

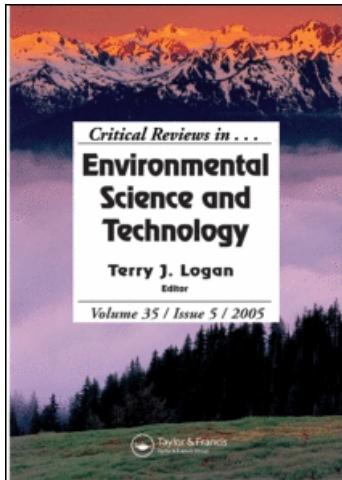
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Effects of Agricultural Drainage on Aquatic Ecosystems: A Review

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The extensive development of surface and subsurface drainage systems to facilitate agricultural production throughout North America has significantly altered the hydrology of landscapes compared to historical conditions. Drainage has transformed nutrient and hydrologic dynamics, structure, function, quantity, and configuration of stream and wetland ecosystems. In many agricultural regions, more than 80% of some catchment basins may be drained by surface ditches and subsurface drain pipes (tiles). Natural channels have been straightened and deepened for surface drainage ditches with significant effects on channel morphology, instream habitats for aquatic organisms, floodplain and riparian connectivity, sediment dynamics, and nutrient cycling. The connection of formerly isolated wetland basins to extensive networks of surface drainage and the construction of main channel ditches through millions of acres of formerly low-lying marsh or wet prairie, where no defined channel may have previously existed, have resulted in large-scale conversion of aquatic habitat types, from wetland

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mosaics to linear systems. Reduced surface storage, increased conveyance, and increased effective drainage area have altered the dynamics of and generally increased flows in larger streams and rivers. Cumulatively, these changes in hydrology, geomorphology, nutrient cycling, and sediment dynamics have had profound implications for aquatic ecosystems and biodiversity.

KEY WORDS: water quality, wetlands, nutrients, hydrology, fish, macroinvertebrates

INTRODUCTION

The North American landscape has been profoundly altered to promote agricultural development since European settlement (“settlement”) at the end of the nineteenth century. More than 98% of the North American prairie and vast areas of forest have been replaced with croplands. Bringing much of this land into production under modern agricultural systems has been associated with extensive modifications to natural drainage networks. Extensive networks of surface ditches and subsurface drains (“tiles”) have been constructed to remove excess water from the field soil surface or soil profile (Spaling & Smit, 1995). By 1987, the most recent year for which survey data were collected, more than 17% of U.S. cropland (up to 30% in the Upper Midwest) had been altered by artificial surface or subsurface drainage (Pavelis, 1987).

Extensive portions of landscapes in Europe have also been modified by drainage to increase agricultural production, including 34% of farmland in Northwest Europe, and as much as 50% in Scotland (Abbot & Leeds-Harrison, 1998). Subsurface drainage systems are less widespread in the developing world. Just 4% of agricultural land in southeast Asia, for example, and 2% of irrigated acres in Iran has extensive subsurface drainage (Abbott & Leeds-Harrison, 1998; Sohrabi et al., 1998). Worldwide, Smedema and Ochs (1997) estimated that drainage systems are in place on just one-third of the total land area where natural drainage constrains agricultural development or production.

The environmental impacts of agricultural drainage are significant, and have been the subject of study and debate in North America from the beginning of European settlement, when conflicts arose over the replacement of and/or alteration of wetland and floodplain ecosystems by agriculturally drained land (Hey & Phillipi, 1995; Thompson, 2002). Effects of drainage on aquatic ecosystems include both direct and indirect effects. Direct effects include habitat loss due to stream channelization and conversion of wetlands to croplands. Indirect effects include water quality and habitat impacts of sediment, phosphorus, nitrogen and other contaminants in agricultural runoff, as well as hydrologic alteration in the form of altered volume and timing of runoff. Alteration of flow regimes in turn drives a complex of

interrelated changes in stream morphology, instream and riparian habitats, nutrient cycles, and biota.

Drainage activities are ongoing in many agricultural regions, influenced by economic and climatic factors. For example, in Minnesota's Red River Basin, where 66% of soils are classified as "poorly drained," installation of subsurface drainage accelerated during the 1990s driven by wetter conditions. In many parts of the Upper Midwest, replacement of aging subsurface drainage systems is also driving increased drainage activity. The development and maintenance of subsurface drainage systems on agricultural lands continues to generate vigorous debate about the hydrologic and environmental impacts of subsurface drainage, between proponents who emphasize the potential benefits of reduced sediment and phosphorus in runoff and the addition of soil water storage, and skeptics who focus on the cumulative impacts of wetland drainage, hydrologic alteration, and nitrate loss to surface and ground waters.

The purpose of this paper is to present a comprehensive review of research findings of agricultural drainage on aquatic ecosystems, with specific reference to subsurface drainage. However, because land conversion, wetland drainage, installation of surface and subsurface drainage networks, and other modifications of terrestrial and aquatic ecosystems on behalf of agriculture tend to occur in tandem, it is difficult to separate subsurface drainage effects from these other changes. Artificial drainage activities almost always occur concurrently with changes in land use and cropping systems or in response to climatic trends. Intensive water management in many agricultural basins has also increased the demand for downstream flood management, resulting in other water management-related consequences for water resources and aquatic biota (Simpkins et al., 2004). The effects of these suites of changes are cumulative, interrelated, and tend to compound across different spatial and temporal scales (Aadland et al., 2005; Spaling & Smit, 1995). The contribution of subsurface drainage to aquatic ecosystem effects may be difficult to isolate relative to other agricultural impacts. However, many of the field and landscape-scale conservation designs and best practices under evaluation for their potential to mitigate the impacts of subsurface drainage may be effective simultaneously in addressing related conservation problems.

DRAINAGE STATUS AND HISTORY IN THE UNITED STATES

A significant portion of the world's most productive agricultural lands are located in regions and on soils that were wetlands. In the upper Midwest of the United States, much of the landscape was originally "swamp" due to the dominance of fine, glacially-derived soils with poor internal drainage (Eidem et al., 1999; Rodvang & Simpkins, 2001; Skaggs et al., 1994). For example, it is estimated that there were 45 million acres of wetlands in the

Mississippi River Basin at the time of settlement, representing at least 10% of the land surface, compared to 19 million today (Dahl, 1990; Hey & Philippi, 1995). Many settlers initially drained floodplains and low-lying wet areas for farming, rather than clearing forested slopes (Prince, 1997, Thompson, 2002; see Figure 1).

The main purpose of artificial drainage has been to increase crop production on lands converted to agriculture. By removing excess water from

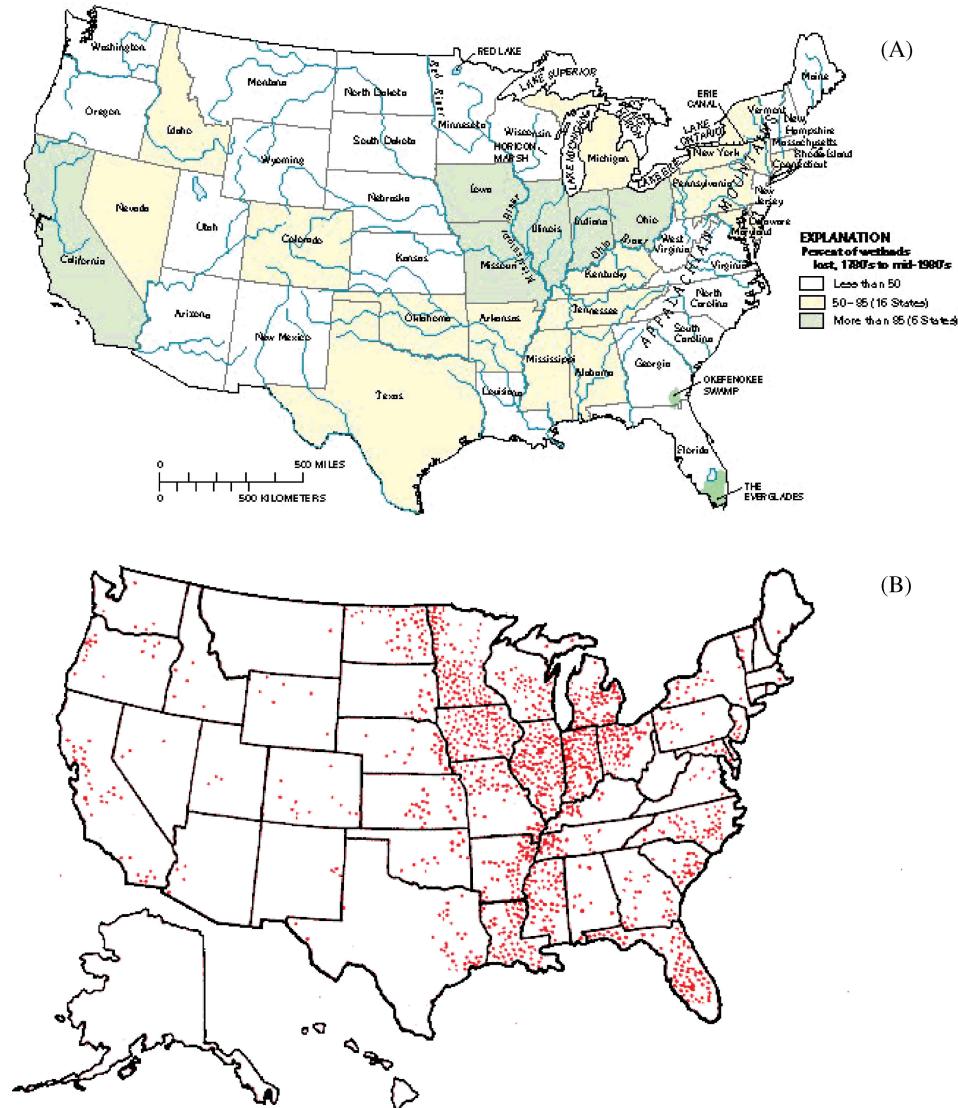


FIGURE 1. (a) States with notable wetland loss, 1780s to mid-1980s, in relation to (b) extent and location of artificially drained agricultural land in the United States, 1985. Modified from Dahl, 1990. (*Courtesy of U.S. Department of Agriculture Economic Research Service*)

the soil and creating an environment that allows greater plant uptake of nutrients, drainage significantly increases crop growth and productivity (Zucker & Brown, 1998). Drainage can reduce the risk of crop loss from excess water stress and provide more uniform crop production amidst climate variability. Farmers have more control over field operations, including earlier planting, drier harvest conditions, more efficient use of machinery, less soil compaction, and a wider choice of crops and crop varieties (Spaling & Smit, 1995). Greater control over moisture also reduces crop susceptibility to pests and disease that can reduce the need for and use of pesticides and fungicides.

