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EFFECTS OF TURFGRASS ESTABLISHMENT METHOD AND MANAGEMENT ON THE QUANTITY AND NUTRIENT AND PESTICIDE CONTENT OF RUNOFF AND LEACHATE

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ABSTRACT

The environmental impacts and human health effects of agricultural chemical use have come under intensive investigation in recent years. In particular, the occurrence of pesticides and nutrients in surface and ground water resources have attracted the attention of the scientific community, as well as the general public and the media. The application of these materials for the purpose of maintaining urban and suburban landscape plantings has received a great deal of unscientific scrutiny, and is presently being considered by legislators in various states as a target of increased regulation. Very little research data are available that support or refute the concerns of either the opponents or proponents of these nonagricultural uses of pesticides and nutrients.

A research facility was developed at Penn State for the purpose of studying the water quality impact of pesticide and nutrient use in the urban landscape. Turfgrass was established on sloped plots by either sodding with 100% Kentucky bluegrass or seeding with either of two species mixtures. Site preparation and turfgrass maintenance were consistent with methods employed by professional contractors in the northeastern U.S., including regularly scheduled applications of nutrients and pesticides. Irrigation was used before and after each chemical application to produce runoff and leachate samples for quality analyses, as well as to provide previously unavailable hydrologic data for high quality turfgrass sites. In addition, all natural events were monitored and sampled, when possible. During the period August 1985 to September 1988, evaluations were made of various turfgrass quality and

soil physical properties that could affect the hydrologic or chemical attenuating response of the various turfgrass types.

In general, runoff volumes from all treatments were less than expected or indicated by the literature. Natural precipitation events did not produce detectable runoff during the experimental period. Seeded plots yielded average runoff volumes between 10 and 15 percent of a 152 mm/hr event for one-hour irrigation; average runoff from sodded plots was less than 1 percent. Compared statistically to the seeded treatments, sodded plots had a significantly higher infiltration capacity, which resulted in both the lower runoff volumes from sod and a longer lag time between the onset of precipitation and runoff detection. Reasons for the differences were not readily apparent, but greater thatch development and possibly better soil structure associated with sod were considered likely. Initial differences were attributed to differences in vegetative cover; however, with eventual dense cover on seeded plots, they still had runoff rates that exceeded the sodded areas.

The average concentrations of NO_3^--N, PO_4^--P, and K in runoff and leachate samples were less than 5 mg/L; 2,4-D acid, 2,4-DP acid, and dicamba were less than 35 ug/L. The herbicide pendimethalin and the insecticide chlorpyrifos were not detected in any sample; nor were the esters of 2,4-D and 2,4-DP. Nutrient concentrations were typically about the same as those observed in tap water. Transport of pesticides and nutrients was greatest for events immediately following applications, and was limited to less than 5 percent of nutrients and 2 percent of pesticides for the worst case events. Partitioning into runoff or leachate appeared to be related to the relative solubility of individual materials.

Based on the hydrologic nature of the turf plots and the chemical transport estimates, losses to surface runoff of those products applied does not appear to pose a significant hazard at this site. Downward, rather than overland, movement of these materials is more likely to occur; however, various decomposition and attenuation processes exist that should lessen the chance of hazardous levels reaching groundwater resources.

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INTRODUCTION

During the past several years, a great deal of the public's attention has been directed at the use and possible effects of chemical fertilizers and pesticides that are employed by many different types of agriculture. Many individuals and groups have expressed concern over the safety of the foods that are produced using these products, and the extent to which they might be mobile or persistent in the environment.

One particular use of nutrients and pesticides that has been harshly criticized is for the maintenance of turf and ornamentals in urban and suburban landscapes. Rapid growth of the professional landscaping industry and a high degree of visibility to the urban populace have made lawn care applicators a favorite target of a sometimes less than understanding media. As a result, legislation is pending in many states with the aim of closely regulating these applications. Industry professionals, however, fear that many of the proposed regulations will make it difficult, if not impossible, for them to operate their business.

The objectives of the research addressed by this thesis were threefold: 1) to determine the effect of turfgrass establishment methods on hydrologic characteristics; 2) to assess the mobility of nutrients and pesticides common to professional turfgrass management from application sites; and 3) to estimate the combined impact of objectives 1 and 2 on water quality.

Chapter 1

LITERATURE REVIEW:
PESTICIDES AND NUTRIENTS
IN URBAN/SUBURBAN WATER RESOURCES

Much of the development in the agricultural chemical industry has occurred since the end of WWII [Brinkley, 1964; Kenyon, 1964]. Since then, the increase in use of chemical fertilizers and pesticides in U.S. agriculture has largely coincided with the development of the petrochemical industry. Throughout this period, the quantity, utility, and chemistry of agrochemicals have increased phenomenally. The 1987 issue of the Farm Chemicals Handbook lists hundreds of fertilizer, pesticide, and miscellaneous agricultural chemical substances [Meister, 1987].

Since the 1970's, the U.S. Environmental Protection Agency (EPA) has been assessing the environmental impact and human health effects of agrochemicals registered for use in this country. The EPA has been cooperating with various state and federal agencies to monitor the occurrence of pesticides and nutrients in surface and groundwater resources throughout the United States since about 1975 [EPA, 1986].

In a nationwide review of the problem of pesticides in groundwater, the EPA reported that "By 1985, 17 pesticides had been detected (in groundwater) in 23 states (due to normal applications to land surfaces). As many as 50 others have been detected, but the sources have not been identified" [EPA, 1986]. Such figures indicate the extent of pesticide occurrence in water resources, but do not reveal much about the frequency or manner in which pesticides may have gotten there. Aside from routine applications, the EPA recognizes the potential for contamination of ground or surface water at any time during the life of a pesticide. These include the manufacturing process, bulk transport and storage of concentrates, mixing, and disposal of excess spray material or pesticide containers [EPA, 1986].

Studies of the Chesapeake Bay and its tributary streams have identified nutrient enrichment (nitrate and phosphate) as one of the leading causes of lowered water quality and ecological degradation of that receiving body [EPA, 1982]. Detectable levels of pesticides were also noted in the Chesapeake Bay; however, their specific impact on estuarine species has not been deemed ecologically significant [EPA, 1982]. In a national survey of public water systems, McCabe et al. [1970] reported that 19 of approximately 635 supplies examined contained nitrate levels in excess of the public health limit. More recently, unsafe nitrate levels in groundwater supplies have become a concern.

The toxicities of registered pesticides have been determined, or are in the process of being determined for reregistration, as required by the EPA. Using these data and human exposure models, health risks have been assessed in order to establish tolerances for food crop residues. Tolerances for water resources, however, have been more difficult to determine due to a lack of consistent methodology for risk assessment [EPA, 1986]. Excessive nitrate concentrations in drinking water have been responsible for methemoglobinemia in infants [National Research Council, 1978], and nutrient loading to surface waters often results in their eutrophication [EPA, 1984].

