Review

# A Review of Turfgrass Fertilizer Management Practices: Implications for Urban Water Quality

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Summary. **Urban watersheds include extensive turfgrass plantings that are associated with anthropocentric attitudes toward landscapes. Native and construction-disturbed urban soils often cannot supply adequate amounts of nitrogen (N) and phosphorus (P) for the growth and beauty of landscape plants. Hence, fertilization of landscape plants is practiced. Mismanaged fertilization and irrigation practices represent a potential source of nutrients that may contribute to water quality impairment. This review focuses on turfgrass fertilization practices and their impacts on urban water quality. Research results show that fertilization during active growth periods enhances turfgrass nutrient uptake efficiencies. The major concern regarding the fertilization of turfgrass and landscape plants in urban watersheds, therefore, is selecting the proper combination of fertilizer rate, timing, and placement that maximizes nutrient utilization efficiency and reduces the risk for nutrient loss to water bodies. Encouraging individuals to adopt best management practices (BMPs) is a priority for watershed managers. Research has found that educational programs are an important part of changing fertilization habits and that education needs to be thorough and comprehensive, which is beyond the scope of many seminars and fact sheets currently in use.**

Turfgrass dominated landscapes are prominent features of urban watersheds. Milesi et al. (2005) estimated that turfgrass covers 1.9% of the total U.S. surface area, which is similar to previous areal estimates of 10 to 16 million hectares (Robbins and Birkenholtz, 2003). Robbins and Sharp (2003) discussed several factors that have contributed to the expansion of U.S. turfgrass coverage, including the association of turfgrass aesthetics and function with family, community, and environmental values. Landscape plants such as ornamental species are also associated with increasingly urbanized environments (Amador et al., 2007; Hipp et al., 1993; Shober et al., 2010).

Plant nutrients, such as N and P, are required for the growth and beauty of landscape plants. Numerous nutrient sources already present in urban watersheds can satisfy these needs and these sources, excluding fertilizer, have been discussed in detail (R.O. Carey, G.J. Hochmuth, C.J. Martinez, T.H. Boyer, V.D. Nair, M.D. Dukes, G.S. Toor, A.L. Shober, J.L. Cisar, L.E. Trenholm, and J.B. Sartain, unpublished data). Most native and construction-disturbed urban soils cannot supply adequate amounts of nutrients for normal growth of landscape plants, so fertilizers are often used. To critically analyze plant fertilizer needs relative to potential water quality threats, an understanding of nutrient budgets, especially in relation to fertilizer inputs and losses in urban landscapes, is needed. Fertilization mismanagement of urban vegetation represents a potential source of nutrients that may contribute to water quality impairment. Many states, such as Florida, are seeking to reduce the potential for fertilizer losses through BMPs or state and local regulations.

Water bodies impaired by high nutrient concentrations require water quality management plans outlined by the Total Maximum Daily Load Program [Florida Department of Environmental Protection (FDEP), 2009; U.S. Environmental Protection Agency (USEPA), 2010]. Under the federal Clean Water Act, the FDEP establishes surface water quality standards for the state (FDEP, 2012). For example, the nitrate–nitrogen ($NO_3$–N) standard for Florida springs is 0.35 ppm. These standards become important benchmarks against which to measure nutrient losses from land-based sources, including fertilizers.

Laws and regulations guide urban landscape management practices, but there are no federal laws in the United States specifically targeting urban fertilizer use. However, several existing laws are indirectly applicable

**Units**

| To convert U.S. to SI, multiply by | U.S. unit | SI unit | To convert SI to U.S., multiply by |
|---|---|---|---|
| 0.4047 | acre(s) | ha | 2.4711 |
| 0.3048 | ft | m | 3.2808 |
| 2.54 | inch(es) | cm | 0.3937 |
| 1.1209 | lb/acre | $kg \cdot ha^{-1}$ | 0.8922 |
| 0.001 | ppm | $g \cdot kg^{-1}$ | 1000 |
| 1 | ppm | $mg \cdot kg^{-1}$ | 1 |
| 1 | ppm | $mg \cdot L^{-1}$ | 1 |

to fertilizers, such as the Resource Conservation and Recovery Act, which pertains to recycled wastes used as fertilizers (USEPA, 1999). State agricultural departments are primarily responsible for developing and implementing fertilizer regulations. The Safe Fertilizer Act (1998) in the state of Washington was the first statewide legislation regulating fertilizer contaminants (USEPA, 1999) and other states have implemented similar fertilizer use policies targeting nutrients. In 2005, Minnesota became the first state to restrict fertilizer use on turfgrass to reduce P runoff (Minnesota Department of Agriculture, 2007; Rosen and Horgan, 2005). Soil test surveys revealed that the vast majority (70% to 80%) of lawns in the Twin Cities Metropolitan Area had high soil P levels (Bray P1 test >25 $mg \cdot kg^{-1}$) that did not require additional P inputs to maintain healthy turfgrass (Rosen and Horgan, 2005). Legislative restrictions included using fertilizers without P in Minnesota, although exceptions allowed lawn P fertilization during the first year of establishment. These restrictions have led to a significant increase in the availability, at the retail level, of fertilizers without P (Rosen and Horgan, 2005). The Florida Department of Agriculture and Consumer Services also published the Urban Turf Fertilizer Rule [5E-1.003(2) Florida Administrative Code] in 2007 to establish standards for N and P content in fertilizers (State of Florida, 2007), but did not remove P from landscape fertilizers.

Local governments have also implemented fertilizer regulations in an attempt to improve water quality. For example, in Florida, county and municipal fertilizer ordinances have included rules addressing application timing [e.g., N blackouts during the wet (summer) season], nutrient composition (e.g., percentage of slow- vs. quick-release N sources), restricted areas (e.g., buffer zones surrounding water bodies), and impervious surfaces (e.g., intentional or unintentional fertilizer application) (Hartman et al., 2008; Hochmuth et al., 2009; Tampa Bay Estuary Program, 2008). In Michigan, Lehman et al. (2009) reported a decrease in total P concentrations in the Huron River watershed after the city of Ann Arbor enacted an ordinance limiting P application to lawns. However, the ordinance was only one component of a multifaceted approach to improve water quality in the region. The effectiveness of the ordinance could not be isolated, for example, from educational program effectiveness.

