Water Quality Impacts Associated with Converting Farmland and Forests to Turfgrass

K. W. King, J. C. Balogh

Abstract. *Three to four hundred new or renovated turfgrass systems are constructed in the U.S. each year. Many of these systems (golf courses, city parks, and residential and institutional lawns) are constructed in agricultural and silvicultural environments. However, knowledge of the water quality impact in transitioning from an agricultural or silvicultural landscape to a turfgrass landscape is at best limited. Using the Soil Water Assessment Tool (SWAT) water quality model, 99–year simulations for three locations were completed for a continuous corn (*Zea mays *L.) agricultural rotation (AGR), a forested environment (FST), a golf course built in a previously agricultural setting (AGR–G), and a golf course constructed in a previously forested (FST–G) setting. Hydrologic, nitrate–nitrogen, and pesticide (2,4–Dichlorophenoxyacetic acid) impacts were evaluated. The hydrologic balance associated with AGR was significantly different from those for AGR–G, FST–G, and FST. Transition from FST to FST–G increased the loading and risk potential of surface runoff losses for both nitrate and 2,4–D and significantly increased ($\alpha = 0.05$) the potential for percolate losses of 2,4–D. Converting AGR to AGR–G significantly reduced the loading and risk potential for nitrate and 2,4–D losses. However, the addition of housing developments and increased impervious areas, which generally follow turfgrass land developments, were not considered, so the actual risk potential is probably higher than shown with this model. In addition to the impacts assessed, this study shows the SWAT model and associated simulation and analysis strategy to be a useful tool in evaluating risk assessments associated with land use conversions.*

Keywords. *Urban expansion, Land use change, Nitrate, 2,4–D, Agriculture, Silviculture.*

Many agricultural and forest ecosystems are being converted to urban landscapes, which contain a combination of impervious surfaces and turfgrass ecosystems. This urban sprawl is often associated with a demand for residential developments, retail centers, and recreational turfgrass systems. Between 1992 and 1997, the United States Department of Agriculture reported that approximately 1.3 million hectares of privately held, undeveloped property was developed for residential communities (Montaigne, 2000). This is twice the rate of development observed between 1982 and 1992. This conversion of agricultural land, forest ecosystems, or other natural ecosystems to urban and suburban uses (urban sprawl) is considered by many environmental groups as a critical environmental issue in the United States (McKinney and Murphy, 1996; Montaigne, 2000). Public objections to urban sprawl are often a policy issue regarding land use and land use changes. The focus of environmental reviews and assessments regarding specific land use conversion projects is often on the associated issues of water quantity and quality (Balogh and Watson, 1992; Biradar and Rayburn, 1995; Carrow, 1994; McKinney and Murphy, 1996; Racke and Leslie, 1993).

The management of new turfgrass areas (including residential, parks, and golf courses) is generally more intense than was previous landscape management. The water quality changes associated with the turfgrass portions of urban development have not been well documented on a watershed scale. Previous studies have addressed the environmental effects of turfgrass; however, these studies have often focused on small areas, from plot–size to individual greens or fairways (Balogh and Walker, 1992; Cohen et al., 1999; Clark and Kenna, 2000). Generally, water quality studies of turfgrass systems have not been directly compared with agricultural or native ecosystems. Small plot studies or field–scale computer simulations of turfgrass systems are valuable for assessing management impacts on water quality (King and Balogh, 1999); however, plot assessments do not represent the diversity and ecosystem interrelations associated with turfgrass systems on a watershed level.

Over 16,000 golf courses are in operation in the United States, with approximately 1.1 new or renovated courses opening every day (National Golf Foundation, 1999). Public demand is increasing for golf course superintendents to protect the quality of water and soil in the vicinity of these facilities while maintaining quality turfgrass (Balogh et al., 1992; Beard and Green, 1994). Turfgrass management, especially for new golf courses, is often the focal point of intense public debates regarding water use and water quality issues (Balogh et al., 1992). The objective of this work is to quantify the water quality impact associated with transitions from typical agricultural and forest environments to

Article was submitted for review in July 2000; approved for publication by the Soil & Water Division of ASAE in January 2001

The authors are: **Kevin W. King,** ***ASAE Member Engineer,*** Agricultural Engineer, USDA–ARS, Temple, Texas, and **James C. Balogh**, Soil Scientist, Spectrum Research Inc., Duluth, Minnesota. **Corresponding author:** Kevin W. King, USDA–ARS, 808 E. Blackland Road, Temple, TX 76502; phone: 254–770–6616; fax: 254–770–6561, e–mail: king@brc.tamus.edu.

recreational turfgrass systems while demonstrating a modeling strategy for risk assessments. The methodology presented in this study can be used to objectively evaluate the relative changes in water quality associated with land use conversion, the focus of many local debates.