The EPA and U.S. Geological Survey (USGS) have identified agriculture as the largest contributor of nonpoint source (NPS) nutrient and pesticide loads to receiving waters. Urban runoff is also recognized by these same agencies as a significant contributor of NPS pollution, including nutrients and pesticides [EPA, 1984]. Sources and loading potentials of contaminants within the urban landscape appear to be complex, poorly understood, and in need of further study. The vastly different characteristics of the two ecosystems requires separate investigation, because only limited and very careful extrapolation of findings from one to the other is appropriate.

The issue of nutrient and pesticide usage for urban/suburban (from hereon referred to as "urban") vegetation management has become an emotional one in many areas of the country. Recently, popular articles have addressed the potential health and environmental risks of chemicals used by commercial landscapers and "do-it-yourself" home owners [Romeo, 1986; Weidener, 1987; Giles, 1987]. Legislation to closely regulate the activities of commercial pesticide applicators exists or is pending in some states [Turgeon, 1985; Pordum, 1987; Wilkinson, 1988; Anonymous, 1988]. Little data exist to support or dispute the safety of urban pesticide and nutrient use. The purpose of this review is to examine the spectrum of information regarding pesticides, nutrients, and water quality as a means to better understand the <u>potential</u> impact of chemical vegetation management on urban watersheds.

Nonpoint Source Pollution in the U.S.

In 1984, the EPA identified "five significant nonpoint (pollution) sources" in the United States. These include, in order of relative importance: agriculture, urban lands, mining, silviculture, and construction [EPA, 1984]. Because of their relationship to urban landscape management, some key aspects of the agricultural and urban contributions are detailed here.

Nonpoint Source Pollution From Agricultural Lands

The variation in types of agricultural land use and the individualistic manner in which farm operators manage their acreage make it difficult to generalize about the impact of agriculture on water quality. Agricultural activities (including dryland and irrigated cropping, pasture and rangeland, and livestock holding facilities) contribute a variety of contaminants to surface and groundwater. These include sediment, nutrients, pesticides, salts and minerals, organic matter, ammonia, and fecal bacteria [EPA, 1984].

Most of the cultivated cropland in the U.S. is subject to soil erosion, with an average annual loss of about 11 metric tons per hectare. Between 25 and 40 percent of this potential sediment load finds its way to bodies of water [EPA, 1984]. Beasley et al. [1984] reported that the USGS measured an average 97 million metric tons of sediment per year flowing past St. Louis, MO, in the Mississippi River between 1964 and 1972. They also identified sediment as the second largest carrier of fertilizer and pesticide pollutants, after water [Beasley et al., 1984].

Nutrient loading (nitrogen and phosphorus) is associated with both cropping and livestock management. Nationwide, agricultural lands contribute 68 percent of the total nitrogen and phosphorus load to surface waters [EPA, 1984]. The EPA attributed between 50 and 55 percent of the total annual nutrient load to the Chesapeake Bay to nonpoint sources. Agriculture was determined to be the major source of nitrate. Phosphorus loading is mainly from point sources (urban sewage) in the Chesapeake Basin [EPA, 1982]; however, phosphate enriched sediment is a common loading source nationwide [Beasley et al., 1984]. In the Chesapeake Bay, much of the potential sediment load from the Susquehanna River was retained in impoundments along the lower reaches of the river [EPA, 1982]. In a related study in Lancaster County, PA, the NPS contributions of specific land uses were determined. Similar relationships were determined for nitrate and sediment loading from residential versus agricultural land uses [USGS, 1983].

An estimated 378 thousand metric tons of pesticides were used for agricultural production in the United States in 1980, including 199 thousand and 137 thousand metric tons of herbicides and insecticides, respectively [EPA, 1984]. In general, concentrations in surface and groundwater have been quite low, with a few specific exceptions [EPA, 1984]. Temporal variations in loading rate and concentration are common, with higher incidences of contamination occurring immediately following applications and/or major precipitation events [USGS, 1983; EPA, 1984; EPA, 1982].

Nonpoint Source Pollution from Urban Lands

Urbanization results in an increase in the percentage of impervious land surface (pavement, rooftops, etc.) within a watershed, which affects both its hydrologic response and pollution loading potential. The hydrologic effect is an increased runoff peak discharge rate and total runoff volume and a decrease in the runoff peak lag time [Delleur, 1982]. The pollution loading potential of a particular urban watershed is a function of precipitation patterns, population density, land use, and sanitary practices. Interpretation of this mathematical relationship as presented by Kibler [1982] suggests that pollution loading to stormwater increases with degree of urbanization, i.e., population density and intensity of land use. Sources of contaminants include air pollution fallout, automobile exhaust and tire deposition, animal waste, miscellaneous litter, lawn and garden chemicals, and sediment from the land surface; septic liquid from stormwater catch basins; and wastewater components in combined sewer flows [Roesner, 1982].

Within the urban watershed, pervious and impervious surfaces differ in their pollution loading potential [Roesner, 1982]. Many of the more noxious contaminants (e.g. metals and organics), by nature of their origin, are deposited on the impervious fraction. Nearly 100 percent of the precipitation striking impervious surfaces contributes to the washoff and transport of contaminants. For storms of insufficient intensity and duration to effectively flush the watershed, dissolved and suspended materials can be concentrated and stored in depressions until the occurrence of a major event [Pitt and Bozeman, 1982].

The frequency of pollutant washoff from pervious surfaces varies widely between sites, but, in general, it is much less than that from impervious areas. Roesner [1982] suggested that a modified version of the Universal Soil Loss Equation may be appropriate for the estimation of suspended solid loading rates. The transport of dissolved materials from the pervious landscape is a complex process affected by the chemical properties of the individual pollutant and the hydrologic characteristics of the landscape [Pionke and Chesters, 1973].

The nature of substances emanating from the urban land surface contrasts considerably with those from nonurban sources. For the determination of general water quality (urban, agricultural, forest, etc.), the EPA evaluates ten "standard pollutants" that characterize suspended solids, oxygen demand, nutrients, and heavy metals [EPA, 1983]. Indicative of the types of contaminants found in the urban environment, the EPA-coordinated Nationwide Urban Runoff Program (NURP) recognized 129 "priority pollutants" that included pesticides, metals and inorganics, and eight classes of organic compounds [EPA, 1983].