Although fertilizers have been targeted in watershed management programs, the relationship between fertilizer use and water quality impairment in urban watersheds is complicated by multiple factors such as fertilizer inputs, fertilizer nutrient management and cycling, and nutrient losses. For example, additional research is still needed to quantify direct water quality benefits of P fertilizer restrictions in Minnesota (Minnesota Department of Agriculture, 2007). The objectives of this review were to 1) summarize the major fertilizer sources and specific fertilizer management practices that can affect N and P cycling and exports from turfgrass and vegetated landscapes in urban watersheds, 2) discuss water quality impacts associated with nutrient runoff and leaching, and 3) identify critical socioeconomic factors associated with fertilizer management in urbanized areas. Finally, certain research gaps are identified where further information is needed on the overall contribution of fertilizers to nutrient exports from urban watersheds.

## Fertilizer application rates to turfgrass

Normal plant growth and reproduction are impaired when soils lack sufficient quantities of essential elements. Turfgrass N applications occur more frequently and in larger quantities than any other fertilizer-supplied nutrient because plants require more N and it is typically the most yield-limiting nutrient. Phosphorus, another essential plant macronutrient, is required for energy reactions and is also commonly applied as fertilizer. Both agricultural and urban areas use fertilizers, but compared with the extensive research documenting nutrient losses from agricultural soils (Allen et al., 2006; Sims et al., 1998), relatively few studies have analyzed losses from urban landscapes (Bierman et al., 2010; Easton and Petrovic, 2004; Erickson et al., 2005; Soldat and Petrovic, 2008).

Cultural management practices influence turfgrass growth, quality, and nutrient exports (Beard and Green, 1994; Bell and Moss, 2008; Linde et al., 1995). Recycling grass clippings, for example, can improve turfgrass nutrient sequestration in the landscape (Hull and Liu, 2005; Qian et al., 2003; Starr and DeRoo, 1981) because clippings typically represent the largest N sink in established turfgrass, storing 25% to 60% of applied N (Petrovic and Easton, 2005). Kopp and Guillard (2002) showed that returning clippings without adjusting fertilization rates, increased dry matter yields for a turfgrass mixture containing kentucky bluegrass [*Poa pratensis* (35%)], creeping red fescue [*Festuca rubra* (35%)], 'Cutter' perennial ryegrass [*Lolium perenne* (15%)], and 'Express' perennial ryegrass (15%). If clippings are returned, a reduction in N fertilization rates (50% to 75%) may not adversely impact turfgrass quality (Heckman et al., 2000; Kopp and Guillard, 2002). Clippings management (removing or returning) affects N dynamics in turfgrass systems, but this practice does not significantly affect P transport from the landscape (Bierman et al., 2010; Kussow, 2008). In addition to factors that increase the potential for runoff (e.g., slope, precipitation rate, etc.) and sediment movement (Linde et al., 1995; Steinke et al., 2007), fertilizer sources and application rates also influenced P losses (Bierman et al., 2010; Easton and Petrovic, 2004).

Effects of fertilizer sources and rates on turfgrass growth have been studied for many years in the United States. In general, terminology is consistent among scientists, but several terms are clarified for use in this

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article. Turfgrass quality is a term that relates to any or all of the following characteristics: greenness in terms of hue, turfgrass density in terms of soil surface coverage or number of tillers per unit area of land, amount of observed brown or yellow leaf discoloration, disease damage, and general growth rate (Krans and Morris, 2007; Morris and Shearman, 2012). Most studies (unless specifically defined) relate turfgrass "quality" to greenness on a visual scale of 1 (yellow or brown) to 9 (dark green). Controlled-release fertilizer (CRF) and slow-release fertilizer (SRF) often have been used interchangeably, unless specifically defined by the author.

Turfgrass research has been conducted with several forms of fertilizers, especially for N. These forms include soluble fertilizers and CRFs. Soluble forms include ammonium nitrate, urea, ammonium sulfate, and potassium nitrate, among others. Over the years, many types of CRFs have been evaluated for effects on turfgrass growth and quality and effects on nutrient losses to the environment (Easton and Petrovic, 2004; Skogley and King, 1968; Spangenberg et al., 1986). These include slowly degraded formulations such as isobutylidine diurea (IBDU) and urea-formaldehyde. In addition, CRFs have been formulated as a soluble N source coated with materials to slow the solubility and release of nutrients. These formulations include sulfur-coated urea (SCU) and polymer-coated urea. Obreza and Sartain (2010) reviewed the major sources of fertilizers used to enhance N use efficiency in horticultural crops, including turfgrass. Some examples of recent research conducted with various nutrient forms are summarized below.

Sartain (1981) showed that water soluble ammonium sulfate produced a quicker maximum turfgrass color response in 'Tifway' hybrid bermudagrass (*Cynodon dactylon* × *C. transvaalensis*) and 'Derby' ryegrass (seeded over 'Tifway' hybrid bermudagrass) than slow-release sources (e.g., IBDU). Carrow (1997) investigated the effects of different N sources on 'Tifway' hybrid bermudagrass and concluded that long-term (61–95 d) turfgrass response (i.e., turfgrass quality, turfgrass shoot growth rate, etc.) was dependent on sulfur, polymer content, or both in coated fertilizers. The thickness and nature of polymer-coatings affect N-release properties (Carrow, 1997). Slow-release N sources may reduce clipping yields compared with quick-release fertilizers (Heckman et al., 2000), but Cisar et al. (2001) found no differences in turfgrass quality and clipping yields when equal amounts of N were supplied to 'Tifgreen' hybrid bermudagrass by CRFs and ammonium sulfate. After investigating 'Floratam' st. augustinegrass (*Stenotaphrum secundatum*) and hybrid bermudagrass response to organic and inorganic fertilizers, Trenholm and Unruh (2005) suggested that N application rates were more important than N sources for optimizing turfgrass quality.