Per acre, subsurface drainage systems are generally more expensive than surface drainage. Although requiring a significant initial investment (\$300–600/acre), subsurface drainage can increase yields on-farm by 5–25% annually, with attendant economic benefits that accrue not just to the farm but to rural and regional economies (Eidman, 1997). Where farmers depend heavily on irrigation for adequate water supply, subsurface drainage can also be used to prevent buildup of salts in the soil profile, flushing them into drainage waters or keeping them deeper in the soil profile.

Drainage pipes are typically installed at a depth of 0.6–1.2 m, and at a spacing of 10–30 m, depending on site-specific soils, crop type, and cost (Pavelis, 1987; Skaggs & van Shilfegaarde, 1999; see Figure 2). In some areas, surface inlets or intakes (risers extended from underground pipes to the surface) remove surface water from depressed areas in fields. Most drainage networks discharge directly to an open ditch or stream. Drain spacing, depth, and outlets have a significant impact on cost, as well as hydrology and water quality of receiving streams, which will be discussed later. Traditionally, subsurface drains were used strategically to drain wet areas of fields where surface water tended to pond. Increasingly, drainage installations are designed to lower the water table throughout whole fields, a practice termed “pattern tiling” (Hubbard, 2005).

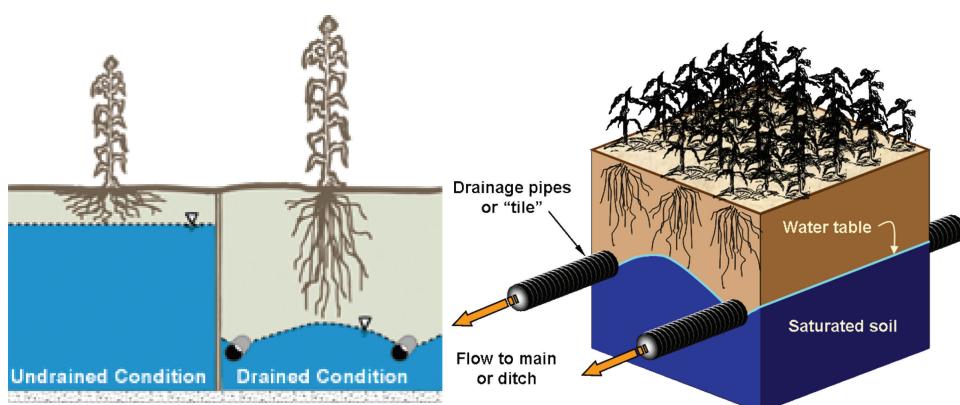


FIGURE 2. Subsurface drainage lowers the water table to improve crop root growth in soils with poor internal drainage. (Courtesy of University of Minnesota Extension Service)

In the midwestern United States (Midwest), subsurface drainage lines began to be installed in the late 1800s (earlier in New England). Most of the “tile” drains (pipes or sections of clay, concrete, or wood) in the region were installed between 1870 and 1920 and again between 1945 and 1960 (Zucker & Brown, 1998). In the 1970s, drainage pipes made from perforated polyethylene tubing began to replace concrete or clay “tile” (Pavelis, 1987). Subsurface drainage pipes underlie an estimated 18–28 million hectares (40–70 million acres) of the 3 million km² Mississippi basin (Mitsch et al., 2001). These estimates on the extent of subsurface drainage are considered very approximate because data are not currently kept by any central authority or agency.

Subsurface drainage both catalyzed and occurred coincidently with other crop production and management changes in modern agriculture (Dinnes et al., 2002), which now occurs across 80% of the landscape in the Midwest. The construction of extensive surface drainage networks facilitated plowing the soil for annual row-crop production. Early drainage efforts focused on straightening and enlarging streams to carry water more rapidly from wet areas of farms to nearby surface waters, as well as creating ditches to drain isolated or undrained basins and connecting them to surface water networks. Floodplains were extensively leveed, drained, and converted to agriculture as rivers were dammed and channelized for navigation. By 1987, more than 20 million ha in the states of Illinois, Indiana, Iowa, Ohio, Minnesota, Michigan, Missouri, and Wisconsin had been artificially drained (or 10–60% of cropland acreage by state; Pavelis, 1987; Zucker & Brown, 1998) in contrast to approximately 2 million ha that was irrigated in the same states (USDA Natl. Agric. Stat. Serv., 1999). Skaggs and colleagues (1994) estimated that drainage could increase crop production on 31 million ha that are already partially drained and an additional 12 million ha of undrained cropland.

Societal views of drainage broadened through time to recognize the importance of protecting water quality, and later, ecosystems and ecosystem services. Yet even from the start, there was minority opposition and social conflict over land drainage and mutually incompatible uses of rivers, floodplains, and wetlands (Thompson, 2002). From 1879 through the 1920s, as wetlands were drained and levees constructed in the floodplains of major rivers, such as the Illinois and Mississippi, naturalists and sportsmen denounced the elimination and modification of wetland ecosystems, fish, and wildlife habitats. Conflicts were typically resolved in favor of agriculture, commercial navigation, and urban and industrial use, which were viewed as more critical to the economic and industrial development of a growing world power. Farm, business, and urban interests organized public works and exercised eminent domain to secure farmland from overflow, seepage, and runoff.

In the wake of the Great Depression and the Dust Bowl, the U.S. federal government began to take a larger role in conservation of natural resources. Long-standing debates over flood control and water storage policy with respect to wetlands originated in this era (Anfinson, 2003). In the 1950s, the USDA and the Army Corps of Engineers debated the relative merits of different approaches to flood storage, with the USDA in favor of many small dams distributed throughout the watershed (headwaters and upper and middle reaches) and the Corps in favor of large projects (Hey & Philippi, 1995).

By the mid-1980s, concern about the high rate of wetlands loss—largely due to agricultural drainage—prompted the passage of federal wetlands protection programs such as the Wetland Reserve Program and the Swampbuster provisions of the 1985 and 1990 Farm Bills (Dahl, 1990, 2000). These programs helped to reduce the annual rate of wetland losses and decrease agriculture's share as a cause of wetland loss. However, freshwater emergent wetlands in agricultural areas, especially those that are partially drained, continue to be lost (Dahl, 2000). Research priorities at land grant universities have tracked the evolution of societal priorities, from technical developments in drainage systems through the early part of the century to increasing interest in the environmental impacts of drainage (Spaling & Smit, 1995; Walters & Shrubsole, 2003, 2005). In the 1960s and 1970s, research focused on the basic understanding of water quality implications, and has increasingly been expanded to synthesize broader ecosystem impacts of drainage as well as determining the most effective and efficient management and policy responses (Wilson, 2000).

IMPACTS OF AGRICULTURAL DRAINAGE ON AQUATIC ECOSYSTEMS

Agricultural development occurring over the past 150 years has significantly modified habitats, resulting in biodiversity loss and impacts to aquatic ecosystems at landscape scales. Changes in land use from agriculture have disrupted natural water and nutrient cycles and significantly altered natural regimes to which aquatic communities are adapted. In North America and throughout the world, land drainage and water management for agriculture has significantly altered the hydrology of wetland, stream, river, and floodplain riparian ecosystems.

Conceptual diagrams have been developed to illustrate the general relationship between drainage and environmental impacts. In Figure 3, we adapt heuristic conceptual models reviewed by Spaling and Smit (1995) and others to represent the dominant pathways by which surface and subsurface drainage influences aquatic ecosystems discussed in this review, as well as to

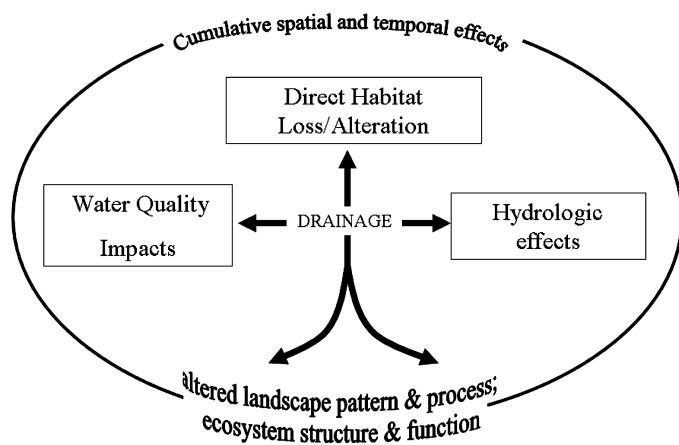


FIGURE 3. Conceptual model of dominant effects of drainage on aquatic ecosystems.

represent the spatial and temporal accumulation of environmental change. The accumulation of these direct environmental changes through time and across space interacts with the effects of altered landscape pattern, process, structure, and function, with long-term implications for local and regional biodiversity. Figure 4 is a more detailed representation of the range of direct and indirect effects of drainage, designed to illustrate the complexity of interactions. Figure 4 also separates the dominant pathways for surface drainage, on the left side of the diagram, from subsurface drainage on the right.

The severity of impacts of agricultural drainage on aquatic ecosystems can be characterized in proportion to the degree to which drainage alters local and regional ecological structure, function, and disturbance regimes to which natural communities have adapted. The most critical factors include hydrologic, chemical, geomorphic, and temperature dynamics; cycling of nutrients and organic matter; and habitat connectivity (Braun et al., 2003). Both the magnitude and direction of changes in flow affect the degree to which stream channels and habitats are destabilized. The impacts of nutrient enrichment are also generally proportional to the degree to which nutrient losses from agricultural ecosystems disrupt or contrast with patterns and timing of natural nutrient cycling (Dodds & Welch, 2000).

Despite agreement on the importance of these factors in the aquatic science literature, there is little quantitative predictive guidance available regarding how much alteration of key factors can occur without precipitating biodiversity loss; which specific agricultural activities cause the most change; how to develop, assess, and promote ecologically effective conservation and management practices; and the level and mixture of conservation and management practices needed to meaningfully address these threats (Braun et al., 2003; Rabalais 2002).