Twenty-one pesticides or degradation products were included in the EPA's list of priority urban pollutants. Nineteen of the 21 are organochlorine insecticides, many of which have long been discontinued or restricted for use only as subterranean termite controls around structures. One, acrolein, is a discontinued aquatic herbicide [Meister, 1987]. Eleven of the 21 pesticides or derivatives in the NURP review were detected with less than 10 percent frequency. Six were never detected. Four have been detected with greater than a 10 percent frequency [EPA, 1983]. Only one of the four, Lindane, is registered for an urban use other than structural pest control. Lindane is available to homeowners and professional applicators for use as an ornamental and garden insect control [Chemical and Pharmaceutical Press, 1988].

Various references have been made in the literature as to the probability of urban pesticide and fertilizer use contributing to the NPS pollution load [Delleur, 1982; EPA, 1984]. To date, however, the EPA has not included most of the commonly used lawn and garden pesticides in its NPS reviews, so the incidence or extent of their occurrence in urban runoff is not documented. About 80 different turf product formulations are listed in the 1988 Crop Protection Chemicals Reference [Chemical and Pharmaceutical Press, 1988]. Annual home and garden pesticide use is projected to be around 42.4 thousand metric tons for the next five years [Fertig, 1987]. Numerous sources of nutrients were also acknowledged, including animal wastes, various sewage and septic origins, fertilizers, and atmospheric deposition.

Environmental Fate of Pesticides and Nutrients and Entry into Water Resources

The environmental fate and persistence of any substance is dependent upon chemical, physical, and biological interactions it undergoes with its surroundings. The primary goal of pesticide and nutrient manufacturers is to develop products that utilize those characteristics to maximize efficacy and minimize nontarget effects [Hansberry, 1966]. Formulations are based on the chemical nature of the active ingredient, mode of action, delivery requirements, and environmental behavior.

For a substance to affect water quality, it must be capable of entering the hydrologic system and reaching a resource in significant quantity. There are numerous ways in which this may occur, including direct contact with a water body, surface runoff of water and sediment, subsurface interflow (lateral), and deep percolation to groundwater. The fate and persistence of a compound is likely to vary according to the pathways it follows in the environment.

Direct Contact With Water

Direct contact of agrochemicals with water can occur accidentally, as drift, atmospheric deposition of volatilized material [Pionke and Chesters, 1973], or a spill, or intentionally, as aquatic applications. Nearly all pesticide labels contain a precautionary statement pertaining to drift and their disposal near water. In addition, about 25 different pesticides are registered for aquatic or drainage ditch applications [Chemical and Pharmaceutical Press, 1988].

Land Applications

Most agrochemicals are applied directly to soil or vegetation on the land surface, either as soil or foliar treatments. Some may be subjected to rapid volatilization (urea fertilizer) [Tisdale et al., 1985] or decomposition by ultraviolet radiation (s-triazine herbicides) [Anderson, 1983], unless incorporated to maintain their effectiveness. A portion may be absorbed by crops or pests and metabolized. The fraction that remains on the surface is either adsorbed to soil particles and organic matter, or is dissolved by water.

Subsurface Pathways of Agrochemicals

Under conditions favorable for water percolation, these chemicals move downward into the soil profile. Some, such as paraquat, become so tightly bound to the soil as to be essentially unavailable for dissolution in water or biological degradation [Weed Science Society, 1983]. Others, like nitrate, are highly mobile and can quickly leach if not utilized by plants or microorganisms [Tisdale et al., 1985]. Most chemicals continue to dissipate via root uptake, microbial degradation, and chemical reactions [Anderson, 1983]. The retention of a particular chemical species in the soil matrix is affected by its affinity for various components of the soil-water system. Nonionic pesticides tend to adsorb to the organic soil fraction, and cationics are attracted to negatively charged exchange sites. Anionic species are only weakly adsorbed to most clay minerals. Solubility is highly pH dependent for acidic or basic compounds. Leaching is greatest for weakly adsorbed molecules in conditions that favor high solubility [Pionke and Chesters, 1973].

The mechanisms by which pesticides and nutrients reach groundwater are difficult to study and hence poorly understood. Little is known about the chemical or possible biological nature of the vadose zone that lies between the biologically active soil and parent bedrock. It is thought by some that chemical contaminants that percolate deeply are not subject to the many degradation processes that exist in the upper 3 to 6 feet of "soil," and may be free to reach the groundwater "intact."

Surface Pathways of Agrochemicals

Surface movement of pesticides and nutrients is generally associated with overland flow of rain or irrigation water, or soil erosion by wind or water. Soil conservation techniques have been developed that are effective in keeping erosion processes to a minimum [Beasley et al., 1984]. Wauchope [1978] reviewed the incidence of about 50 pesticides in agricultural runoff, and concluded that greatest runoff losses occurred (1) from the initial runoff-producing rainfall events after pesticide applications; (2) on slopes greater than three percent; (3) of water-insoluble formulations applied as emulsions; and (4) of unincorporated pesticides. Greatest losses of nitrate in runoff, and ammonium and phosphate on eroded sediment, have been noted when events occur before crop plants can utilize the nutrients [Timmons et al., 1968]. Pionke and Chesters [1973] examined the partitioning of pesticides between sediment and the aqueous phase after their entry into receiving waters. They concluded that the same interactions between the pesticide, sediment and water occur as in the terrestrial environment, but the equilibriums can shift due to the change in aqueous and ionic concentrations.

Soil and Water Management Considerations

The objective of most soil and water management plans is to provide for the maximum reduction of surface water flow and its associated sediment loss. The erosion potential of a soil is a function of its inherent erodibility, precipitation patterns, slope length and steepness, land use, and conservation practices [Beasley et al., 1984]. Factors influencing runoff include precipitation intensity and duration, infiltration capacity, subsurface permeability, evapotranspiration rate, surface cover, topography, and geology [Schwab et al., 1981].

Erosion Processes

The force associated with falling water droplets provides the kinetic energy required to detach soil particles from larger aggregates. A low, dense vegetative canopy or a thick surface mulch are the most effective means of protecting the soil surface from this form of structural degradation. Incorporation of organic matter can increase cohesive forces in the soil that resist the breakdown of structural aggregates [Brady, 1974]. Once dispersed, sediment is suspended in surface runoff sheet flows or concentrated flow channels, and transported downslope to be redeposited or enter receiving bodies of water [Foster et al., 1985].

Runoff Processes

Precipitation reaching the earth's surface runs off when other hydrologic abstractions have been satisfied. Vegetative and structural surfaces can intercept and store water until it is evaporated back to the atmosphere. After reaching the soil, water infiltrates the surface and begins to percolate into the subsoil. If precipitation exceeds infiltration capacity, water is stored in localized surface depressions [Viessman et al., 1972].