METHODS AND PROCEDURES

MODEL SELECTION

The Soil and Water Assessment Tool (SWAT) (Arnold et al., 1998) model was the selected simulation tool for adaptation to this study. SWAT was selected because it is public domain, user friendly, robust, widely accepted, and validated in an array of geographical locations. Other advantages of SWAT include a comprehensive crop growth model and the ability to vary management schemes and practices throughout the watershed. SWAT has also been modified and tested for use in turfgrass environments (King, 2000).

SWAT is a comprehensive watershed scale model developed to predict management impacts on water, sediment, and chemical yields for ungaged basins. SWAT operates on a daily time step with the option of using either curve number (USDA–SCS, 1972) or Green–Ampt (1911) excess rainfall methods when measured data are available. SWAT can simulate long periods of time (100+ years), which is a requirement for evaluating long–term impacts of land use change.

SWAT validation and application efforts for predominantly agricultural environments have been completed for water yield (Arnold et al., 1993; Srinivasan and Arnold, 1994; Rosenthal et al., 1994; King et al., 1999), sediment yield (Bingner et al., 1996), nutrient loss (Jacobson et al., 1995), and pesticide fate and transport (Arnold et al., 1995). As with all models, inherent weaknesses associated with process simplification and model assumptions are present with SWAT. However, the previously cited studies show that, when appropriate parameters are used, SWAT simulations are reasonable. For comparing the type of systems addressed here, SWAT is the most appropriate model available. If different systems are compared other models may be more appropriate, but the same procedures are applicable.

EXPERIMENTAL DATA

A hypothetical 1.74 km^2 golf complex (fig. 1) was used as the foundation for this study and basis for determining model inputs. The same hypothetical design was used for each geographic location. The four treatments modeled were: typical agriculture, in this case continuous corn (*Zea mays* L.) (AGR); undisturbed forest (FST); a golf course cut out of the forest (FST–G); and a golf course cut out of the agricultural system (AGR–G). These systems were simulated for three locations (Dallas, Texas; Columbia, Missouri; and Minneapolis–St. Paul, Minnesota) for 99 years. The three locations were chosen in an effort to mirror a range of urban expansion with an array of climatic conditions that will affect management and turfgrass varieties.

Simulated management for continuous corn (AGR) included fertilizer and pesticide application, tillage, and harvesting. The simulated management for the golf course areas (AGR–G and FST–G) included fertilizer and pesticide

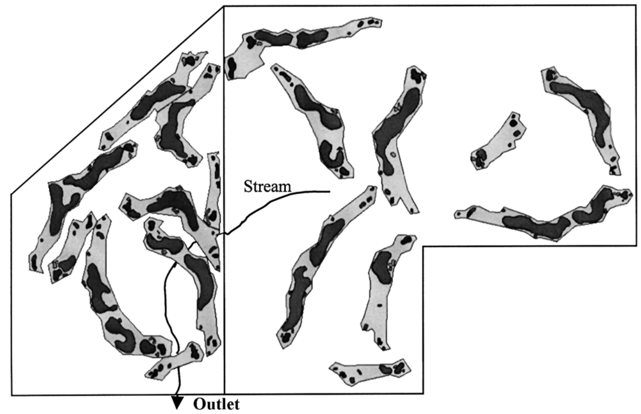

Figure 1. Layout of study area with golf course areas superimposed.

application, irrigation, mowing, and thatch control. No management activities were simulated for the forest (FST) condition. In the case of the forest with golf course (FST–G) land use, the forested areas were assumed undisturbed. In the AGR–G condition, the preexisting continuous corn areas not used for the golf course were converted to non–managed pasture/rangeland to mirror a real world scenario. Of the 1.74 km^2 area, tee boxes comprise 0.90 ha, greens 0.97 ha, fairways 11.44 ha, roughs 24.19 ha, and 136.54 ha were either forest or pasture/rangeland depending on treatment.