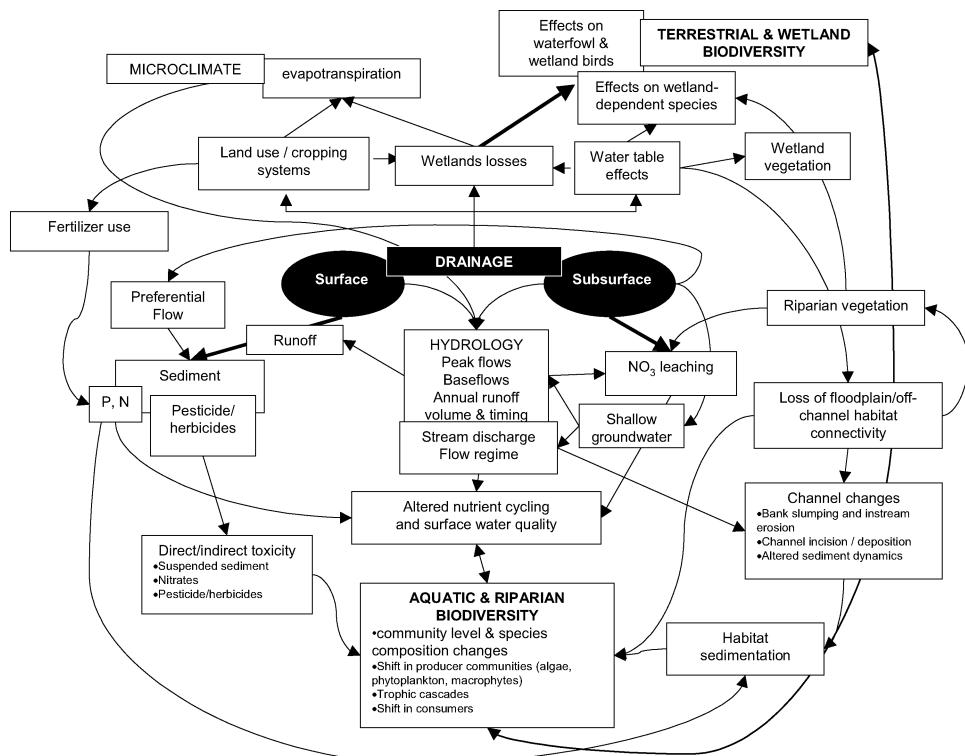


FIGURE 4. Conceptual model of surface and subsurface drainage effects on aquatic ecosystems.

Both surface and subsurface drainage result in land use and cover change, substantial loss of “leaked” agricultural nutrients and contaminants to surface and ground waters, and complex changes in hydrology and geomorphology relative to pre-drained conditions. Nutrient inputs and hydrologic alteration rank as two of the top three threats to 135 imperiled freshwater fishes, crayfishes, dragonflies and damselflies, mussels, and amphibians in the United States (Richter et al., 1996, Stein & Flack, 1997). Nutrient flows from agricultural drainage systems are considered to be the key water quality problem in the Mississippi River Basin, driving hypoxia in the Gulf of Mexico (Rabalais & Turner, 2001; Randall & Gross, 2001).

Considerable research has been devoted to understand the impacts of drainage on field- and watershed-scale hydrology and water quality. In recent years, a comprehensive picture has begun to emerge in reviews of the hydrologic and water quality implications associated with subsurface drainage. The effects of nutrient enrichment on aquatic ecosystems are also increasingly better understood (Rabalais, 2002). However, the effects of specific practices on aquatic communities in agricultural ecosystems are difficult to quantify and predict (Watzin & McIntosh, 1999).

Because the mechanisms differ by which subsurface versus surface drainage systems remove water from the landscape and deliver it to surface waters, there are differing implications for water quality, as well as for the direction and magnitude of hydrologic impacts. For example, losses of phosphorus, sediment, and other pollutants that sorb to soils tend to be more significant with surface drainage, where a relatively higher proportion of discharge occurs as overland runoff. However, a subset of environmental problems are unique to subsurface drainage. Subsurface drainage tends to substantially increase losses of nitrate and other soluble contaminants that leach into water through the soil profile. Subsurface drainage also contributes to increased potential for total N loss because the cropping systems required to provide a return on drainage investment are often “leakier” (Baker et al., 2004). In the U.S. Midwest, for example, subsurface drainage encourages the planting of high-value crops such as corn and soybeans relative to crops such as small grains and alfalfa typically associated with lower nitrate losses.

In more arid regions or areas with high background levels of salts in soils, diversion of freshwater inflows for irrigation and agricultural use may result in insufficient flows for aquatic ecosystems along with substantially reduced water quality of remaining instream flows due to return flows from irrigation (Smakhtin, 2001). The accumulation of toxic trace elements and salts in sediments and in biota, particularly selenium, is a potential concern in closed basins (i.e., basins lacking a permanent direct connection to a stream network) receiving subsurface agricultural drainage waters from irrigated cropland (Lemly, 1993). Migratory birds have been poisoned by toxic elements in drainwater—primarily selenium—on at least six national wildlife refuges in both the Pacific and Central U.S. flyways (Lemly, 1994; Presser et al., 1994).

Agricultural drainage has reduced water storage in the soil and increased conveyance, leading to higher concentrations of nutrients and greater sediment loads entrained and transported downstream. Subsurface drainage also has implications for hydrology of adjacent and downstream wetlands and riparian areas. Lowered local water tables on agricultural lands influence water table dynamics in adjacent natural areas and riparian zones that alter the composition of plant communities and their habitat value for wildlife.

Agricultural drainage contributes cumulatively to environmental change at catchment (i.e., the entire upstream area drained by a wetland, stream, or lake), landscape, regional, and global scales. The addition of subsurface drainage to lands already drained by surface drainage may result in field- and catchment-scale changes in hydrology and water quality, as well as cumulative effects due to changes in cropping systems, production acreage, basin evapotranspiration, and water budgets. Nutrients lost from agricultural systems contribute significantly to altered regional and global biogeochemical cycling. Cropping and agricultural management systems associated with significant drainage infrastructure contribute to decreased habitat diversity

and heterogeneity at the landscape scale, with consequences for biodiversity at regional scales.

The most significant aquatic ecosystem impact of drainage historically has been the direct loss and alteration of wetland and riparian habitats (Goodwin et al., 1997). The extent of agricultural drainage is often inversely proportional to wetland, riparian, and aquatic habitat loss. The role of habitat connectivity at differing spatial and temporal scales is a fundamental and well-understood principle of population biology (Meffe & Carroll, 1997). Extensive habitat loss can fragment remnant habitat, such that threshold effects on plant and animal populations are possible, in which remaining habitat is too patchy, isolated, or disconnected to maintain populations (Askins, 1995; Fahrig, 2002; Kennedy et al., 2003; Lehtinen et al., 1999; Stein & Ambrose, 2001). Several recent assessments of aquatic biodiversity in agricultural landscapes have suggested that the combination of land use change, habitat and population fragmentation, and past disturbance has created an extinction debt that is likely to drive continuing biodiversity declines even if current threats are ameliorated (Hanski & Ovaskainen, 2002; Jenkins et al., 2003; Poole & Downing, 2004; Tilman et al., 1994).

Often ecological change is the result of cumulative effects, and it is not possible to fully account for the relative importance of different factors (Bedford & Preston, 1998). Because it is difficult to separate the independent effects of subsurface drainage from cumulative effects of other hydrologic alterations resulting from agriculture, including land conversion, surface drainage, surface and groundwater withdrawals for irrigation, impoundments, and channelization and straightening of watercourses, streams, and rivers, this review covers mechanisms that influence changes in aquatic communities as well as empirical evidence of change. Despite the abundance of drainage studies, the independent effects of subsurface drainage on aquatic communities have not been well studied. Yet even where agricultural drainage is not the primary cause of habitat loss or changes in distribution and abundance of species, the additional stress of nutrients, chemicals, or sediments in drainage waters is potentially significant.

Direct Habitat Loss Due to Drainage

The most salient effect of artificial drainage has been the direct elimination of wetland and riparian habitats, with attendant consequences for loss of wetland services, alteration of stream and wetland hydrology, and nutrient cycles. Less than half of the 89.5 million ha (221 million acres) of wetland acres estimated to have been present in the conterminous United States at the time of European settlement remain (Dahl, 1990; see Figure 1). Most of these historic losses of wetlands are attributable to drainage for agriculture. In many agricultural areas, local wetland losses—particularly for certain

wetland types—often exceed 90% (see Cowardin et al., 1979, for an overview of wetland types and terminology). The loss of wetlands has resulted in the reduction of a variety of economically and ecologically significant wetland services, including direct loss of habitat for wetland-dependent species, significant alteration of biogeochemical and hydrologic cycles, loss of flood storage and water quality functions of wetlands, and elimination of nutrient and sediment sinks and other buffering capacities of wetlands in relation to adjacent upland and riparian ecosystems (Pinay et al., 2002; Tiner, 2005; Zedler, 2003). Wetland losses in the Prairie Pothole Region (a grassland ecoregion of regional and global significance for migratory birds, among other fauna, that extends across the central United States and Canada; see Figure 5) are estimated to have reduced populations of wetland and wetland-dependent wildlife by at least 50% (NRC, 1992), as well as causing local extirpation of many species (Weller, 1981). Jenkins et al. (2003) estimated, based on demographic simulation modeling, that drainage of wetland habitats in Illinois over the 100-year period from 1850–1950 is likely to have driven the extinction of 8–9 crustacean species (or roughly 10% of the original species). Model results further indicated that, due to lag times in the effects of habitat fragmentation and isolation, future extinction rates for crustaceans of ephemeral wetlands in Illinois will be even greater.

Drainage has also altered the landscape pattern and hydrology of remaining wetlands, from a mosaic of numerous, diverse, clustered wetlands to fewer, more isolated, and more permanent wetlands (Gibbs, 2000; Krapu et al., 2004). This has implications both for the structure and function of



FIGURE 5. The Prairie Pothole Region of North America. Modified from Euliss et al. (1999).

wetland vegetation, communities and food webs (Galatowitsch & van der Valk, 1996; Greenwood et al., 1995), as well as for dispersal of small wetlands organisms that rely on occasional migration to sustain local populations (Semlitsch, 1998). Average dispersal distances are generally <0.3 km for frogs, salamanders, and small mammals (Semlitsch & Bodie, 1998), and <0.5 km for reptiles (Joyal et al., 2001). Reduced wetland availability and connectivity also impact wider ranging birds and mammals, such as raptors and other predators that rely heavily on wetland-dependent prey (Murphy, 1997; Richards & Cable, 2003).