Mathematical treatments of overland flow processes have been quite rigorous. Advancements in computer simulation have significantly increased modeling capabilities for both developed and undeveloped watersheds. Until relatively recently, engineers were mainly concerned with the quantitative aspects of surface water management. Although it is now a major research priority, less progress has been made in the area of modeling water quality [EPA, 1983; Roesner, 1982].

The goal of most quantitative models is to relate peak runoff discharge rate, runoff peak lag time, and total runoff volume to a given design rainfall. Watershed yields are expressed in the form of graphical or tabular hydrographs, and model accuracy is determined by comparison to outflow hydrographs from gaged watersheds [Viessman et al., 1972]. Careful analysis of watershed surface characteristics are required, particularly its ability to infiltrate or otherwise store precipitation. The infiltration and storage capacities are influenced by land use patterns and inherent soil properties [Schwab et al., 1981].

Infiltration Processes

Infiltration of water into the soil is a function of the surface condition and subsurface permeability. Soil texture, surface aggregate stability, vegetation or surface mulch, pore size and continuity, flow limiting subsurface horizons, and antecedent moisture conditions all affect infiltration capacity of a given soil. Under "normal" conditions, i.e., an unsaturated profile with a porous surface, infiltration rate tends to initially be high and decrease rapidly to a limiting value. This trend is due to a decrease in capillary suction as the soil profile receives water, as well as the formation of an impeding crust resulting from the progressive deterioration of surface soil structure and subsequent sealing of conducting pores by fine particles [Hillel, 1982; Jennings et al., 1988]. Hence, surface management practices that promote or preserve aggregation, e.g. dense vegetative cover and application of mulches or physical amendments, will tend to reduce runoff and erosion by increasing infiltration. For the purpose of runoff prediction, hydrologic researchers have empirically derived coefficients that reflect the infiltration capacity of different soils and the effects of various land use management practices on soil surface conditions. Generally, coarse-textured soils and turfgrass or forest land are least conducive to runoff.

Effects of Construction and Urbanization on Soil and Water Management

Construction associated with urbanization can result in the alteration of a soil's inherent infiltration characteristics, and hence its runoff response. Excavating, transport, and regrading with heavy equipment disrupts natural profile horizonation, degrades structural aggregates, creates discontinuous soil layers, and excessively compacts the soil [Kelling, 1972]. McSweeney and Jansen [1984] observed that the rate and degree of soil restructuring that occurred after mineland reclamation activities varied widely across a given site with the degree and method of handling. Bullock et al. [1985] noted that the compaction of agricultural soils reduced macropore (> 0.6 um dia.) volume from 8 to 3.5 percent (v/v) in the upper 5 cm of soil. When left undisturbed, macropore regeneration to 13 percent (v/v) was achieved after 18 months through wetting/drying (shrinking/swelling) processes. Developed areas, however, are often subject to intense traffic (compaction), which increases bulk density and moisture retention, and hence reduces infiltration capacity [Waddington, 1965]. Kelling [1972] determined average infiltration rates for different land uses near Madison, WI and found that natural forest and prairie sites were considerably more pervious than vegetated sites that were previously disturbed by urbanization or agriculture. Active construction sites exhibited the lowest infiltration rates. Infiltration rates on home lawns varied considerably. The lowest rates were generally attributable to lawns of poor quality, compacted sites, and disturbed profiles. The highest rates were comparable to undisturbed prairie and forest lands.

Effects of Turfgrass Culture on Soil and Water Management

Turgeon [1980] defined turfgrasses as "plants that form a more or less contiguous ground cover"; and turf as "an interconnecting community of turfgrasses and the soil adhering to their roots and other below-ground organs." The latter definition implies the interactive nature of the perennial turfgrass community with its environment, which includes the atmosphere, the thatch-soil complex, pest species, and man.

The impact of turf on soil and water is a function of vegetative characteristics, interactions with environment, and the type and intensity of use. The benefit of turfgrasses for soil and water conservation has long been recognized. Young et al. [1980] observed 81 and 61 percent volume reductions in feedlot runoff passing through 36-m buffer strips comprised of orchardgrass and sorghum-sudangrass, respectively. Entrained solids and their associated nutrient and bacteriological components were similarly reduced, with the exception of nitrate, which is readily soluble.

Turfgrasses reduce runoff and sedimentation by enhancing conditions that promote infiltration. The dense, low growing foliage intercepts precipitation and absorbs the impact energy that would otherwise lead to structural degradation and crusting of the soil [Beasley et al., 1984]. The density, and in some cases interconnection, of turfgrass plants creates a tortuous overland flow path for water that reduces its velocity and increases its surface residence time. Surface litter and the macroporous nature of the thatch layer can absorb and facilitate entry of water into the system [Street, 1988]. Micro- and macrobiological activity associated with the degradation of this organic fraction contributes to the structural aggregation of the underlying soil, which maintains porosity. The profuse fibrous root system increases the ratio of macropores to micropores and maintains the surface layer in a loose, friable condition.

Under conditions of practical usage, stresses imparted to turf (particularly wear and compaction) and the activity of pests can reduce its quality and counteract its positive influence on infiltration. Most of the practices associated with turf management, such as irrigation, fertilization, pest control, and mechanical cultivation, have been developed to maintain the turfgrasses at an optimum level of quality while being subjected to the stresses created by intensive use or poor soil conditions on disturbed and revegetated areas [Schmidt and Blaser, 1969]. Presumably, heavily used turf that is highly maintained can continue to positively influence infiltration.

The Turfgrass Industry in the U.S.

Roberts and Roberts [1988] estimated that there are between 25 and 30 million acres of turfgrass in the U.S., with over 81 percent of this acreage in lawns. They reported that approximately 500 thousand people are employed by the turfgrass management industry in the U.S. Uses of turfgrass include lawns (residential, corporate, parks, schools, etc.), cemeteries, athletic fields, golf courses, bowling greens, airport and roadside right-of-ways, soil stabilization, and more [Nutter and Watson, 1969].

Initially, turfgrass management professionals and researchers were employed or supported by the golf industry. A recent boom in non-golf turfgrass management in the U.S. has paralleled an increase in single family housing and discretionary income. The proliferation of professional landscape management services and retail suppliers is testimony to the growth of the industry. Between 1965 and 1982, the economic value of the turfgrass industry increased from 4.3 to 25 billion dollars annually. The fastest growing sector of the industry, chemical lawn services, reported a 40 percent increase in revenues from 1983 to 1984 [Roberts, 1986]. A more recent survey estimated average industry revenue growth from 1986 to 1987 at 15 percent [Maras, 1988]. Growth is expected to slow around 1990, and level off early in the 21st century. During the 1980's, the "do-it-yourself" homeowner market has been growing 14 percent annually. During this period, sales of lawn/garden chemicals and fertilizers have increased six percent annually [Roberts, 1986]. These statistics suggest that the use of pesticides and nutrients in the urban environment should be projected to increase.