Topographic features from the hypothetical layout were used as input for SWAT and held constant for all locations. Baseline curve numbers were adapted from the USDA–SCS (1972) recommendation based on hydrologic soil classification (table 1) for the 3 locations. Daily curve numbers were adjusted in SWAT for both physical and managerial conditions based on a curve number modification suggested by Williams and LaSeur (1976). Average annual fertilizer applications (table 1) varied by land use and location and were comprised of slow release and fast release formulations applied at various times throughout the year. The slow release formulations followed the procedure of King and Balogh (2000). The pesticide used for AGR, AGR–G, and FST–G treatments was 2,4–D (2,4–Dichlorophenoxyacetic acid). 2,4–D has generally not been used on corn as often as it once was, but it is still labeled for pre–plant use on corn. Therefore, 2,4–D was chosen as the pesticide to simulate since it could be used on all treatments. Application rates (table 1) varied by management unit. The chemical properties for 2,4–D are: adsorption coefficient (50), water solubility (900 mg L^{-1}), wash off fraction (0.45), foliar half life (9 days), soil half life (10 days), maximum contaminant level (MCL) (70 $\mu g\ L^{-1}$), and the LC_{50} for amphibious species (110 $\mu g\ L^{-1}$). Irrigation for the turf areas of the golf course was modeled based on available water capacity (AWC). Irrigation was performed based on 75% AWC to conserve water use. When the AWC dropped below the 75% level, irrigation was initiated to bring the AWC to field capacity. Deficit irrigation has been found to be beneficial for maintaining high quality turfgrass when the availability of irrigation water is limiting (Balogh and Watson 1992; Carrow 1994).

DATA ANALYSIS

Comparison of modeling results to regulatory standards is one of the primary methods used to communicate risk to the

Table 1. Baseline parameter values and properties used for SWAT simulations.

Parameter	Dallas, Texas	Columbia, Missouri	Minneapolis–St. Paul, Minnesota
SCS curve numbers			
Greens	55	55	55
Tees	55	55	55
Fairways	61	80	61
Roughs	61	80	61
Corn	76	86	76
Forest	55	77	55
Pasture/Range	70	85	70
Native soil[a]			
Name	Silawa (fine–loamy, siliceous, thermic ultic Haplustalfs)	Mexico (fine, montmorillonitic, mesic aeric vertic Epiaqualfs)	Hayden (fine–loamy, mixed, mesic typic Hapludalfs)
USDA classification	B	D	B
Sand (%)	65.3	26.3	63.6
Silt (%)	19.7	52.7	26.4
Clay (%)	15.0	21.0	10.0
Sat. cond. (mm h^{-1})	67.1	7.2	65.5
Grass varieties	Bermudagrass (*Cynodon Dactylon* (L.) Pers.)	Kentucky Bluegrass (*Poa pratensis* L.) and Colonial Bentgrass (*Agrostis tenuis* Sibth.)	Colonial Bentgrass (*Agrostis tenuis* Sibth.) and Kentucky Bluegrass (*Poa pratensis* L.)
Fert. application[b] (kg ha^{-1} yr^{-1})			
Greens	758	538	342
Tees	624	391	245
Fairways	303	220	157
Roughs	49	49	49
Corn	180	180	180
2,4–D application[c] (kg ha^{-1} yr^{-1})			
Greens	1.35	1.23	1.23
Tees	0.56	0.49	0.49
Fairways	0.67	0.49	0.49
Corn	1.12	1.12	1.12

[a] Soils for greens and tees follow USGA recommendations (U.S. Golf Association, 1993).
[b] Actual nitrogen application amounts.
[c] Active ingredient.

public (Kamrin et al., 1995). One method of evaluating human health risks is to use the U.S. Environmental Protection Agency's Maximum Contaminant Levels (MCL). These human health advisory levels are used by the U.S. EPA for risk evaluation of potential chemical exposure in drinking water (EPA, 1986). The MCL is the highest amount or concentration of a compound allowed by the EPA in water supplied by a municipal water system and forms the basis of national primary drinking water standards. The drinking water standard or MCL for nitrate–nitrogen is 10 mg L^{-1} (EPA, 1986), while the MCL for 2,4–D is 70 μg L^{-1} (Kamrin, 1997).

The EPA makes a presumption of acceptable water quality risk for human health when the concentration of a contaminant in water does not exceed the MCL. The average annual number of days when the concentration exceeds the MCL is reported using a daily concentration frequency count (total number of days exceeding the target concentration divided by the number of years of simulation). This is one of the typical assessment formats used during an environmental review process. Frequency occurrence of concentrations on a daily basis is a reasonable approach considering the validity of SWAT for water balance (Arnold et al., 1993; Srinivasan and Arnold, 1994; Rosenthal et al., 1994; King et al., 1999) and mass load predictions (Jacobson et al., 1995; Arnold et al., 1995). Depending upon the current status of the water body in question, an appropriate range of concentrations to compare to the modeling results can be selected.