Cumulative wetlands losses due to agricultural drainage in the Midwest have disproportionately affected small, ephemeral, hydrologically variable wetland types (Johnston, 1994; Snodgrass et al., 2000; Tiner, 2003). Kuehner (2004) examined aerial photographs, presettlement maps, and survey notes for a small agricultural watershed in southeastern Minnesota, as well as mapping surface and subsurface drainage modifications and cropping system shifts. His analysis revealed that 50% of the watershed had once been covered by wetlands, of which 88% had been drained and converted to cropland. Almost half of the wetlands losses occurred prior to 1938. From 1938 to 1985, an additional 40% loss was incurred in conjunction with the construction of 40 km of drainage ditches and nearly 1000 km of subsurface drainage systems. Total corn and soybean acreage within the watershed increased from 30% to 96% from 1938 to 1985. An unpublished analysis of wetlands losses in a five-county area in southwestern Minnesota showed that although wetlands losses are only 50% in terms of acreage, 87% of the basins have been lost by number, indicating that it is overwhelmingly the numerous small basins that have been lost (Johnson, Rex, U.S. Fish and Wildlife Service, unpublished analysis).

Although wetland losses due to agricultural drainage have slowed considerably due to the combination of federal and state legislation designed to protect many types of wetlands, additional losses and impacts are still of concern due to the landscape-scale effects of habitat loss and the impact of drainage on wetland hydrology, particularly of small, temporary wetlands. Small, temporary wetlands are not protected under current legislation in many jurisdictions (Gibbs, 2000; Semlitsch & Bodie, 1998). Naugle et al. (2001) found the proportion of protected wetlands in South Dakota was highest for semi-permanent (32.3%), intermediate for seasonal (25.6%), and lowest for temporary wetlands (15.8%). Smaller, more isolated wetlands have traditionally received little management or conservation emphasis relative to larger wetland complexes, and are not protected by current legislation in many jurisdictions (Gibbs, 2000; Semlitsch & Bodie, 1998; Snodgrass et al., 2000).

An important contemporary impact of subsurface drainage is the removal of temporary and seasonal ponded water depressions (i.e., ephemeral

wetlands) on cultivated land. Agricultural wetlands, including temporary ponded depressions in fallow fields, idled cropland, and conservation reserve program (CRP) lands, often receive high avian use relative to larger natural emergent or permanent wetlands (Heard et al., 2000; Krapu, 1974; LaGrange & Dinsmore, 1989). Tillage has been found to decrease the diversity of invertebrate prey available to wetland avifauna, when tilled and untilled wetlands were compared (Euliss & Mushet, 1999; Freeland et al., 1999). Nevertheless, emergent, temporal seasonal wetlands in tilled cropland have been reported to be the preferred habitat of killdeer (*Charadrius vociferus*) and several other species of shorebirds (Hands et al., 1991; Hubbard, 2005), and may receive significant use by migrating waterfowl as well (LaGrange & Dinsmore, 1989).

Continued hydrologic loss of small and/or ephemeral wetlands should be of conservation concern, particularly in the context of historic wetland losses, because it directly impacts the connectivity and quality of remaining wetland habitats and therefore the viability of wetland-dependent populations (Gibbs, 1993; Johnston, 1994; Semlitsch & Bodie, 1998). Although wetland area is positively related to wetland species richness for a wide variety of aquatic and semi-aquatic species, small wetlands also have unique ecological functions. Small wetlands with short hydroperiods that favor periodic drying are important to amphibians, because large, permanent wetlands are more likely to support predatory fish and invertebrates that prey on or compete with amphibian larvae (Pechmann et al., 1989). The hydrology of these ephemeral wetlands provides a unique set of habitat conditions for food resources (e.g., amphibians, aquatic invertebrates, seeds and tubers of aquatic plants) used by birds during migration and breeding (Gibbs, 1993; Rivers & Cable, 2003; Snodgrass et al., 2000). Water depth and the recession of water into mudflats directly influences the availability of prey to shorebirds (Davis & Smith, 1998). The daily and seasonal recession of water levels provides a habitat analogous to natural mudflats associated with annual spring floods and high water, with which aquatic insects and other prey species have synchronized their life cycle (Williams, 1998, 2005). The burst in abundance and availability of food resources during emergence provides the patchy, short-term, concentrated high energy resources needed by birds to sustain the large energy expenditures needed during migration (Askins, 1995; Skagen & Knopf, 1993).

Heterogeneity of wetland types within an overall wetland complex creates greater habitat diversity and higher vertebrate species richness (Fairbairn & Dinsmore, 2001; Porej, 2004). Many wetland species forage and/or loaf in aquatic habitats but require specific habitat features for successful breeding or roosting. During migrations, both waterfowl and shorebirds require frequent stopover feeding locations to replenish their energy needs (Skagen & Knopf, 1993). The Great Plains flyway differs from the coastal routes in the

unpredictability of stopover sites (Skagen & Knopf, 1993). Migratory birds of the U.S. central flyway have been shown to shift their migration routes in response to landscape-scale habitat availability, with density of birds settling directly related to the availability of temporal wetlands (Fairbairn & Dinsmore, 2001). At the landscape scale, availability of feeding habitats and their use by migratory birds shifts in response to changing hydrologic conditions throughout the spring (Skagen & Knopf, 1993). During drought periods, migration patterns tend to shift to the wetter eastern and northern range of the central prairie pothole region, where wetlands are more abundant (Johnson et al., 2005).

The availability of seasonal wetland habitats imposes an important constraint on breeding waterfowl populations (Kantrud et al., 1989; Fairbairn & Dinsmore, 2001; Rivers & Cable, 2003; Van der Valk, 1989) as well as on migratory birds of the central flyway. Successful breeding requires the availability of a variety of wetlands because no single wetland or basin provides for all food and resource needs of waterfowl and shorebirds throughout the breeding season (April through September in the northern hemisphere; Swanson & Duebbert, 1989). Haig et al. (1998) reviewed numerous studies showing the importance of frequent within-season movements of birds among wetlands for foraging, nesting, territory switching, and habitat shifts associated with brood development.

Cumulative habitat losses may lead to threshold effects in which otherwise suitable habitats are rendered unusable (Kennedy et al., 2003; Skagen & Knopf, 1993; Sorenson et al., 1998). For example, using a landscape scale habitat suitability model, Naugle et al., (2001) predicted the removal of wetlands < 0.5 ha would result in a 21% decrease in the number of wetlands suitable for northern pintails, a highly mobile species of duck that uses multiple wetlands within a season.

Migration, breeding, or feeding “bottlenecks” (i.e., events that reduce reproduction or increase mortality by 2–10 or more, across significant portions of the landscape) can dramatically affect waterfowl and wetland-dependent bird populations (Johnson et al., 2005). The Prairie Pothole Region (PPR) of the United States and Canada represents only 10% of North America’s waterfowl breeding area but produces half of the continent’s waterfowl in an average year (Tiner, 2003). Modeling exercises that explored the implications of climate change suggest that under likely climate conditions—shortened hydrological cycles with increased variability in wet-dry conditions across much of the Midwest—additional wetland habitat loss may result in significant declines in waterfowl populations in the central North American flyway (Johnson et al., 2005; Poiani & Johnson, 1991; Sorenson et al., 1998). Sorenson et al. (1998) estimated that under dryer conditions (35–55% habitat loss), the north central U.S. duck population could be cut in half. Population declines below thresholds or historic levels could limit the availability

of waterfowl populations to rebound quickly in wetter years. Therefore, it is important to understand the potential impact of subsurface drainage on temporary ponded depressions on croplands, as well as on the hydrology of adjacent undrained wetlands.

Hydrologic Effects of Surface and Subsurface Drainage

FIELD AND CATCHMENT-SCALE HYDROLOGY

Extensive artificial drainage significantly alters the natural hydrology of agricultural watersheds, where it is used extensively. In many parts of the world, increased frequency and intensity of flooding have been attributed to large-scale land conversion and intensification of agriculture (Wiskow & van der Ploeg, 2003). However, the hydrologic effects of drainage typically occur in conjunction with land use changes that also play a significant role. Compared to land in natural forest or perennial grassland, conversion and drainage of land for agriculture usually increases peak runoff rates, sediment, and pollutant loads to surface-water resources (Hill, 1976; Lemly, 1993; Miller, 1999; Robinson & Rycroft, 1999; Skaggs et al., 1994, citing seven studies; Wiskow & van der Ploeg, 2003). Croplands in intensively managed agricultural landscapes typically store less water, and runoff is higher and more flashy (i.e., characterized by spiking peak flows followed by a rapid return to base flow) than lands in perennial vegetation, such as the native prairie of the North American Midwest (Knox, 2001; Robinson & Rycroft, 1999; Schilling & Libra, 2003). In northern prairie, evapotranspiration (ET), rather than drainage, is the major water loss in northern prairie wetlands, accounting for 70–80% precipitation in years of normal rainfall (Poiani et al., 1996; Woo & Rowsell, 1993). Although ET rates on cropland during the peak of the growing season may exceed that of natural prairie, most runoff in the upper Midwest occurs during early spring before crops are planted, when ET rates are typically higher for lands in perennial crops or native vegetation (Brye et al., 2000; Schaffer, 2005).

A coherent picture of hydrologic and water quality effects of subsurface drainage has begun to emerge in comprehensive reviews by Robinson (1990), Skaggs et al. (1994), Robinson and Rycroft (1999), and others. These reviews that evaluated research, experimental studies, and computer simulations throughout the world confirmed that despite improved understanding, the effects of subsurface drainage are complex and vary in response to many local factors. The magnitude and direction of effects on peak flows and total runoff depends on a number of factors, including the timing and amount of precipitation, antecedent soil moisture, soil type, depth to water table, topography, and configuration of catchment drainage networks, as well as management factors such as tillage practices; drainage system design, depth

and spacing; and location in relation to the point of impact assessment (Skaggs et al., 1994).

A distinction can be made between improving subsurface drainage on land already used for agriculture versus draining undrained soils or converting undrained wetlands. Relative to undrained land, both surface and subsurface drainage produce significant changes in on- and off-site hydrology, due primarily to more rapid conveyance of water and "flashier" hydrographs. However, where land has already been converted to agricultural production, the addition of subsurface drainage may reduce runoff, peak outflow rates, and sediment losses (Konyha et al., 1992; Robinson, 1990; Skaggs et al., 1994). Relative to cropland drained by surface drainage alone, subsurface drainage can create increased temporary storage capacity in the upper layer of soil, allowing water to infiltrate and spread through the soil over a longer period (Fraser & Fleming, 2001; Mason & Rost, 1951; Moore & Larson, 1980; Skaggs & Brodhead, 1982).

EFFECTS OF AGRICULTURAL DRAINAGE ON PEAK FLOWS

Historically, the combined effect of agricultural surface and subsurface drainage, channelization, and land use change has been to increase streamflow peaks and shorten catchment response times (Robinson & Rycroft, 1999; Wiskow & van der Ploeg, 2003). Sixty percent (60%) of streams or rivers in the United States have experienced major changes in high or low flow (>75% change) or in the timing of these flows (>60 day shift) in the 1970s through 1990s compared to a 1930–reference period (Heinz Center, 2002). Although many of these changes are most directly related to extensive development of dams and water control structures, the contribution of watershed hydrologic changes to altered flow regimes is significant.