Nutrient uptake efficiencies, however, vary for different turfgrass species, cultivars, and growing conditions such as soil types (Table 1) (Bowman et al., 2002; Pare et al., 2006; Trenholm et al., 1998). Nitrogen fertilizer application rates for cool-season turfgrasses [e.g., kentucky bluegrass, perennial ryegrass, creeping bentgrass (*Agrostis palustris*), etc.] range from 50 to 200 kg·ha$^{-1}$ per year, while rates for warm-season turfgrasses [e.g., st. augustinegrass, zoysiagrass (*Zoysia* sp.), centipedegrass (*Eremochloa ophiuroides*), etc.] are sometimes higher (50 to 300 kg·ha$^{-1}$ per year) (Branham, 2008). Dudeck et al. (1985) vegetatively established nine bermudagrass cultivars and investigated establishment rates using multiple N treatments. Establishment rates varied for different cultivars, and although a monthly application rate of 49 kg·ha$^{-1}$ N resulted in maximum establishment, establishment rates were reduced at 59 kg·ha$^{-1}$ N. Trenholm et al. (1998) evaluated 'FloraDwarf' and 'Tifdwarf' bermudagrass (*C. dactylon*) cultivars during establishment in a glasshouse and found differences in growth and quality because of N rate, cultivar, and photoperiod.

Different fertilizer requirements for turfgrass species and cultivars, coupled with variable turfgrass response to soluble- and slow-release sources, can lead to opportunities for inappropriate fertilizer management practices. Several studies have suggested that soluble fertilizer sources are associated with increased nutrient losses, but this is dependent on application rates, irrigation, climate, and particular turfgrass characteristics (Barton and Colmer,

**Table 1. Nitrogen (N) recovered and leached from turfgrass systems receiving variable fertilizer application rates.**

| Study | Species | Application rate (kg·ha$^{-1}$ N)[z] | Samples after treatment (no.) | Avg N recovered (kg·ha$^{-1}$)[y] | Avg N in leachate (kg·ha$^{-1}$) |
|---|---|---|---|---|---|
| Frank (2008) | Kentucky bluegrass | 24.5 | 7 (15–637 d) | 19.11 | 0.15 |
| | | 49 | | 36.26 | 2.65 |
| Miltner et al. (1996) | Kentucky bluegrass | 39.2 | 7 (18–748 d) | 30.99 (Spring) | 0.002 |
| | | | | 34.87 (Fall) | 0.011 |
| Bowman et al. (2002) | Centipedegrass | 50 | 1 (≈30 d) | 35.5 | 5.35 |
| | St. augustinegrass | 50 | | 37 | 0.5 |
| | 'Meyer' zoysiagrass | 50 | | 31.5 | 6.45 |
| | 'Emerald' zoysiagrass | 50 | | 38.5 | 3.4 |
| | 'Tifway' bermudagrass | 50 | | 42 | 1.3 |
| | Common bermudagrass | 50 | | 35 | 4.6 |
| Starr and DeRoo (1981) | Kentucky bluegrass/red fescue | 180 | 1 (≈188 d) | 116 (no clippings) | trace |
| | | | | 137 (clippings) | trace |
| Engelsjord et al. (2004) | Kentucky bluegrass | 48.8 | 4 (2–365 d) | 39.79 | — |
| | Perennial ryegrass | 48.8 | | 36.29 | — |

[z]1 kg·ha$^{-1}$ = 0.8922 lb/acre.
[y]N recovered includes plant tissues and soil except Bowman et al. (2002), which only includes plant tissues.

2006; Easton and Petrovic, 2004). Mixtures of soluble and CRF materials may be a good approach where nutrient loss potential is high. Returning clippings to turfgrass landscapes further reduces N fertilizer requirements, but nutrient uptake efficiencies, recycling rates, or both among turfgrass species and cultivars (within a range of environmental conditions) require further assessment. Most state land-grant universities provide fertilizer management recommendations for turfgrass. It is beyond the scope of this article to summarize these recommendations for all states. An example of recommendations for warm-season turfgrass can be found for Florida (Sartain, 2007) and a recent example for cool-season turfgrass is provided for Wisconsin (Kussow et al., 2011).

## Turfgrass nutrient uptake in response to fertilizer timing and placement

Increasing turfgrass nutrient use efficiency depends on optimal timing and placement of fertilizer so that plant utilization is maximized and losses to the environment are minimized. Turfgrass nutrient uptake rates are higher during periods of active root development (Mangiafico and Guillard, 2006; USEPA, 2005; Wherley et al., 2009). Educating homeowners about geological and environmental factors specific to different turfgrass types, such as cool-season vs. warm-season grasses, may help control nutrient leaching and runoff (Varlamoff et al., 2001). Accounting for differences in the rate of nutrient uptake between turfgrass and other landscape vegetation is another concern. Fertilizer application techniques must match the type of vegetation to minimize nutrient losses because N uptake rates for turfgrass are typically greater than for ornamental species (Cisar et al., 2004; Erickson et al., 2001). Therefore, fertilizer application rates for turfgrass can be inappropriate for other landscape plants (Broschat et al., 2008).

**Fertilizer timing.** Fertilization during the turfgrass establishment period can lead to potential nutrient mismanagement. Erickson et al. (2010) compared different fertilizer schedules on muck (organic soil) and sand-based sod and found that delayed fertilization (at least 30 d after installation) significantly reduced both $NO_3$–N and orthophosphate–phosphorus ($PO_4$–P) in leachate, with muck sod retaining more nutrients during establishment. Calcium and magnesium ratios in manure and manure-impacted soils can also potentially improve P retention (Nair et al., 2003). Growing sod on manure amended soils increased P retention, eliminated the need for additional P applications, accelerated turfgrass establishment, and reduced dissolved P in runoff (Vietor et al., 2004).