Turfgrass Management and Water Quality

The dramatic increase in the incidence of urban agrochemical use has generated research interest in their possible environmental impact. Much of this work is in its early stages. Very little data have been published to date, especially with respect to the mobility of pesticides applied to home lawns.

Kelling and Peterson [1975] observed inorganic nitrogen and orthophosphate concentrations of 0.8 to 2.1 mg/L and 0.5 to 0.7 mg/L (equivalent to 0.5 to 1.1 and 0.5 to 1.4 percent of applied), respectively, in runoff from home lawns shortly after application and watering in. Concentrations were considerably higher when watering in was not practiced or application rates were increased. Smallest losses were observed where infiltration rates were highest and runoff was delayed. Chichester [1977] found that inorganic nitrogen concentrations (ranging between 0.2 and 4.5 mg/L) in percolate from bluegrass meadow reflected fertilizer application rates and seasonal soil moisture flux patterns, but remained well below the U.S. Public Health Service drinking water standard of 10 mg/L NO_3^--N [National Research Council, 1978]. Nitrate concentrations in runoff samples were above the federal recommendation, but runoff was generally observed to be <1 cm/yr from the meadow sites. Morton et al. [1988] measured inorganic nitrogen concentrations between 0.2 and 5.6 mg/L in leachate from home lawn type plots established on a sandy loam soil and maintained under a range of watering and fertilization regimes. They observed the highest losses when plots were fertilized and/or watered excessively. Only two runoff events, both less than 0.2 cm, were recorded for the experimental period.

Both occurred under unusual conditions (frozen and saturated soil profiles), and nitrogen concentrations found were in the same range as the leachate samples.

It has been shown that nitrogen mobility can be managed by the correct choice of nitrogen source. Brown et al. [1982] noted that even on high sand golf greens, nitrogen losses were minimized by the use of organic sources that required conversion to nitrate for availability, in lieu of easily mobilized inorganic sources. In addition, the presence of nitrogen in the soil profile does not necessarily constitute a groundwater hazard. Shuford et al. [1977] observed negligible nitrate concentration changes in a well structured silt loam soil after a 27 cm deluge. Their studies suggest that mobile ions present in capillary pore spaces may not be easily leached, since the bulk of gravitational water drainage in saturated soils occurs in macropores.

These studies indicate that while the potential for inorganic nitrogen (and leachable pesticides) to leach from turfgrass managed in a typical fashion exists, concentrations in leachate may not necessarily be a health or environmental hazard. The possible interactions of soil types, fertilization and irrigation practices, topography, turf quality, etc. are too numerous to address individually. In general, the greatest potential for leaching appears to be associated with coarse-textured soils, inefficient fertilizer or pesticide use, and high rates of irrigation or rainfall throughout the growing season. Overall, the literature indicates that runoff events from good quality turf should be minimal.

Chapter 2

METHODS AND MATERIALS

The facilities for this project are located at the Landscape Management Research Center near the University Park campus of The Pennsylvania State University. The site, located on a variable slope (9 to 14 percent), was formerly utilized for soil erosion research and was allowed to return to a natural state for nearly 40 years before being renovated to accommodate this project. The soil is a Hagerstown series (Typic Hapludalf) originating from limestone residuum and typical of the karst topography found in the Ridge and Valley province of central Pennsylvania [National Cooperative Soil Survey, 1981]. The surface soil was texturally classified as clay (23% sand, 36% silt, 41% clay), based on a particle size analysis at the time of tillage. This textural feature is more typical of subsurface horizons in the Hagerstown series, suggesting that significant erosion of the surface horizon(s) occurred as a result of the previous research conducted at the site.

Site Development and Plot Design

Renovation of the site took place from 1982 to 1985 and included grading; installation of individual plot irrigation systems; installation of lysimeters in the upper and lower portions of the plot slopes; restoration of collection weirs; fabrication of flow monitor and subsampling equipment; and linkage of automated datalogging and computer systems. Twelve plots were constructed, and nine were used in this study. Surface preparation for turfgrass establishment consisted of rototilling (10 cm depth), stone removal, rolling, and leveling by hand raking. Plots were 6.5 m by 19 m and were separated by a plastic edging material (Edg-King^{TM}) that extended 10 cm into the soil. The purpose of the edging was to eliminate interplot surface and near surface movement of water or applied chemicals. Each plot (Fig. 1) contained 21 pop-up sprinkler irrigation heads (Weathermatic^{TM} 34-P) with head to head coverage calibrated to uniformly deliver water at a rate of 76 mm/hr during 1985. In 1986, the sprinklers were upgraded with #520 nozzles and calibrated to deliver 152 mm/hr thereafter [Telesco, 1986]. Output and uniformity of the irrigation system were checked periodically by randomly placing cans in the plots to capture and measure water. Positioned at the bottom of each slope was an epoxy-coated concrete weir that intercepted runoff water. The runoff was directed through a galvanized steel chute into a building that housed the flow monitoring and subsampling apparatus (Fig. 2). Pan lysimeter-type subsurface sampling devices (Fig. 3) were installed 150 mm below the soil surface for the purpose of capturing percolating water. The depth capacity of the samplers was 38 mm.

The lysimeters were constructed from round, high density polyethylene containers (265 mm diameter x 150 mm depth) that were filled with 16 mm diameter glass marbles as ballast material. A piece of polyester geotextile material separated the glass ballast from the overlying soil and prevented soil particles from entering the lysimeters. Polyethylene fittings installed at the top and bottom of the containers facilitated the venting and emptying of the samplers. Water samples were

