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Influence of turfgrass coverage on nutrient and pesticide transport as affected by water and sediment displacement during surface runoff

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INFLUENCE OF TURFGRASS COVERAGE ON NUTRIENT AND PESTICIDE TRANSPORT AS AFFECTED BY WATER AND SEDIMENT DISPLACEMENT DURING SURFACE RUNOFF

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DEDICATION

I dedicate this dissertation to my wife Golda, my parents Gina and Steve, my brothers Justin, and Stewart, and the rest of my loving family.

It is not the talents we possess so much as the use we make of them that counts in the progress of the world"

-Brailsford Robertson

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ABSTRACT

Turfgrass systems such as roadsides, home lawns, and golf course roughs can vary in surface coverage. Nutrient and pesticide applications applied to these systems may pose an increased risk to adjacent water supplies from surface runoff. Therefore the objectives of this research were: 1) determine the effect surface coverage has on nutrient, pesticide, and sediment runoff and, 2) evaluate the impact of pesticide solubility on runoff losses at varying turfgrass coverages. Surface runoff research commenced in 2010 and 2011 with experimental units consisting of six turfgrass coverages (0, 25, 50, 75, and 100%) and untreated bare soil. Coverages each received a granular fertilizer, atrazine, azoxystrobin, monosodium methyl arsenate (MSMA), pendimethalin, simazine, and *S*-metolachlor treatment. Simulated rainfall was applied at 7.38 cm hr^{-1} with runoff collected and analyzed for dissolved nitrogen (DN), total nitrogen (TN), dissolved phosphorus (DP), total phosphorus (TP), total solids (TS) and each pesticide applied. Runoff volumes and TS loading decreased as turfgrass coverage increased from 0 to 100% turfgrass coverage. Total solids decreased from 1078 and 873 kg ha^{-1} from 0% coverage to 35 and 14 kg ha^{-1} at 100% turfgrass coverage in 2010 and 2011, respectively. Turfgrass coverage did not affect TN and TP with 5.76 kg N ha^{-1} and 5.60 kg P ha^{-1} lost at 100% turfgrass coverage. As turfgrass coverage increased DN and DP losses accounted for greater portions of total losses with decreasing sediment bound nutrients losses. Greater than 56% of DN and DP occurred during the first 15 min after the onset of runoff (AOR). Higher water soluble pesticides were more susceptible to loss during surface runoff. MSMA losses of 25.7% at 100% turfgrass coverage were 7 times greater than the highest pendimethalin losses observed. Similar to soluble nutrients, > 56% of dissolved pesticides were lost in the first 15 min AOR. Total atrazine losses of 12.5 and18.3% compared to total simazine losses of 10.1and 5.7% for 0

and 100% coverage respectively, indicate the importance of maintaining greater turfgrass coverage for reducing sediment bound pollutants. However, turfgrass coverage may not be as effective for reducing dissolved pollutant transport.

CHAPTER 1: LITERATURE REVIEW

Introduction

Agriculture and urban environments have been cited as significant sources of non-point surface water pollution (USEPA, 2002). Sediment, nutrient, and pesticide runoff from these systems have become an environmental concern in the United States (Haith and Rossi, 2003; Kaufmann III and Watschke, 2007; Kramer et al., 2009). These growing concerns have led to Major Farm Bill legislation implemented to protect the environment and decrease erosion and runoff from farming systems (LeBaron et al., 2008).

Turfgrass comprise more than 16 million hectares, representing the single largest crop in the United States (Kramer et al., 2009; Milesi et al., 2005). Given this high amount of land area and the associated maintenance requirements turfgrass systems may pose a risk to surface water systems from runoff (Gross et al., 1990). The application of fertilizers to maintain healthy grass swards (Balogh and Watson, 1992; Branham et al., 2005; Witteveen and Bavier, 1999) increases this potential for surface waters to be negatively impacted by eutrophication (Kohler et al., 2004; Schmidt, 2006; Shuman, 2002). Along with nutrients, pesticides can also be routinely applied to turfgrass systems.

A study conducted by Cohen et al. (1999) sampling water sources adjacent to golf courses detected 31 different pesticides at various concentrations. According to Haith and Rossi (2003) concentrations of turfgrass-applied chemicals chlorothalonil (trachloroisophthalonilnitrile), iprodione (3-(3,5-dichlorophenyl)-*N*-isopropyl-2,4-dioxoimidazolidine-1-carboxamide), quintozene (pentachloronitrobenzene), and trichlorfon (dimethyl (*RS*)-2,2,2-trichloro-1-hydroxyethylphosphonate) came near or exceeded LC_{50} levels for rainbow trout (*Salmo gairdneri*) and/or water flea (*Daphnia magna*) in surface runoff from golf greens. Runoff carrying water pollutants from turfgrass systems can flow into streams and rivers ultimately affecting surface water supplies and shallow alluvial aquifers (Burkart et al., 1997). Therefore, a better understanding of nutrient and pesticide transport from turfgrass systems would be beneficial for designing more effective best management practices (BMP).

Turfgrass Conservation Systems

Grasses are often utilized in conservation systems because of their ability to impede sediment and nutrient losses during surface runoff. Currently, grasses are utilized extensively by the United States Department of Agriculture (USDA) and the Soil Conservation Service for controlling surface runoff (Anderson et al., 1989; Bennett, 1979; Gross et al., 1990; Hayes et al., 1979; USEPA, 1983; Welterlen et al., 1989). Vegetative buffer strips are bands of native or planted vegetation, typically grasses, placed down-slope of agricultural systems and are used to filter nutrients, sediment, and pesticides during surface runoff (Dillaha et al. 1989; Rankins Jr. et al., 2005). Vegetative buffer strips are an effective erosion control practice and considered a major BMP for the protection of surface water quality (Anderson et al., 1989; Barfield et al., 1979; Hayes et al., 1979).

As the conversion of agricultural landscapes to urban landscapes increases, there is an increasing need for turfgrass buffer strips to hinder runoff from impervious neighboring water sources (Steinke et al., 2009). Moss et al. (2006) investigating graduated bermudagrass (*Cynodon dactylon* L.) buffer strips observed average losses of 1.5 and 0.5% of applied nitrogen (N) and 5.5 and 3.3% of applied phosphorus during a 51 mm h^{-1}, 60 min rainfall event across all treatments. They concluded graduated buffers established along turf areas would significantly reduce N and P losses from surface runoff.

Many processes enable turfgrasses to impede and reduce pollution transport from surface runoff. Turfgrass leaves lessen the impact force of rain drops decreasing splash erosion, while the fibrous root structure and dense organic matter layer enhances soil stability and increases nutrient and pesticide binding. According to Carrow et al. (2001), runoff losses are usually lower on established turfgrass sites because of their dense canopy. Easton et al. (2005) concluded soil infiltration could be increased from 7 to 21 cm h^{-1} when turfgrass shoot density increased from 60,000 to 120,000 shoots m^2. Therefore, managed turfgrasses such as bermudagrass, are preferred over less dense cultivars. McFarland and Hauck (2004) examined potential P runoff from bermudagrass and sorghum dairy waste fields high in soil extractable P. Bermudagrass reduced 51 and 61% of incoming PO_4-P and total phosphorus (TP) compared to sorghum (*Sorghum bicolor*).

Once surface runoff is initiated, dense vegetative growth provides an indirect pathway to obstruct water flow, reduce water velocity, trap suspended sediment, and increase infiltration (Linde et al., 1995, 1998). In addition, turfgrass canopies remove significant amounts of soil-water through evapotranspiration, thus potentially increasing soil water holding capacity (Ebdon et al., 1999). According to Easton and Petrovic (2004), evapotranspiration in turfgrass ecosystems is generally the greatest source of water removal and through this process adds to the removal of soil solution nutrients (Petrovic, 1990).

Turfgrass Nutrient Inputs

Intense management practices can increase the potential for turfgrass managers to over apply pesticides and nutrients, thus, making these areas more susceptible to pollution transfer. Pesticide and nutrient runoff may be more substantial when applications are made to exposed soil; when a heavy rainfall occurs immediately following applications especially on a fine-textured or compacted soil; on sites with considerable slopes; and where surface water from nearby land areas runs across the turf site (Carrow et al., 2001).

Newly established turfgrass swards require continuous management to accelerate canopy coverage. Rapid turfgrass establishment can aide in resisting weed encroachment through competition (Busey, 2003). Higher fertility is utilized to accelerate turfgrass establishment from seedlings, plugs, sprigs, and stolons (Turgeon, 2008). An increase in turfgrass establishment and plant vigor from fertilization gives a competitive advantage over undesirable weed species. However, increased inputs during establishment and at lower turfgrass coverage could potentially increase nutrient losses. Butler et al. (2006, 2007) evaluated mixed tall fescue (*Festuca arundinacea* Schreb.) and dallisgrass (*Paspalum dilatatum* Poir.) at different coverage percentages (bare soil, 45, 70, and 95%) and concluded 45% vegetative cover was necessary to reduce N and P losses 34% and 36%, respectively, compared to bare ground. Burwell et al. (2011), investigating N losses during common bermudagrass establishment on levee embankments and reported expected erosion and runoff reduction from the addition of N inputs did not occur, but alternatively increased N losses.

Substantial fertilizer inputs are sometimes required to maintain healthy turfgrass (Balogh and Watson, 1992; Branham et al., 2005; Witteveen and Bavier, 1999). These high fertilization requirements for establishment and maintenance increase the potential for surface losses and detrimental effects to surface water quality (Kohler et al., 2004; Schmidt, 2006; Shuman, 2002). In light of this, runoff from urban areas and golf courses are often presumed to contribute to surface water pollution (Kohler et al., 2004). However, previous research conducted (Gross et al., 1990, 1991; Harrison et al., 1993; Kohler et al., 2004; Linde et al., 1994; Morton et al., 1988) has generally concluded runoff from mature turfgrass systems to be negligible when proper maintenance and BMPs are followed.

Eutrophication is the leading cause of surface water body impairment in the U.S. (USEPA, 2002). Previous research has concluded nitrate concentrations of 0.1 mg L^{-1} (Mallin and Wheeler, 2000) and P concentrations ≥ 0.024 mg L^{-1} (Owens et al., 1998) can lead to eutrophication. The United States Environmental Protection Agency (USEPA) maximum contaminant levels are 10 mg L^{-1} NO_3-N and 0.1 mg L^{-1} for PO_4-P for human drinking water. Along with agriculture systems, urban environments have been cited as major contributors of P to surface waters (USEPA, 2002). Phosphorus levels of 0.5 to 1 mg L^{-1} have been found in runoff waters from non-treated turfgrass plots (Shuman, 2002) indicating the potential for turfgrass ecosystems as surface water contaminators.

Turfgrass Pesticide Inputs

Preemergent weed control is utilized to suppress weed encroachment before the establishment of a turfgrass sward. Herbicide applications are made at this timing to create species uniformity and increase the rate of establishment. Butler et al. (2006) concluded Tifton 85 bermudagrass could be established at a faster pace when unwanted weed species were controlled. Turfgrass fungicide applications can also be made at early growth state to ensure health and vigor. However, pesticide applications made at establishment or at an early growth stage could be applied to bare soil increasing the potential for surface runoff. These pesticides off-site movement via surface transport can have detrimental effects on surface water supplies and shallow alluvial aquifers (Burkart et al., 1997).

Turfgrass differs from traditional row-crop agriculture in that there is a fixed layer of organic matter on top of the soil profile (Gardner et al., 2000). Previous research has indicated that turfgrass leaves and thatch strongly sorb organic compounds and thus should have a significant impact on fate of pesticides applied to the system (Dell et al., 1994; Gardner et al., 2000; Lickfeldt and Branham, 1995). In soil, a fairly mobile herbicide is subject to plant uptake, sorption onto soil particles and organic matter, volatilization, microbial and chemical degradation, and solubilization by water (Hixson et al., 2009). Although pesticide concentrations in the environment are determined by numerous interacting processes and conditions (Smith and Bridges, 1996) one risk factor commonly assessed is pesticide water-solubility.

A study conducted by Smith and Bridges (1996) investigated highly water soluble 2,4-D, dicamba, and mecoprop movement from simulated golf course greens and fairways. Researchers found that 75% of applied herbicides were transported from plots during the first rainfall. During this event the greatest losses of 2,4-D ((2,4-dichlorophenoxy)acetic acid), dicamba (3,6-dichloro-o-anisic acid), and mecoprop ((*RS*)-2-(4-chloro-o-tolyloxy)propionic acid) were lost with corresponding concentrations of 811, 279, and 820 µg L^{-1}, respectively. Conversely, Lee et al. (2000) evaluated a low water soluble herbicide pendimethalin (*N*-(1-ethylpropyl)-2,6-dinitro-3,4-xylidine) and concluded little environmental carryover would occur in turfgrass systems. Total losses from 20 cm of rainfall for 10 days reached 0.81 and 1.22% of initial applications at 2.25 and 4.50 kg/ha, respectively. Lee et al. (2000) reported 90% of pendimethalin remained at the top 10 cm of soil 90 days after initial treatments.

Along with pesticide water solubility, physical partitioning processes such as soil-organic-carbon partition coefficient (K_{OC}) and *n*-octanol-water partition coefficient (K_{OW}) have been utilized in prediction models for pesticide mass transport (Rice et al., 2010). Low water soluble pesticides such as pendimethalin typically equate with high K_{OC} values indicating a strong affinity for soil and organic matter sorption. These pesticides can be susceptible to off-target movement via sediment loss during surface runoff. Lee et al. (2000) posited croplands that receive pendimethalin applications could accrue greater runoff losses though TS movement.

Similar to pendimethalin, simazine (6-chloro-*N*2,*N*4-diethyl-1,3,5-triazine-2,4-diamine) and (dithiopyr (*S*,*S*'-dimethyl 2-difluoromethyl-4-isobutyl-6-trifluoromethylpyridine-3,5-dicarbothioate) are two commonly applied turfgrass preemergent herbicides that have high potential for retention within thatch, mat, and surface soils (Schleicher et al., 1995).

Hong and Smith (1998) investigated surface movement of EC formulation dithiopyr from golf course fairways. They concluded > 2% of applied dithiopyr was lost in runoff water 11 days after application. The highest concentration observed in runoff was 39.27μg L^{-1} after 38 mm of continuous simulated rainfall. This concentration was much lower compared to the higher water soluble pesticides evaluated by Smith and Bridges (1996).

Hixson et al. (2009) concluded that as turfgrass systems age, simazine leaching potential decreases due to a large capacity for biodegradation and binding to organic matter. Compared to pesticides with high water solubility, lower water soluble pesticides have less potential for initial surface runoff from rainfall events.

Stewart et al. (1975) identified high water solubility, low soil-organic-carbon partition coefficients (K_{OC}), long environment half-lives, and high leaching potentials as characteristics most common to chemicals routinely found in groundwater.

Pesticide Runoff Modeling

Several studies have evaluated turfgrass hydrology, chemical transport, and hydrological transport modeling (Kramer et al., 2009) with numerous computer based models for agriculture chemical transport modeling. EPIC (King and Balogh, 1997; King and Balogh, 1999), EXPRESS (Roy et al., 2001), GLEAMS (Ma et al., 1999a; Smith and Tilloston, 1993), Opus (Ma et al., 1999b), PRZM (Durborow et al., 2000; Ma et al., 1999a), RZWQM (Shwartz and Shuman, 2005), and SWAT(King and Balogh, 2001), have been utilized for predicting chemical transport in turf (Kramer et al., 2009). However, these models were created for predicting chemical losses in agricultural systems and may not be as suitable to characterize chemical transport in turfgrass systems. Currently, TurfPQ (Haith, 2001; Haith, 2002) is the only model developed exclusively for turfgrass systems (Kramer et al., 2009). Research (Haith and Duffany, 2007; Haith and Rossi, 2003; Kramer et al., 2009; Vincelli, 2004) evaluating the efficacy of TurfPQ to predict pesticide movement have resulted in varying degrees of accuracy.

TurfPQ is based on a curve number calculation for runoff volume and linear partitioning of pesticide into absorbed and dissolved components during a precipitation or irrigation event (Haith, 2001). Recent research conducted by Kramer et al. (2009) observed an underestimation of pesticide runoff when TurfPQ was compared to field generated data. Kramer et al. (2009) suggested more research should be conducted evaluating TurfPQ compared to actual pesticide runoff data in order to continue refining the model's capabilities. They suspected chemical transport underestimations were the result of infiltration and runoff inaccuracies. Models such as TurfPQ rely on numerous equations and environmental conditions to predict runoff potential of a given chemical. Therefore, further research regarding model inputs is required to investigate TurfPQ and its ability to accurately predict chemical transport.

Based on erosion research evaluating vegetative coverage effects, the hypothesis that turfgrass coverage percentage has a direct effect on surface runoff should translate into an effect on chemical and nutrient movement (Burwell, 2010). Previous modeling research utilizing TurfPQ has focused on mature, complete turfgrass canopies. No research has specifically evaluated the potential for TurfPQ to predict chemical transport in turfgrasses of varying coverage.

Turfgrass Coverage Effect on Nutrient and Pesticide Runoff

Research evaluating sediment, nutrient and pesticide runoff from turfgrass environments has primarily focused on bare soil and/or established turfgrass (>75% coverage) sites (Haith, 2001; Haith and Rossi, 2003; Hong and Smith, 1997; Kauffman III and Watschke, 2007; Kramer et al., 2009; Lee et al., 2000; Moss et al., 2005; Smith and Bridges, 1996; Steinke et al., 2009; Vincelli, 2004). Although these conditions may represent areas such as intensively managed golf courses and athletic fields, they may not accurately characterize the majority of grassed areas in the United States.

The effect of increasing cover to reduce runoff volume and TS movement is well documented in the scientific literature (Burwell et al., 2011; Butler et al., 2006, 2007; Gross et al., 1990, 1991; Krenitsky et al. 1998; Kussow 2008; Linde and Watschke, 1997). Many grassed areas are not highly managed in terms of fertilization, pesticide application, frequency of cultural practices, and irrigation compared to golf courses and athletic fields. Less intensely managed turfgrasses tend to equate to reduced turfgrass canopy cover (Turgeon, 2008). Easton and Petrovic (2008) concluded high maintenance turfgrass areas reduced runoff volume two times that of low maintenance turfgrass areas.