Spring and summer fertilizer applications are most effective for established warm-season grass performance, but nutrient uptake rates peak during spring and fall for cool-season grasses (USEPA, 2005). Fertilization dates consequently influence nutrient leaching and runoff potential. For example, inappropriately timed N applications to both warm- and cool-season grasses in North Carolina reduced the uptake of applied fertilizers (Osmond and Hardy, 2004). Mangiafico and Guillard (2006) investigated fertilizer-timing effects on $NO_3$–N leaching and turfgrass quality for a Connecticut lawn comprised of kentucky bluegrass (90%) and creeping red fescue (10%). The authors applied soluble N sources [ammonium–N ($NH_4$–N) and urea] on different dates during the fall (September to December). Fall fertilization improved turf quality, but late fall applications (after September) did not affect color, shoot density, or root mass. Percolate water also contained greater $NO_3$–N concentrations and mass losses were greater following late fall applications (Mangiafico and Guillard, 2006). Other studies comparing fall fertilization effects have reported similar results for late-season applications (Grossi et al., 2005; Guillard and Kopp, 2004; Petrovic, 1990). Wehner and Haley (1993) evaluated fertilizer timing effects on kentucky bluegrass using three different sources [urea, SCU, and activated sewage sludge] and found turfgrass color ratings and clipping yields were higher for urea applications after October. The turfgrass was grown in a sandy loam soil that could potentially limit nutrient losses, but the authors recommended the use of CRFs (SCU or sludge) or split urea applications (late fall/early spring) to reduce N leaching.

Fertilization during periods of reduced turfgrass growth can increase nutrient losses. Bierman et al. (2010) noted that in temperate climates, adding P to soils during fall leads to an increase in the P runoff potential because of possible frozen-soil conditions. The authors compared kentucky bluegrass plots in Minnesota under frozen and nonfrozen conditions for 3 years and runoff from frozen soils accounted for 80% of runoff P. Turfgrass tissues can be the dominant source of P in runoff (Kussow, 2008) as the amount of soluble P leached from fresh vegetation can be much lower than losses from frozen or dried tissues (Kussow, 2004; Steinke et al., 2007). In addition to potential nutrient runoff or leaching, fertilization during periods of turfgrass dormancy may increase the prevalence of diseases, thatch problems, and weed pressure. Busey (2003) reviewed several studies investigating N fertilizer timing and turfgrass competitiveness against weeds. For example, Dunn et al. (1993) reported an increase in winter weeds associated with late-season fall fertilization of 'Meyer' zoysiagrass (*Zoysia japonica*). Turfgrass species with slow growth rates, such as zoysiagrass, are especially susceptible to increased weed encroachment because of fertilization (Busey, 2003).

The length of time between fertilizer applications and irrigation or rainfall also determines the extent of nutrient losses from turfgrass systems (Bell and Moss, 2008; Kenna, 2008; Petrovic and Easton, 2005). In a lysimeter study, creeping bentgrass irrigated 5 d after N application had reduced leaching losses up to 90%, compared with irrigation 1 d after fertilization (Bowman et al., 1998). Using simulated rainfall events and multiple P rates (0, 5, and 11 $kg\cdot ha^{-1}$), Shuman (2002) reported greater dissolved P losses in runoff from bermudagrass at 4 h after fertilizer applications (0.5–7.5 $mg\cdot L^{-1}$) than at 24 h after fertilization (0.5–1.75 $mg\cdot L^{-1}$). Light irrigation or "watering-in" of fertilizers decreases runoff by enhancing P dissolution and movement into soils (Shuman, 2004).

**Fertilizer placement.** Studies have suggested that broadcast fertilization of turfgrass may limit the development of deeper root networks (Bowman et al., 1989; Murphy and Zaurov, 1994). In a greenhouse study, Murphy and Zaurov (1994) compared

'Gettysburg' perennial ryegrass response to fertilizer applications at multiple soil depths (0, 5, 10, and 15 cm) and concluded that fertilizer placement can affect turfgrass growth and water use efficiency. Subsurface fertilizer applications in two soils (sandy clay loam and sandy-peat mixture) increased shoot and root development compared with surface fertilization. Water use efficiency for perennial ryegrass subjected to subsurface fertilization in both the sandy clay loam (11% increase at 10 and 15 cm depths) and sandy-peat mixture (21% increase at 5 cm) was greater than with the surface fertilization treatment (Murphy and Zaurov, 1994). Incorporation of fertilizers into soil by methods such as core cultivation and topdressing alters turfgrass response to applied nutrients (Waddington and Duich, 1976), especially under conditions where the downward movement of nutrients is restricted (e.g., compacted soils) (Agnew and Christians, 1993). Johnson et al. (2006) topdressed established kentucky bluegrass using composted manure to evaluate the effects on nutrient losses. During the 2-year study, $NO_3$–N and P concentrations in runoff and leachate did not increase with manure use; however, $NH_4$-N concentrations in runoff increased.

Totten et al. (2008) investigated fertilizer effects on creeping bentgrass and found greater N uptake efficiency for 100% liquid fertilizers (through foliar absorption) compared with 100% granular fertilizers. Gross et al. (1990) compared nutrient runoff and soil percolate losses from tall fescue (*Festuca arundinacea*) and kentucky bluegrass fertilized with liquid and granular urea fertilizers (Table 2). Although total N runoff losses from plots receiving fertilizers were significantly higher than an unfertilized control plot, nutrient concentrations were low with all treatments. There were no differences in runoff losses between turfgrass fertilized with liquid and granular fertilizers. However, at depths greater than 0.9 m, soil $NO_3$–N concentrations with the granular treatment exceeded the liquid treatment. The fertilizers were applied in split applications throughout the year during active plant growth periods, and soil percolate was collected monthly from lysimeters. Gross et al. (1990) suggested that higher $NH_3$ volatilization following the liquid N treatment contributed to lower soil $NO_3$–N concentrations. Turfgrass species and cultivars