Results and Discussion

Hydrology

Average annual simulated precipitation was 878 mm for Dallas, Texas; 892 mm for Columbia, Missouri; and 701 mm for Minneapolis–St. Paul, Minnesota (fig. 2). The simulated average annual hydrologic response of the 4 treatments (fig. 3) indicated significantly ($\alpha = 0.05$) more surface runoff and percolation for AGR when compared to AGR–G, FST–G, and FST, regardless of location. Evapotranspiration was significantly lower for AGR than all other treatments. This result was expected because there is less vegetative cover in AGR and no irrigation was used. Native soil composition (table 1) was also a factor in the water balance of the watershed complexes. For example, evidence of a clayey soil influence can be seen in the large amount of surface runoff and evapotranspiration, along with reduced amounts of percolate and lateral flow, at Columbia, Missouri.

Nitrate–Nitrogen

Average annual total losses of nitrate (table 2) followed a geographic pattern. Losses in the south and central locations

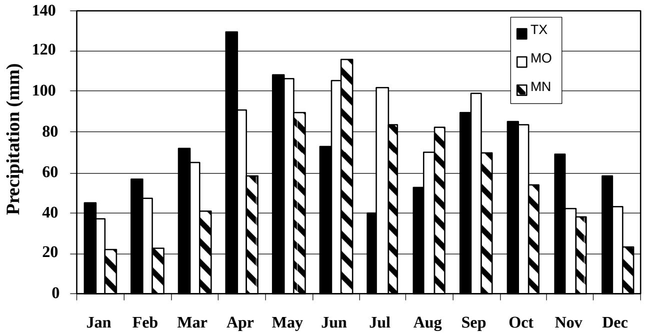

Figure 2. Simulated monthly precipitation distribution for the three study sites (TX = Dallas, Texas; MO = Columbia, Missouri; and MN = Minneapolis–St. Paul, Minnesota).

were attributed to the longer growing seasons and the additional nitrogen applied during that extended season on golf course areas. Total nitrate applications for AGR–G and FST–G were calculated at approximately 34 kg ha^{-1} in Dallas, Texas, 26 kg ha^{-1} in Columbia, Missouri, and 20 kg ha^{-1} in Minneapolis–St. Paul, Minnesota (table 1). Nitrate application for corn (AGR) was 180 kg ha^{-1} and held constant across all locations. When comparing treatments, AGR produced significantly ($\alpha = 0.05$) more nitrate in the surface runoff and percolate than all other treatments regardless of location. Nitrogen is only applied to approximately 22% of the total area in the case of AGR–G and FST–G, so the losses from the turfgrass managed areas are masked by surface runoff and percolate from non–managed turfgrass areas. However, nitrogen is applied to the whole area in the case of AGR, resulting in a much larger area–weighted amount. When the masking is taken into account (table 3), the losses from an individual fairway management unit (FRWY) lie between FST and AGR but are still significantly lower than the losses associated with AGR. This result suggests that the system as a whole acts as a natural buffer system. The buffering effects of pasture and forest vegetation found within the turfgrass management system should be accounted for when evaluating nitrogen losses.

The monthly distributions of surface nitrate losses (fig. 4) and percolate nitrate losses (fig. 5) generally follow the periods of greatest precipitation (fig. 2). Multiple nitrogen applications and slow release formulations associated with the fairway management and aggressive uptake by turfgrass cultivars limit the availability of nitrogen for transport via surface and percolate pathways. Even though nitrogen is applied to corn using a split application, the fast release formulation coupled with the relatively large amount of nitrogen leaves a substantial amount available for transport when rainfall occurs. In the case of AGR, between 5% and 8% of the applied nitrogen was recovered in the combined runoff and percolate on an annual basis for all locations (table 4), which is not uncommon for agricultural land (Nolan et al., 1997; Kolpin, 1997; Keeney, 1986). Nitrate losses leaching past the root zone from the golf course systems ranged from 1% to 5% and decreased from southern to northern locations.

The frequency of days when nitrate–nitrogen concentrations in surface water exceeded the water quality limit of 10 mg L^{-1} is low in all locations for AGR and is zero for the AGR–G, FST–G, and FST treatments (table 4). The potential risk of nitrate concentrations in percolate passing beneath the root zone is greatest for AGR, intermediate for AGR–G and FST–G, and lowest for FST. This result is consistent with plot–level reports by other investigators (Walker and Branham, 1992; Cohen et al., 1999). Conversion of land from agriculture to turfgrass or forest use significantly reduces the potential for nitrate contamination of groundwater. Conversely, conversion of forest land to turfgrass moderately increases the risk of groundwater contamination with nitrate. The evaluation used for this study focused on the EPA drinking water criteria of 10 mg L^{-1}. The same approach could be used for lower concentrations, depending upon the relative value of a water resource (e.g., assessing the impacts on a fishery).