Drastic differences in surface runoff sediment losses have been reported between bare soil and established turfgrass environments. Gross et al. (1991) reported soil losses of 15 kg ha^{-1} for bare soil and 225 kg ha^{-1} for an established tall fescue stand after a 30 minute simulated rainfall event. Previous research discussed indicated the greater runoff potential with soil movement for low soluble, strongly adsorbing herbicides. Lang (1979) concluded water channeling can occur at 70% vegetative coverage and similar losses may be observed when compared to bare soil. If this were to occur after pesticide and nutrient applications, losses at 70% turfgrass coverage may be substantial.

A study evaluating the effect of turfgrass grow-in on levee systems by Burwell et al. (2011) determined N losses and TS loading did not decrease during bermudagrass establishment with additional N inputs. Butler et al. (2006, 2007) concluded 45% vegetative cover was necessary to reduce N and P losses 34% and 36%, respectively, compared to bare ground. Interestingly, Burwell et al. (2011) and Butler et al. (2006, 2007) did not observe decreases in N and P losses among higher vegetative covers.

Greatest nutrient and pesticide losses have been reported to occur during the first runoff event post nutrient and pesticide application (Easton and Petrovic, 2004; Gaudreau et al., 2002; Kelling and Peterson, 1975; Rector et al. 2003a, 2003b). If managers rely on natural rainfall events, especially in low managed areas such as but not limited to roadsides, home lawns, golf courses and utility areas, runoff losses could be substantial from turfgrassed areas.

Given the increase in local and state regulations concerning nutrient application to turfgrasses (Rosen and Horgan, 2005; Throssell et al., 2009) and growing concerns for sediment and pesticide runoff (Haith and Rossi, 2003; Kaufmann III and Watschke, 2007; Kramer et al., 2009) an investigation is required to assess turfgrass coverage influence on pesticide and nutrient runoff. This information would provide greater insight and aide practitioners develop more environmentally sound management practices and improve environmental stewardship.

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CHAPTER 2: INFLUENCE OF TURFGRASS COVERAGE ON NITROGEN AND PHOSPHORUS TRANSPORT AS AFFECTED BY WATER AND SEDIMENT DISPLACEMENT DURING SURFACE RUNOFF

Introduction

Sectors such as agriculture and urban environments have been cited as significant sources of non-point surface water pollution (USEPA, 2002). In the United States, grassed areas comprise more than 16 million hectares, representing the single largest crop (Kramer et al., 2009; Milesi et al., 2005) and potentially a significant contributor to surface water impairment. The application of fertilizers to maintain healthy grass swards (Balogh and Watson, 1992; Witteveen and Bavier, 1999; Branham et al., 2005) increases this potential for surface waters to be negatively impacted by eutrophication (Shuman, 2002; Kohler et al., 2004; Schmidt, 2006). Nitrate concentrations of 0.1 mg L^{-1} (Mallin and Wheeler, 2000) and P concentrations ≥ 0.024 mg L^{-1} (Owens et al., 1998) can lead to eutrophication. Increased nutrient losses from agricultural systems has led to current USEPA regulations requiring water contaminant levels be below 10 mg L^{-1} NO_3-N and 0.1 mg L^{-1} PO_4-P (USEPA, 2002).

Scientists have generally concluded N and P runoff from established grass systems to be negligible (Gross et al., 1990; Harrison et al., 1993; Kohler et al., 2004; Linde et al., 1994; Morton et al., 1988) due to dense turfgrass canopies and fibrous root structures (Carrow et al., 2001). It has been proposed that dense vegetation which turfgrass provides is an indirect pathway for lateral water flow that reduces water velocity, traps suspended solids, and increases water infiltration (Butler et al., 2007; Gross et al., 1990 and 1991; Linde et al., 1995, 1998). Vegetation, such as grasses, also reduce soil water through evapotranspiration to increase soil infiltration capacity (Ebdon et al., 1999); a factor that can influence surface runoff occurrence. The efficacy of grass swards to limit runoff and reduce erosion has resulted in their adoption in conservation systems established by the USDA and the Soil Conservation Service (Anderson et al., 1989; Bennett, 1979; Gross et al. 1990; Hayes et al., 1979; Welterlen et al., 1989; USEPA, 1983).

To date, most research examining surface runoff losses from turfgrasses have been conducted on highly managed turfgrass canopies (Kelling and Peterson, 1975; Moss et al., 2006 and 2007; Shuman, 2002). Although these conditions may represent areas such as intensively managed golf courses and athletic fields, they may not accurately characterize other grassed areas in the United States. For example, Burwell et al. (2011) investigated N losses during common bermudagrass establishment on levee embankments with slopes ranging from 23 to 33% and reported erosion and runoff reduction expected from additional N inputs did not occur, but alternatively increased N losses. Butler et al. (2006, 2007) evaluated cover percentages (bare soil, 45, 70, and 95%) of mixed tall fescue (*Festuca arundinacea* Schreb.) and dallisgrass (*Paspalum dilatatum* Poir.) vegetation on N and P runoff from animal grazing pasture. They concluded 45% vegetative cover was necessary to reduce N and P losses 34% and 36%, respectively, compared to bare ground. Interestingly, Burwell et al. (2011) and Butler et al. (2006, 2007) did not observe further decreases in N and P losses among higher vegetative covers.

The effect of increasing cover to reduce runoff volume and total solids (TS) movement is well documented in the scientific literature (Butler et al., 2006, 2007; Burwell et al., 2011; Gross et al., 1990, 1991; Krenitsky et al. 1998; Kussow 2008; Linde and Watschke, 1997). However, the relationship between the percentage of grass coverage and potential nutrient runoff is not clearly defined. Most grassed areas are not as highly managed in terms of fertilization, pesticide application, frequency of cultural practices, and irrigation compared to golf courses and athletic fields. As a result, these less intensely managed turfgrasses tend to equate to reduced turfgrass canopy cover (Turgeon, 2008). Easton and Petrovic (2008) concluded high maintenance turfgrass areas reduced runoff volume two times that of low maintenance turfgrass areas. Therefore, fertilizers applied in accordance with anticipated precipitation to non-irrigated turf may be more susceptible to nutrient offsite transport as a direct result of lower runoff resistance. Nutrient losses are typically highest from the first runoff event post fertilizer application (Easton and Petrovic, 2004; Gaudreau et al., 2002; Kelling and Peterson, 1975), therefore a greater emphasis should be placed on practices that reduce initial nutrient transport.

Given the increase in local and state regulations concerning nutrient application to turfgrasses (Throssell et al., 2009), information concerning the interaction of N and P with turfgrass coverage would provide greater insight for developing more environmentally sound fertility management practices for non-irrigated and less intensely managed turfgrass areas. Therefore, the primary objective of this study was to examine the effect turfgrass coverage has on applied N and P losses from surface runoff during a single-rainfall event. The second objective was to evaluate N and P losses during 30-mins of continuous runoff.

Materials and Methods

Site Characterization and Treatments

Surface runoff experiments commenced 9 Sept. 2010 and 17 Mar 2011 on 10% sloped embankments located at the Louisiana State University Agricultural Center Burden Research Facility in Baton Rouge, LA. The soil texture was a silt loam consisting of 13.25 % sand 63.6 % silt, and 23.05% clay. Slope vegetation consisted of St. Augustinegrass [*Stenotaphrum secundatum* (Walt.) Kuntze] in 2010 and perennial ryegrass (*Lolium perenne* L.) in 2011. General maintenance included mowing to 7.62 cm weekly with clippings collected. No fertilizers were applied within 6 months of each experiment. Soil tests were collected and analyzed by the Louisiana State University Agricultural Center Soil Testing and Plant Analysis Lab (LSU STPAL) prior to each experiment that and resulted in soil pH 6.7 and 64 kg P ha^{-1} and 138 kg K ha^{-1}.

Experimental units consisted of six treated turfgrass coverages (0, 25, 50, 75, and 100%) and included an unfertilized bare soil control. Coverages were arranged in a randomized complete block design with 3 replications in 2010 and 2011. Replications were blocked by day rainfall simulation occurred. All simulations occurred within a 6 d period. Four to six weeks prior to rainfall simulation, turfgrass coverages were achieved by randomly removing turfgrass plugs within each treatment using a 10 cm diameter probe. Efforts were made to remove as little soil as possible during turfgrass harvesting with uneven areas immediately leveled with the same soil texture.

All turfgrass coverage treatments with the exception of the unfertilized bare soil control were fertilized at 48.8 kg N ha^{-1} and 48.8 kg P_2O_5 ha^{-1} using a commonly available granular fertilizer (19-19-19, LESCO®, Cleveland, OH). The N source was comprised of 11.7% urea-N and 7.3% ammoniacal-N. Fertilizer was applied by hand 24 hrs prior to rainfall simulation using shaker jars to insure even distribution. No irrigation was applied post-fertilizer application in order to simulate fertilization of non-irrigated turfgrass.

Rainfall Simulation

Rainfall simulations protocols adhered to the USDA National P Research Project protocol for rainfall simulation (USDA, 2008). Stainless steel runoff trays (15.25 cm x 208.5 cm x 75 cm) with an internal area of 1.5 m^2 were inserted into the top 8 cm of soil 2 to 3 days prior to rainfall simulation. At the base of each tray, stainless steel troughs directed runoff water gravimetrically through 2.54 cm diameter exit ports and into collection reservoirs.

A Tlaloc 3000 rainfall simulator (Joern's Inc., West Lafayette, IN) based on designs of Miller (1987) and Humphry et al. (2002) was used for rainfall simulations. Rainfall was simulated using a specialized spray nozzle (Spraying Systems Co. Fulljet ½HH SS 50WSQ) with a spray angle of 104° (±5%) (USDA, 2008) mounted 3 m above the soil surface. To approximate local conditions, rainfall intensity for a two-year, one-hour precipitation extreme of 7.32 cm h^{-1}, was applied during rainfall simulations (LA Office of State Climatology, 2009). A municipal water source was utilized with samples collected and analyzed for electrical conductivity, pH, NH_3-N, NO_3-N, Total Kjeldahl N (TN), dissolved reactive P (DP), TP and cation concentrations (Appendix A.1).

Rainfall simulations were initiated 24 hours post-fertilizer application. Initiation of surface runoff from each treatment was demarked at continuous water flow into collection reservoirs. Runoff was collected *in toto* for 30 min.

Data Collection

Immediately preceding rainfall simulation, volumetric soil moisture (m^3 m^{-3}) content was recorded for each experimental unit utilizing a portable TH_2O probe (Dynamax Inc., Houston, TX)(Appendix A.2). For each simulated rainfall event, total runoff volume (L) for the 30 min runoff event was collected with 1 L composite samples analyzed. In 2011, runoff was collected every minute for the first 10 min followed by a collection interval of every 5 min the remaining 30-min duration. This change in collection protocol was employed to calculate water flow for 0, 50, and 100% turfgrass coverages. In addition, 1-liter subsamples were collected every 5 min for 30 min and analyzed for TS and nutrient concentrations. Water was agitated in each collection reservoir to ensure the sample was well mixed allowing for a representative sample to be collected. All samples were immediately stored on ice in the field before being transported to the laboratory and stored at 4°C.

Total Solids, Nitrogen, and Phosphorus Analyses

Water samples were analyzed for NH_3-N, NO_3-N, TN, DP, TP, and TS within 48 hrs after collection. Total solids analysis was performed by centrifuging 800 mL of each sample at 3500 rpm for 15 min using an Algera-6 table-top centrifuge (Beckman Coulter Inc., Brea, CA). Liquid was decanted and the remaining soil was dried for 7 d before mass was recorded.

Nitrogen (NH_3-N, NO_3-N, and TN) analyses were performed following EPA Method 351.2 and 976.06. Total N samples (25 mL) were digesting in a sulfuric acid-mercuric sulfate-potassium sulfate solution at 408 C to convert all N to NH_4-N for analysis. Soluble N, NH_4-N and NO_3-N, were determined following a modified version of EPA Method 351.2 that excluded digestion procedures. Total and soluble N were quantitated through titanous chloride reduction of NO_3-N to NH_3-N prior to automated colorimetric analysis (Quickchem 8500 FIA, Lachat Instruments, Milwaukee, WI, USA) at a detection limit of 0.01 mg L^{-1}. Soluble N data for NH_3-N and NO_3-N are combined and presented as dissolved N (DN).

Phosphorus analyses were performed in accordance to EPA Methods 3052, 200.7 and 985.01. Potassium persulfate digestions as described by Pierzynski (2000) were used to prepare samples for total phosphorus (TP) analysis. Samples analyzed for dissolved phosphorus (DP) were prepared by filtering samples through a 0.45 µm filter (Whatman International Ltd., Maidstone, England). Dissolved P and TP were analyzed using Inductively Coupled Plasma Atomic Emission Spectroscopy (ICP-AES) at a detection limit of 56 µg L^{-1}.

Statistical Analysis

Groundcover treatments were arranged in a randomized complete block design with three replications in 2010 and 2011. Data were analyzed according to the Analysis of Variance (ANOVA; α=0.05) following the general linear method using the SAS (SAS Institute, 2000) statistical software. Post-hoc testing was performed on cumulative means for DN, TN, DP, and TP using Fisher's protected least significant difference (LSD; $\alpha = 0.05$). Nutrient data recorded over time were analyzed using repeated measures with means separated using standard errors. Runoff volume, water flow, and TS losses were regressed against turfgrass coverage with standard errors calculated for water flow data. Data for all measurements were pooled across years when interaction terms had a p-value ≥ 0.20.

Results

Effect of Turfgrass Coverage on Total Runoff Volume, Flow, and Total Solid Losses

Surface runoff volume losses were correlated to turfgrass cover for each rainfall simulation in order to assess percentage turfgrass cover effects on runoff resistance. In 2010 and 2011, 30-min composite surface runoff volumes exhibited negative quadratic trends across turfgrass coverages (Figure 2.1). Turfgrass coverage at 0% up to 50% had the highest runoff volumes ranging between 33 and 38 in 2010 and 26 to 34 L in 2011 with average declines to 27 and 26 L in 2010 and 2011, respectively, at 75% turfgrass coverage. Lowest runoff volumes occurred at 100% turfgrass coverage with average losses of 14 and 23 L in 2010 and 2011, correspondingly. Turfgrass coverages of 0 and 50% exhibited the greatest increase in water flow within the first 10 min after the onset of runoff (AOR) compared to 100% turfgrass coverage. Thirty-min AOR static water flows of 1.08 and 1.09 L min^{-1} for 0, and 50% respectively, were greater than 0.82 L min^{-1} observed for 100% turfgrass coverage.

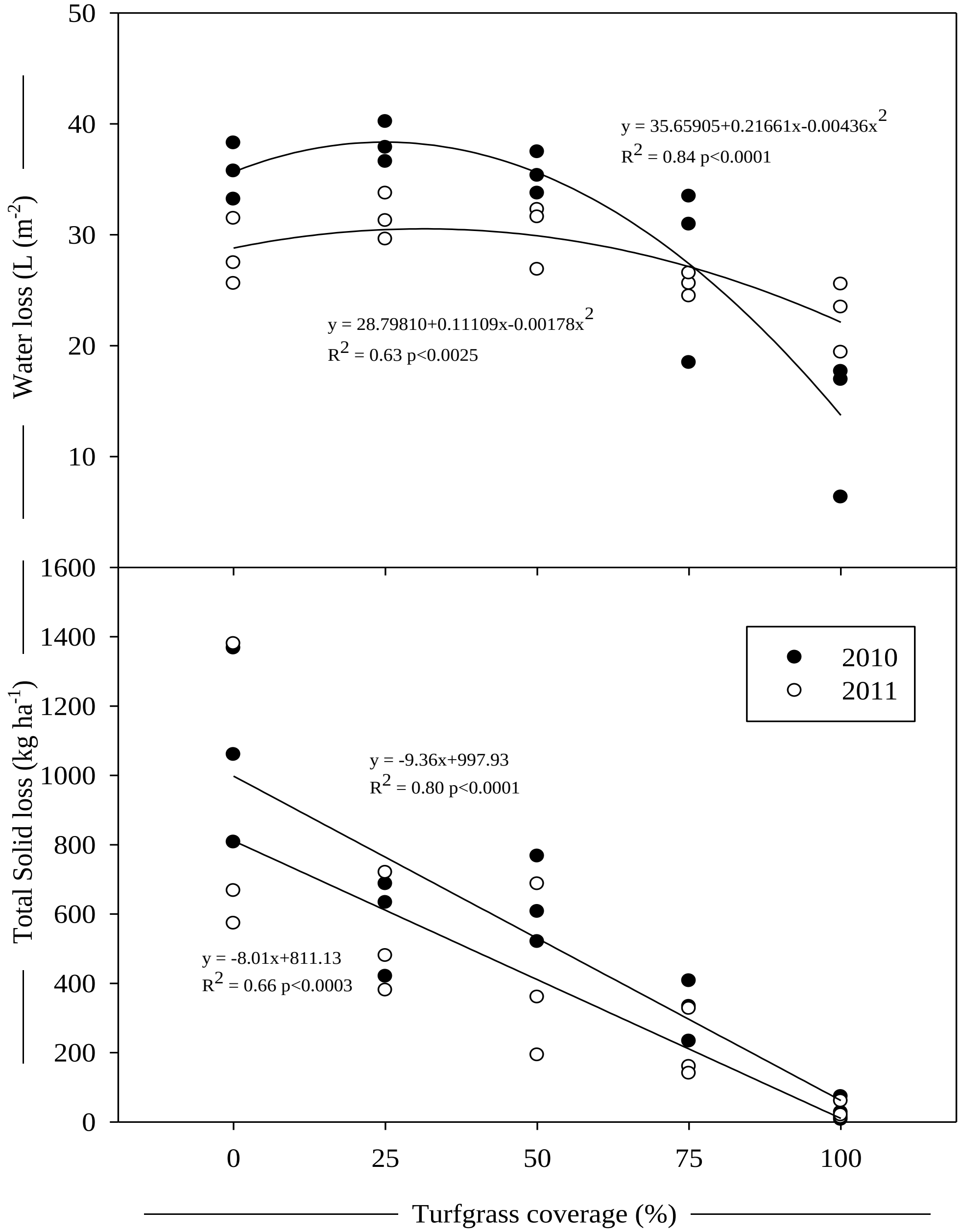

Figure 2.1. Influence of turfgrass coverage on runoff volume and sediment loading during thirty minutes of continuous runoff from simulated rainfall at 7.32 cm h^{-1} in 2010 and 2011.