**Table 2. Comparison of nutrient exports ($kg\cdot ha^{-1}$ of total watershed area) from turfgrass, agricultural, forest, and other urban land uses.**

| Study | Land use | Area (ha)[z] | Duration | Nutrient exports ($kg\cdot ha^{-1}$ per yr)[y] | | | | |
|---|---|---|---|---|---|---|---|---|
| | | | | $NO_3$–N | $NH_4$–N | TN | DRP | TP |
| King et al. (2008) | Golf course: Creeping bentgrass/ annual and kentucky bluegrass | 21.8 | 2 years | 0.59 | 0.11 | 2.79 | 0.14 | 0.27 |
| Schwartz and Shuman (2005) | 'Tifway' bermudagrass | 0.003 | 4 years | 3.05 | — | — | — | — |
| King et al. (2001) | Golf course: (Bermudagrass/perennial ryegrass): storm events | 29 | 13 mo. | 2.10 | — | — | 0.33 | — |
| | Golf course: baseflow | | | 4.30 | — | — | 0.05 | — |
| Coulter et al. (2004) | Agricultural (95% agriculture; 5% urban) | 327 | 1 year | 20.40 | 0.34 | — | 0.28 | 1.13 |
| | Mixed (43% agriculture; 57% urban) | 506 | | 10.80 | 0.95 | — | 0.12 | 1.14 |
| | Urban (1% agriculture; 99% urban) | 226 | | 5.97 | 0.52 | — | 0.07 | 0.66 |
| Kaushal et al. (2008) | Forest | 41 | 5 years | 0.11 | — | 0.84 | — | — |
| | Forest/suburb | 381 | | 5.62 | — | 5.66 | — | — |
| | Agricultural | 8 | | 20.58 | — | 22.88 | — | — |
| Kaushal et al. (2008) | Suburb | 1065 | | 8.54 | — | 9.68 | — | — |
| | Urban | 1429 | | 4.78 | — | 8.02 | — | — |
| Steinke et al. (2007) | Prairie (frozen soil) | 0.002 | 2 years | — | — | — | 1.30 | 1.92 |
| | Prairie (nonfrozen soil) | | | — | — | — | 0.02 | 0.04 |
| | Kentucky bluegrass blend (frozen soil) | | | — | — | — | 1.36 | 2.11 |
| | Kentucky bluegrass blend (nonfrozen soil) | | | — | — | — | 0.01 | 0.01 |
| Gross et al. (1990) | Tall fescue/kentucky bluegrass (liquid fertilizer) | 0.001 | 2 years | 0.05 | 0.07 | 0.17 | — | — |
| | Tall fescue/kentucky bluegrass (granular fertilizer) | | | 0.06 | 0.11 | 0.18 | — | — |
| Line et al. (2002) | Residential (lawns/impervious surfaces) | 2.5 | 1–3 years | 3.20 | 2.40 | 23.90 | — | 2.30 |
| | Golf course | 1.75 | | 4.80 | 3.00 | 31.20 | — | 5.30 |
| | Pasture | 6.23 | | 1.20 | 0.40 | 6.70 | — | 4.30 |
| | Construction-I (clearing/grading) | 4.05 | | 1.40 | 0.60 | 8.30 | — | 3.00 |
| | Construction-II (roads/storm drains/ building phase) | 4.05 | | 7.30 | 4.10 | 36.30 | — | 1.30 |
| | Wooded | 3.32 | | 3.60 | 0.30 | 11.40 | — | 1.00 |

[z] 1 ha = 2.4711 acres.
[y] Nitrate–nitrogen ($NO_3$–N), ammonium–nitrogen ($NH_4$–N), total nitrogen (TN), dissolved reactive phosphorus (DRP), total phosphorus (TP); $1\ kg\cdot ha^{-1} = 0.8922$ lb/acre.

also respond differently to liquid and granular fertilizers. For example, Steinke and Stier (2003) compared turf quality (color and density) for creeping bentgrass, kentucky bluegrass, and supina bluegrass (*Poa supina*) using different fertilizers on shaded golf course tees. Turf quality for creeping bentgrass improved with application of liquid urea, kentucky bluegrass had a better response with granular urea, and supina bluegrass response to both fertilizers was seasonally dependent (spring: granular; summer/fall: liquid).

Turfgrass assimilates nutrients faster than landscape plants (Cisar et al., 2004; Erickson et al., 2001), but uptake efficiencies are dependent upon fertilizer placement and seasonal absorption differences between species. To evaluate these differences further, long-term datasets on the performance of cool and warm-season grasses in geographically diverse watersheds are needed (King and Balogh, 2008).

## Nutrient leaching and runoff from turfgrass and landscape plants

**Turfgrass characteristics, fertilizer sources, and nutrient losses.** Species-specific characteristics of turfgrass, and the types of fertilizer applied, influence nutrient losses from turfgrass landscapes. Bowman et al. (2002) compared six warm-season turfgrass species and identified species selection as an important factor contributing to potential $NO_3$–N leaching. Lower root length density contributed to greater $NO_3$–N losses from 'Meyer' zoysiagrass (55 $kg \cdot ha^{-1}$ per year) compared with st. augustinegrass (3 $kg \cdot ha^{-1}$ per year). Pare et al. (2006) compared N uptake for 11 annual bluegrass (*Poa annua* var. *reptans*) ecotypes and three bentgrass species (*Agrostis canina*, *Agrostis castellana*, and *Agrostis stolonifera*). Leaching losses were greater for annual bluegrasses (21% to 78% of applied N) compared with the bentgrass species (6% to 11%). However, root development and $NO_3$–N leaching varied within bluegrass ecotypes and among different bentgrass species.

Easton and Petrovic (2004) investigated N and P losses in leachate and runoff by applying synthetic organic fertilizers (readily available urea and SCU) and natural organic amendments (compost and biosolids) to kentucky bluegrass and perennial ryegrass at two different N rates, 50 and 100 $kg \cdot ha^{-1}$ (200 $kg \cdot ha^{-1}$ per year). The N:P ratio of the nutrient sources determined the amount of applied P because turfgrass is generally fertilized to satisfy N requirements. Increased application rates for the synthetic organic fertilizers and natural organic amendments resulted in greater N and P losses from turfgrass, respectively (Easton and Petrovic, 2004).