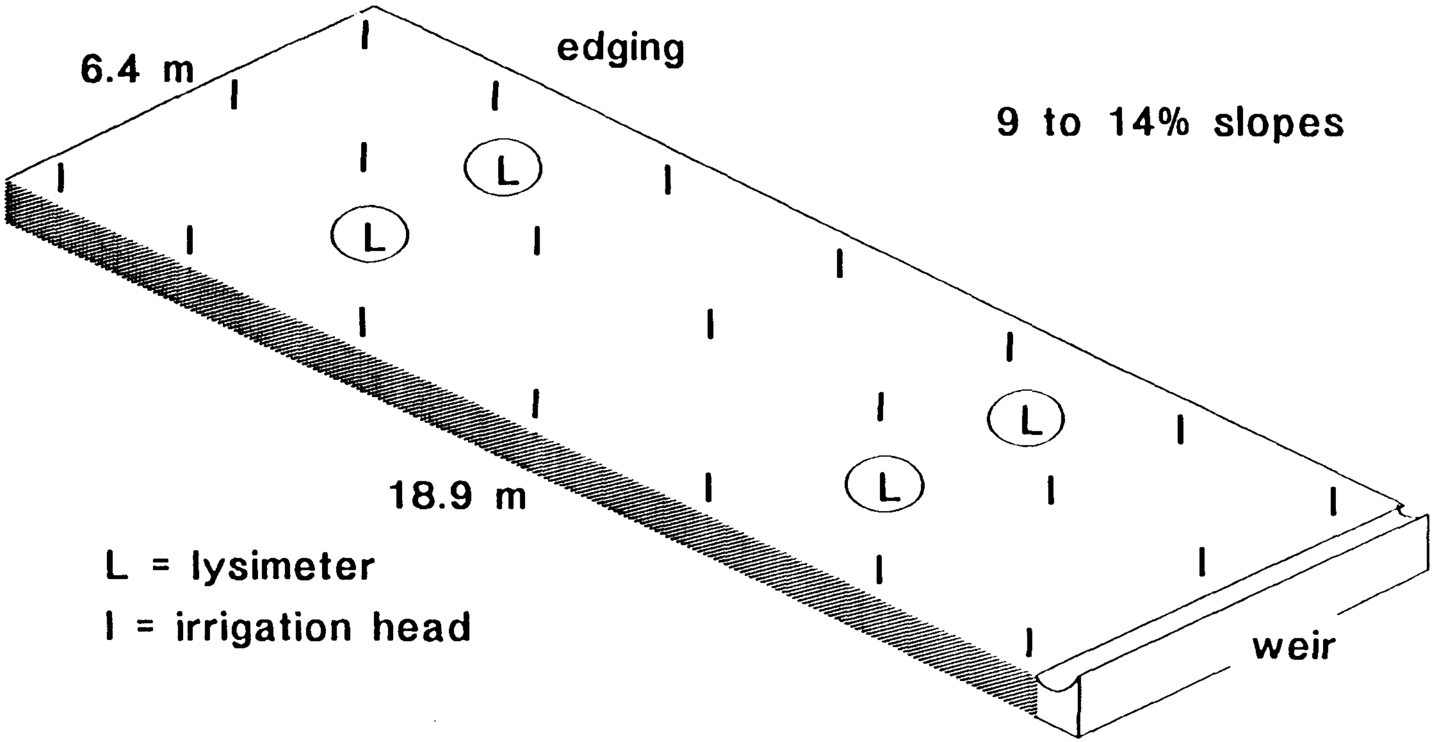

Figure 1. Plot schematic.

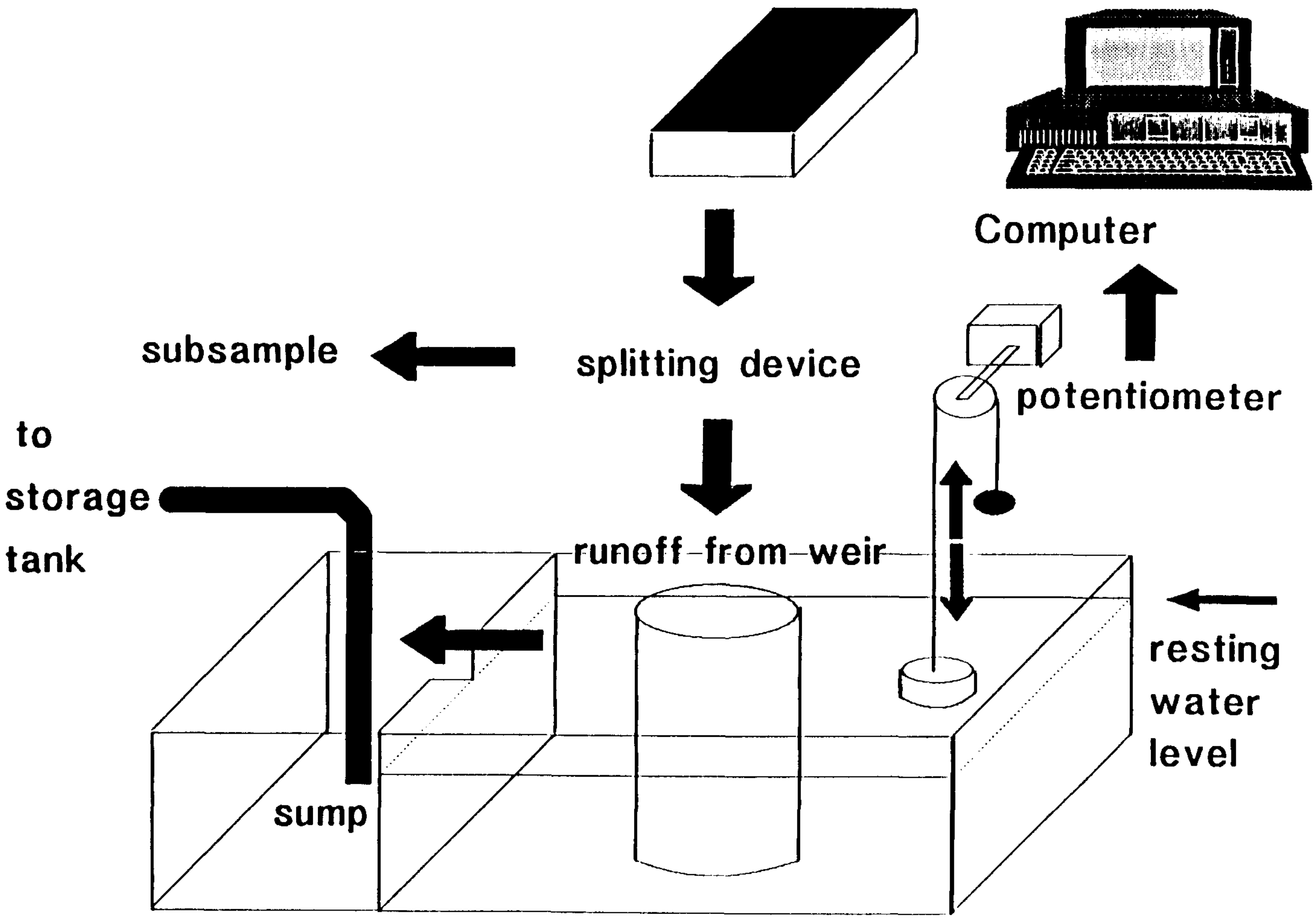

Figure 2. Flow monitoring/subsampling apparatus.