The effect of increasing turfgrass coverage to reduce runoff volume and water flow was also observed between turfgrass coverage and TS loading. Total suspended solid losses were highest for 0% at 1078 and 873 kg ha^{-1} in 2010 and 2011, respectively. As turfgrass coverages increased to 50 and 100%, TS losses declined linearly to 631 and 35 kg ha^{-1}in 2010 and 413 and 31 kg ha^{-1}

in 2011. The effect of turfgrass coverage to reduce TS losses was more effective at lower turfgrass coverages of 25 and 50% compared to the 75% turfgrass coverage necessary for reducing runoff.

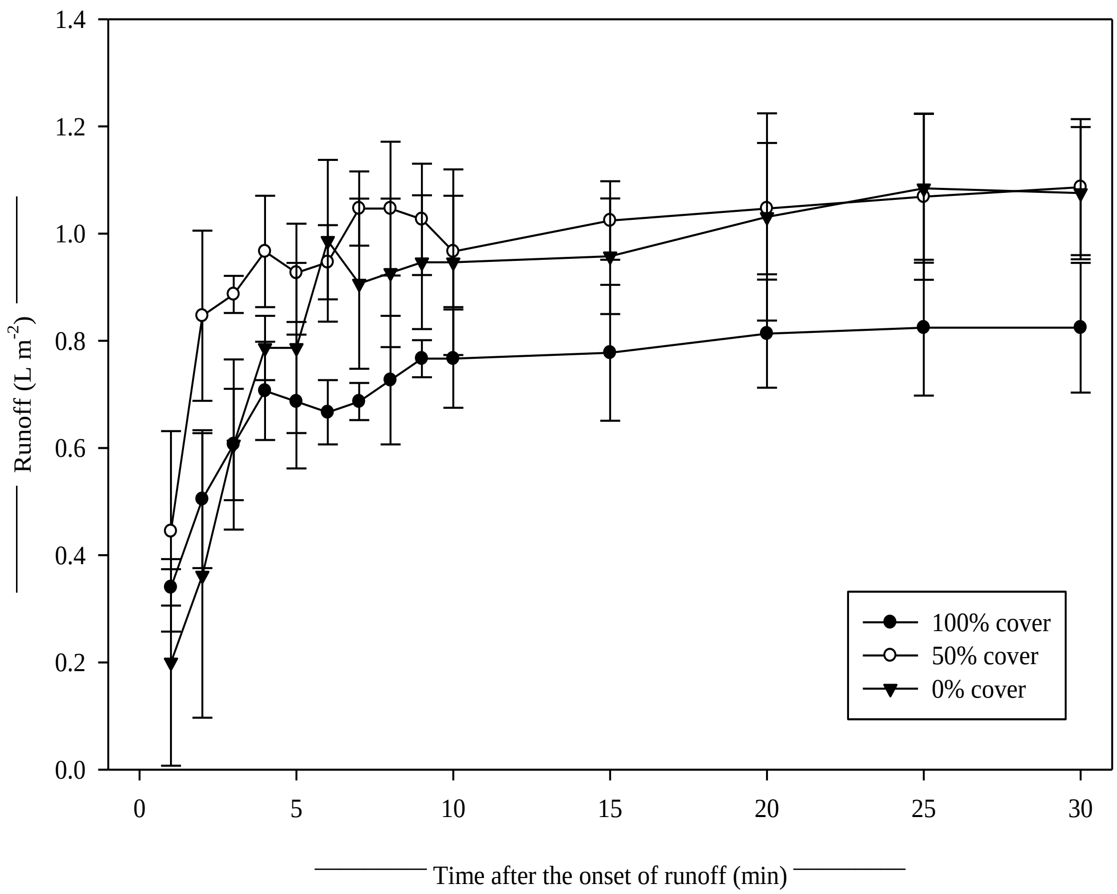

Figure 2.2. Influence of turfgrass coverages (0, 50, and 100%) on water flow during thirty-minutes of continuous runoff from simulated rainfall at 7.32 cm h^{-1} in 2011.

Nitrogen and Phosphorus Runoff Losses

Nitrogen and P losses were similar in 2010 and 2011. Therefore, N and P data for total and dissolved fractions per nutrient were pooled across experiments (p-value < 0.2) to more clearly characterize the effect turfgrass coverage has on nutrient transport during a single surface runoff event. Composite TN losses did not differ between turfgrass coverage treatments (Figure 2.3). Losses of 3.5, 5.7, 4.6, 5.8, and 5.7 kg total N ha^{-1} corresponded to 0, 25, 50, 75, and 100%

turfgrass coverage with DN accounting for 45, 64, 70, 68, and 71% of TN losses, respectively. In 2011, DN losses were analyzed over the 30-min runoff event for 0, 50, and 100% turfgrass coverages. Complete turfgrass coverage exhibited the highest DN losses of 2.25 kg N ha^{-1} 5 min AOR compared to 1.95 and 0.54 kg N ha^{-1} for 50% turfgrass coverage and 0%, respectively (Figure 2.4). At 20 min AOR, DN losses for 0 and 50% were between 0.14 and 0.25 kg N ha^{-1} for the duration of the rainfall simulation compared to a continued decline in DN of 0.76 to 0.42 kg N ha^{-1} at 20 and 30 min AOR for 100% turfgrass coverage.

Nitrogen loss patterns were also evaluated to compare DN:TN losses for 0, 50 and 100% coverage during the 30-min runoff period (Figure 2.5). Each coverage (0, 50, and 100%) exhibited a different pattern of DN:TN loss. Initially 5 min AOR, 50 and 100% turfgrass coverage resulted in 3.01 and 3.21 kg N ha^{-1} respectively, with DN comprising 65 and 70% of the TN lost. Total N and DN values continued to decline for 100% turfgrass coverage in conjunction with DN losses. At 30 min AOR TN was comprised of 37% DN for 100% turfgrass coverage. The 50% turfgrass coverage exhibited continued declines in TN and DN losses until 20min AOR at which point DN losses were static but TN losses were maintained at 1.5 kg N ha^{-1} for the duration of the runoff period. Conversely, TN losses with 0% remained static at 1.0 kg N ha^{-1} for the entire runoff event with DN dropping from 51% TN at 5 min AOR to 14% TN at 30 min AOR. The greatest N losses for all coverages occurred in the first 15 min AOR with 57.0, 63.6, and 68.1% of TN lost for 0, 50 and 100% turfgrass coverage. Coverage of 0% exhibited the lowest composite TP loss at 3.2 kg P ha^{-1} compared to 5.2, 4.8, 4.5, and 5.6 kg ha^{-1} for 25, 50, 75 and 100% turfgrass coverage (Figure 2.6). Similar to N, but at higher percentages, the majority of P losses occurred as DP at 73, 86, 87, 93 and 97% for 0, 25, 50, 75, and 100% turfgrass coverage, respectively.

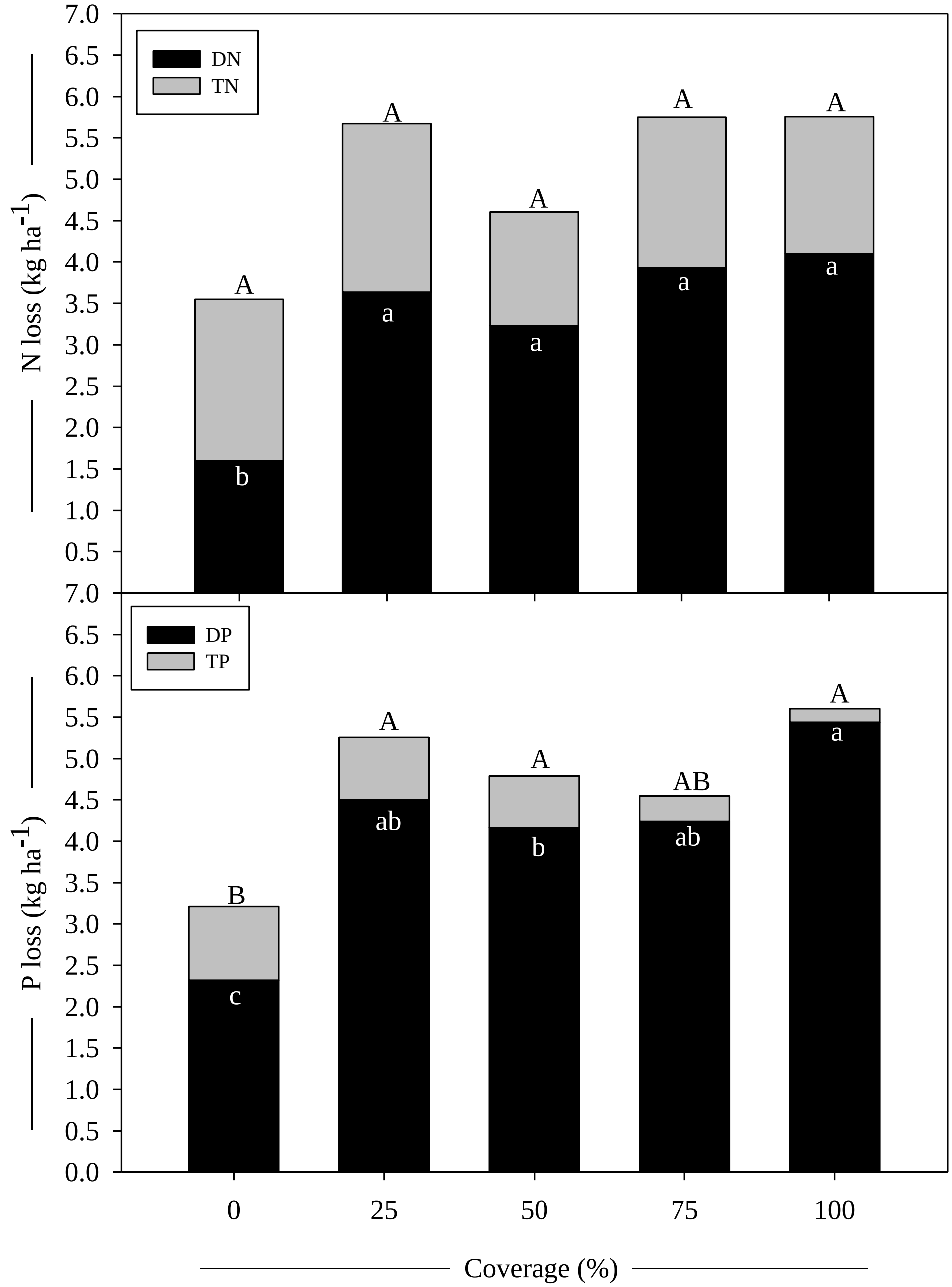

Figure 2.3. Effect of turfgrass coverage on total and dissolved nitrogen (N) and phosphorus (P) losses from thirty-minutes of continuous runoff. Data were pooled for 2010 and 2011 ($p \leq 0.20$). Means for total and dissolved N and P losses are separated using Fisher's protected least significant difference ($p \leq 0.05$). Lowercase letters denote differences in dissolved N and P losses across turfgrass coverages; uppercase letters denote differences between total N and P losses across turfgrass coverages.

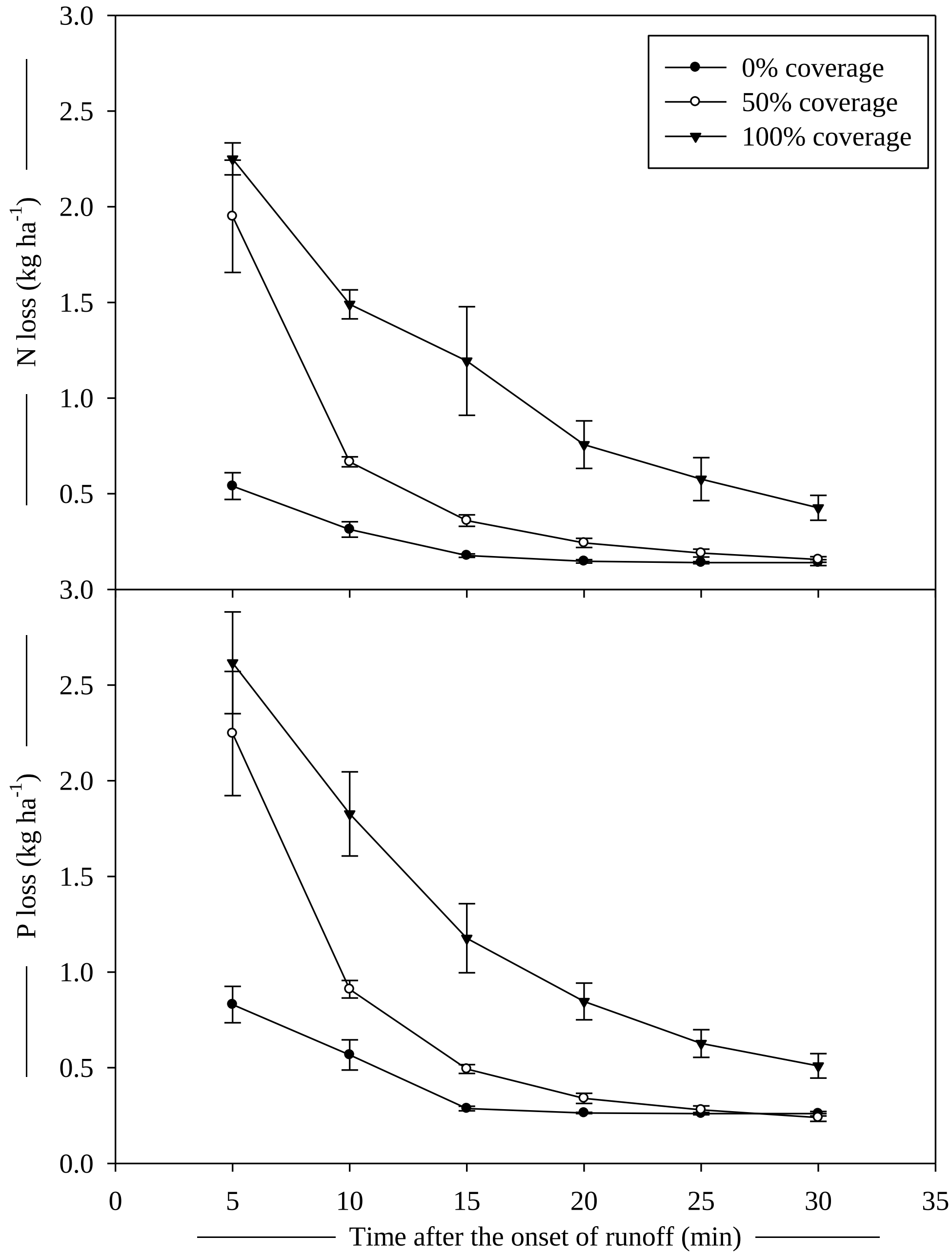

Figure 2.4. Influence of turfgrass coverage (0, 50, and 100%) on dissolved nitrogen (DN) and dissolved phosphorus (DP) losses from thirty minutes of continuous runoff in 2011. Standard errors were calculated for mean comparisons. Each point represents a mean of three replicates.

When DP losses were analyzed over the 30-min runoff period, 50 and 100% turfgrass coverage exhibited continued declines in DP from 2.25 and 2.61 kg P ha^{-1} 5 min AOR to 0.36 and 0.51 kg P ha^{-1} 30 min AOR, respectively. Conversely, 0% exhibited DP losses of 0.8 to 0.6

kg P ha^{-1} at 5 and 15 min AOR followed by static DP losses of 0.15 kg P ha^{-1} at 20 min AOR and for the remainder of the runoff period.

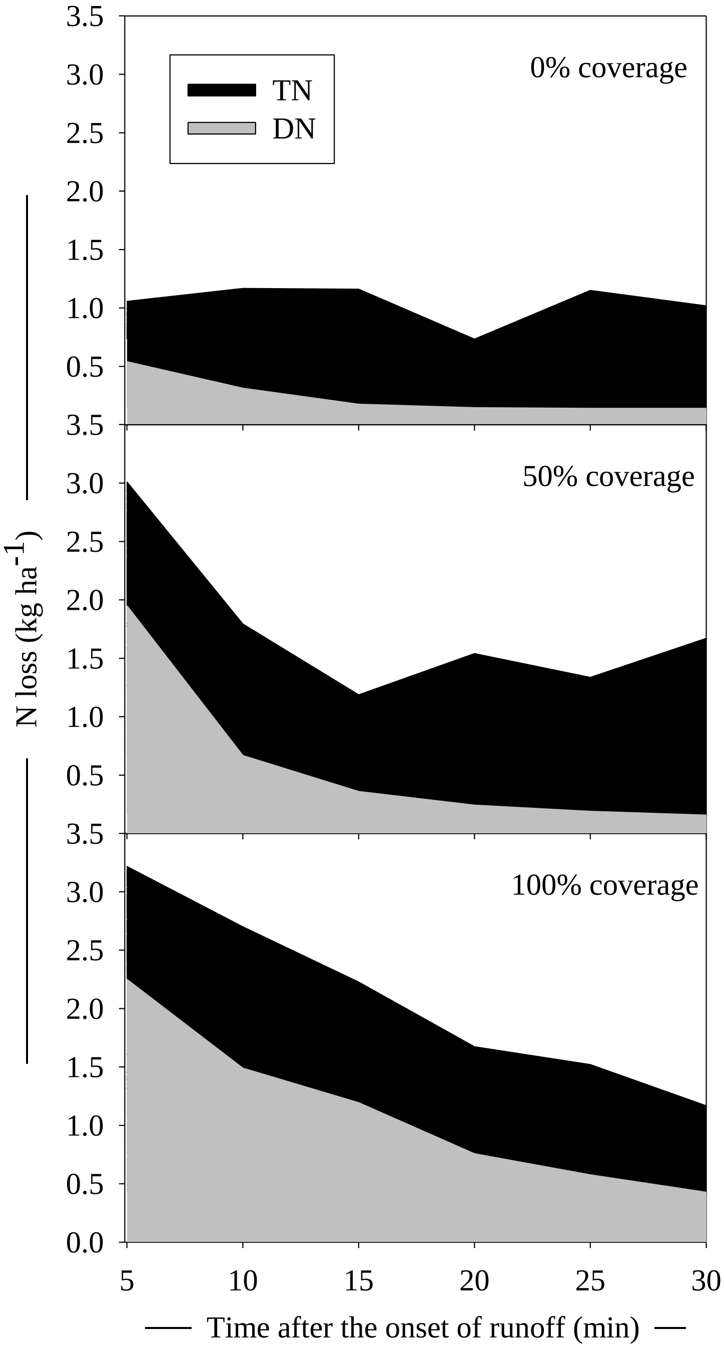

Figure 2.5 Influence of turfgrass coverage (0, 50, and 100%) on dissolved nitrogen (DN) as a component of total nitrogen (TN) losses from thirty minutes of continuous runoff in 2011.