Nitrogen leaching can be minimized by applying water soluble fertilizers frequently at relatively low rates and CRFs less frequently at relatively higher rates (Barton and Colmer, 2006; Engelsjord and Singh, 1997; Snyder et al., 1984). On golf greens in Georgia, water soluble N (potassium nitrate, urea, and ammonium phosphate) resulted in significantly greater $NO_3$–N concentrations in leachate than CRFs (poly- and sulfur-coated N) at application rates ranging from 0 to 49 $kg \cdot ha^{-1}$ N (Shuman, 2001). Sartain (1992) compared leaching losses from N sources in Florida and reported that noncoated ammonium sulfate leached 50% more N than coated ammonium sulfate. In another study conducted over a 112-d period in sand-filled lysimeters without vegetation, the percentage of applied N lost through leaching increased relative to the solubility of fertilizer sources: IBDU (17%), Nitroform [28% (Agrium, Calgary, AB, Canada)], SCU (50%), Nutralene [58% (Agrium)], coated ammonium sulfate (62%), and ammonium sulfate (80%) (Sartain, 1995, 1996).

Fertilizer formulations often include both slow- and quick-release N sources to enhance turfgrass growth. Guillard and Kopp (2004) measured N leaching losses from a New England study site containing kentucky bluegrass, perennial ryegrass, and creeping red fescue that were fertilized with multiple N sources. Leaching losses, reported as a percentage of applied N, for ammonium nitrate [$NH_4NO_3$ (16.8%)], polymer-coated SCU (1.7%), and an organic fertilizer (0.6%) led the authors to suggest that fertilizers used in New England should contain greater percentages of slow-release N sources to minimize $NO_3$–N losses. Frank et al. (2006) also indicated that applying soluble fertilizer sources on mature kentucky bluegrass at 49 $kg \cdot ha^{-1}$ N would enhance leaching losses. In contrast, Cisar et al. (2004) reported that average $NO_3$–N concentrations in leachate were less than 0.32 $mg \cdot L^{-1}$ when fertilizers containing predominantly soluble sources (only 37.5% SCU) were applied to st. augustinegrass at a rate of 49 $kg \cdot ha^{-1}$ N. St. augustinegrass is the most common residential turf species in Florida. The above $NO_3$–N concentration in the leachate is less than the nutrient criterion of 0.35 $mg \cdot L^{-1}$ for $NO_3$–N mentioned earlier in this article. In fact, inorganic N concentrations in leachate from well-managed st. augustinegrass sod can be lower than typical concentrations found in southern Florida precipitation (Erickson et al., 2008).

**Nutrient losses from landscape vegetation compared with other urban land uses.** Appropriate fertilization and irrigation management practices for both turfgrass and landscape plants reduce the potential for nutrient runoff and leaching compared with other land uses (Table 2) (Augustin and Snyder, 1984; Easton and Petrovic, 2004; Exner et al., 1991). Gold et al. (1990) analyzed $NO_3$–N in leachate from multiple land uses in southern New England; both fertilized cornfields (66 $kg \cdot ha^{-1}$ per year) and septic systems (48 $kg \cdot ha^{-1}$ per year) had greater leachate $NO_3$–N loads than fertilized (6 $kg \cdot ha^{-1}$ per year) and unfertilized lawns (1.4 $kg \cdot ha^{-1}$ per year). Groffman et al. (2009) also reported greater average soil $NO_3$–N content for row crop agriculture (8.7 $mg \cdot kg^{-1}$) compared with turfgrass (1.2 $mg \cdot kg^{-1}$) in Maryland. Davis and Lydy (2002) analyzed water quality in ponds receiving runoff from a golf course in Kansas before and after implementing nonstructural and structural BMPs (i.e., reducing fertilization rates, using slow-release N fertilizers, and establishing buffer zones). Initial nutrient concentrations (1.4 $mg \cdot L^{-1}$ $NO_3$–N, 1.1 $mg \cdot L^{-1}$ total P) declined significantly (0.2 $mg \cdot L^{-1}$ $NO_3$–N, 0.5 $mg \cdot L^{-1}$ total P) in subsequent years after implementation of recommended practices. Management practices using turfgrass buffers can also reduce nutrient concentrations in runoff. This is dependent on factors such buffer length, mowing height, antecedent soil moisture, and frozen or nonfrozen soil (Cole et al., 1997; Moss et al., 2006; Steinke et al., 2007).

Once turfgrass and other landscape plants are established, the likelihood of water contamination from nutrient runoff is reduced (Bowman et al., 2002; Shober et al., 2010; Soldat and Petrovic, 2008). Less than 5% of applied N is typically leached from established turfgrass systems that receive moderate (200–300 kg·ha$^{-1}$ per year) amounts of fertilizer (Barton and Colmer, 2006), suggesting that properly fertilized and irrigated turfgrass has minimal water quality impacts (Cisar et al., 2004). Easton and Petrovic (2004) measured greater nutrient runoff and leachate concentrations from unfertilized plots compared with established kentucky bluegrass and perennial ryegrass receiving fertilizer treatments. The characteristic dense groundcover of established turfgrass contributes to reduced sediment and P losses in runoff (Linde et al., 1995; Linde and Watschke, 1997). Easton and Petrovic (2008) measured runoff losses from different types of land uses in urban areas and on various soils ranging from low to high P storage. Particulate P losses were consistently greater from wooded and barren areas compared with fertilized lawns. Dissolved P losses in runoff from fertilized lawns on shallow soils were generally higher than from other land uses, but P losses from fertilized lawns declined on soils with greater P storage (Easton and Petrovic, 2008). Even when runoff P concentrations from turfgrass were higher compared with other unmanaged landscapes, the reduction in total runoff volume associated with established turfgrass (Beard and Green, 1994) resulted in lower overall P loads (Kussow, 2004; Steinke et al., 2007).