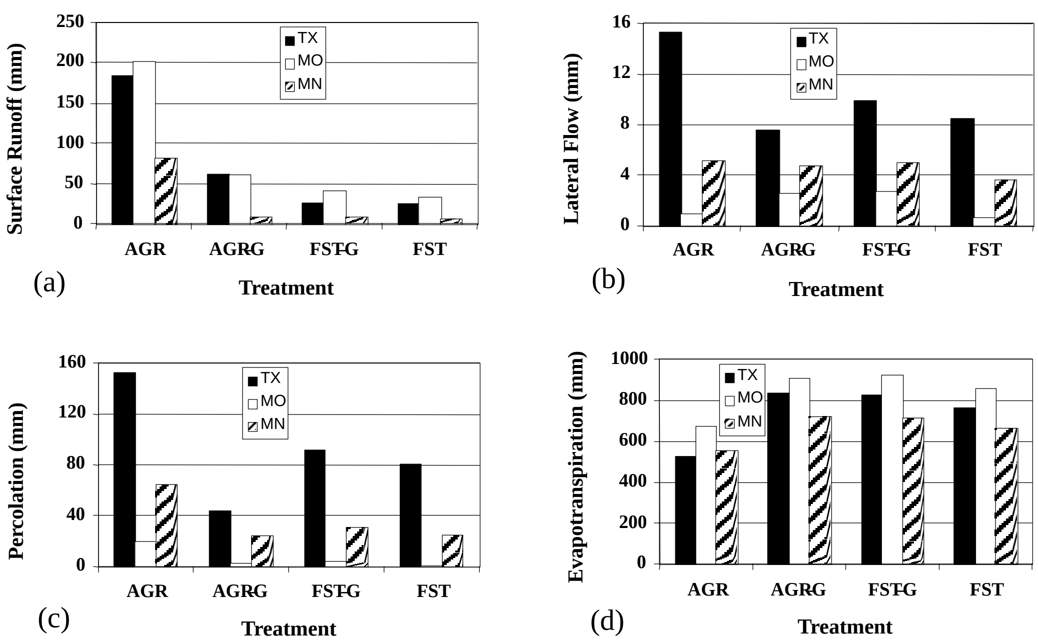

Figure 3. Simulated average annual watershed hydrologic response by treatment and location for (a) surface runoff, (b) lateral flow, (c) percolation, and (d) evapotranspiration.

Table 2. Simulated average annual (n = 99) watershed nitrate losses and 2,4–D losses by site and weighted by treatment.

Treatment	Nitrate Surface Losses (kg ha^{-1})	Nitrate Percolation Losses (kg ha^{-1})	Nitrate Lateral Flow Losses (kg ha^{-1})	Nitrate Crop Uptake (kg ha^{-1})	2,4–D Surface Losses (mg ha^{-1})	2,4–D Percolate Losses (mg ha^{-1})
			Dallas, Texas			
AGR–G	0.21 a[a]	1.23 ac	0.43 a	31.7 a	92 a	172 a
AGR	2.62 b	8.15 b	1.97 b	156.6 b	25741 b	153 a
FST–G	0.10 c	1.64 a	0.44 a	31.4 a	92 a	172 a
FST	0.06 c	0.70 c	0.08 c	8.0 c	0 a	0 b
			Columbia, Missouri			
AGR–G	0.29 a	0.01 a	0.05 a	29.1 a	660 a	17.1 a
AGR	4.44 b	9.16 b	0.16 b	151.9 b	51596 b	0.2 b
FST–G	0.26 a	0.33 a	0.05 a	28.7 a	660 a	17.1 a
FST	0.20 a	0.32 a	0.01 c	8.1 c	0 a	0 b
			Minneapolis–St. Paul, Minnesota			
AGR–G	0.14 a	0.08 a	0.06 a	25.2 a	12 a	179 a
AGR	1.67 b	7.79 b	1.08 b	154.0 b	10701 b	510 b
FST–G	0.16 a	0.42 a	0.08 c	24.9 a	13 a	184 a
FST	0.14 a	0.04 a	0.02 d	6.8 c	0 a	0 c

[a] Means, within a column by location, followed by the same letter are not significantly different using Tukey's pairwise comparison (α = 0.05).

Table 3. Simulated average annual (n = 99) nitrate losses and 2,4–D losses by site, assuming a homogenous land use.