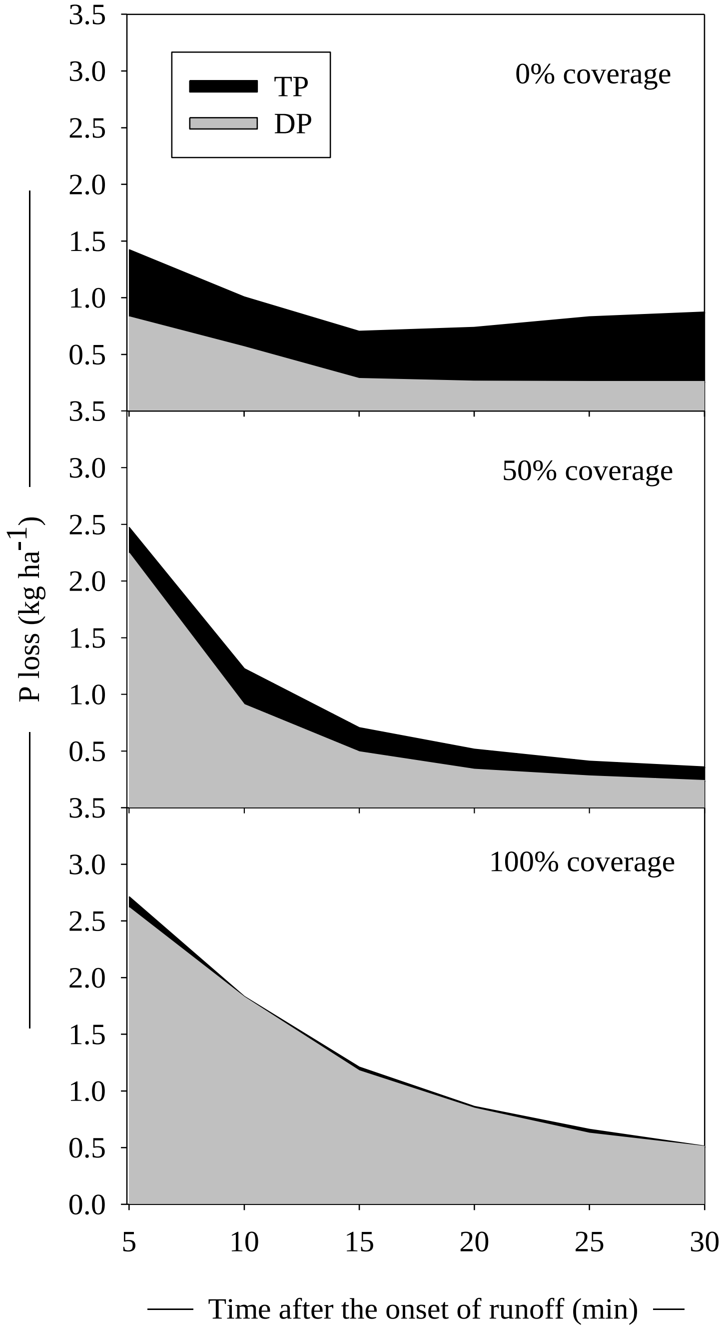

Figure 2.6. Influence of turfgrass coverage (0, 50, and 100%) on dissolved phosphorus (DP) as a component of total phosphorus (TP) losses from thirty minutes of continuous runoff in 2011.

Total P losses for 0, 50 and 100% turfgrass coverage declined from 1.4, 2.5, and 2.7 kg P ha^{-1} 5 min AOR to 0.8, 0.4, and 0.5 kg P ha^{-1} 30 min AOR, respectively. Total P losses for 0%

coverage decreased from 5 to 15 min AOR with losses thereafter becoming static and losing P around 0.8 kg P ha^{-1} for the remainder of runoff. Evaluations of portioned P losses indicated TP losses for 0, 50, and 100% turfgrass coverage consisted of 58, 91, and 97 % DP, correspondingly 5 min AOR. As time progressed DP concentrations of TP losses decreased for 0 and 50% turfgrass coverage and at 30 min AOR consisted of 30 and 67% of TP losses respectively. Different from 0 and 50%, 100% turfgrass coverage DP concentrations of TP losses did not decrease below 95% for any sample timing. Similar to N, the greatest P losses for all coverages occurred in the first 15 min AOR with TP losses of 56.2, 77.4, and 73.8% for 0, 50 and 100% turfgrass coverage.

Discussion

Grass swards delay the onset of runoff and reduce runoff volumes (Easton and Petrovic, 2004; Linde et al., 1995, 1998; Moss et al., 2006) by creating a permeable barrier at the soil surface. Easton et al. (2005) characterized this relationship between turfgrass coverage and surface runoff by correlating plant densities to infiltration rates. They observed an increase in soil infiltration from 7 to 21 cm h^{-1} when shoot density increased from 60,000 to 120,000 shoots m^2. Therefore, the decrease in runoff severity observed in this study with reduced water losses 30 min AOR of 0.82 L min^{-1} for 100% turfgrass coverage compared to 1.09 and 1.08 L min^{-1} for 0 and 50% turfgrass coverage, respectively, adhered to past runoff research results.

Higher turf biomass (e.g. shoots, leaves, and stems) impedes water flow, increases water infiltration, disrupts raindrop splash erosion, and filters suspended solids (Easton and Petrovic, 2004; Easton et al., 2005). Although, comparable flow rates were observed between 0 and 50% turfgrass coverage, the higher than expected flow rate from 50% turfgrass coverage is likely a result of water channeling. Lang (1979) made a similar observation from 70% vegetative coverage compared to bare soil and concluded the higher runoff volume was the result of bare areas connecting to one another.

The lack of differences in water flow rates between 0 and 50% coverage lead to similar cumulative runoff losses for 0, 25, and 50% turfgrass coverage for both years of the study. Cumulative runoff volumes for the 30-min runoff periods only decreased once turfgrass coverage exceeded 75% with further decreases in cumulative runoff to 20.5 L in 2010 and 34.2 L in 2011 for 100% turfgrass coverage. Similar reductions in cumulative runoff volumes have been reported between complete turfgrass canopies and bare soil (Gross et al., 1990; Linde et al., 1994). Although factors such as growth habit (e.g. stoloniferous and tillering) have been shown to affect cumulative runoff volumes (Linde et al., 1995), data from this study indicate differences in runoff losses between species may not occur until turfgrass coverage approaches 100% or thatch development is significant.

The effect of increasing turfgrass coverage to restrict water flow and reduce runoff volumes lead to decreased TS loading. Studies evaluating TS losses from surface runoff have overwhelming reported increases in plant coverage reduces TS transport during surface runoff (Burwell et al., 2011; Butler et al., 2006, 2007; Gross et al., 1990, 1991; Krenitsky et al., 1998; Kussow, 2008; Linde and Watschke, 1997). The data from this study clearly exhibits a linear decline in TS loading from 0 to 100% turfgrass with reductions in TS loading of 96.8 and 96.4% in 2010 and 2011, respectively. Dense vegetation not only absorbs raindrop kinetic energy to prevent splash erosion and sediment dislodging, but also acts as a filter to trap suspended solids (Elwell and Stocking, 1976; Gross et al., 1990, 1991; Linde and Watschke, 1997). Interestingly, turfgrass coverages of 25 and 50% were able to reduce TS loading compared to 0%, but were not as effective in reducing cumulative runoff volumes. This difference in efficacy suggests the filtering capacity of turfgrasses is an important component in reducing TS transport but may not be as effective in reducing dissolved pollutant transport.

In fact, TN and TP losses did not decrease regardless of turfgrass coverage. Nutrient losses were between 4.61 and 5.76 kg N ha^{-1} and 4.54 and 5.60 kg P ha^{-1} for all turfgrass coverages. These total losses accounted for 9.4 to 11.8% and 9.3 and 11.5% of applied N and P, respectively. With the exception of 0%, the highest fraction of TN and TP losses occurred as dissolved constituents with P having the highest dissolved fraction losses. Dissolved N comprised 44% of TN losses for 0% coverage but increased to ≥ 63% for turfgrass coverages of 25 to 100% with corresponding DP of 72 and ≥86%. The effect of turfgrass coverage to affect nutrient pollutant transfer appeared to hinge on the form in which the nutrient was lost. Moe et al. (1968) observed a similar response with greater soluble N losses for areas vegetated with sod compared to bare soil. They attributed the increase in soluble N losses to reduced sediment bound losses and the interference of a turfgrass canopy with fertilizer granule-soil interaction. Moe et al. (1968) concluded that although bare soils retained surface applied soluble N more effectively than sod, N losses via sediment may pose a greater environmental risk over time if an area remains unvegetated.

More interesting were the shifts in dissolved N and P versus sediment bound nutrient losses that occurred over the 30-min runoff period. Initial nutrient losses were composed of >50% dissolved nutrients for all turfgrass coverages 5 min AOR. As time progressed, a greater percentage of N and P losses shifted from soluble losses to sediment bound losses. As dissolved N and P losses decreased, total N and P losses decreased for 50 and 100% turfgrass coverages however, it is clear that beyond a 30 min runoff period the majority of N and P losses would be sediment bound with soil losses having more effect on P losses due to its increased affinity for soil adsorption compared to N. However, nutrient sediment bound losses are greatly curbed with increasing turfgrass coverage.

A shift in dissolved nutrient losses to sediment bound losses was less prominent for 0% coverage which exhibited high N and P sediment bound losses within 5 min AOR for N and 10 min AOR for P. Again, nutrient losses beyond a 30-min runoff period would be influenced more by sediment transport. Similar results have been reported in studies evaluating N and P losses in annual agronomic commodities (Romkens et al., 1973). Johnson et al. (1979) reported higher dissolved N losses from no-till cultivation compared to conventional tillage practices. However, higher sediment bound losses occurred with traditional cultivation. They concluded the interaction of nutrients with soil greatly affect the forms of nutrient loss.

Understanding turfgrass coverage influence on runoff occurrence and nutrient loss forms should help devise more site-specific strategies to reduce potential losses into adjacent surface water bodies. Several studies evaluating nutrient losses from agronomic commodities have reported incorporation of nutrients can decrease N and P losses (Pote et al., 2006). Although this may help during planting, cultivation such as tilling is not a viable option for existing turfgrass areas. Other options include light irrigation post fertilization or increased soil interaction through aerification. Kelling and Peterson (1975) and Shuman (2004) concluded nutrient losses were reduced up to fivefold when a light irrigation was applied prior to rainfall events. However, practices such as aerification have been shown to have little effect to no effect on runoff volume and N and P movement (Moss et al., 2007). Unfortunately, the use of irrigation and to some extent aerification to reduce runoff and nutrient transport may not be feasible for large utility turfgrass areas, therefore strategies involving fertilizer type may be more efficacious.

Turfgrass practitioners have generally relied on rainfall to incorporate fertilizers or initiate dissolution. Burwell et al. (2011) reported increasing turfgrass cover delayed the onset of runoff. However, if rainfall rates exceed soil infiltration, nutrient runoff occurs (Eason and Petrovic, 2004; Gross, 1991) as dissolved nutrients regardless of turfgrass coverage as purported by this research. Previous literature has reported the highest losses of nutrients occur during the first precipitation event following application (Daniel et al., 1979; Easton and Petrovic, 2004; Gaudreau et al., 2002; Kelling and Peterson, 1975). During this study, once runoff was initiated TN losses within the first 15 min for 0, 50, and 100% turfgrass coverages accounted for 57.0, 63.6, and 68.1% of the total losses and 59.9, 79.0, and 73.96% of TP observed for 30-min runoff period, respectively.

The unpredictability of rainfall intensity and inability to provide wide-scale irrigation necessitates the implementation of strategies focused on reduced application rates, application of less water-soluble N and P sources, and integration of soil testing. In a study conducted by Shuman (2004), N and P transport directly correlated with application rate; therefore, if soil testing dictates adequate levels of nutrients additional inputs pose a greater risk of higher runoff losses. Easton and Petrovic (2004) concluded proper fertilization practices should mitigate runoff losses from turfgrass systems. Because dissolved N and P losses were the major source of nutrient losses during this study, applying less water soluble N and P sources could be used to mitigate nutrient losses. Vietor et al. (2004) and Burwell et al. (2011) both reported higher N losses when water soluble N source fertilizers were applied to grasses, but nutrient losses could be reduced with sources that are less water-soluble sources. Further research is needed to evaluate nutrient sources as well as determine long-term effects with different turfgrass coverages in order to improve BMP.

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CHAPTER 3: INFLUENCE OF PESTICIDE SOLUBILITY DURING SURFACE RUNOFF AS AFFECTED BY TURFGRASS COVERAGE

Introduction

Pesticide runoff from turfgrass is an environmental concern in the United States (Haith and Rossi, 2003; Kaufmann III and Watschke, 2007; Kramer et al., 2009) because of its potential impact on surface waters used for aesthetics, fisheries, habitats, recreation, industry and consumption. Turfgrasses comprise more than 16 million hectares, representing the single largest crop in the United States (Kramer et al., 2009; Milesi et al., 2005). Given this vast land area and associated maintenance practices, grassed areas may be a significant source of non-point pollution (Gross et al., 1990). Non-target pollution via surface transport has been shown to have detrimental effects on surface water supplies and shallow alluvial aquifers (Burkart et al., 1997).

As urbanization of rural lands increases management of grassed areas are expanded and the amount of impervious surfaces (sidewalks, roadways, parking lots, etc.) is increased, populated areas will likely become greater contributors to non-point water pollution. For example, a sampling study conducted by Cohen et al. (1999) identified 31 different pesticides at various concentrations in ponds and streams adjacent to golf courses. All pesticides were identified as being routinely applied to turf. In another study, sampling for organophosphate insecticides from agricultural and residential storm water runoff, Penderson et al. (2006) developed a total exposure equivalency parameter based on total concentrations of six insecticides. When the total exposure equivalencies were compared to LC_{50} values, they determined 73% of runoff samples were potentially toxic to the referenced macroinvertabrate, *Pteronarcys californica*.

Compared to traditional production agriculture, turfgrass systems have been characterized as having lower risk for chemical losses because they contain a fixed layer of organic matter in the upper soil profile that interacts with chemicals to reduce movement (Gardner et al., 2000). Leaves and thatch strongly sorb organic compounds to significantly impact pesticide fate (Dell et al., 1994; Gardner et al., 2000; Lickfeldt and Branham, 1995). Hixson et al. (2009) demonstrated decreased simazine movement in mature turfgrass systems due to increased capacity for biodegradation and binding to organic matter. However, organic matter and its interaction with pesticides can vary greatly in grass systems due to species, coverage, management regimens, and environmental conditions.

Although several factors affect pesticide movement, one risk factor that is commonly assessed includes pesticide water-solubility. Lee et al. (2000) posited low water-soluble pesticides have little environmental carryover in turfgrass systems. They confirmed this hypothesis evaluating the movement of pendimethalin (*N*-(1-ethylpropyl)-2,6-dinitro-3,4-xylidine), a low water-soluble pesticide, via surface runoff. After 20 cm of rainfall for 10 d, 0.81 and 1.22% of applied pendimethalin at application rates of 2.25 and 4.50 kg/ha, respectively, were lost. Lee et al. (2000) suggested pendimethalin losses would most likely have increased for croplands where high sediment loading occurs due to pendimethalin's affinity for soil binding and that fact 90% of the pendimethalin applied remained in the top 10 cm of soil 90 days after application. In another study examining highly water-soluble pesticides, Smith and Bridges (1996) reported high losses from surface runoff of 811, 279, and 820 $\mu g\ L^{-1}$ for 2,4-D ((2,4-dichlorophenoxy)acetic acid), dicamba (3,6-dichloro-o-anisic acid), and mecoprop ((*RS*)-2-(4-chloro-o-tolyloxy)propionic acid), respectively, from simulated golf course greens and fairways.

They noted the first runoff event post pesticide application resulted in 75% of total pesticides lost for each pesticide over the course of the experiment.

The majority of research evaluating sediment, nutrient and pesticide runoff from turfgrass systems has primarily focused on bare soil and/or highly managed well-established turfgrass sites (Haith, 2001; Haith and Rossi, 2003; Hong and Smith, 1997; Kauffman III and Watschke, 2007; Kramer et al., 2009; Lee et al., 2000; Moss et al., 2006; Smith and Bridges, 1996; Steinke et al., 2009; Vincelli, 2004). Singular comparisons such as these may not accurately characterize turfgrass effects on pesticide transport during surface runoff given the effect of turfgrass to alter runoff severity. Burwell et al. (2011) and Easton and Petrovic (2004), each reported a direct relationship between increasing turfgrass coverage and density to decreasing runoff volumes and total solid (TS) movement. In light of this, a greater emphasis on understanding the interaction of water solubility on pesticide transport from various turfgrass coverages would be beneficial for developing more site-specific best management practices (BMP). Therefore, the objective of this research was to evaluate the effect turfgrass coverage has on surface runoff losses of pesticides with differing water-solubilities during a single-rainfall event.

Materials and Methods

<u>Site Characterization and Treatments</u>

Surface runoff experiments commenced 9 September 2010 and 17 March 2011 on 10% sloped embankments located at the Louisiana State University Agricultural Center Burden Research Facility in Baton Rouge, LA. The silt loam soil texture consisted of 13.25 % sand 63.6 % silt, and 23.05% clay. Slopes were grassed with St. Augustinegrass [*Stenotaphrum secundatum* (Walt.) Kuntze] in 2010 and perennial ryegrass (*Lolium perenne* L.) in 2011. General maintenance included mowing to 7.62 cm weekly with clippings collected. No fertilizers were applied within 6 months of each experiment. Soil tests were collected and analyzed by the Louisiana State University Agricultural Center Soil Testing and Plant Analysis Lab (LSU STPAL) prior to each experiment that and resulted in soil pH 6.7 and 64 kg P ha^{-1} and 138 kg K ha^{-1}.

Experimental units consisted of five turfgrass coverages (0, 25, 50, 75, and 100%) and an untreated bare soil control. Coverage treatments were arranged in a randomized complete block design with 3 replications in 2010 and 2011. Replications were blocked by the day rainfall simulation occurred with all simulations occurring within a 6-d period. Four to six weeks prior to rainfall simulation, turfgrass coverages were achieved by randomly removing turfgrass plugs within each treatment using a 10 cm diameter probe. Efforts were made to remove as little soil as possible during turfgrass harvesting with uneven areas immediately leveled with the same soil texture.