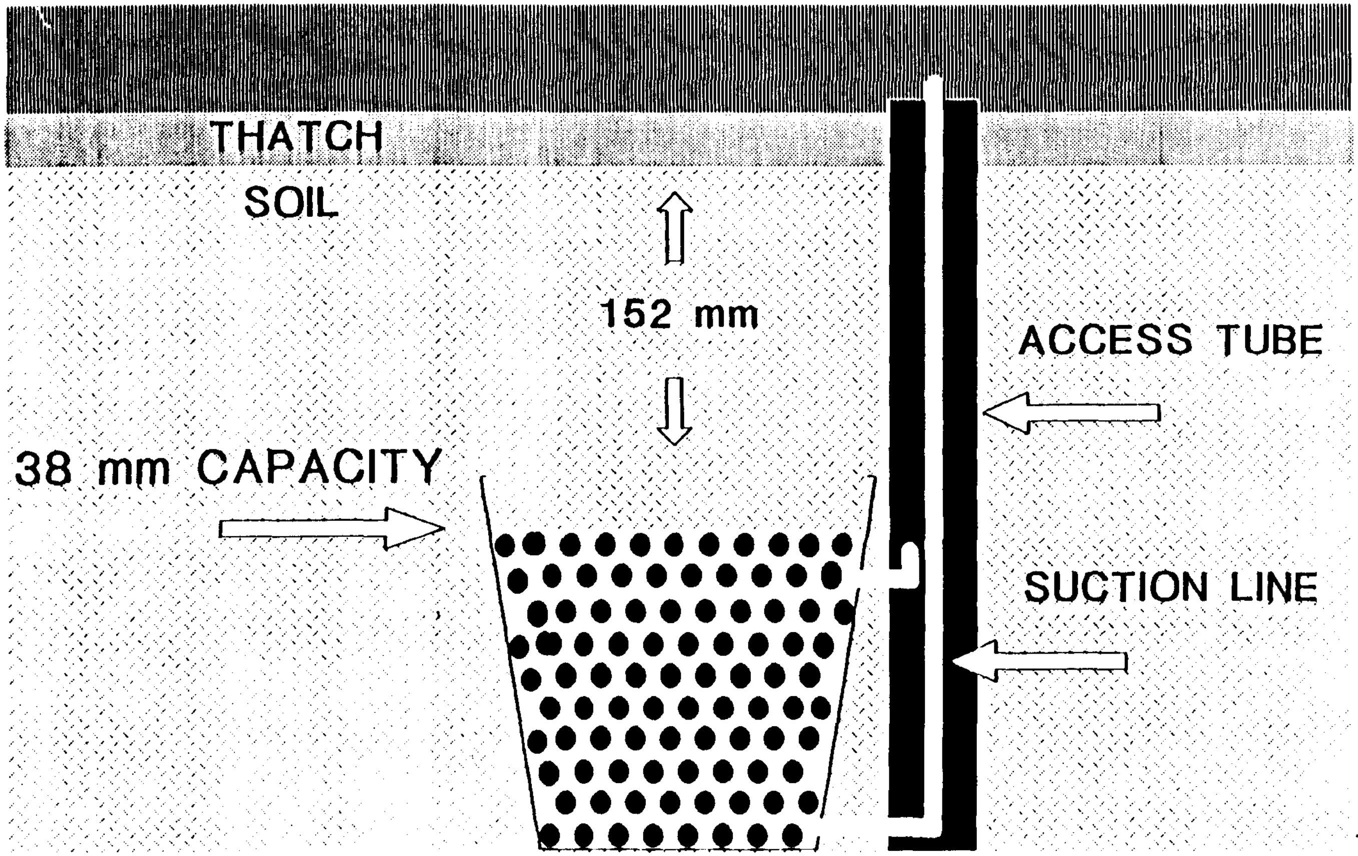

Figure 3. Lysimeter design and installation.

withdrawn through Tygon™ tubing located in a 10 cm access tube adjacent and downslope to the collectors with a centrifugal pump.

Inside the building, water from the chute flowed through a polyethylene splitting chamber (for subsample collection) and into a partitioned, 0.6 m by 1.2 m by 0.3 m deep steel tank. A length of 20 cm corrugated PVC pipe was suspended below the splitter to act as a baffle to minimize wave formation in the tank. Water accumulating in the receiving side of the tank flowed out through a standard 90 degree hydrologic V-notch into the exit chamber and was pumped to a storage/disposal tank. A float and counterweight assembly was positioned in the receiving side of the partitioned tank and was banded to a pulley attached to a potentiometer. As the float assembly responded to changing water levels in the tank (a function of runoff flow rate), it turned the potentiometer and produced a voltage signal uniquely associated with that water level and flow rate. The voltage signal in each building was read every 60 seconds by a microprocessor-equipped datalogger (ACUREX Autocalc Data Acquisition System) in an adjacent laboratory. The voltage signals were converted into flow rates and the data recorded on a bulk storage tape drive, accessible by PC communication software. The data collection system could be activated manually, or automatically by the detection of rainfall at an adjacent weather station.

Runoff water for quality analyses was subsampled continuously from the splitting chamber over the course of each runoff event. Water was transferred at a rate of 16 ml/min to a one liter high density polyethylene bottle (Nalgene™ 2114) through 0.635 mm diameter Tygon™

tubing via a peristaltic pump (Masterflex^{TM} Model No. 7017, Cole Palmer Instrument Co.).

Experimental Procedures

Three turfgrass types (treatments) were established in late June of 1985. The three experimental treatments, replicated three times in a randomized complete block design, were: (1) a seed mixture of perennial turfgrass species (CLASSIC), consisting of 25 percent 'Merit' Kentucky bluegrass, (Poa pratensis L.), 25 percent 'Julia' Kentucky bluegrass, 20 percent 'Shadow' chewings fescue (Festuca rubra ssp. commutata Gaud.), and 30 percent 'Citation' perennial ryegrass (Lolium perenne L.); (2) a "contractor's" seed mixture (CONTRACT) containing 60% annual ryegrass (Lolium multiflorum Lam.), 20 percent common Kentucky bluegrass, and 20 percent creeping red fescue (Festuca rubra L.); and (3) a three-year-old Pennsylvania Certified 100 percent Kentucky bluegrass sod (KBG SOD) grown from the following seed mixture: 'Adelphi' (25%), 'Baron' (25%), 'Fylking' (25%), and 'Nassau' (25%). Seed was applied at the rate of 19.6 gm/m^2 (CONTRACT) or 14.9 gm/m^2 (CLASSIC) with a drop spreader and mulched with clean wheat straw immediately afterward. All treatments received a complete fertilizer (according to soil test recommendation) at planting which supplied 48.6, 190.3, and 40.6 kg/ha of elemental nitrogen, phosphorus, and potassium, respectively (equivalent to 48.6, 437.4, and 48.6 kg/ha of N, P_2O_5, and K_2O, respectively). Approximately 89 percent of the phosphorus component was applied as superphosphate (46% P_2O_5) and incorporated during the rototilling operation. The remaining phosphorus and the nitrogen and potassium were added using a 15-15-15 (N-

P_2O_5-K_2O) fertilizer, which was applied and raked into the soil surface just prior to seeding or sodding. Soil pH was 7.0 and no lime was applied.