Field-scale surface drainage typically increases peak flows by reducing surface storage (Robinson & Rycroft, 1999). The effects of subsurface drainage on peak flows at the field scale, however, have been found to be variable, depending on local soil properties as well as antecedent moisture conditions. Subsurface drainage reduces both peak outflows and the frequency of surface runoff events at sites characterized by high water tables or prolonged surface saturation ("ponding") in the undrained condition (Robinson, 1990; Robinson & Rycroft, 1999). The majority of studies show reduced peak flows in response to rainfall/runoff events relative to lands drained by surface drainage alone, as high as 60% (Konyha & Skaggs, 1992; Robinson & Rycroft, 1999; Wiskow & van der Ploeg, 2003). Skaggs et al. (1994) cited more than 17 studies where systems with improved subsurface drainage had less runoff and lower peak outflow rates than systems that depend primarily on surface drainage.

However, on more permeable soils, where infiltration, soil water storage capacity, and lateral conductivity/seepage is large enough to handle a given storm event, subsurface drainage may have the opposite effect for an

equivalent magnitude event, increasing peak flows by increasing the speed of subsurface discharges. Increased peak flows result because for storm events within a given range of magnitude and intensity, subsurface drainage may not substantially affect infiltration, whereas the rate of subsurface flow through the soil profile may increase over that prior to installation of artificial drainage (Robinson, 1990; Wiskow & van der Ploeg, 2003).

At larger scales and event magnitudes, the effects of subsurface drainage on peak flows tend to be dominated by other variables, including the pattern, magnitude, and timing of precipitation, the design and layout of surface and subsurface drainage networks, and the capacity and conveyance of the surface drainage network (Moore & Larson, 1980; Robinson & Rycroft, 1999).

EFFECTS OF AGRICULTURAL DRAINAGE ON MEAN ANNUAL RUNOFF AND BASIN YIELD

Changes associated with surface and subsurface drainage—including land use, cropping systems, and increased drainage area and density—affect catchment water budgets by altering the relative proportion of precipitation that is evapotranspired, stored in soil water and groundwater, or exported. Subsurface drainage typically alters the total water yield from a field or small watershed, not just the timing and shape of the hydrograph. The increase in total runoff tends to be relatively minor (~10%) but occurs because subsurface drainage may increase the proportion of total annual precipitation that is discharged to surface waters via subsurface flow relative to the amount that is stored semi-permanently, evaporated, or transpired (Magner et al., 2004; Moore & Larson, 1980; Serrano et al., 1985; Tomer et al., 2005).

Changes in evapotranspiration typically accompany the conversion of land from prairie grassland and forest to agricultural land use (Brye et al., 2000). The hydrologic impact of changes in ET also varies seasonally. As noted earlier, ET rates for cropland during the peak growing season can exceed some natural ecosystem types. However, most runoff in the upper Midwest occurs during early spring before crops are planted, when ET rates and soil storage are typically higher for lands in perennial crops or native vegetation (Brye et al., 2000; van der Kamp et al., 2003). Melesse et al. (2006) used remote sensing to estimate average increases in annual ET of 9–25% for five basins in northwestern Minnesota that were restored from cropland/pasture to native prairie.

The connection of once hydraulically isolated drainage basins to surface water drainage networks through surface and subsurface drainage systems can also influence total streamflow. Most of the wetlands in the Prairie Pothole Region of the United States and Canada historically lacked natural surface drainage outlets (Tiner, 2003). For example, more than 80% of a lowland region in eastern South Dakota drains into closed basins (Johnson & Higgins, 1997). Although considered “isolated,” many of these basins are hydrologically connected through shallow groundwater to regional water

tables and stream baseflows. In extensively drained landscapes, such as the agricultural Midwest of the United States, the connection of isolated basins has inflated total surface water discharge and increased the density of linear drainage networks (Haitjema, 1995; Magner et al., 2004; Ter Haar & Herricks, 1989).

EFFECTS OF AGRICULTURAL DRAINAGE ON BASEFLOWS

The effect of subsurface drainage is generally to increase baseflows (i.e., that portion of streamflow that derives from seepage or shallow groundwater, as opposed to surface runoff), regardless of whether peak flows are increased or decreased (Moore & Larson, 1980; Robinson, 1990; Schilling & Libra, 2003). Baseflows are directly related to the shift in the proportion of precipitation that is not evaporated or transpired, but rapidly conveyed into subsurface drain flow. Subsurface drainage flows now constitute the majority of base flow in many agricultural portions of the Midwest. For example, drain tiles contributed 86% of the flow in an 48,000 ha agricultural watershed in Illinois from 1995 to 1996 (Xue et al., 1998).

Under some conditions, tile drainage systems may reduce shallow groundwater recharge by lowering local water tables or diverting a higher proportion of water into surface drainage networks (Leopold, 1968). By lowering water tables, drainage may lower the base level for groundwater flows (Skaggs et al., 2005). In arid regions where instream flows often reach critically low levels in late summer and irrigated agriculture has elevated evapotranspiration (ET) losses, drainage may lead to reduced flows and exacerbate late-season water shortages. Drainage of river valley bottoms can lower groundwater tables, as well as reduce the near-channel storage that otherwise sustains lateral drainage during dry periods (Bullock & Acreman, 2003; Riggs, 1976; Smakhtin, 2001). Changes to the vegetation regime in valley bottom areas can affect evapotranspiration loss from riparian soils, affecting gains or losses to bank or alluvial storage (Smakhtin, 2001; Swank et al., 1988).

ECOLOGICAL EFFECTS OF DRAINAGE-RELATED HYDROLOGIC ALTERATION

The role of hydrology and flow regime in aquatic ecology. The flow regime is increasingly viewed as the key driver of the ecology of wetlands, lotic (moving water) systems, and associated floodplains (Bunn & Arthington, 2002; Mazeika et al., 2004; Nilsson & Svedmark, 2002; Poff & Allan, 1995; Poff et al., 1997; Pringle et al., 2000; Richter et al., 1996; Sparks, 1995; Townsend & Riley, 1999). Critical components of the flow regime that regulate ecological processes include the magnitude, frequency, duration, timing, and rate of change of hydrologic conditions (Poff et al., 1997; Richter et al., 1997).

The alteration of flow regimes affects ecosystem structure and function, which may shift the dominance in native community assemblages and facilitate the invasion and success of exotic and introduced species (Bunn & Arthington, 2002). Maintaining, restoring, or mimicking natural system

hydrology has become a major tenet of aquatic ecosystem science, management, and conservation (Fausch et al., 2002; Frothingham et al., 2002; Gilvear & Heal, 2002; Sprenger et al., 2001).

Variation in flow creates dynamic habitat conditions in streams (Aadland, 1993). High flows shape the river channel, deliver woody debris to the channel, flush fine sediments that would otherwise smother gravel substrates, and provide periodic connectivity with floodplain and backwater habitats, whereas low flows place temporal constraints on the seasonal and annual availability of aquatic habitat (Larimore et al., 1959; Schlosser, 1985; Smakhtin, 2001). Physical conditions in pools and riffles change with flow stage, creating temporally diverse habitats upon which many species are dependent for spawning, resting, and feeding (Aadland, 1993; Delucchi, 1988; Kemp et al., 1999; Poff & Allan, 1995; Poff et al., 1997; Rabeni & Jacobson, 1993). Flow variability also affects the pattern and availability of pool and riffle habitats, the connectivity of main channel flows to oxbows and off-channel habitats, and inundation of shallow, slow water habitat (Poff et al., 1997). Flow regime also strongly influences water quality and thermal regime (Manley, 1999).

Aquatic, semi-aquatic, and riparian-dependent species—whether plant, invertebrate, or fish—have evolved life history strategies in response to particular flow regimes and the habitat conditions created by them (Lytle & Poff, 2004). Numerous empirical studies have demonstrated that the structure and functional organization of species assemblages in lotic systems varies across a gradient of hydrological stability (Aadland, 1993; Bain et al., 1988; Frenzel & Swanson, 1996; Horwitz, 1978; Mion et al., 1998; Poff & Allan, 1995; Schlosser, 1985). Some species are adapted to stable flows, whereas others can tolerate extreme fluctuations in flow (Bain et al., 1988; Poff & Allan, 1995; Poff et al., 1997; Schlosser, 1985). Lytle and Poff (2004) recently reviewed more than 30 studies of life history, behavioral, or morphological adaptations to flow variability in fish and aquatic invertebrates. Their review explored how differences in these modes of adaptation influence the vulnerability of organisms to different kinds of flow regime alterations, as well as the implications for effective management and restoration of lotic ecosystems.

The relationship between flow regime and geomorphology. Changes in flow regime have important geomorphologic consequences for stream habitats (Frothingham et al., 2002; Kemp et al., 1999; Poff et al., 1997). Stream-flow controls depth, width, water velocity, and substrate composition (Poff & Ward, 1989). Spatial variation in channel form produces variability in mean velocity, flow depth, and substrate characteristics, which, in turn, creates large-scale and fine-scale habitat template for aquatic communities (Aadland, 1993; Fisher, 1997; Frissell et al., 1986; Southwood, 1977). The modification of geomorphological conditions in fluvial systems that drives homogenization of habitats is one of the major threats to aquatic biodiversity (Frothingham et al., 2002; TerHaar & Herricks, 1989).

Alterations in basin yield, peak flows, low flows, and total annual discharge have significant implications for stream channel geometry and form, and by extension, stream habitats. Changes in flow regime or sediment loads tend to produce compensatory adjustments in channel morphology as stream channels seek to establish new equilibria (Poff et al., 1997). Physical responses to increased magnitude and frequency of high flows include bank erosion and channel widening (Hammer, 1972).

Increased flashiness of flows due to urbanization and increased impervious surface is a well-known phenomenon (Hammer, 1972; Leopold, 1968). Increased flashiness of flows can lead to channel widening, channel incision, loss of the channel-floodplain connection, and headward erosion of stream channels (Leopold, 1968; Prestegaard, 1988).

Incised stream channels are laterally confined and increasingly disconnected from a functioning riparian corridor and floodplain (Simon & Darby, 1999). Such streams are unstable and redistribute sediment downstream. Faster travel times and increased water velocities reduce bank stability and carry greater loads of suspended and bedload sediment. Lowered water table levels in the vicinity of a stream can lead to streambank erosion and channel downcutting after loss of vegetation that stabilizes streambanks (Kondolf & Curry, 1986). Channel incision thus results in elevated rates of streambank erosion (Zaines et al., 2004).