Treatment	Nitrate Surface Losses (kg ha^{-1})	Nitrate Percolation Losses (kg ha^{-1})	Nitrate Lateral Flow Losses (kg ha^{-1})	Nitrate Crop Uptake (kg ha^{-1})	2,4–D Surface Losses (mg ha^{-1})	2,4–D Percolate Losses (mg ha^{-1})
			Dallas, Texas			
AGR	2.7 a[a]	7.92 a	1.87 a	157.4 a	23128 a	150 a
FRWY	0.5 b	3.26 b	0.18 b	238.6 b	1049 b	2896 b
FST	0.1 c	0.89 c	0.09 b	8.1 c	0 b	0 a
			Columbia, Missouri			
AGR	4.5 a	10.0 a	0.16 a	151.1 a	56429 a	0.10 a
FRWY	1.3 b	0.0 b	0.00 b	181.8 b	9783 b	2.00 b
FST	0.3 c	0.05 b	0.00 b	8.1 c	0 b	0 a
			Minneapolis–St. Paul, Minnesota			
AGR	1.5 a	7.4 a	1.03 a	154.7 a	13284 a	607 a
FRWY	0.3 b	0.7 b	0.00 b	169.2 b	111 b	2523 b
FST	0.1 b	0.1 b	0.00 b	7.0 c	0 b	0 c

[a] Means, within a column by location, followed by the same letter are not significantly different using Tukey's pairwise comparison (α = 0.05).

Pesticide (2,4–D)

Total 2,4–D applications for AGR–G and FST–G were approximately 0.05 kg ha^{-1} in Dallas, Texas, and 0.04 kg ha^{-1} in Columbia, Missouri, and Minneapolis–St. Paul, Minnesota (table 1). The 2,4–D application for corn (AGR) was 1.12 kg ha^{-1} and held constant across all locations. Average annual losses of 2,4–D were dominated by surface losses associated with AGR (table 2). Surface losses of 2,4–D from AGR–G and FST–G were not significantly different (α = 0.05) from that of FST, where 2,4–D was not applied. In the case of percolate losses of 2,4–D, losses associated with AGR–G and FST–G were significantly greater than zero for all locations. As with the nitrate analysis, losses of 2,4–D from the turfgrass systems within AGR–G and FST–G are masked by dilution with percolate from untreated areas. When comparing 2,4–D losses from a fairway condition (FRWY) to AGR and FST, there is an obvious difference in analyzing the system as a whole (table 3). The surface losses from FRWY are still significantly smaller than those of AGR and not significantly different from FST, but the magnitude of those losses has risen approximately 8–15 fold depending on location. FRWY percolate losses are significantly greater than AGR and FST. This result suggests the necessity of evaluating the turfgrass systems within the whole watershed complex, rather than on a plot– or small–scale basis.

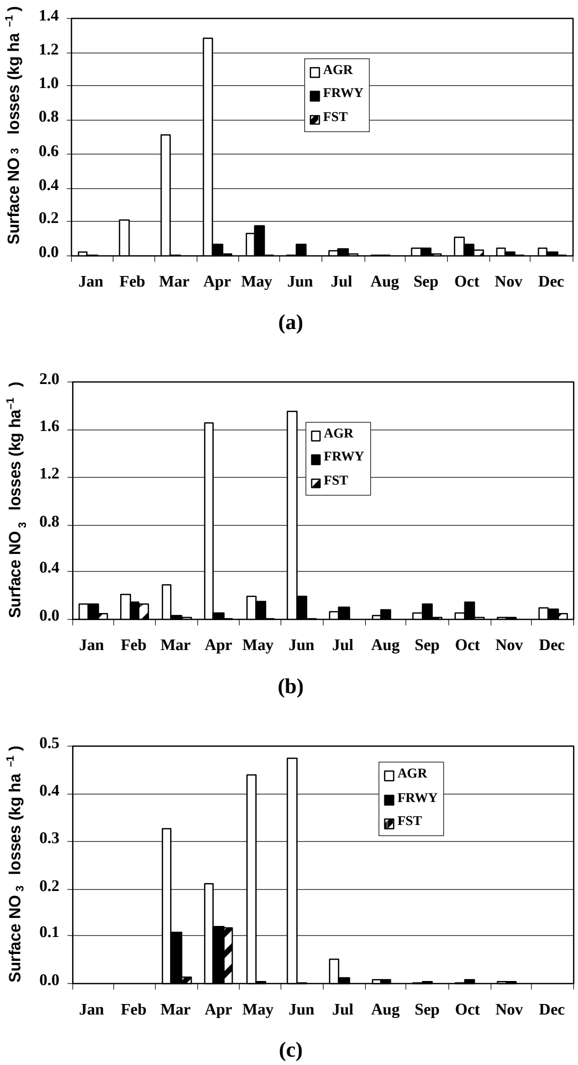

Figure 4. Simulated monthly distribution of nitrate losses in the surface runoff for (a) Dallas, Texas, (b) Columbia, Missouri, and (c) Minneapolis–St. Paul, Minnesota.

