from the control and treatment watersheds during the calibration and treatment periods for a dairy heavy use area in Connecticut

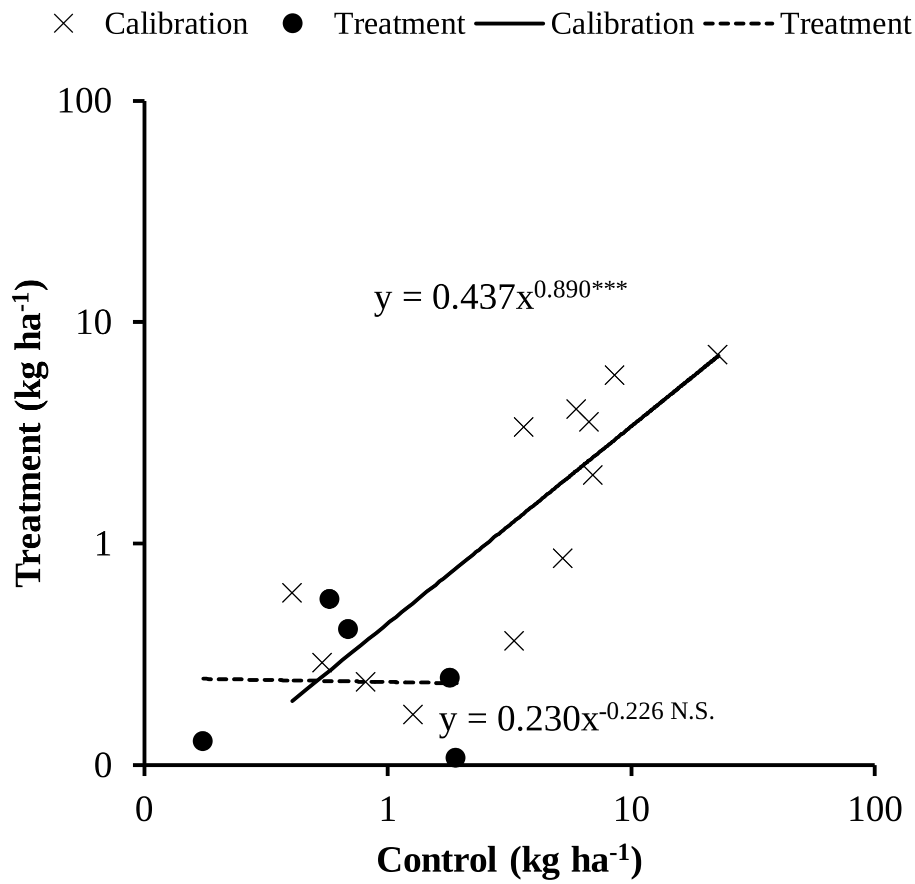

Figure E5: Paired total nitrogen mass export from the control and treatment watersheds during the calibration and treatment periods for a dairy heavy use area in Connecticut