Experimental units were treated with azoxystrobin (methyl (*E*)-2-{2-[6-(2-cyanophenoxy)pyrimidin-4-yloxy]phenyl}-3-methoxyacrylate), *S*-metolachlor (mix of: (a*RS*,1*S*)-2-chloro-6'-ethyl-*N*-(2-methoxy-1-methylethyl)acet-o-toluidide and 20–0% (a*RS*,1*R*)-2-chloro-6'-ethyl-*N*-(2-methoxy-1-methylethyl)acet-o-toluidide), MSMA (sodium hydrogen methylarsonate), and pendimethalin applied only in 2011 (Table 3.1).

Table 3.1. Pesticide water solubility and soil sorption carbon partition coefficients. †

	Application rate	Water solubility (20°C)	K_{OC}‡
Pesticide	— kg ha^{-1} —	— mg L^{-1} —	— ml g^{-1} —
MSMA	**2.24**	**580,000**	**250-7000§**
***S*-metolachlor**	**2.54**	**530**	**200**
Azoxystrobin	**1.12**	**6.7**	**589**
Pendimethalin	**4.84**	**0.33**	**17,581**

†http://sitem.herts.ac.uk/aeru/footprint/en/index.htm, Sensemen, 2007.
‡Soil organic carbon partition coefficient.
§MSMA K_{OC} dependent on soil characteristics.

Pesticides were applied with a CO_2 pressurized backpack sprayer calibrated to deliver 280 L ha^{-1} and equipped with 8002 XR TeeJet® (Spraying Systems Co. Wheaton, IL) flat-fan nozzles. Pesticides were applied 24 hr prior to rainfall simulation, a period that met or exceeded manufacturer's pesticide labeling for rain-fastness. No irrigation was applied post-pesticide application. Additionally, all turfgrass coverage treatments with the exception of the untreated bare soil control were fertilized at 48.8 kg N ha^{-1} and 48.8 kg P_2O_5 ha^{-1} using a commonly available granular fertilizer (19-19-19, LESCO®, Cleveland, OH). Fertilizer was applied by hand 24 hrs prior to rainfall simulation using shaker jars to insure even distribution. No irrigation was applied post-fertilizer application in order to simulate fertilization of non-irrigated turfgrass.

Rainfall Simulation

Rainfall simulations protocols adhered to the USDA National P Research Project protocol for rainfall simulation (USDA, 2008). Stainless steel runoff trays (15.25 cm x 208.5 cm x 75 cm) with an internal area of 1.5 m^2 and were inserted into the top 8 cm of soil 2 to 3 days prior to rainfall simulation. At the base of each tray a stainless steel trough gravimetrically directed water through a 2.5 cm diameter exit port and into collection reservoirs.

A Tlaloc 3000 rainfall simulator (Joern's Inc., West Lafayette, IN) unit, based on designs of Miller (1987) and Humphry et al. (2002) was used for rainfall simulations. Rainfall was simulated using a specialized spray nozzle (Spraying Systems Co. Fulljet ½HH SS 50WSQ) with a spray angle of 104° (±5%) (USDA, 2008) attached 3 m above the soil surface. To approximate local conditions, rainfall intensity for a two-year, one-hour precipitation extreme of 7.32 cm h^{-1}, was applied (LA Office of State Climatology, 2009). A municipal water source was used for each rainfall simulation. Municipal water samples were collected and analyzed for electrical conductivity, pH, cations, and azoxystrobin, MSMA, *S*-metolachlor, and pendimethalin concentrations.

Rainfall simulations were initiated 24 hours post-pesticide application. Initiation of surface runoff from each plot was demarked at the start of a continuous water flow into collection reservoirs. Runoff was collected *in toto* for 30 min.

<u>Data Collection</u>

Immediately preceding rainfall simulation, volumetric soil moisture ($m^3 m^{-3}$) content was recorded for each experimental unit utilizing a portable TH_2O probe (Dynamax Inc., Houston, TX)(Appendix A.2.). For each simulated rainfall event total runoff volume (L) for the 30-min runoff event was collected with 1 L composite samples analyzed. In 2011, 1-liter subsamples were collected every 5 min for 30 min and analyzed for pesticide concentrations at 0, 50, and 100% coverage. Prior to all sample collections, water was agitated in each collection reservoir to prevent TS settling. Samples were immediately placed on ice before being transported to the laboratory and frozen.

<u>Pesticide Extraction from Water and Sediment Excluding MSMA</u>

All pesticides were extracted and quantified using the Louisiana Department of Agriculture and Forestry Chemistry Department's laboratory facilities in Baton Rouge, La. Pesticide analysis in water was performed according to EPA method 525 for the determination of azoxystrobin, *S*-metolachlor, and pendimethalin. Eight hundred mL samples were centrifuged (Algera-6, Beckman Coulter Inc., Brea, CA) at 3500 rpm for 15 min. Liquid was decanted with 500 mL liquid-liquid partitioned twice against 75 mL methylene chloride. Extraction solutions were filtered through sodium sulfate and placed in a 50°C water bath and evaporated. Analytes were reconstituted in 10 mL hexane and filtered through sodium sulfate to remove any excess water. If more concentrated samples were required for gas chromatography mass spectroscopy (GC-MS) analysis, samples were concentrated under high purity dry nitrogen and reconstituted in lower hexane volumes.

Pesticide extraction from soils was conducted on soil from the 800mL decanted centrifuge tubes. Soils were dried under a ventilation hood with total mass recorded. Soils were extracted in 100 mL ethyl acetate on an orbital table shaker for 4 h. Extracting solutions were filtered (no. 4, Whatman International Ltd, Maidstone, England) and reduced in a 50°C water bath. Analytes were reconstituted in 10 mL ethyl acetate and filtered through sodium sulfate to remove any excess water. If more concentrated samples were required for analysis, samples were concentrated under high purity dry nitrogen and reconstituted in lower ethyl acetate volumes.

Pesticide Analysis Excluding MSMA

Samples were analyzed with a Hewlett-Packard (HP) 6890 GC (Agilent Technologies Inc. Santa Clara, CA) equipped with an autoinjector, split-splitless front inlet, and a single RTX-35SIL MS capillary column (30 m x 0.25 mm i.d. x 0.25 µm film thickness). The autoinjector delivered 2.0µL sample injections. The GC was equipped with an Agilent 5975 C mass selective detector (MSD). Column oven temperatures were as follows: initial 120ºC for 2 min, ramp at 30ºC min^{-1} to 340 ºC and held for 3 min for a total run time of 12.33 min. The carrier gas was ultra-high pure helium with an inlet pressure of 17.55 psi, 20.0 psi pulse pressure and initial injector temperature of 250 ºC. Azoxystrobin, *S*-metolachlor, and pendimethalin were retained on the column for 11.317, 7.221, and 7.494 min, respectively. All pesticides were analyzed at levels of detection (LOD) of 12.5 µg L^{-1} for water and 12.5 mg L^{-1} LOD for soil. During separation and quantitation, blank water samples, laboratory spiked deionized water, a duplicate sample for every 10 samples, and a random matrix spike for every 10 samples were also analyzed for quality assurance.

Pesticide Extraction and Analysis of MSMA

Pesticide analysis for the determination of MSMA followed the microwave assisted digestion of siliceous and organically based matrices in EPA method 3052. Samples were digested by adding 9 ml of concentrated nitric acid and 1ml of concentrated hydrochloric acid to 0.5 g of soil samples and radiated. Samples were analyzed using Inductively Coupled Plasma-Atomic Emission Spectroscopy following the methods outlined by EPA method 200.7. An Optima 4300 DV ICP-OES (PerkenElmer Inc. Waltham, MS) was utilized for soil and water MSMA quantitation. Arsenic was analyzed with 20 µg L^{-1} LOD for both water and soil. Natural soil arsenic concentrations and losses from unfertilized controls were quantified and accounted for in calculation of MSMA losses.

Statistical Analyses

Groundcover treatments were arranged in a randomized complete block design with three replications in 2010 and 2011. Data were analyzed according to the Analysis of Variance (ANOVA; α=0.05) following the general linear method using the statistical software SAS (SAS Institute, 2000). Post-hoc testing was performed on cumulative means for azoxystrobin, *S*-metolachlor, MSMA, and pendimethalin using Fisher's protected least significant difference (LSD; $\alpha = 0.05$). Data for total pesticide losses are reported as a mass of applied pesticide per active ingredient with data separated by year due to the addition of pendimethalin in 2011. Runoff volume and water flow were regressed against grass coverage.

Results

<u>Effect of Turfgrass Coverage on Total Runoff Volume and Total Solid Losses</u>

Surface runoff volume losses were correlated to turfgrass cover for each rainfall simulation in order to assess turfgrass coverage effects on runoff resistance. In 2010 and 2011, 30-min composite surface runoff volumes exhibited negative quadratic pattern across turfgrass coverage (Figure 3.1). Turfgrass coverage at 0% up to 50% had the highest runoff volumes ranging between 33 and 38 in 2010 and 26 to 34 L in 2011 with average declines to 27 and 26 L in 2010 and 2011, respectively, at 75% turfgrass coverage. Lowest runoff volumes occurred at 100% turfgrass coverage with average losses of 14 and 23 L in 2010 and 2011, correspondingly. A similar effect of increasing turfgrass coverage to reduce runoff volume was also observed for TS loading. Total solids losses were highest for 0% coverage at 1078 and 873 kg ha^{-1} in 2010 and 2011, respectively. However, as turfgrass coverage was increased to 50 and 100%, TS losses declined linearly to 631 and 35 kg ha^{-1}in 2010 and 413 and 31 kg ha^{-1} in 2011. Interestingly, the effect of turfgrass coverage to reduce TS losses was more effective at lower turfgrass coverages of 25 and 50% compared to the >75% turfgrass coverage necessary for reducing runoff.

<u>Pesticide Runoff Losses</u>

Patterns for pesticide losses were similar in 2010 and 2011. However, data were not pooled across years due to the addition of pendimethalin as an herbicide treatment in 2011. MSMA exhibited the highest total percent of applied losses (POA) compared to *S*-metolachlor and azoxystrobin (Figure 3.2) Losses of MSMA were 40.1 and 53.1% POA from 0 and 25% turfgrass coverage and declined to 25.7% at 100% turfgrass coverage. Losses of azoxystrobin and *S*-metolachlor exhibited similar patterns as MSMA losses for turfgrass coverages of 25 to

100%. The highest total losses were 19.6 and 16.6% POA azoxystrobin and *S*-metolachlor, respectively, at 25% turfgrass coverage, with reductions to 5.3 and 4.5% at 100% turfgrass coverage. Azoxystrobin and *S*-metolachlor losses did not differ for 50, 75, and 100% turfgrass coverages.

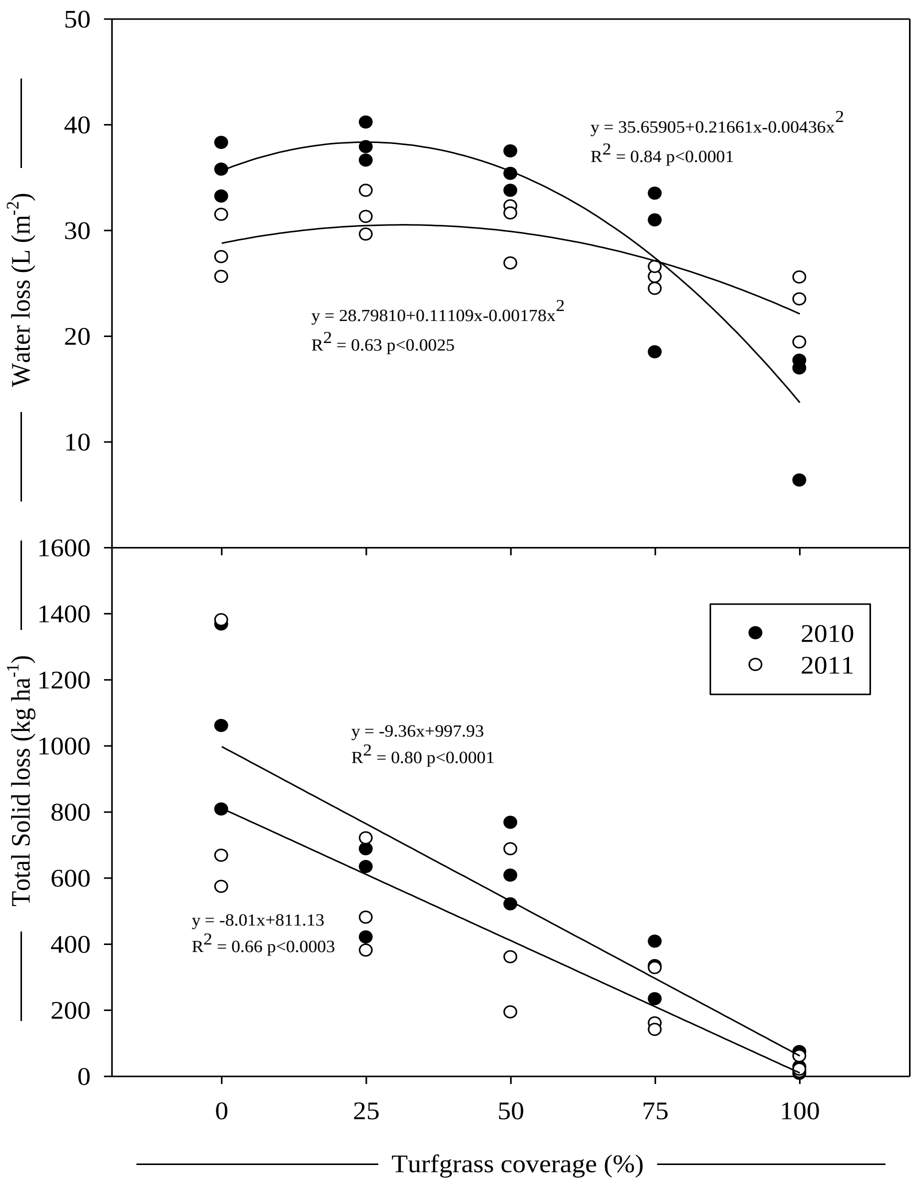

Figure 3.1. Influence of turfgrass coverage on runoff volume and sediment loading during thirty minutes of continuous runoff from simulated rainfall at 7.32 cm h^{-1} in 2010 and 2011.

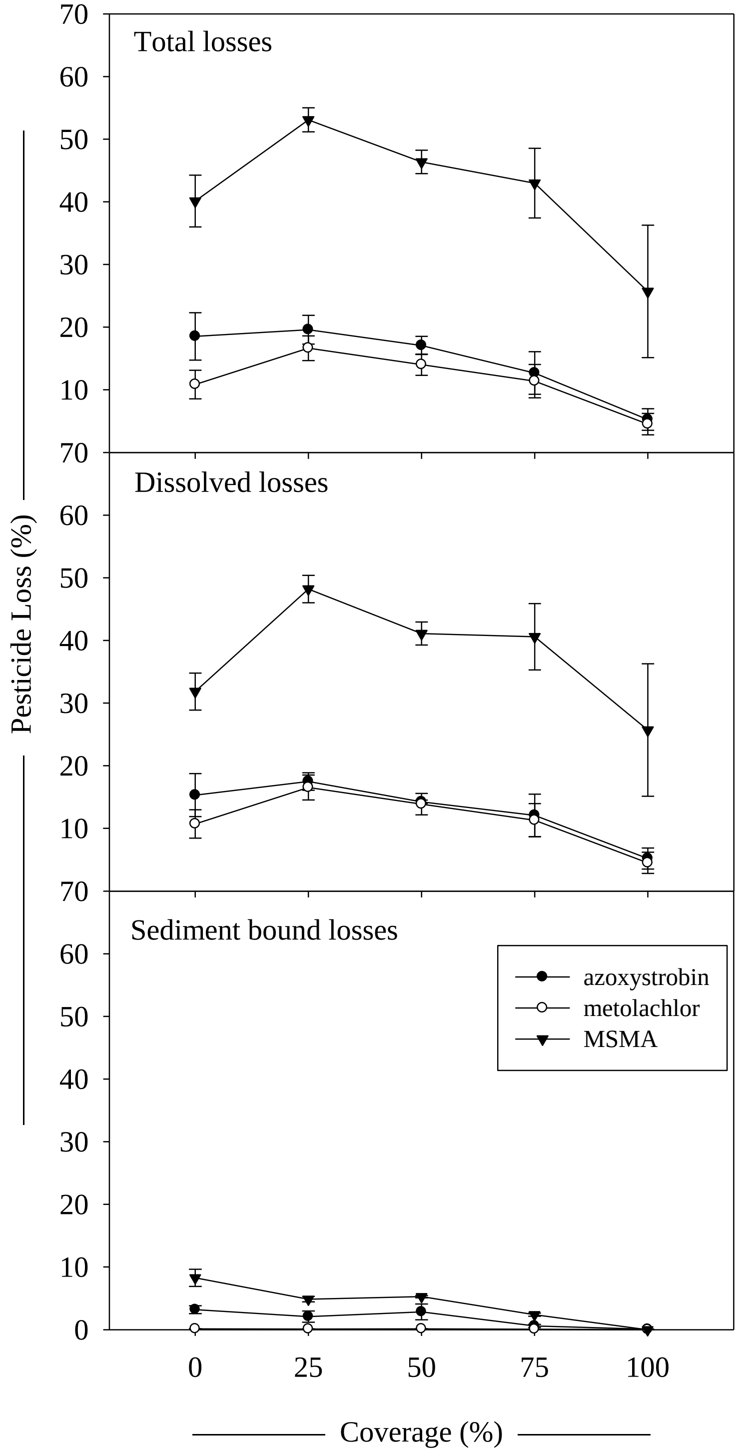

Figure 3.2. Influence of turfgrass coverage (0, 25, 50, 75, and 100%) on total and partitioned losses of azoxystrobin, metolachlor, and MSMA during thirty minutes of continuous runoff in 2010. Standard errors were calculated for mean comparisons. Each point represents a mean of three replicates.