The severity of geomorphologic effects depends on the relative magnitude of changes in flow and to a lesser extent the resistance of riparian soils and bed materials to erosion. Through the time stream, systems tend to develop a dynamic equilibrium between the movement of water and the movement of sediment (Dunne & Leopold, 1978). The channel-forming flow, also known as the effective discharge, occurs at or just below the bankfull stage with a recurrence frequency of 1–2 years (Leopold, 1994). Below this flow, a stream has less power to cause streambed or streambank erosion. Flows above bankfull occur infrequently, and when they do occur, their energy is typically dissipated by the floodplain if the channel is not incised.

Any increase in stream power (e.g., due to change in peak flows or increased frequency of bankfull flows) will generate an increase in sediment load (Lane, 1955). For a given flow, an increase in suspended sediment load will cause the streambed to aggrade, and a decreased load will cause erosion or incision. Both increases or decreases in sediment loads have implications. Impacts that reverse depositional tendencies along active deltas and floodplains may also accelerate rates of change, causing wetlands, floodplains, and other landscape sinks, where materials have accumulated over decades or centuries to become large net exporters rather than modest net importers, resulting in additional nutrient and sediment loading to downstream aquatic ecosystems (Brinson, 1988).

Geomorphic data throughout the Midwest that although many channels have been entrenched through ditching and straightening, natural channels have incised in response to increased discharge associated with the changes in land use and artificial drainage (Brookes, 1988; Faulkner, 1998; Fitzpatrick et al., 1999; Knox, 1987; Magner & Steffen, 2000). Modern subsurface drainage has altered contributing drainage area by increasing drainage density, decreasing micro-landscape storage, and connecting isolated basins (Magner et al., 2004; Moore & Larson, 1979). Tributary streams now convey floods and sediment downstream more efficiently than in the past (Faulkner, 1998).

Analyzing field morphological features for 25 USGS gaging stations in five tributaries across the Minnesota River basin, Magner and Steffen (2000) found evidence of recent channel incision. Analysis of USGS stream gage station data revealed that annual peak flows for the 1- to 2-yr recurrence intervals have increased by 20 to 206% over the past 25 years, a period corresponding with renewed installation of subsurface drainage (Magner & Steffen, 2000; Magner et al., 2004). Increased flows were attributed primarily to the increased volume of annual runoff associated with higher drainage density and wetland and prairie-lake conversions to cropland. The independent effects of subsurface drainage versus surface drainage on sediment and geomorphology are more difficult to isolate and summarize, and likely vary with the relative impact of subsurface drainage on the frequency and duration of bankfull flows.

Stream channel adjustments to a less variable flow regime and reduction in overbank sedimentation can, however, occur rapidly (Faulkner, 1998; Knox, 2001). Appropriately designed subsurface drainage in conjunction with other management practices designed to increase infiltration and reduce runoff can contribute to restoration of more stable, natural hydrologic regimes in catchments that have been dramatically transformed by large-scale agricultural land conversion in the past (Magilligan & Stamp, 1997).

Instream ecosystem effects of flow alterations. Because flow is the dominant physical process within the channel, changes in basin hydrology affect fish and other aquatic organisms through almost all hydrogeomorphic processes (Hupp, 1992). Disruption of hydrologic regimes results in loss of riparian habitat and connectivity, altered sediment transport, and either magnified or reduced differences between baseflow and flood stages (Sprenger et al., 2001). The combination of changes in flow, sediment dynamics, and habitats drives shifts in fish community structure, often from specialists to a few competitively-dominant generalists (Rahel, 2000). Changes in species composition alter nutrient uptake and energy flows within and across trophic levels (i.e., phytoplankton, macroinvertebrate, and fish communities) via competition, predation, and consumption (Dent et al., 2001; Hershey et al., 1988; Niyogi et al., 2004; Poff et al., 1997; Schlosser, 1985; Shieh et al., 2002).

The specific local impacts of hydrologic alteration on aquatic communities vary depending on the magnitude and direction of changes in flow from pre-disturbance conditions. Aquatic communities that have evolved in variable environments, such as headwater reaches and upstream environments, tend to be better adapted to stochastic disturbance and more tolerant of thermal and hydrologic extremes (Schlosser, 1990). Fishes also tend to alter their habitat use daily, seasonally, and/or interannually in response to shifting flow conditions. Flow alterations that reduce flow variability and/or homogenize habitats tend to cause displacement of habitat specialists by species that are competitively dominant under the altered habitat conditions. Reductions in flow variability can also facilitate invasions by non-native species that out-compete or displace native species under novel or homogenized conditions (Fausch et al., 2002; Magnuson, 1991). In large river systems, where biotic factors tend to dominate fish community structure, the most significant driver of ecological change is often the reduction in disturbance regimes and homogenization of habitats associated with navigation, flood control, and other water-level management activities (Freeman et al., 2001; Sparks, 1992).

In groundwater- or baseflow-dominated systems (e.g., trout streams), fish communities tend to be adapted to fairly stable flow regimes and specific instream microhabitats. Alterations that lead to greater thermal or hydrologic instability will drive fish communities toward more generalist and tolerant species, adapted to a wider range of flow conditions and associated habitats. Counterintuitively, in coldwater streams, increased diversity can even be an indicator of reduced biological integrity, as generalist species become more abundant under altered thermal or hydrologic regimes. Flow alterations associated with increased sedimentation cause fish assemblages to shift toward species tolerant of low velocity and silt, rather than silt-intolerant species (Poff & Allan, 1995).

Stream channel incision generally leads to reduced spatial habitat heterogeneity and greater temporal instability instream (Shields et al., 1994; Simon & Darby, 1999). Small fish species with restricted microhabitats were eliminated from a study site with high flow fluctuation, whereas the relative abundance of habitat generalists that could tolerate deep, fast flows peaked at sites with greatest flow variability (Shields et al., 1994, 1998). In Michigan streams, increased channel incision was associated with reduced biomass of total, game, and intolerant fish species (Infante, 2001; Infante et al., 2004). McRae et al. (2004) found that sites with the richest mussel assemblages in Michigan streams had greater flow stability, lower percentage of fine sediments in the streambed substrate, and lower channel incision.

Lowered water tables that result from channel incision or subsurface drainage of lands adjacent to wetlands or riparian lands changes water table and riparian-instream interaction. Downcutting due to channel incision that disconnects a stream from its floodplain and riparian border can disrupt ecological functioning in the riparian zone as well, limiting potential for both

floodplain storage and denitrification in saturated soils (Magner et al., 2004). Responses of vegetation communities to changes in hydroperiod are well-documented (Hupp, 1992; Miller & Zedler, 2003; Williams, 2005). Lowered water tables in Britain, due to land drainage, have negatively affected aquatic-dependent species in adjacent natural habitats, and played a role in the extinction and near extinction of two damselflies (*Coenagrion armatum* and *Lestes dryas*) and the loss of the large copper butterfly, *Lycaena dispar* (Moore, 1976, 1980; Williams, 2005).

As the major determinant of physical form and habitat in streams, the flow regime also determines the successional evolution of riparian plant communities and ecological processes (Nilsson & Svedmark, 2002.) Plants that require bare soil and access to shallow water tables rely on the periodic disturbance from flow variation. Stable flows allow aquatic communities to undergo succession—from communities dominated by pioneer, colonist species to communities in which competitive interactions play a significant role in structuring assemblages. Stabilization of historically variable flow regimes that reduces frequency of overbank flows can generate successional changes leading to community change. In southern Ontario, Toner and Keddy (1997) found evidence that a more stable flow regime led to encroachment of woody vegetation into herbaceous wetlands. In the Platte River of Nebraska, flow stabilization and vegetative colonization of sandbars has caused the river channel to narrow in some places by up to 85% (Johnson, 1994). The narrower channels reduced nesting habitat for piping plover (*Charadrius melanotos*) and the least tern (*Sterna antillarum*) (Sidle et al., 1992) and affected use of the river flats by migrating sandhill cranes (*Grus canadensis*).

Early descriptions of prairie streams prior to extensive land conversion for agriculture and artificial drainage describe sinuous clear brooks flowing through tall overhanging grasses and willows with overhanging sod banks. Streams arose from perennially flowing springs in marshy headwaters and flowed through prairie country alternating with forested areas (Aadland et al., 2005; Meek, 1893; Menzel et al., 1984). In the Midwest, the most significant impacts on stream channels, alluvial valleys, and stream communities occurred during the initial period of land conversion to agriculture during European settlement. Enormous amounts of topsoil were moved into channels and deposited in lower valley floodplains in association with clearing of land for agriculture, such that several meters of sediment were deposited in tributary valleys throughout the Mississippi River Basin. Aquatic communities were dramatically affected, with local extirpations of many populations of stream fishes widely reported (Karr et al., 1985).

Changes in aquatic ecosystems since settlement have been complex in response to cumulative stresses and impacts. In general, large-scale conversion to agriculture accompanied by artificial drainage and channel alteration has increased water and sediment delivery rates and volumes, altering instream habitat through sedimentation, channel entrenchment, reduced water

quality, and loss of bank and instream cover. Separating the direct effects of modified flow regimes from impacts associated with land-use and water quality changes that have accompanied drainage and water quantity management continues to be a challenge for ecological assessment.

Water Quality Effects of Surface and Subsurface Drainage

SEDIMENT LOSSES IN AGRICULTURAL DRAINAGE WATERS

It has long been established that conversion of lands from native, perennial vegetation to croplands generally results in elevated sediment losses (Crosson, 1995). Despite significant improvements in soil management practices, erosion rates on U.S. croplands are still measured in Mg/ha. Many Midwestern prairie-agricultural streams now transport annual sediment loads up to an order of magnitude greater than prior to agricultural conversion (Menzel et al., 1984). Based on analysis of bottom sediments from Lake Pepin on the Mississippi River above lock and dam 3, sediment loads from the Minnesota River, a predominantly agricultural river basin with extensive subsurface drainage, have risen 12-fold over historic levels, and 7-fold from the headwater Mississippi and St. Croix Rivers combined (Engstrom & Almendinger, 1997, Kelley & Nater, 2000). Likewise, Johnson et al. (1980) attributed alluvial sediment deposits up to 16 feet thick for a small stream basin in central Oklahoma to a three-fold increase in peak discharge volume following moderate storm events relative to pre-settlement. Sediment footprints consistently suggest that the most significant sources of eroded sediment in surface waters are cropped floodplains, followed by cropped lands bordering the floodplain (Wilkin & Hebel, 1982).