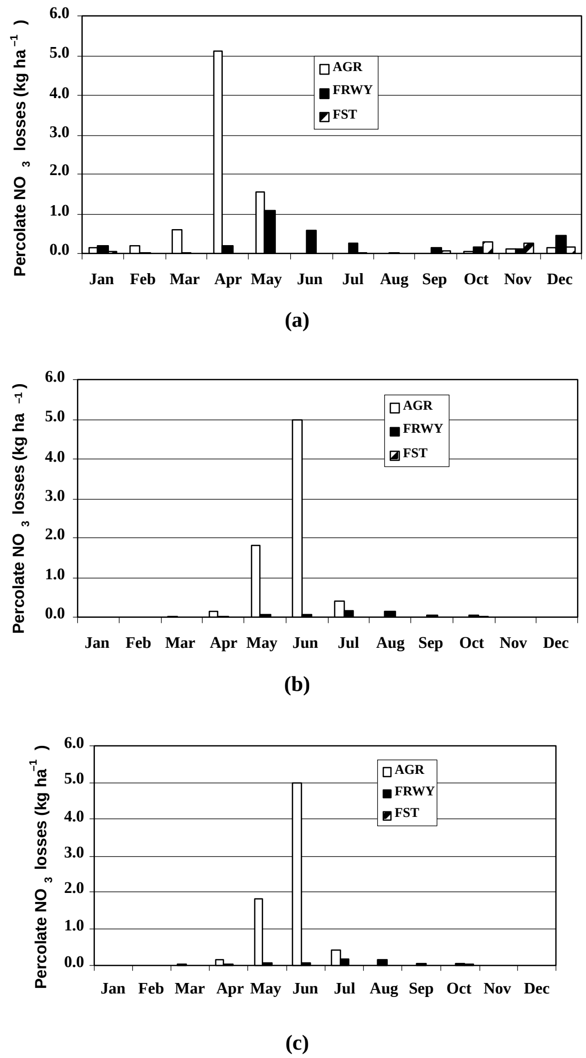

Figure 5. Simulated monthly distribution of nitrate losses in the percolate for (a) Dallas, Texas, (b) Columbia, Missouri, and (c) Minneapolis–St. Paul, Minnesota.

The frequency of days when 2,4–D concentrations in surface water exceeded the water quality limit of 70 μg L^{-1} is low in all locations for AGR and is zero for the AGR–G, FST–G treatments (table 5). However, detectable levels of 2,4–D in surface water exist for all systems in which 2,4–D is used. Using a more conservative number of 5% of the MCL (3.5 μg L^{-1}) increases the likelihood of exceeding the water quality limit (table 5). Chemicals with greater water quality concerns such as 2,4–D require the evaluation of impacts at lower concentrations to account for possible chronic versus acute exposures. The potential risk of 2,4–D concentrations in percolate passing beneath the root zone is greatest for AGR, followed by moderate to low risk for AGR–G and FST–G.

The soils at the different locations have a significant affect on leaching potential. The fine–textured soils dominant in Columbia, Missouri, significantly reduce the risk of groundwater contamination but increase the risk associated with surface water contamination (table 5). This is consistent with plot–scale findings by other investigators (Balogh and Anderson, 1992; Cohen et al., 1999). Conversion of land from agriculture to turfgrass has little impact on the potential for 2,4–D contamination of groundwater. Conversion of forest land to turfgrass significantly increases the risk of surface water contamination if 2,4–D is used for weed control on golf courses. In both the agricultural and turfgrass management systems, the detection of 2,4–D in surface water and groundwater is both a human health and water quality risk. Land use planners and policy makers must consider the risks and alternative management practices when making decisions regarding potential development of either agricultural or forest land.

Summary and Conclusions

Urban expansion and demand for recreational turfgrass systems have resulted in the conversion of both agricultural and forest lands to other uses. When conversion of forest land to turfgrass is considered, there exists a loading and risk potential for increases in the surface losses of both nitrate and 2,4–D, and a significant potential exists to increase the percolate losses of 2,4–D. When considering a transition

Table 4. Average annual frequency analysis of nitrate–nitrogen by treatment and location.