Total solids bound pesticides constituted no more than 21% of total pesticides lost with MSMA exhibiting the highest sediment bound losses. Sediment bound POA pesticide losses at 0% coverage were 8.3, 3.2, and 0.1% for MSMA, azoxystrobin, and *S*-metolachlor, correspondingly. Different from pesticide water losses, sediment losses decreased at 25% coverage corresponding losses of 4.9 % POA MSMA compared to 2.1 and 0.1% POA azoxystrobin and *S*-metolachlor, respectively. Pesticide losses from TS were < 2.4% of applied per pesticide for turfgrass coverages of 75 and 100%.

In 2011, the similar losses patterns observed between pesticides in 2010 occurred with MSMA total losses higher than azoxystrobin, *S*-metolachlor, and pendimethalin (Figure 3.3). Total pesticide losses for 0% coverage were 52.0, 20.1, 9.6, and 3.5% of applied for MSMA, azoxystrobin, *S*-metolachlor, and pendimethalin, respectively. Similar to MSMA total losses in 2010, MSMA losses decreased from 59.9 to 34.1% when turfgrass coverage increased from 25 to 100%. However, increases in turfgrass coverage did not result in decreasing total azoxystrobin and *S*-metolachlor losses. The lowest observed losses for azoxystrobin and *S*-metolachlor of 14.1 and 10.7% of applied at 25 and 100% turfgrass coverage, respectively, with differences between the two pesticides only occurring for 0% coverage. Aside from MSMA, pendimethalin losses steadily decreased from 2.3 to 0.5% as turfgrass coverage increased from 25 to 100% with pendimethalin exhibiting the lowest losses of all pesticides evaluated.

Greater than 81% of MSMA, azoxystrobin, and metolachlor losses were contained in the water fraction in 2011. Similar to 2010, pesticide lost in the water fraction mimicked total losses for all pesticides with the exception of pendimethalin. Pendimethalin water losses fluctuated between 0.7 and 0.4% for all turfgrass coverages. Interestingly, as coverage increased water losses comprised greater amounts of the total losses observed for pendimethalin. Pendimethalin water losses were 20% of the total losses observed for 0% turfgrass coverage compared to 97% of total losses at 100% turfgrass coverage.

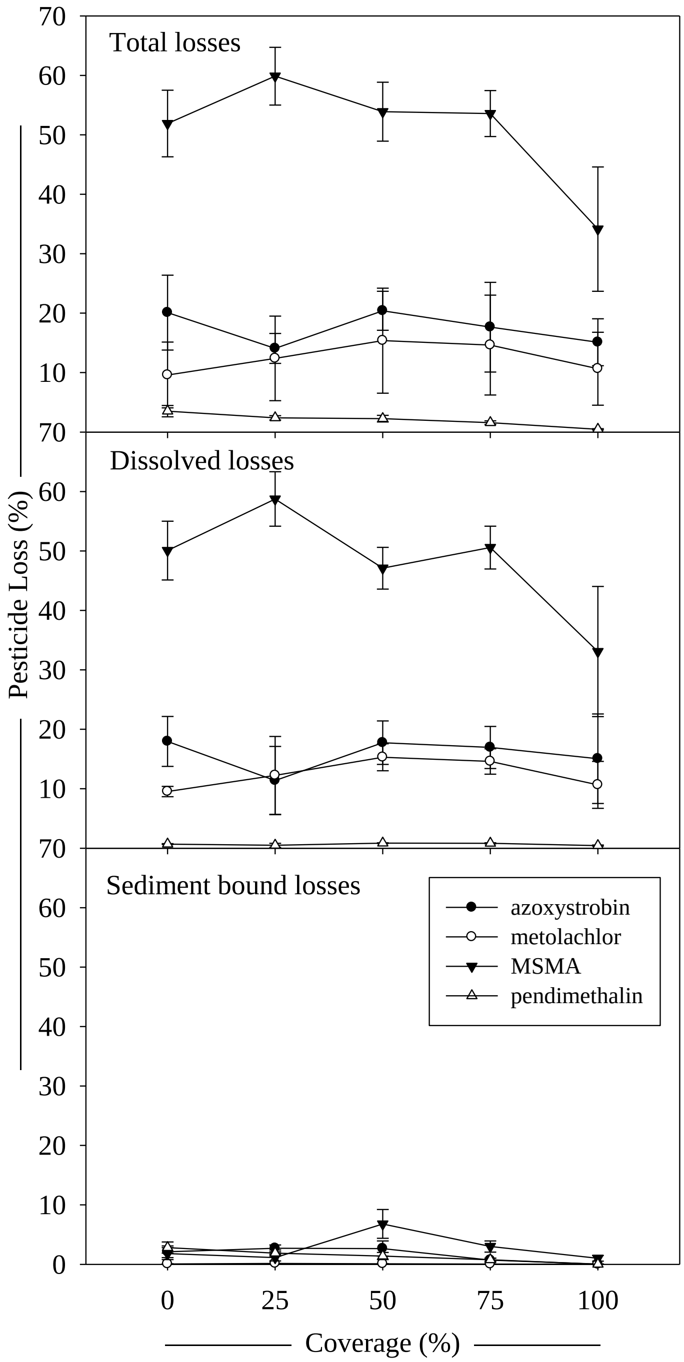

Figure 3.3. Influence of turfgrass coverage (0, 25, 50, 75, and 100%) on total and partitioned losses of azoxystrobin, metolachlor, and MSMA during thirty minutes of continuous runoff in 2011. Standard errors were calculated for mean comparisons. Each point represents a mean of three replicates.

Total solids bound pesticide losses accounted for 4, 11, 1, and 80% of the total losses of applied MSMA, azoxystrobin, metolachlor, and pendimethalin, respectively. The highest TS losses observed occurred at 50% turfgrass coverage with 6.8% of applied MSMA lost. Otherwise, < 3% of applied MSMA, azoxystrobin, metolachlor, and pendimethalin were lost via TS for 0, 25, 75, and 100% turfgrass coverages. As seen in 2010, pesticide losses from higher to lower concentrations followed decreasing pesticide water solubility, MSMA > azoxystrobin > *S*-metolachlor > pendimethalin.

Pesticide Losses over Time in The Water Fraction

In 2011, pesticide losses were analyzed over the 30-min runoff event for 0, 50, and 100% turfgrass coverages (Figure 3.4). Greater than 56% of pesticide losses occurred within the first 15 min after the onset of runoff (AOR) with 0% coverage exhibiting the highest total pesticide losses. The most water-soluble pesticide, MSMA, had the highest losses over time. Greatest losses for MSMA, azoxystrobin, metolachlor, and pendimethalin occurred at 50% turfgrass coverage with corresponding losses of 17.8, 4.7, 6.7 and 0.1% of applied at 5 min AOR. At 0% coverage losses became static 20 min AOR whereas pesticide losses for 50 and 100% turfgrass coverage continued to decline. Total losses for 0% coverage were total losses were 11.7, 3.9, 2.8, and 0.1% at 5 min AOR for MSMA, azoxystrobin, metolachlor and pendimethalin, correspondingly, with reductions to 8.8, 2.0, 1.0, and 0.1%, 30 min AOR. Compared to 0% coverage, greater losses occurred from 5 to 30 min AOR with 50 and 100% turfgrass coverage however, initial losses at 5min AOR were greater than those observed with 0% coverage. For example, at 100% turfgrass coverage 14.6, 4.2, and 3.7% of MSMA, azoxystrobin, and *S*-metolachlor were lost 5 min AOR. MSMA, azoxystrobin and *S*-metolachlor total losses were reduced to 2.2, 1.2, and 0.7%, respectively, 30 min AOR. Pendimethalin losses at 100% turfgrass coverage did not exceed 0.02% at any timing.

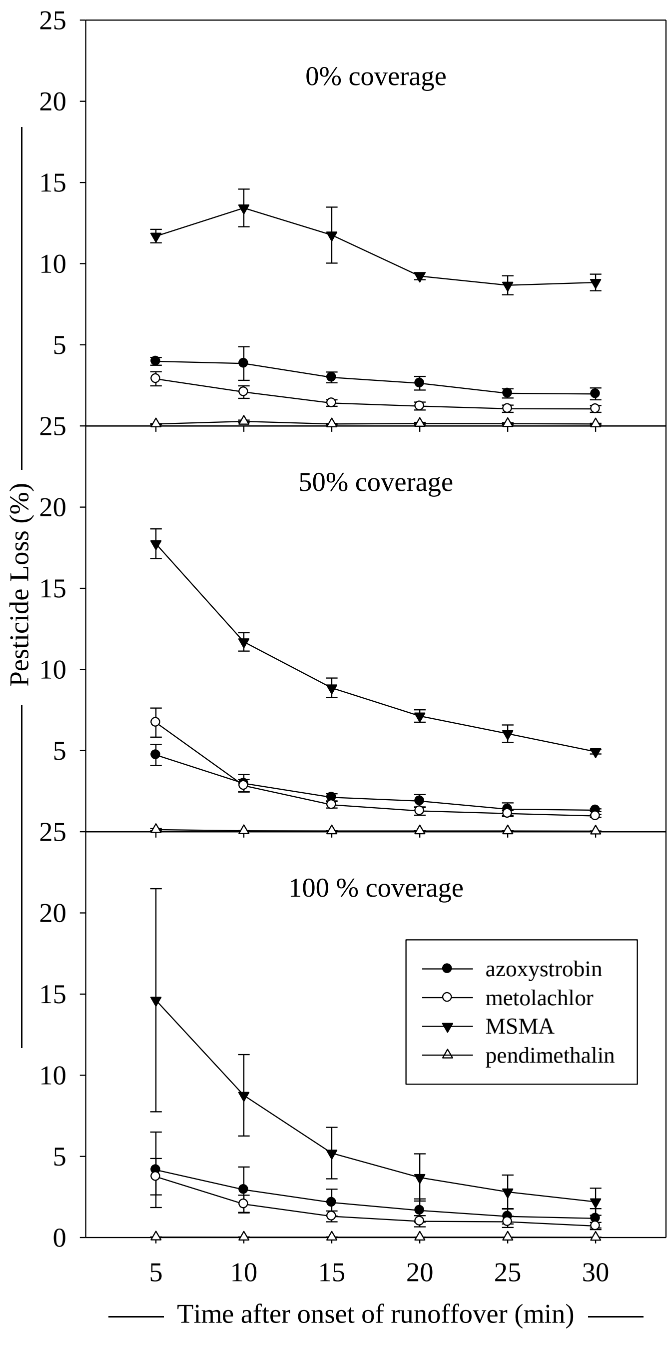

Figure 3.4. Influence of turfgrass coverage (0, 50, and 100%) on dissolved pesticide losses during thirty-minutes of continuous runoff in 2011. Standard errors were calculated for mean comparisons. Each point represents a mean of three replicates.

Discussion

Pesticide losses from surface runoff were highly influenced by water-solubility and to a much lesser extent turfgrass coverage. During both years, the most water soluble pesticide, MSMA, resulted in total losses of 51.9 to 25.7% POA compared to losses for the more moderate water soluble pesticides, azoxystrobin and *S*-metolachlor of 20.1 to 5.3 and 15.4 to 4.5%, respectively and the least water soluble pesticide, pendimethalin, at > 3.5%. Physical partitioning processes such as soil-organic-carbon partition coefficient (K_{OC}) and *n*-octanol-water partition coefficient (K_{OW}) along with pesticide water solubility have been utilized in prediction models for pesticide mass transport (Rice et al. 2010b). During this research, pesticide losses for MSMA, azoxystrobin, *S*-metolachlor, and pendimethalin in 2010 and 2011 compare with previous research conducted by Smith and Bridges (1996) and Rice et al. (2010b) with quantities of loss mimicking pesticide solubility from highest to lowest. According to Lickfeldt and Branham, (1995) pesticide solubility and K_{OC} influence pesticide distribution between leaves, soil, and thatch however, factors such as post treatment irrigation, spray volume and pesticide formulation can alter pesticide fate.

Research evaluating pesticide losses from turfgrasses have primarily focused on dense or mature turfgrass canopies (Armburst and Peeler, 2002; Linde et al. 1995; Smith and Bridges, 1996; Gardner et al., 2000; Rice et al. 2010). Although it was hypothesized turfgrass coverage would have a greater effect on pesticide losses because higher turfgrass coverages and densities have been correlated to decreased runoff volumes and total solid movement (Burwell et al. 2011; Easton and Petrovic, 2004), turfgrass coverage had little effect on total pesticide losses for turfgrass coverages below 75%. Rather turfgrass coverage affected pesticide partitioning between water and TS fractions. Dense vegetation absorbs raindrop kinetic energy to prevent splash erosion and sediment dislodging, as well as acts as a filter to trap suspended solids (Elwell and Stocking, 1976; Gross et al., 1990, 1991; Linde and Watschke, 1997). Erosion was substantial for turfgrass coverages < 50% and corresponded to pesticide losses of < 8, 3.5, 1 and 3% of applied MSMA, azoxystrobin, metolachlor, and pendimethalin losses respectively. As turfgrass coverage increased dissolved pesticide losses accounted for a higher proportion of total pesticide losses with ≥ 97% for all pesticides at complete turfgrass coverage. Therefore, dissolved pesticide losses were the primary mode of transport for all pesticides for all turfgrass coverages with the exception of pendimethalin at lower turfgrass coverages in 2011.

Increased losses of pesticide in the dissolved fraction may be partially explained by examining turfgrass coverage effects on runoff dynamics. Even though higher turfgrass coverages lead to reduced runoff volumes and TS loading (Burwell et. al, 2011), patterns of loss between each parameter and turfgrass coverages greatly differed. Runoff volumes were similar for 0 to 50% turfgrass cover and declined at 75 and 100% turfgrass coverages with average losses reducing from 47.9 to 27.3 L plot^{-1} from 0 and 100% turfgrass coverage respectively. In comparison, TS loading declined linearly with average losses of 554, 522, 267, and 33 kg TS ha^{-1} for 25, 50, 75, and 100% turfgrass coverage. These data clearly demonstrate the filtering capacity of turfgrasses is an important component in reducing TS sorbed pesticide transport but may not be as effective in reducing dissolved pesticide movement. Pesticide concentrations in the environment are determined by numerous interacting processes and conditions (Smith and Bridges, 1996).

In 2011, dissolved losses for all pesticides with the inclusion of pendimethalin were evaluated over the 30 min rainfall simulation in order to characterize pesticide losses during surface runoff. Smith and Bridges, (1996) observed greatest pesticide losses occurred with the first rainfall post pesticide application. In this study > 56% of dissolved pesticides were lost in the first 15 min AOR. Interestingly as turfgrass cover increased, greater loads of dissolved pesticides were lost in the first 15 min AOR. These data suggest the ability of turfgrasses to delay the onset of runoff is an important characteristic that could influence results if studies were completed under natural rainfall. Easton et al. (2005) characterized the relationship between turfgrass coverage and surface runoff by correlating plant densities to infiltration rates. They observed an increase in soil infiltration from 7 to 21 cm h^{-1} when shoot density increased from 60,000 to 120,000 shoots m^{2-1}. However, at higher coverages turfgrass foliage could act as a barrier and limit pesticide penetration into the thatch layer and soil surface to subsequently increase the potential for dissolved pesticide transport in surface runoff. When evaluating MSMA losses from cotton, McDowell et al. (1985) determined 50% of applied pesticides could be washed from plant tissue with a 7-8 mm rainfall event. Compared to > 56% of pesticides lost 15 min AOR for 0% coverage, > 66% of pesticides were lost at 100% turfgrass cover. Different from 50 and 100% turfgrass coverage, 0% coverage losses did not continually decrease during the runoff event. In fact, losses became static at 15 min AOR compared to steady decreases observed with 50 and 100% turfgrass coverage. Total losses of pesticides were relatively lower for 0% coverage compared to 25% turfgrass coverage, long-term sediment losses could pose a serious impact to adjacent water bodies if an area remained unvegetated.

With increasing public awareness for the potential of pesticide pollution, increased understanding of factors that affect pesticide movement should be used to develop more site specific best management practices. Data from this study clearly demonstrate pesticide selection based on water-solubility could be used to help reduce pesticide movement regardless of turfgrass coverage. Although, one would posit pesticides that sorb to TS would have greater movement at lower coverage due to increased erosion over the long-term, the lowest total MSMA loss of 25.7% of applied at 100% turfgrass cover compared to the highest total pendimethalin losses of 3.5% observed for 0% coverage. In other words, bare soil treated with pendimethalin would have to be subjected to 7 surface runoff events at an intensity of 7.38 cm hr^{-1} to equal the losses of MSMA observed in this study from one surface runoff. Though these chemistries are utilized for entirely different management scenarios, their loss potentials do stress the importance of pesticide selection. In the case of herbicides for example, atrazine and simazine control similar weed species however differ in water solubility. Proper selection could mitigate potential losses from areas subject to surface runoff. Therefore, more research is needed in order to evaluate losses of pesticides used for similar control strategies and their fates during surface runoff from differing turfgrass coverages.

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CHAPTER 4: EVALUATION OF ATRAZINE AND SIMAZINE TRANSPORT DURING SURFACE RUNOFF AS AFFECTED BY TURFGRASS

Introduction

In the United States pesticide runoff from turfgrasses is an environmental concern (Haith and Rossi, 2003; Kaufmann III and Watschke, 2007; Kramer et al., 2009). Non-target pollution of water supplies can be a result of extensive pesticide use (Selim, 2003) and given the vast acreage of managed grassed areas in the U.S. (Kramer et al. 2009; Milesi et al., 2005) turfgrasses could pose a significant threat to adjacent water supplies (Burkart et al., 1997; Cohen et al., 1984; Hixson et al. 2009).

A sampling study conducted by Cohen et al. (1999) identified 31 different pesticides at various concentrations in ponds and streams adjacent to golf courses with all pesticides identified as being routinely applied to turfgrasses. Triazine herbicides such as atrazine (6-chloro-*N*2-ethyl-*N*4-isopropyl-1,3,5-triazine-2,4-diamine) are among the most commonly identified pesticide pollutants in water bodies (Giroux, 2002). As a result of triazine herbicide prevalence in water supplies, the USEPA has developed maximum contamination levels of 3 and 4 $\mu g\ L^{-1}$ for atrazine and simazine (6-chloro-*N*2,*N*4-diethyl-1,3,5-triazine-2,4-diamine), respectively.

Atrazine and simazine are applied in agricultural and turfgrass systems for the suppression and control of broadleaf and grassy weeds (Rector et al., 2003; Hixson et al., 2009; Caron et al., 2010). In the southern United States, atrazine and simazine are routinely applied to home lawns to control broadleaf weed infestations or annual bluegrass (*Poa annua* L.) in dormant bermudagrass (Cox et al. 2003). Atrazine and simazine applications are also utilized for weed suppression during centipedegrass (*Eremochloa ophiuroides*) establishment (Gannon et al., 2004).