Subsurface drainage influences sediment in surface waters, both directly through the delivery of sediment loads in drainage effluent and indirectly through hydrologic and geomorphologic influences on sediment dynamics. In general, subsurface drainage reduces sediment loss from agricultural watersheds relative to surface drainage (Bengtson et al., 1995; Skaggs et al., 1994, citing 15 studies; Thomas et al., 1995). However, a slate of recent studies has demonstrated that subsurface drains have the potential to be significant conduits of sediment and associated agrochemicals in a wide variety of environments (Chapman et al., 2005; Kronvang et al., 1997; Laubel et al., 1999). The primary pathway for sediment loss in subsurface drainage waters is discharge of surface runoff to subsurface drainage pipe via surface inlets or preferential flow during high-flow episodic storm events. The use of surface tile inlets to drain closed depressions provides a direct and significant pathway for sediment, solids, and nutrients in agricultural runoff to enter waterways via subsurface drainage systems. Episodic transport of sediment in subsurface drainage could be responsible for a disproportionate share of sediment yield relative to erosion losses, as much sediment reported as

lost from fields does not in any case reach streams, but is stored in upland areas of catchments (Trimble & Crosson, 1987). Sediment that finds its way into subsurface drains—whether via macropore flow or surface tile inlets—is discharged directly to surface waters, bypassing riparian filters and upland sediment sinks. During large runoff or snowmelt events, ponding and back-pressure at tile inlets can result in settling of suspended sediments, reducing some of the total phosphorus (TP) and total solid (TS) load; however, such ponding may simultaneously increase dissolved P (DP) losses (Ginting et al., 2000).

The movement of sediments in channel, streambank, valley floodplains, and bottom sediments has long-term implications for stream habitats throughout a channel network (Magilligan & Stamp, 1987). The spatial distribution of alluvial sediment reflects the influence of watershed size, valley-bottom width, and historical channel incision, depending on how recently channels have been incised (Faulkner, 1998). Sources, sinks, and fluxes vary widely over time and space (Trimble, 1999). Alluvial deposits from periods of high sediment aggradation may require timescales of 100 years to migrate downstream. In river basins of southwestern Wisconsin, for example, geomorphologic investigations have demonstrated that the rate of alluvial sedimentation has greatly decreased since the period of maximum erosion during the 1930s, when conversion to agriculture generated a 3–5 fold increase in annual flood volumes. However, sediment yield has remained fairly constant as streams have redistributed bedload downstream, and most sediment has moved only short distances (Knox, 1977; Trimble, 1999).

Because the zone of impact from sedimentation is typically displaced in space and time from the source, it is often difficult to identify the sources of sediment responsible for observed instream effects. Riparian vegetation may reduce bank erosion at a site to the extent that it dissipates stream power and reduces velocities and sediment loads in surface runoff, and may even cause fine sediments to settle out. A pulse input of fine sediment into a steep channel may be rapidly transported downstream, but persist in a lower-gradient reaches (Montgomery & MacDonald, 2002). Increased sediment loads are typically deposited in wider downstream reaches. Silts and clays tend to be transported further downstream as suspended sediment (Magner & Steffen, 2000).

The link between hydrology and channel geomorphology means that even if field erosion is largely eliminated, increased instream sediment loads from channel and bank erosion processes may still be indirectly traceable to agricultural land use or drainage. Because the transport of sediment by flowing water is a function of both flow velocity and volume, channel incision resulting from increased effective discharge has important implications for stream sediment budgets (Leopold, 1994). Stream bank erosion can supply 50% or more of the sediment load in streams (Kronvang et al., 1997;

Laubel et al., 2003; Lawler et al., 1999). Furthermore, if waters discharging into streams from drainage tiles are relatively sediment-free relative to their sediment transport capacity, they may contribute to additional channel and bank erosion as they re-mobilize sediment (Hubbard, 2005; Leopold, 1994). Downcutting and mass wasting of stream banks are well-known responses to the creation of “sediment-hungry” waters by impoundments. This phenomenon has not been well-studied specifically as it relates to subsurface drainage, and the role of low-sediment water from tile drains in causing downcutting may be difficult to separate from the effect of increased effective discharge.

Aquatic ecosystem effects of agriculturally derived sediment. Sediment is the major nonpoint source pollutant of surface waters in North America both by mass and effect (Waters, 1995; Zaimes et al., 2004). While some sediment in streams is natural, excessive sediment is a pollutant with deleterious effects on aquatic biota (Berkman & Rabeni, 1987; Pimentel et al., 1995; Rabeni & Smale, 1995; Waters, 1995). Soil erosion, besides being a major threat to long-term agricultural productivity, results in sedimentation of aquatic habitats and water quality impairment (Waters, 1995). In addition to phosphorus, eroded sediment in agricultural runoff may carry pesticides, pathogens, heavy metals, and other pollutants. Sediment can eliminate or smother benthic organisms, driving shifts in macroinvertebrate abundance and composition, and alter substrates for macrophytes (aquatic vegetation), degrading habitats for fish spawning, feeding, and cover. Siltation linked to agricultural land use intensity and drainage alterations also threatens freshwater mussels, the most endangered category of aquatic taxa, and is cited as a contributor to declines in 40–50% of the rich molluscan fauna of the southeastern United States (Stansbery, 1971), as well as more recent declines in mussel species richness and distribution in Iowa between 1984 and 1998 (Poole & Downing, 2004). Sedimentation also has significant economic impacts on reservoirs, drinking water, and floodplain lands.

Suspended sediment and turbidity induce physiological stress in many aquatic organisms, reduce water clarity/visibility, and reduce the amount of sunlight available to aquatic biota (Newcombe & Jensen, 1996). Many studies have reported altered spawning behavior, reduced larval survival, decreased foraging efficiency, decreased growth, and reduced diversity of fishes in response to sedimentation and turbidity (Burkhead & Jelks, 2001; Johnston & Shute, 1997; Mion et al., 1998; Newcombe & Jensen, 1996; Page et al., 2005; Pimentel et al., 1995; Sweka & Hartman, 2001; Wichert & Rapport, 1998). The severity of lethal and chronic effects on aquatic organisms varies by species in response to both concentration and duration of exposure (Newcombe & Jensen, 1996; Newcombe & MacDonald, 1991). Although salmonid species (e.g., trout) are sensitive to acute effects of sediment, nonsalmonid species in warm-water stream environments are also vulnerable, particularly after extended periods of exposure (Newcombe & Jensen, 1996).

Although surface runoff and soil erosion receive the bulk of attention as the major cause of instream sediment problems, the role of stream geomorphic change is often under-emphasized. Channel incision and bank slumping may contribute significantly to suspended sediment loads and turbidity, sedimentation of spawning and feeding habitats, and altered temperature regimes that negatively impact fish and aquatic invertebrates (Waters, 1995). Ultimately, the instream ecosystem effects of subsurface drainage depend on upstream conditions that influence the magnitude and direction of hydrologic and sediment effects, in the context of recent land use history.

NUTRIENT LOSSES IN AGRICULTURAL DRAINAGE WATERS

Nutrient enrichment of aquatic ecosystems, primarily with nitrogen (N) and phosphorus (P), is one of the most significant water quality and ecological problems throughout the world. Humans have more than doubled the rate of N fixation in the biosphere (Vitousek et al., 1997), while increased inputs of P eroded from the landscape and carried from human wastewater into the world's rivers have increased global fluxes of P to the oceans almost threefold, from historic levels of ~8 million metric tons per year to current loadings of ~22 million metric tons per year (Howarth et al., 1995).

Agriculture is one of the primary contributors to anthropogenic alteration of natural biogeochemical cycles of N and P. Despite the fact that subsurface and surface drainage differ substantially in both the magnitude and pathways by which they deliver nutrients to surface waters, enrichment of surface waters with both N and P is common wherever intensive agriculture constitutes a significant portion of the landscape (Daniel et al., 1998). Conventional agricultural systems rely on fertilizer inputs to maintain high yields, but a substantial proportion of these inputs is lost to the environment rather than taken up by crops. Fertilizer is the dominant contributor of net anthropogenic inputs of N in most basins draining to the North Atlantic Ocean (Howarth et al., 1996). Anthropogenic deposition rates of oxidized (NO_x) and reduced (NH_y) forms of N are estimated to exceed preindustrial inputs by several fold (Howarth et al., 1996). Approximately 15% of anthropogenic N is exported to the world's rivers in the form of nitrate, the most soluble and mobile form of N (Caraco & Cole, 1999). N loading by riverine input into estuarine systems has increased by 6–50 times from historic conditions to present, whereas the P load has increased 18–180 times (Conley, 2000).

Nitrogen export from the Mississippi River Basin has increased 2- to 7-fold over the last century. Agriculture's contribution to N export in the Mississippi River is about 2–3 kg per hectare. Typical direct losses from agricultural lands are much higher, from 1–50 kg/ha/year in surface runoff (Downing et al., 1999) compared to 2–100 kg/ha/year from subsurface drainage (see Table 1). The largest riverine N fluxes are observed from basins draining agricultural regions dominated by corn-soybeans with extensive subsurface

TABLE 1. Studies reporting nitrogen concentrations and loads in agricultural drainage waters

Citation	Study site	General description/variable	Land use/cropping system	Soil type and slope	Fertilizer	Annual precipitation	Drainage volume	Flow weighted mean N	Annual N loads	Notes and conclusions
Logan et al., 1994	Surface runoff Subsurface runoff	Corn						0.009–2 kg/ha 0.009–76.5 kg/ha		99% nitrate loss during spring 25% of total fertilizer N applied lost
Patni et al., 1996	Tillage effects	Corn, 3 ha	Loamy				Flow was significantly higher ($P < 0.05$) under NT than CT, mainly due to flow difference during the snowmelt and spring periods.	>10 mg/L for NO ₃ -N in more than 90% of the 1010 tile effluent samples analyzed.	N lost as nitrate during the study period represented more than 20% of the amount applied as anhydrous ammonia.	Most loss occurred during the spring period when tile flow was highest. Over the 40-month study period, NO ₃ -N loss in tile effluent was not significantly different for the two treatments. It was significantly ($P < 0.05$) higher under NT than CT only in one crop-year due to significantly higher flow under NT. There was no tillage treatment effect on tile effluent pH and specific conductivity.
Jaynes et al., 1998	Illinois	5130 ha watershed, IL	86% row crop				>10 mg/L during May–July	4–66 kg/ha/yr, 6–115% of N applied as fertilizer		(Continued on next page)

TABLE 1. Studies reporting nitrogen concentrations and loads in agricultural drainage waters (*Continued*)

Citation	Study site	General description/variable	Land use/cropping system	Soil type and slope	Fertilizer	Annual precipitation	Drainage volume	Flow weighted mean N	Annual N loads	Notes and conclusions
Borah et al., 2003	Illinois	Effect of stormflows	Extensively tile drained watershed			Two-season spring runoff	Reported in relation to flow	Average 31 kg/ha Range 16–54 kg/ha	Nitrate-N showed inverse relationships, whereas phosphate P, atrazine, and metolachlor had positive relationships with discharge; [1]	highly correlated with peak of the hydrograph
Cambardella et al., 1999	Illinois	Walnut Creek, 5130 ha watershed, 80% cropland	Tile drained corn and corn-soybean	Clarion soil, Okoboji and Canisteo soil.			>10 mg/L for 12 months between 6 and 9 mg/L for 32 months over four years	1993: 51.3 kg/ha 1994: 4.9 kg/ha	*Denitrification rates 2x higher in Okoboji surface soil than in Clarion surface soil	*Highest denitrification potential was observed for deep unoxidized loess.