	AGR–G	AGR	FST–G	FST
Dallas, Texas				
NO_3–N runoff events per year (>10 μg L^{-1})[a]	337.8	338.9	330.6	309.7
Max. conc. in runoff for simulation period (mg L^{-1})	2.9	31.5	3.4	1.1
NO_3–N runoff events (>10 mg L^{-1})[b]	0.0	1.2	0.0	0.0
NO_3–N percolate events per year (>10 μg L^{-1})	113.1	184.5	149.1	110.9
Max. conc. in percolate for simulation period (mg L^{-1})	35.2	58.4	35.2	25.1
NO_3–N percolate events (>10 mg L^{-1})	4.3	27.7	4.1	0.4
Percent of applied mass recovered in runoff	0.6	1.5	0.3	NA
Percent of applied mass recovered in percolate	3.6	4.5	4.8	NA
Columbia, Missouri				
NO_3–N runoff events per year (>10 μg L^{-1})	338.5	353.8	334.8	283.6
Max. conc. in runoff for simulation period (mg L^{-1})	12.9	34.2	9.1	10.0
NO_3–N runoff events (>10 mg L^{-1})	0.0	2.1	0.0	0.0
NO_3–N percolate events per year (>10 μg L^{-1})	5.1	58.9	1.1	0.8
Max. conc. in percolate for simulation period (mg L^{-1})	0.2	146.5	30.1	43.7
NO_3–N percolate events (>10 mg L^{-1})	0.0	58.9	1.0	0.8
Percent of applied mass recovered in runoff	1.1	2.5	1.0	NA
Percent of applied mass recovered in percolate	0.0	5.0	1.2	NA
Minneapolis–St. Paul, Minnesota				
NO_3–N runoff events per year (>10 μg L^{-1})	217.6	262.9	232.3	196.0
Max. conc. in runoff for simulation period (mg L^{-1})	11.8	21.7	9.1	11.7
NO_3–N runoff events (>10 mg L^{-1})	0.0	0.5	0.0	0.0
NO_3–N percolate events per year (>10 μg L^{-1})	34.2	95.2	47.3	27.2
Max. conc. in percolate for simulation period (mg L^{-1})	6.9	67.7	48.1	30.1
NO_3–N percolate events (>10 mg L^{-1})	0.0	43.7	0.8	0.4
Percent of applied mass recovered in runoff	0.6	0.9	0.7	NA
Percent of applied mass recovered in percolate	0.3	4.3	2.0	NA

[a] 10 μg L^{-1} (ppb) represents detectable limit.
[b] 10 mg L^{-1} (ppm) EPA drinking water standard (EPA, 1986).

from agricultural lands to a turfgrass system, the loading and risk potential for nitrate and 2,4–D losses is significantly reduced. The watershed–based approach also revealed how the existing landscape can act as a natural buffer.

These results are important for land use planners and managers to consider when deciding to make long–term changes in the existing landscape. It should also be noted that the discussion in this study is only indicative of the changes specific to the conversion of agriculture or forest lands to golf courses and does not consider the impacts of added urban expansion that often accompany golf course developments. The use of the SWAT model and frequency analysis approach demonstrates a modeling strategy and evaluation method typical for risk assessments regarding changes in land use.

Table 5. Average annual frequency analysis for 2,4–D by treatment and location.

	AGR–G	AGR	FST–G	FST
Dallas, TX				
2,4–D runoff events per year[a] (> 10 ng L^{-1})	19.4	40.8	22.8	NA
2,4–D runoff events[b] > 70 μg L^{-1}	0.0	1.6	0.0	NA
2,4–D runoff events > 3.5 μg L^{-1}	0.4	10.0	0.6	NA
2,4–D percolate events per year (> 10 ng L^{-1})	56.8	77.8	60.7	NA
2,4–D percolate events > 70 μg L^{-1}	0.0	0.0	0.0	NA
2,4–D percolate events > 3.5 μg L^{-1}	4.1	0.0	4.0	NA
Columbia, MO				
2,4–D runoff events per year (> 10 ng L^{-1})	45.5	49.6	50.5	NA
2,4–D runoff events > 70 μg L^{-1}	0.0	2.6	0.0	NA
2,4–D runoff events > 3.5 μg L^{-1}	0.7	13.4	0.9	NA
2,4–D percolate events per year (> 10 ng L^{-1})	0.0	0.0	0.0	NA
2,4–D percolate events > 70 μg L^{-1}	0.0	0.0	0.0	NA
2,4–D percolate events > 3.5 μg L^{-1}	0.0	0.0	0.0	NA
Minneapolis–St. Paul, MN				
2,4–D runoff events per year (> 10 ng L^{-1})	10.7	31.6	10.6	NA
2,4–D runoff events > 70 μg L^{-1}	0.0	1.2	0.0	NA
2,4–D runoff events > 3.5 μg L^{-1}	0.1	7.1	0.1	NA
2,4–D percolate events per year (> 10 ng L^{-1})	14.0	69.0	16.7	NA
2,4–D percolate events > 70 μg L^{-1}	0.0	0.0	0.0	NA
2,4–D percolate events > 3.5 μg L^{-1}	4.0	0.4	4.3	NA

[a] 10 ng L^{-1} (ppt) represents detectable limit
[b] 70 μg L^{-1} (ppb) MCL standard for 2,4–D (Kamrin, 1997)

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