Offsite movement has been researched for both atrazine (Rector et al., 2003a 2003b; Gaynor et al. 2001; Selim, 2003; Caron et al., 2010; Wauchope, 1978; Glenn and Angle 1986) and simazine (Hixson et al., 2009; Glenn and Angle 1987; Edwards, 1972; Liu and O'Connell; 2003; Glotfelty et al. 1983) and seasonal runoff losses of 15.9% for atrazine (Wauchope, 1978) and 3.5% for simazine (Edwards, 1972) have been reported in agricultural systems. According to Caron et al. (2010) moderate to high water soluble pesticides such as atrazine are transported via surface runoff compared to sediment bound transport for weakly soluble pesticides like simazine. This suggests vegetative groundcover should affect atrazine and simazine losses from surface runoff as a result of decreasing total solid (TS) movement with denser canopies.

Compared to agricultural systems, turfgrasses contain a fixed layer of organic matter in the upper soil profile (Gardner et al., 2000) that is hypothesized to reduce pesticide runoff incidences. Turfgrass leaves and thatch strongly sorb organic compounds to significantly alter pesticide fate (Dell et al., 1994; Gardner et al., 2000; Lickfeldt and Branham, 1995). However, the majority of research evaluating pollutant transport during surface runoff from turfgrass systems has primarily focused on bare soil and complete turfgrass swards (Haith, 2001; Haith and Rossi, 2003; Hong and Smith, 1997; Kauffman III and Watschke, 2007; Kramer et al., 2009; Lee et al., 2000; Moss et al., 2006; Smith and Bridges, 1996; Steinke et al., 2009; Vincelli, 2004). Given the high variability in turfgrass coverage and difference in water solubility between atrazine and simazine, transport by surface runoff could differ across turfgrass coverages. Therefore, the objective of this research was to evaluate the effect turfgrass coverage has on atrazine and simazine runoff losses during a single surface runoff event. This would allow for a site-specific evaluation of two commonly applied herbicides and assist with developing best management practices (BMP) for low maintenance and/or home lawn turfgrass systems.

Materials and Methods

Site Characterization and Treatments

Surface Runoff experiments were conducted in 2010 and 2011 on 10% sloped grassed embankments located at the Louisiana State University Agricultural Center Burden Research Facility in Baton Rouge, LA. The silty loam soil texture consisted of 13.25 % sand 63.6 % silt, and 23.05% clay. General maintenance included mowing to 7.62 cm weekly with clippings collected. No fertilizers were applied within six months of each simulated rainfall event. Soil tests were collected and analyzed by the Louisiana State University Agricultural Center Soil Testing and Plant Analysis Lab (LSU STPAL) prior to each experiment and resulted in soil pH 6.7 and 64 kg P ha^{-1} and 138 kg K ha^{-1}.

Experimental units consisted of five turfgrass coverages (0, 25, 50, 75, and 100%) and an untreated bare soil control. Coverage treatments were arranged in a randomized complete block design with 3 replications in 2010 and 2011. Replications were blocked by the day rainfall simulation occurred with all simulations occurring within a 6 d period. Four to six weeks prior to rainfall simulation, turfgrass coverages were achieved by randomly removing turfgrass plugs within each treatment using a 10 cm diameter probe. Efforts were made to remove as little soil as possible during turfgrass harvesting with uneven areas immediately leveled with the same soil texture.

Herbicides were applied with a CO_2 pressurized backpack sprayer calibrated to deliver 280 L ha^{-1} and equipped with 8002 XR TeeJet® (Spraying Systems Co. Wheaton, IL) flat-fan nozzles. Atrazine and simazine were applied at recommended label rates 24 hrs prior to rainfall simulation, a period that met or exceeded manufacturer's herbicide labeling for rain-fastness (Table 4.1). No irrigation was applied post-herbicide application. Additionally, all turfgrass coverage treatments with the exception of the untreated bare soil control were fertilized at 48.8 kg N ha^{-1} and 48.8 kg P_2O_5 ha^{-1} using a commonly available granular fertilizer (19-19-19, LESCO®, Cleveland, OH). Fertilizer was applied by hand 24 hrs prior to rainfall simulation using shaker jars to insure even distribution. No irrigation was applied post-fertilizer application in order to simulate fertilization of non-irrigated turfgrass.

Table 4.1. Herbicide water solubility and soil sorption carbon partition coefficients. †

	Application rate	**Water solubility (20ºC)**	**K_{OC}‡**
Herbicide	**— kg ha^{-1} —**	**— mg L^{-1} —**	**— ml g^{-1} —**
Atrazine	**2.24**	**580,000**	**250-7000§**
Simazine	**2.24**	**6.7**	**589**

† Sensemen, 2007
‡Average soil organic carbon partition coefficient.

Rainfall Simulation

Rainfall simulation methodology followed those used by the USDA National phosphorus Research Project protocol for rainfall simulation (USDA, 2008). Stainless steel runoff trays (15.25 cm x 208.5 cm x 75 cm) with an internal area of 1.5 m^2 and were inserted into the top 8 cm of soil 2 to 3 days prior to rainfall simulation. At the base of each tray a stainless steel trough gravimetrically directed water through a 2.54 cm diameter exit port and into collection reservoirs.

A Tlaloc 3000 rainfall simulator (Joern's Inc., West Lafayette, IN) unit, based on designs of Miller (1987) and Humphry et al. (2002) was used for rainfall simulations. Rainfall was simulated using a specialized spray nozzle (Spraying Systems Co. Fulljet ½HH SS 50WSQ) with a spray angle of 104° (±5%) (USDA, 2008) attached 3 m above the soil surface. To approximate local conditions, rainfall intensity for a two-year, one-hour precipitation event of 7.32 cm h^{-1}, was applied during rainfall simulations (LA Office of State Climatology, 2009). A municipal water source was used for rainfall simulations. Municipal water samples were collected and analyzed for electrical conductivity, pH, cations, atrazine, and simazine

Rainfall simulations were initiated 24 hours post-herbicide application. Initiation of surface runoff from each plot was demarked at the start of a continuous water flow into collection reservoirs. Runoff was collected *in toto* for 30 min.

Data Collection

Immediately preceding rainfall simulation, volumetric soil moisture ($m^3\ m^{-3}$) content was recorded for each experimental unit utilizing a portable TH_2O probe (Dynamax Inc., Houston, TX)(Appendix A.2). For each simulated rainfall event total runoff volume (L) for the 30-min runoff event was collected with 1 L composite samples analyzed. In 2011, 1-liter subsamples were collected every 5 min for 30 min and analyzed for water-soluble herbicide concentrations for 0, 50, and 100% coverages. Prior to all sample collections, water was agitated in each collection reservoir to prevent TS settling. Samples were immediately placed on ice before being transported to the laboratory and stored at 4°C.

Herbicide Extraction Procedure from Water and Sediment Matrixes

The two herbicides were extracted and quantified using the Louisiana Department of Agriculture and Forestry Chemistry Department's laboratory facilities in Baton Rouge, La. Herbicide analysis in water was performed according to EPA method 525 for the determination of atrazine and simazine. Samples were centrifuged (800 mL) with an Algera-6 table-top centrifuge (Beckman Coulter Inc., Brea, CA) at 3500 rpm for 15 min. Liquid was decanted with 500 mL liquid-liquid partitioned against 75 mL methylene chloride twice. Extraction solutions were filtered through sodium sulfate before being placed in a 50°C water bath and evaporated.

Analytes were reconstituted in 10 mL hexane and again filtered through sodium sulfate to remove any excess water. If more concentrated samples were required for analysis on the gas chromatorgraphy mass spectroscopy (GC-MS), samples were concentrated under high-purity dry nitrogen and reconstituted in lower hexane volumes.

Herbicides were extracted from TS obtained from the 800mL decanted centrifuge tubes. Total solids were dried for 3 d under a ventilation hood with total mass recorded for each sample. Total solids were then extracted in 100 mL ethyl acetate on an automated table shaker for 4 h. Extracting solutions were filtered (no. 4, Whatman International Ltd, Maidstone, England) and reduced in a 50°C water bath. Analytes were reconstituted in 10 mL ethyl acetate and filtered through sodium sulfate to remove any excess water. If more concentrated samples were required for analysis, samples were concentrated under high-purity dry nitrogen and reconstituted in lower volumes of ethyl acetate.

Herbicide Analysis

Samples were analyzed with a Hewlett-Packard (HP) 6890 GC (Agilent Technologies Inc. Santa Clara, CA) equipped with an autoinjector, split-splitless front inlet, and a single RTX-35SIL MS capillary column (30 m x 0.25 mm i.d. x 0.25 µm film thickness). The autoinjector delivered 2.0µL sample injections. The HP 6890 GC was equipped with an Agilent 5975 C mass selective detector (MSD). Column oven temperatures were as follows: initial 120ºC for 2 min, ramp at 30ºC min^{-1} to 340 ºC and held for 3 min for a total run time of 12.33 min. The carrier gas was ultra-high pure helium with an inlet pressure of 17.55 psi, 20.0 psi pulse pressure and initial injector temperature of 250 ºC. Atrazine and simazine had retention times of 6.499 and 6.559 min respectively. Both herbicides were analyzed with 12.5 ug L^{-1} level of detection (LOD) for water and 12.5 mg L^{-1} LOD for soil. During separation and quantitation, blank water samples, laboratory spiked deionized water, a duplicate sample for every 10 samples, and a random matrix spike for every 10 samples were also administered for quality assurance.

Statistical Analyses

Vegetative treatments were arranged in a randomized complete block design with three replications in 2010 and 2011. Data were analyzed according to the Analysis of Variance (ANOVA; α=0.05) following the general linear method using the statistical software SAS (SAS Institute, 2000). Data for total herbicide losses are reported as a mass of applied herbicide per active ingredient. Post-hoc testing was performed on cumulative means for atrazine and simazine using Fisher's protected least significant difference (LSD; $\alpha = 0.05$). Atrazine and simazine data recorded over time were analyzed using repeated measures with means separated using standard errors. Data for all measurements were pooled across years when interaction terms had a p-value ≥ 0.20.

Results

Effect of Turfgrass Coverage on Surface Runoff Volume and Total Solid Losses

The influence of turfgrass coverage on runoff resistance, total runoff volume and total solids (TS) were assessed for each rainfall simulation. In 2010 and 2011, turfgrass coverage decreased total runoff volume and total (Figure 4.1). Turfgrass coverage at 0% up to 50% had the highest runoff volumes ranging between 33 and 38 in 2010 and 26 to 34 L in 2011 with average declines to 27 and 26 L in 2010 and 2011, respectively, at 75% turfgrass coverage. Lowest runoff volumes occurred at 100% turfgrass coverage with average losses of 14 and 23 L in 2010 and 2011, correspondingly.

Total solids losses were highest for 0% coverage and ranged from 1380 to 573 kg ha^{-1} during both years. As turfgrass coverage, increased to 50 and 100%, TS losses declined linearly resulting in reductions ranging between 7 and 73 kg ha^{-1} at 100% turfgrass coverage. Interestingly, the effect of turfgrass coverage to reduce TS losses was more effective at lower turfgrass coverages compared to the >75% turfgrass coverage necessary for reducing total runoff volume.

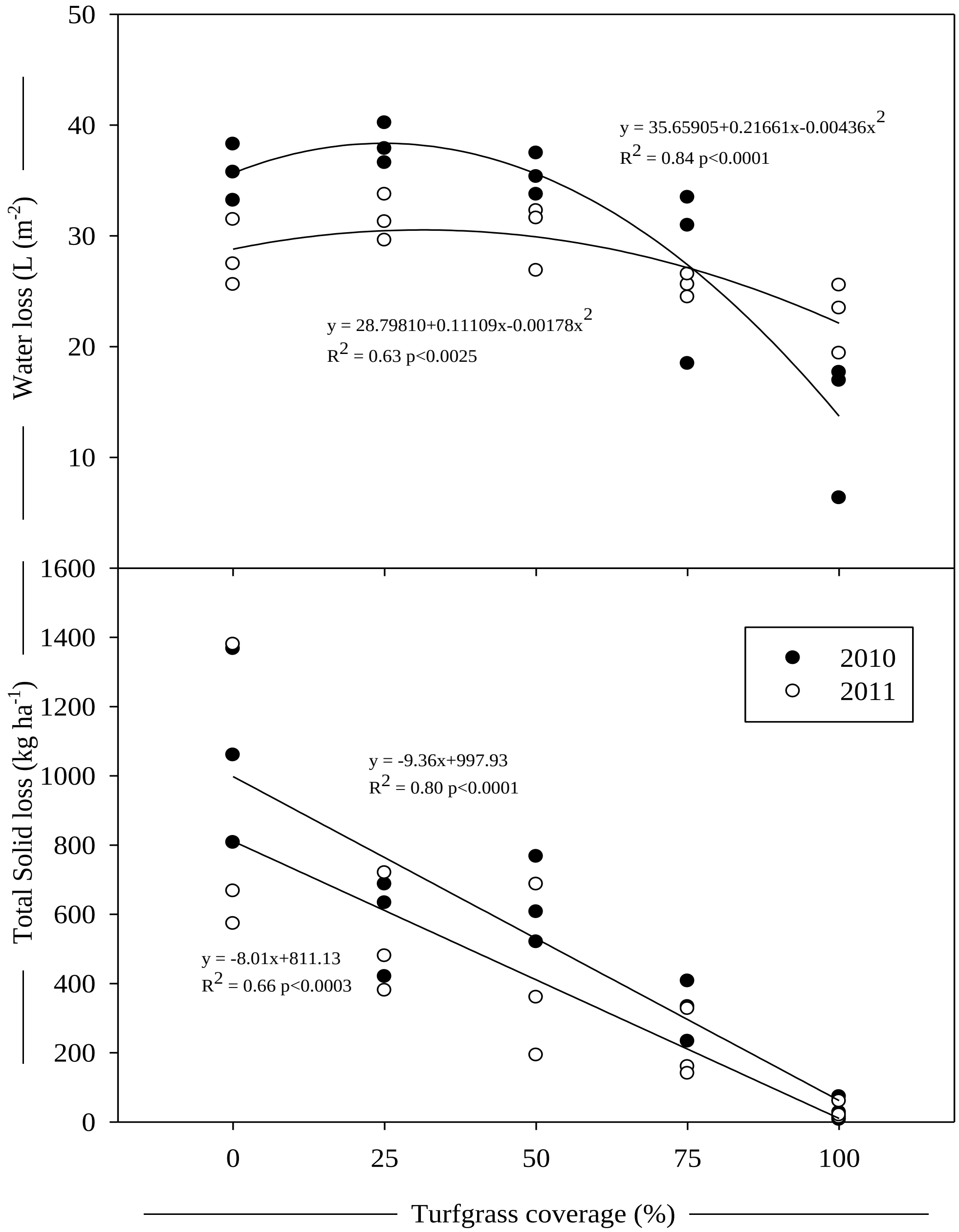

Figure 4.1. Influence of turfgrass coverage on runoff volume and sediment loading during thirty minutes of continuous runoff from simulated rainfall at 7.32 cm h^{-1} in 2010 and 2011.

Atrazine and Simazine Losses

Patterns of atrazine and simazine losses over turfgrass coverages were similar between years. Therefore, data for water and soil fractions per herbicide were pooled across experiments (p-value < 0.2) to more clearly characterize the effect turfgrass coverage has on atrazine and simazine transport during a single surface runoff event.

Greater atrazine and simazine losses occurred as turfgrass coverage increased from 0 to 50% (Figure 4.2). Total atrazine losses were 12.5% of applied (POA) for 0% coverage and increased to 15.6, 23.2, 20.1, and 18.3 POA at 25, 50, 75, and 100% turfgrass coverages, respectively. Similar to atrazine, simazine losses were highest for 50% turfgrass coverage at 14.6 POA compared to 10.1 POA for 0% coverage, but declined to 10.4 and 5.7 POA at 75 and 100% turfgrass coverages, respectively. Total simazine losses declined 29 and 45 % at 75 and 100% turfgrass coverages compared to corresponding declines in atrazine losses of 13 and 9%. Atrazine and simazine did not differ between 0 and 25% turfgrass coverage.

Herbicide losses were also partitioning into water and TS fractions for analysis. Dissolved atrazine losses constituted >98 % of total atrazine losses for all coverages. Less than 1.9% of total atrazine applied occurred as the result of TS loading regardless of turfgrass coverage. Therefore, dissolved atrazine loss patterns were similar to total atrazine loss patterns for each coverage. In comparison, less dissolved simazine losses occurred and comprised 42, 31, 42, 35, and 55 % of total simazine lost for 0, 25, 50, 75, and 100% coverages. Remaining total simazine losses were recovered with the sediment bound partition. Total solids sorbed simazine losses were 5.9, 8.0, 8.5, and 6.7 POA for 0, 25, 50 and 75% coverage followed by a reduction to 2.6% at 100% turfgrass coverage.