Nutrient losses from ornamental species can be lower than turfgrass after establishment because these landscape plants typically require less fertilizer (Erickson et al., 2005; Erickson et al., 2008). Native plant species have been recommended as alternative landscape options to turfgrass (Hipp et al., 1993), but few studies have investigated runoff and leaching losses from ornamental plants (Shober et al., 2010). Studies describing irrigation requirements for established ornamental plants are also limited (Henson et al., 2006; Shober et al., 2010). The thousands of ornamental species (compared with the number of major turfgrass species, <20) used in urban landscapes contribute to the relative lack of information on ornamentals (Devitt and Morris, 2008). Erickson et al. (2001) demonstrated the importance of including these plants in watershed nutrient analyses when they reported higher $NO_3$–N leaching losses from a mixed ornamental species landscape (48.3 kg·ha$^{-1}$ N) than st. augustinegrass (4.1 kg·ha$^{-1}$ N) during the initial establishment period (12 months). The mixed-species landscape needed 2 years of establishment before $NO_3$–N leaching levels were comparable to losses from st. augustinegrass (Erickson et al., 2008). In another study, data collected over 45 months revealed that cumulative mean P losses from st. augustinegrass (22.9 kg·ha$^{-1}$) were significantly lower than from a mixed-species landscape (37.8 kg·ha$^{-1}$) during establishment on sandy soils (Erickson et al., 2005).

**Soil characteristics and nutrient losses from turfgrass.** Local soil characteristics influence nutrient losses from turfgrass landscapes. Soil texture, for example, affects N leaching rates because it influences infiltration, denitrification, and $NH_4$–N retention (Petrovic, 1990). Coarse-textured soils are especially vulnerable to N losses. Soil amendments that increase soil water-holding and cation exchange capacities (CEC) help limit $NO_3$–N and $NH_4$–N leaching (Barton and Colmer, 2006). For example, adding peat to sandy soils, which typically have high hydraulic conductivities and low CECs, has been shown to reduce N leaching from kentucky bluegrass (Barton and Colmer, 2006; Engelsjord and Singh, 1997). Brauen and Stahnke (1995) conducted a 3-year study to investigate $NO_3$–N concentrations in leachate collected from golf putting greens mixed with peat and sand compared with a pure sand rooting medium. Using three different N rates (195, 391, and 586 kg·ha$^{-1}$), average $NO_3$–N concentrations in leachate—reported as a percentage of applied N lost—were lower from the modified rooting medium (0.3%, 0.4%, and 2.3%) compared with sand (2.7%, 3.2%, and 4.2%).

After investigating P requirements for 'Floratam' st. augustinegrass in sandy soils, Liu et al. (2008) found that a minimum monthly application of 1.4 kg·ha$^{-1}$ P was required to achieve quality turfgrass with acceptable growth rates. The authors recommended that P fertilizers should not be applied if sandy soils had Mehlich-1 P concentrations greater than 10 mg·kg$^{-1}$ or if tissue P levels exceeded 1.8 g·kg$^{-1}$ (dry weight). Soil test results do not always have a strong correlation to soluble reactive P in runoff (Kussow, 2008; Soldat and Petrovic, 2008), but applying P on soils with adequate P levels provides no additional benefit to established turfgrass (Bierman et al., 2010).

## Human dimensions of turfgrass fertilizer management practices

Understanding the socioeconomic factors that influence fertilizer management in turfgrass systems is a critical component of the research into management plans to control urban pollution. Robbins et al. (2001) investigated lawn care practices in Columbus, OH, and found that well-educated people with knowledge about the environmental impacts of excessive chemical inputs were more likely to engage in these negative practices than residents with relatively less awareness. High rates of chemical inputs, even among people with knowledge and awareness about environmental issues, reveal the importance of community influences, or social economies, on lawn management practices (Robbins and Sharp, 2003; Robbins et al., 2001). There is a proportional relationship between urban sprawl and intensive lawn management because people are more likely to use lawn care chemicals if their neighbors engage in similar practices (Robbins et al., 2001; Varlamoff et al., 2001).

People who value water quality may consequently engage in directly contradictory behavior. About 90% of central Florida residents in a telephone survey ($n$ = 660) agreed that landscape practices could negatively affect water quality (Souto et al., 2009). In the same study, 95% of residents ($n$ = 733) agreed that individual landscapes were important to the community. Individuals may therefore have to choose between following or opposing community practices that could impair water quality. Residents often conform to community cultivation and maintenance practices for their yards at the expense of resource-efficient strategies that can limit water pollution (Souto et al., 2009). Actual behaviors or habits, therefore, do not consistently reflect attitudes and values.

**Community norms and fertilizer management practices.** Social norms shape personal and collective behavior, thereby functioning as barriers or incentives for adopting environmentally positive or negative practices (Morton and Padgitt, 2005). Individual lawn management practices (e.g., fertilizer application rates) have a strong social component that is dictated by community-oriented values (Robbins and Sharp, 2003; Robbins et al., 2001; Souto et al., 2009). Nielson and Smith (2005) investigated yard care practices in the Tualatin watershed, OR, and found that residents were not concerned about the environmental costs of their individual actions, such as fertilization and irrigation rates. Residents were primarily concerned about how their actions were perceived by their neighbors. Aesthetic yard care values associated with social pressures are consequently anthropocentric and not ecocentric, emphasizing utilitarian aspects of lawns (Nielson and Smith, 2005). If community norms are based on aesthetics and not water conservation or water quality, reinforcing these behaviors will inevitably result in adverse environmental consequences (Miller and Buys, 2008).

Attitudes, values, and norms among residents need to be accounted for when evaluating watershed management strategies (Morton and Padgitt, 2005). Harwell et al. (1999) developed a socioecological conceptual model of environmental change in southern Florida, and along with population and economic considerations, community values were identified as major societal drivers affecting environmental quality. Schueler (2010) suggested that major reductions in runoff and nutrient loads could be achieved by changing attitudes about what constitutes an ideal lawn in the Chesapeake Bay watershed, where turfgrass covers ≈9.4% (1.5 million ha) of the total land area. Educational programs are essential to changing attitudes and behaviors, both individually and collectively. However, influencing behavior can be a complex process because delayed environmental benefits may appear disconnected from everyday concerns and activities of residents (Jacobson, 2009). Delivery methods are consequently a key aspect of educational programs. Israel et al. (1999) demonstrated that intensive educational programs, such as Florida's Master Gardener training, were more effective for getting residents to adopt recommended landscape practices (e.g., appropriate fertilization and irrigation rates) than other programs (e.g., seminars and publications).