Turf Maintenance Practices

The turf maintenance program developed for the site is representative of practices commonly employed by home lawn care professionals in the northeastern U.S. All turfgrass cover types were maintained in an identical manner.

Plots were mowed weekly to a height of 5 cm (clippings removed) during the growing season. Irrigation was not employed as a routine maintenance practice; however, scheduled irrigations were used to produce runoff and leachate samples. Mechanical cultivation techniques such as core aeration, slicing, or spiking were not used.

Pesticides included in the study were pendimethalin [N-(1-ethylpropyl)-3,4-dimethyl-2,6-dinitrobenzenamine]; 2,4-D ester [(2,4-Dichlorophenoxy) acetic acid]; 2,4-DP ester [2-(2,4-Dichlorophenoxy) propionic acid]; dicamba [2-Methoxy-3,6-dichlorobenzoic acid]; and chlorpyrifos [O,O-Diethyl O-(3,5,6-trichloro-2-pyridyl)-phosphorothioate]. Beginning in 1986, plots were treated with pesticides and fertilizers four times annually as follows:

SPRING - Pendimethalin (1.68 kg/ha) for preemergence control of annual grassy weeds, plus a complete, soluble fertilizer (36.5, 4.2, and 8.0 kg/ha of elemental N, P, and K, respectively);

EARLY SUMMER - 2,4-D, 2,4-DP, and dicamba (1.12, 1.12, and 0.28 kg/ha) for postemergence control of broadleaf weeds, plus urea fertilizer (36.5 kg N/ha);

LATE SUMMER - 2,4-D, 2,4-DP, and dicamba (1.12, 1.12, and 0.28 kg/ha), plus chlorpyrifos (1.12 kg/ha) for the control of insect pest species, plus urea (36.5 kg N/ha);

FALL - 2,4-D, 2,4-DP, and dicamba (1.12, 1.12, and 0.28 kg/ha), plus urea (36.5 kg N/ha).

Water Sampling Procedures

Irrigations were conducted approximately one week prior to and two days after each chemical application, in order to produce runoff and leachate samples for analyses of pesticide and nutrient concentrations. Duration was typically 90 minutes for pre-application events and 60 minutes for post-application events. In addition, all natural precipitation events were monitored for the occurrence of runoff and percolate.

Water samples were collected immediately following precipitation or irrigation events, for subsequent processing and storage. From these, 250 ml aliquots were filtered through Millipore 0.45 um membrane filters and refrigerated at $4^{o}C$ until pH and nutrient determinations were completed. The remainder of the samples were frozen until pesticide concentrations were determined by the Pesticide Research Laboratory at Penn State University.

Water Quality Analyses Procedures

The pH was determined using a Coleman Model 39 Standard pH Meter equipped with Fisher universal glass indicator and reverse sleeve calomel reference electrodes. Nutrient concentrations were determined colorimetrically on a Hach DR/3000 Spectrophotometer. Nitrate-nitrogen (including nitrite) was determined using a cadmium reduction procedure; soluble phosphorus as orthophosphate via amino acid reduction of molybdophosphoric acid; and potassium via the tetraphenylborate method. The nitrate and phosphorus procedures are standard methods [American Public Health Administration, 1985] adapted for the Hach apparatus and are described in the instrument reference manual [Hach, 1986]. The potassium procedure is recommended for the Hach apparatus.

Determinations were made for concentrations of pesticides applied to the plots, including pendimethalin, 2,4-D (ester and acid), 2,4-DP (ester and acid), dicamba, and chlorpyrifos. A procedure was developed by the Pesticide Research Laboratory [Bogus et al., 1988] for the mass analyses of pesticides in two HPLC runs. The following is a general description of the procedures used for the determinations: After thawing, 100 ml aliquots were acidified with concentrated HCl to pH 2.2 and filtered through 0.4 um membrane filters. The acidified samples were drawn through a C-8 solid phase extraction column at a rate of 5-6 ml/min and then vacuum dried at room temperature for 30 minutes. The pesticides were eluted from the dried columns with four 500 uL methanol washes. The resulting 2 ml sample concentrates were processed in two runs on a Waters (Model ALC-GPC 244) HPLC apparatus equipped with a Lambda-Max 480 Variable Wavelength Detector. The system utilizes a 5 um by 4.6 mm by 25 cm C-18 column with a flow rate of 1 ml/min; detection for all seven components was at 230 nm. The mobile phase for dicamba, 2,4-D ester and 2,4-DP ester was a 55:45 mixture of methanol:aqueous 0.005 M Q8 ion pairing reagent (tetra-ammonium phosphate; Regis Chemical Company, Morton Grove, IL); for pendimethalin, the phenoxy acids, and chlorpyrifos, a 82:18 mixture of methanol:deionized water.

Turfgrass Quality Evaluations

Turfgrass quality parameters (color, cover, weeds, and overall quality) were visually estimated periodically throughout the growing season to document the development of the turfgrass and to determine whether stand quality was related to overland flow. Total vegetative cover was determined as a percent of the total area covered by vegetation (as opposed to stand density counts), and reflects the amount of exposed soil associated with each treatment. Weeds were also assessed as a percent of the total area covered by weed species.

Soil Physical Property Determinations

Water infiltration rates were determined for each of the plots during the summers of 1986, 1987, and 1988, utilizing eight, five, and three observations per plot, respectively, in each year. Determinations were made with double ring infiltrometers (20 cm dia. inner ring and 35 cm dia. outer ring, 10 cm height) using 25 mm of falling head [Bertrand, 1965]. Measurements were recorded after saturation with an initial flooding.

In October 1987, ten 5.1 cm diameter by 5.1 cm thick soil/thatch cores per plot were removed for bulk density determinations and thatch measurements. Samples were obtained with a double-cylinder, hammer-driven sampler [Bertrand, 1965]. Compressed thatch thickness of the intact core was measured under a 1 kg weight. After thatch removal, the soil core was sliced into five millimeter horizontal sections. The bulk density of each section was determined gravimetrically.

Statistical Design and Procedures

The three turfgrass cover types/establishment methods were arranged in a randomized complete block design with three replications. Data for vegetative characteristics were analyzed over all dates and statistically treated as a split plot in time, according to a model in Steel and Torrie [1980] designed for repeated evaluations of whole plots.

The three physically fixed water sampling locations did not allow randomization of this main effect, and resulted in a "split-block type" statistical arrangement of these data. Due to major differences in hydrologic conditions for each event, analyses of these data, as well as the hydrologic data, were limited to individual dates.