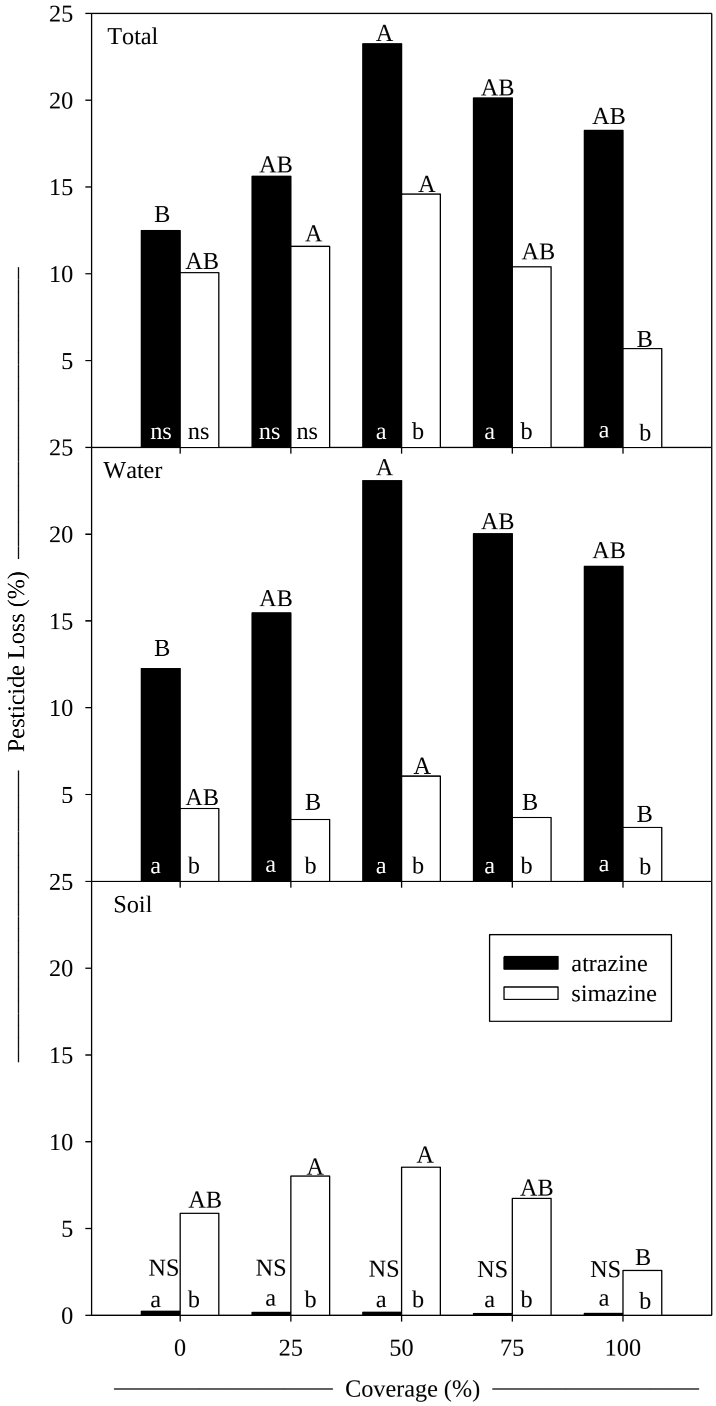

Figure 4.2. Effect of turfgrass coverage on total and portioned pesticide losses from thirty-minutes of continuous runoff. Data were pooled for 2010 and 2011 ($p \leq 0.20$). Means for total and partitioned herbicide losses are separated using Fisher's protected least significant difference ($p \leq 0.05$). Lowercase letters denote differences between atrazine and simazine losses across turfgrass coverages; uppercase letters denote differences between atrazine and simazine at each coverage for each pesticide.

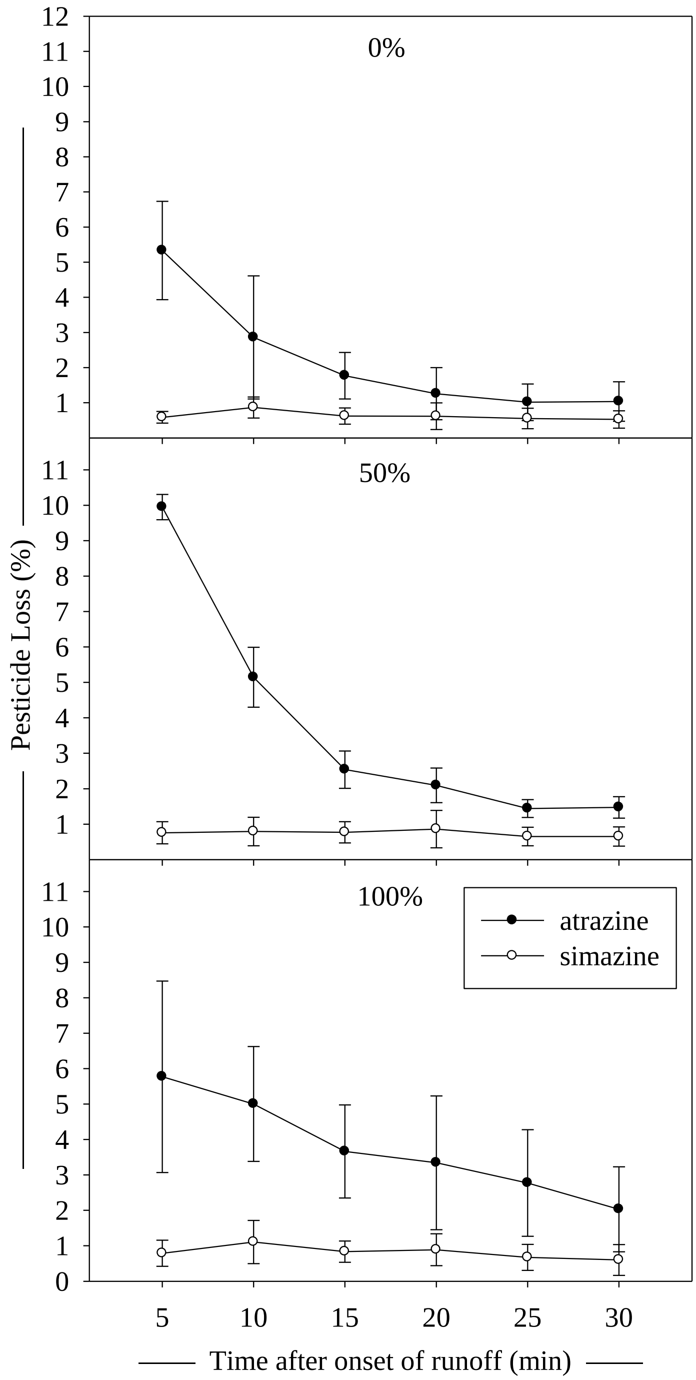

Figure 4.3. Influence of turfgrass coverage (0, 50, and 100%) on dissolved atrazine and simazine losses from thirty minutes of continuous runoff in 2011. Standard errors were calculated for mean comparisons. Each point represents a mean of three replicates.

Atrazine and Simazine Losses over Time

In 2011, herbicide water losses were analyzed over the 30-min runoff event in 5 min intervals for 0, 50, and 100% turfgrass coverages (Figure 4.3). Atrazine losses at 0% coverage decreased from 5.3 to 1.7 POA for 5 and 15 min AOR with subsequent losses of 1.3 to 1.0 POA the remainder of the runoff period. The highest atrazine losses occurred at 50% turfgrass coverage at of 9.9, 2.5, and 1.5 POA for 5, 15 and 30 min AOR, respectively, compared to 5.8, 3.7, and 2.0 POA for complete turfgrass coverage. Regardless of turfgrass coverage, more than 63% of atrazine lost during the 30-min simulations occurred within the first 15 min AOR. Conversely, dissolved simazine losses were similar for 0, 50, and 100% turfgrass coverages with simazine losses never exceeding 1.1 POA at any time.

Discussion

Dense vegetation such as turfgrasses absorb raindrop kinetic energy preventing splash erosion and sediment dislodging, and also acts as a filter to trap suspended solids (Elwell and Stocking, 1976; Gross et al., 1990, 1991; Linde and Watschke, 1997). Data from this research concur with these conclusions as total runoff volume and TS losses decreased with increasing turfgrass coverage. Previous research has indicated this reduction in water and sediment losses can in turn decreases herbicide runoff losses from turfgrass settings and is why they have been adopted as a BMP for runoff (Anderson et al., 1989; Barfield et al., 1979; Hayes et al., 1979). Caron et al. (2010) attribute buffer strip effectiveness to the reduction of surface runoff and sediment loss, however also infer pesticide properties such as solubility, sorption to leaves and soil, and degradation can influence buffer strip efficacy.

Physical partitioning processes such as soil-organic-carbon partition coefficient (K_{OC}) and *n*-octanol-water partition coefficient (K_{OW}) along with pesticide water solubility have been utilized in prediction models for pesticide mass transport (Rice et al. 2010b). According to Lickfeldt and Branham, (1995) pesticide solubility and K_{OC} influence pesticide distribution between leaves, soil, and thatch however, factors such as post treatment irrigation, spray volume and pesticide formulation can alter pesticide fate. Previous research evaluating atrazine and simazine losses in agricultural settings have alluded to pesticide solubility for differing losses observed between the two chemistries (Glenn and Angle, 1986).

Total herbicide loss and loss pattern observed during this study indicate the influence chemical characteristics can have on surface transport. Total atrazine POA runoff losses were greater than total POA runoff losses for simazine at 50, 75, and 100% turfgrass coverage. However, the greatest losses observed for both herbicides occurred at 50% turfgrass coverage 23.2 and 14.6 POA lost for atrazine and simazine, respectively. These higher losses observed for 50% turfgrass coverage were attributed to water channeling on the research experimental units. Lang (1979) made a similar observation from 70% vegetative coverage compared to bare soil and concluded the higher runoff volume was the result of bare areas connecting to one another. Interestingly, no difference in total POA atrazine losses occurred between the different turfgrass coverages. At 100% turfgrass cover 18.2% of total atrazine applied was lost and did not differ from the 12.5% lost for 0% coverage. Higher water soluble pesticides such as atrazine can move into and through the soil profile with less restriction compared to lower water soluble pesticides (Lee et al. 2000). Therefore without the presence of turfgrass to impede soil contact, lower herbicide total losses for 0% coverage were attributed to adsorption and infiltration.

Atrazine a high water soluble herbicide is more susceptible to runoff losses compared to lower water soluble herbicides such as simazine (Glenn and Angle, 1986). Loss patterns observed during this study concur with previous research by Smith and Bridges (1996) and Lee et al. (2000) who concluded pesticide solubility influenced surface transport of pesticides. This similar pattern of dissolved losses has also been reported in nutrient losses during surface runoff. Moe et al. (1968) reported greater soluble N losses for areas vegetated with sod compared to bare soil.

The majority of research evaluating sediment, nutrient and pesticide runoff from turfgrass systems has primarily focused on bare soil and highly managed well-established turfgrass sites (Haith, 2001; Haith and Rossi, 2003; Hong and Smith, 1997; Kauffman III and Watschke, 2007; Kramer et al., 2009; Lee et al., 2000; Moss et al., 2006; Smith and Bridges, 1996; Steinke et al., 2009; Vincelli, 2004). Singular comparisons such as these may not accurately characterize turfgrass effects on pesticide transport during surface runoff given the effect of turfgrass on runoff occurrence and severity.

Previous research has indicated this reduction in water and sediment losses in turn decreases pesticide runoff and thus turfgrass buffers are a BMP for agricultural settings (Barfield et al., 1979). Caron et al. (2010) attribute buffer strip effectiveness to the reduction of surface runoff and sediment loss, however also infer pesticide properties such as solubility, sorption to leaves and soil, and degradation can influence buffer strip efficacy.

Literature evaluating pesticide losses from turfgrasses have primarily focused on complete mature grass systems (Armburst and Peeler, 2002; Linde et al. 1995; Smith and Bridges, 1996; Gardner et al., 2000; Rice et al. 2010) however; pesticide concentrations in the environment are determined by numerous interacting processes and conditions (Smith and Bridges, 1996). Hixson et al. (2009) concluded that as turfgrass systems age, simazine leaching potential decreases due to a large capacity for biodegradation and binding to organic matter.

Therefore, less surface coverage could equate less organic matter ultimately affecting pesticide sorption.

Total simazine and atrazine losses were partitioned into water and sediment adsorbed losses in order to understand more specifically surface coverage influence on runoff losses. Atrazine water losses were greater than simazine water losses at all coverages. Atrazine water losses constituted >98% of total atrazine losses at all coverages compared to highest soluble simazine water losses constituting < 54% of total simazine losses at 100% coverage. Interestingly, simazine water losses for 0, 25, 50 and 75% turfgrass coverage range constituted 31 to 42% of total simazine losses.

Simazine soil losses constituted a majority of simazine total losses and ultimately decreased as surface coverage increased. Sediment simazine losses for bares soil, 25, 50 and 75% turfgrass coverage ranged from 5.9 to 6.7% applied with a decrease to 2.6% observed at 100% turfgrass coverage. This pattern of herbicide loss suggests the filtering capacity of turfgrasses is an important component in reducing TS transport but may not be as effective in reducing dissolved pollutant transport.

The effect of turfgrass coverage to affect herbicide transfer appeared to hinge on the form in which the nutrient was lost. Atrazine and simazine water losses evaluated over the course of the runoff event further indicate the differences in losses between the chemistries. Water losses for simazine were > 1.1% of applied and remained static for 0 50 and 100% turfgrass coverage. Conversely, greatest atrazine losses were observed 5 min AOR with reductions occurring over the 30 min rainfall simulation. Losses occurring from 0 to 15 min AOR were 63% greater than losses from 15 to 30 min AOR. Previous research has indicated greatest pesticide losses occur with the first rainfall post pesticide application (Smith and Bridges, 1996; Glenn and Angle, 1987). Therefore, unincorporated, unabsorbed, residual atrazine on would pose a threat to adjacent water supplies.

Understanding turfgrass coverage influence on runoff occurrence and pesticide loss forms should help devise more site specific strategies to reduce potential losses into adjacent surface water bodies. Several studies evaluating herbicide losses have reported incorporation of simazine and atrazine can decrease losses (Liu and O'Connell, 2003; Harman et al., 2004; Rector et al., 2003a). Although this may help during planting, cultivation such as tilling is not a viable option for existing turfgrass areas. Previous research has also indicated timing atrazine applications around rain and high runoff potential seasons (Rector et al. 2003b). This research evaluated losses occurring 24 h post herbicide application however also indicated the importance of soil and sediment sorption. Timing atrazine and simazine applications around rainfall and increasing residual time between rainfall events could decrease losses. The data presented in this paper also indicate the role herbicide selection can have on mitigating runoff losses and should be considered when creating BMP strategies.

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CHAPTER 5: MITIGATING NUTRIENT AND PESTICIDE SURFACE RUNOFF FROM TURFGRASS SYSTEMS

Surface runoff of applied nutrients and pesticides can pose a risk to adjacent water supplies. The relationship between turfgrass coverage nutrient and/or pesticide movement is not clearly defined in literature. Many grass areas are not complete (100% coverage) stands and are highly managed in terms of fertilization, pesticide application, frequency of cultural practices, and irrigation. Turfgrass systems such as roadsides, home lawns, and golf course roughs can vary in surface coverage however; treating these areas similarly in terms of pesticide and nutrient management strategies could increase chances of non-point water pollution.

Turfgrass coverage had little effect on nitrogen (N) and phosphorus (P) losses during this research. Partitioning total nutrient losses into water carried and sediment bound indicated that the majority nutrients lost from surface runoff predominantly occurred in the water fraction. Evaluating dissolved losses over time indicated that majority of N and P losses occurred during the first 15 min after the onset of runoff (AOR). Turfgrass coverages of 0 and 50% continued to lose N and P after 15 min AOR with transitions from water to sediment bond losses. Different from the water fraction, sediment bound nutrients found in surface runoff decreased as turfgrass coverage increased. This finding indicates the importance of surface coverage to retain suspended solids and in turn sediment bound nutrients.

Nutrient loss patterns were comparable to pesticide loss patterns in that the total amounts of pesticides lost during experiments were dictated by the solubility and dissolved water losses. The highest water soluble compound studied, MSMA, was lost at greater percentages than any other pesticide evaluated in this experiment. Higher water soluble pesticides were lost in greater proportion compared to low water soluble pesticides regardless of turfgrass coverage. When total pesticide losses were partitioned into water and sediment adsorbed, majority of losses occurred within the water partition. However, less water soluble pesticides such as pendimethalin and simazine were primarily lost with total suspended solids (TS). Similar with dissolve nutrient losses, majority of dissolved pesticides losses occurred primarily within the first 15 min AOR. As seen with nutrient losses, as turfgrass coverage increased sediment bound pesticide losses decreased. The lowest pesticide losses observed during this research occurred with pendimethalin, the lowest water soluble pesticide tested. The lowest observed losses for the highest water soluble pesticide, MSMA, were seven times greater than the lowest observed losses for pendimethalin. This difference in losses suggests the filtering capacity of turfgrasses is an important component in reducing TS transport but is not as effective in reducing dissolved pollutant transport.

Turfgrass can delay the onset of runoff however, once runoff is initiated, the majority of pesticide and nutrients are lost within the first 15 minutes. Site specific hydrology (e.g. soil characteristics, turfgrass coverage, slope) all influence potential surface runoff. Site hydrology can directly influence water loss which is the driving force for sediment, nutrient, and pesticide losses. Therefore, best management practices (BMP) should be implemented in order to mitigate potential runoff losses.

Proper nutrient and/or pesticide selection should be considered prior to application in turfgrass systems. As observed in this research, high water soluble pesticides are more prone to surface runoff compared to low water soluble pesticides. Turfgrass coverage can influence pesticide soil and/or thatch contact. Essentially acting as a barrier, turfgrass coverage can retain foliar applied pesticides. If not taken up by the plant and rainfall occurs within 24 hours after application, residual pesticides could be susceptible to surface runoff. Utilizing pendimethalin in a preemergent herbicide program could decrease the necessity for numerous post-emergent herbicide applications decreasing potential for pesticide losses. Using preemptive strategies such as this can decrease the amount of pesticides inputs to a system and in turn decrease the chance of a pesticide runoff event.

Majority of nutrient losses during this research occurred with the dissolved forms of N and P. Selecting a less water soluble nitrogen source could be a BMP for mitigating potential runoff losses. Coinciding with a different nutrient source, fertilizer integration into the soil and/or turfgrass thatch layer should be attempted if possible. Through light irrigation and/or physical incorporation, nutrient runoff losses could be mitigated. However, turfgrass systems unable to provide wide-scale irrigation should avoid relying on rainfall for fertilizer incorporation due to the unpredictability of intensity and duration. These sites should focus on strategies such as reduced application rates and timing fertilizer applications around inclement weather.

As turfgrass coverage increases total runoff volumes and TS loading decreases. However, once runoff occurs, nutrient and/or pesticides located on turfgrass canopies and soil surfaces are susceptible to immediate loss. Therefore, any benefits gained in a turfgrass system from imposed fertilization and pesticide practices must be balanced in terms of potential detriment to surrounding surface waters.

APPENDIX: SUPPLEMENTAL DATA

Table A.1. Source water sample used for rainfall simulations.

Nutrient	Water sample
	— $mg\ L^{-1}$ —
Nitrate	< 0.25
Ammonia	0.31
Dissolved Phosphorus	0.53

Table A2. Soil moisture readings for each turfgrass coverage prior to rainfall simulation pooled across 2010 and 2011.

Coverage	Soil moisture
— % —	— % —
0	25.2c†
25	27.8bc
50	30.0bc
75	33.2a
100	30.5ab

†Numbers followed by the same letter within a row are not significantly different according to Fisher's LSD at α = 0.05.

VITA

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