Researchers found that 80% of Georgia homeowners reported applying fertilizers to their landscapes (Varlamoff et al., 2001) because these homeowners had a perceived need to have a lawn of comparable quality (greenness) to those of their neighbors. While homeowners made the connection between green turfgrass and fertilization, Varlamoff et al. (2001) found that homeowners were receptive to education about landscape practices that reduced the threat to the environment.

Souto et al. (2009) produced data similar to Georgia finding that 84% of homeowners in Florida applied fertilizer to their lawns either themselves or through commercial applicator. Homeowners were found to lack knowledge about amounts of fertilizer applied by hired applicators or amounts applied by the homeowners themselves.

In recent studies of water quality in Roberts Bay, located in the Sarasota Bay–Peace–Myakka watershed in Sarasota County, FL, water quality improved from 1998 to 2007 and Roberts Bay was removed from the state's impaired water list in 2010 (USEPA, 2012). Water quality improvements were attributed to implementation of BMPs by homeowners, monitoring postconstruction loading from nutrient separating baffle boxes, and implementation of educational components by a broad range of stakeholders and the Florida Friendly Landscaping Program™.

**Socioeconomic factors and fertilizer management practices.** People tend to adopt sustainable practices if personal benefits exceed costs and perceived barriers are removed (Jacobson, 2009). Tracking economic indicators such as household income levels provides context for consumption patterns and resource utilization. Zhou et al. (2009) analyzed household characteristics in Maryland and found that income, median home values, age of homes, percent of owner-occupied homes, and education levels were important predictors of lawn care expenditures and overall lawn greenness. Using tax valuations for homes as surrogates for income levels, Osmond and Hardy (2004) reported discrepancies in fertilizer use among high (≥$175,000), medium ($126,000 to $174,000), and low (≤$125,000) income households in North Carolina. Turfgrass fertilizer application rates for high (132 $kg \cdot ha^{-1}$ N) and medium (148 $kg \cdot ha^{-1}$ N) income households were significantly ($P < 0.05$) greater than low (78 $kg \cdot ha^{-1}$ N) income households (Osmond and Hardy, 2004). The relationship between socioeconomic status and fertilization rates is important because the proportion of U.S. residential lots dedicated to lawns is increasing in high-income areas (Robbins and Birkenholtz, 2003).

Additional variables that have been correlated to income levels and home valuations include the age and ethnicity of residents. About 67% of high-income (>$75,000) Ohio respondents to a lawn care survey ($n$ = 417) used lawn chemicals (including fertilizers), but only 29% of respondents with lower incomes (≤$20,000) engaged in this practice (Robbins et al., 2001). When characterized by age, 55% of older respondents (≥60 years old) used chemicals, compared with only 22% of younger individuals (18–29 years old). Social and economic stratification within residential areas may also contribute to behavioral contrasts related to ethnicity or race. Compared with other neighborhoods in Baltimore, MD, residents in predominantly African-American neighborhoods were less likely to spend money on yard expenditures (Troy et al., 2007). However, the age of homes was a confounding factor for landscape management practices. Fertilizer application rates were higher for newer developments in Baltimore (Law et al., 2004), while African-American neighborhoods predominantly reflected legacy vegetation (i.e., plants and trees requiring less maintenance) (Troy et al., 2007).

The research on socioeconomic factors relating to nutrient management reveals several important factors for adopting BMPs. Primary among these factors is that individuals respond to the community norm for turfgrass maintenance. Fertilizer inputs for one resident tend to be related to the practices used by others in the community. Even if an individual acknowledges their understanding about fertilizer BMPs and negative consequences to the environment, they are still likely to engage in negative management practices if this is the community norm. This behavior can have negative consequences for nutrient losses from urban landscapes when mismanagement of fertilizers is the norm. Research has found that educational programs are an important part of changing community fertilization habits and that education needs to be relatively intensive, not simply seminars and fact sheets.

## Conclusions

Urban vegetation such as turfgrass and ornamental plants receive fertilizer treatments to aid growth and development, thereby improving a site's aesthetic value. Best management fertilizer rates have been researched throughout the United States for cool- and warm-season turfgrass species. Fertilizer requirements for ornamental plants are not as well understood, but these species typically require fewer fertilizer applications than turfgrass after establishment. The major concern regarding the fertilization of turfgrass and landscape plants in urban watersheds therefore, is selecting the proper fertilizer rate that maximizes nutrient utilization efficiency and reduces the risk for nutrient loss to water bodies.

In addition to selecting the correct fertilizer rate, cultural practices (e.g., the types of fertilizer used, fertilizer timing and placement, etc.) can potentially impact water quality. Research shows that soluble fertilizers may enhance potential leaching losses compared with slow-release sources, especially when soluble fertilizers are applied at excessive rates. Combinations of CRFs and soluble fertilizers appear to be a good approach to achieving healthy turfgrass and minimizing the potential for nutrient losses. Research also documents the importance of applying N and P fertilizer when the grass is actively growing, when it has the greatest capacity to use fertilizer. Fertilizer applied outside of the active growth period for either cool- or warm-season turfgrass leads to increased potential for nutrient losses. Many studies demonstrate that properly fertilized and irrigated turfgrass results in significantly reduced leaching and runoff losses. However, little is known about nutrient cycling in the turfgrass system after fertilizer application. Research investigating nutrient uptake efficiencies, recycling rates, or both in different turfgrass systems is needed to optimize fertilizer recommendations for geographically diverse watersheds. In particular, watershed-scale studies evaluating the potential contribution of turfgrass to urban nutrient budgets are required.

The long-term success of urban watershed management strategies, measured in terms of quantifiable improvement in water quality, depends on residents adopting proven technologies. Human dimensions research addresses the causes and consequences of environmental change and is particularly relevant in urban watersheds because socioeconomic factors significantly impact sustainability goals. Understanding and targeting the motivations and behaviors of watershed residents is an essential aspect of adopting appropriate fertilizer management practices.

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