*Soil profile NO₃-N concentrations decreased with depth and were the same below 30 cm for fertilized corn and soybean

David et al., 1997	Illinois	Embarra River watershed at Camargo 48173 ha	Tile drained corn and soybean	Clay loam	1995–2000	5–49 mg/L N in 4 monitored drainage tiles; synchronous with Embarras River NO_3^- concentrations.	24.7 kg/ha/yr N 44.2 kg/ha N	High flow events contributed most of the yearly NO_3^- loss (1995; water year where high rainfall events occurred in a low overall precipitation year; in one tile, 21% of the annual load was exported in 1 d). On average, about 49% (average of 1688 Mg N yr) over a 6-yr period of the field inorganic N pool was estimated to be leached through drain tiles and seepage and was exported by the Embarras River Tile export was greatest during high flow events
Gentry et al., 1998	Illinois	Embarra River watershed at Camargo 48173 ha	Tile drained corn and soybean	Silty clay loam, fine-silty, mixed	Typic Haplaqueolls	Applied rate of: 107 kg/ha/yr N 93 kg/ha/yr	16.8 mg/L 10.2 mg/L	River concentrations ranged from a max 10–15 mg/L (spring) to 0–5 mg/L (fall) 14 to 35 kg/ha/yr (representing 14–36% of the nitrogen applied). Losses from the grassed system were 3.8 kg/ha/yr and at the most upstream river station were 11 kg/ha/yr.
Mitchell et al., 2000	Illinois	6 yr study, 3–21 ha plots, different cropping and fertilizer treatments; streams draining watersheds from 6910 to 48 900 ha	Intensive row-crops					(Continued on next page)

TABLE 1. Studies reporting nitrogen concentrations and loads in agricultural drainage waters (*Continued*)

Citation	Study site	General description/ variable	Land use/ cropping system	Soil type and slope	Fertilizer	Annual precipitation	Drainage volume	Flow weighted mean N	Annual N loads	Notes and conclusions
Rladivko et al., 2004	Indiana	Drain spacing (5, 10, and 20 m)	No-till corn soybean rotation	200-300 kg/ha	954 mm/yr rain	1022 mm/yr rain	1986-1988 28 mg/L 1997-1999: 8 mg/L	38 kg/ha 15 kg/ha	Drainflow removed between 8 and 26% of annual rainfall, depending on year and drain spacing.	
Baker and Johnson, 1981	Iowa	Effect of fertilizer	Corn-soybean-corn-oats rotation	N applied to corn: 95 kg/ha vs. 245 kg/ha			20.1 mg/L vs. 40.5 mg/L		Nitrate N concentrations in drainflow did not vary with spacing. Both phases of work showed increasing N NO ₃ -N concentrations with increasing N application rates. There was a lag of about 1 month and 10 cm of flow before differential fertilization affected NO ₃ -N concentrations in tile drainage.	

Baker and Melvin, 1994	Iowa	Effect of fertilizer	Continuous corn, soybean-corn CRP	Economic optimum N rate: ~200 kg/ha (CC) 150 kg/ha (rotation com).	20–40 mg/L NO ₃ , 13–38 mg/L 1–4 ng/L 0.3–4 mg/L	Corn yields increased with N fertilizer rate
Bakhsh et al., 2002	Iowa	Effect of crop: Corn N appl. Rate & method:	Corn soybean Single App late Spring	Loamy 110 kg/ha N 159 kg/ha N 179 kg/ha N	159 mm 178 mm 186 mm 151 mm 145 mm 190 mm	17 kg/ha 15 kg/ha 18 kg/ha 15 kg/ha 15 kg/ha 18 kg/ha
Kanwar et al., 2005	Iowa	Tillage method: Tillage method: No till Tile-drained research site	Chisel plow Corn-soybean Alternative crop system	Loamy 680–1030 mm/yr	0.88 correlation with rainfall 11.2 mg/L N6.5 ng/L N 0.99	13.5 kg/ha N12.6 kg/ha N correlation between drainflow
Weed and Kanwar, 1996	Iowa	1990–1992 effect of crop rotation	Continuous corn Corn-soybean Rotation corn Rotation soybean	195 mm 187 mm	NO ₃ N loss 10–64 mg/L 3–39 ng/L 3–51 kg/ha	13–107 kg/ha 3–51 kg/ha

(Continued on next page)

TABLE 1. Studies reporting nitrogen concentrations and loads in agricultural drainage waters (*Continued*)

Citation	Study site	General description/variable	Land use/cropping system	Soil type and slope	Fertilizer	Annual precipitation	Drainage volume	Flow weighted mean N	Annual N loads	Notes and conclusions
Jaynes et al., 2001	Iowa, central	A 400-ha, tile-drained subbasin	Corn soybean rotation with spring fertilizer app.	Loamy	Three rates: High: 202 kg/ha (1996) 172 kg/ha (1998) Medium: 135 kg/ha (1996) 114 kg/ha (1998) Low: 67 kg/ha (1996) 57 kg/ha (1998)			Exceeded MCL all years, all crops Exceeded MCL years, all crops Exceeded MCL (1029 kg/ha) only in com years	48 kg/ha 35 kg/ha 29 kg/ha	The economically optimum N fertilizer rate for corn was between 67 and 135 kg ha ⁻¹ in 1996 and 114 and 172 kg ha ⁻¹ in 1998
Jaynes et al., 2004	Iowa, central	A 400-ha, tile-drained subbasin in central Iowa	Corn, tilled	A commerce silt loam (fine-silty, mixed, applied, non-acid, thermic, Aeric Fluvaquents).	224 kg N ha ⁻¹ yr (recommended)	Normal (often exceeds 1600 mm) drought (three-way split) before June 10 of each year; 25% at planting, 50% at 30 d after emergence, and the remaining 25% in early June.	255–422 mm (shallow) 270–463 (deep) 40–78 mm (shallow) 38–72 mm (Deep)	11.3 mg N L ⁻¹ (LSNT) 16.0 mg N L ⁻¹ (control)	8.6 kg/ha (surface) 5.7 kg/ha (deep controlled) 2.3 kg/ha (surface) 1.4 kg/ha (deep controlled)	
Grigg et al., 2003	Louisiana Surface drainage only vs. deep controlled drainage (1.1 m)	Surface	Corn, tilled							

Chung et al., 2001	MN, Lam-berton	Observed vs. simulated NO ₃ and flow	Continuous corn Soybean-corn Continuous alfalfa	Clay loam	Long-term average of 211 mm 560–790 105 mm mm	51 kg/ha 45 kg/ha 2 kg/ha
Gast et al., 1978	MN, Lam-berton	1990–1993 Fertilizer rate effects	Continuous corn plots	Applied N at 20, 112, 224, and 448 kg/ha/yr for 3 yr	1st year no difference; second year, by rate, 19, 25, 37, and 65 mg/L; third year were 19, 23, 43, and 81 mg/L	Buildup of NO ₃ -N in the 0 to 3.0-m soil profile for the two highest rates, with 425 and 770 kg/ha present for the 224 and 448 kg/ha fertilization rates,
Strock et al., 2004	MN, Lam-berton	Cropping system	Corn Corn Soybean Corn and rye Rye and soybean	9.4–185 mm 0.25–506 mm 0.10–428 mm 0.30–463 mm	11–13 mg/L 10–19 mg/L 2–14 mg/L 1–12 mg/L	1.2–46 kg/ha 0.04–54 kg/ha 0–46 kg/ha 0–53 kg/ha
Randall and Iragavarapu, 1995	MN, Waseca	Tillage effects	Moldboard plow vs. no-till corn	279 mm 315 mm	15 mg/L 13 mg/L	43 kg/ha N 41 kg/ha N
Randall and Mulla, 2001	MN, Waseca	Fertilizer rate effects	Continuous corn	No fertilizer Spring app: 134 kg/ha 202 kg/ha N Fall applied 134 kg/ha 202 kg/ha	8 kg/ha/yr 21 kg/ha/yr 29 kg/ha/yr 30 kg/ha/yr 38 kg/ha/yr	

(Continued on next page)

TABLE 1. Studies reporting nitrogen concentrations and loads in agricultural drainage waters (*Continued*)

Citation	Study site	General description/variable	Land use/cropping system	Soil type and slope	Fertilizer	Annual precipitation	Drainage volume	Flow weighted mean N	Annual N loads	Notes and Conclusions
Randall et al., 1997	MN, Waseca	Cropping system	CRP Alfalfa Continuous corn Corn-soybeans			Long-term 560–790 mm	0–510 mm 0–320 mm 18–488 mm 20–442 mm	2 mg/L 3 mg/L 24 mg/L 32 mg/L	1.0 lbs/acre N 1.6 lbs/acre N 48.5 lbs/acre N 45.2 lbs/acre N	
Burchell et al., 2005	NC	Shallow drains (0.75 m) Deep drains (1.5 m)	2.5 ha beef cattle pasture	Cape Fear loam, poorly drained with slow permeability and high water table (0–30 cm)		Average 8–16 mg/L	8–27 kg/ha (shallow) kg/ha (deep)	(shallow) 6–37 kg/ha (deep)	Drought year	
Lowrance et al., 1984	NC	Streamflow, drainflow	Mixed cover, 1568 ag watershed		Fertilizer plus N in precipitation of 12 kg/ha/yr	Concentrations of NO ₃ -N from drained row-crop fields	2–13 mg/L (shallow) 1.7–2.3 kg/ha (deep)	kg/ha (deep)	All 10 streamflow sampling sites had maximum NO ₃ -N and SO ₄ -S concentrations in winter and most reached minima during summer. Loads of organic N, Ca, Mg, K, SO ₄ -S, and Cl were from the mixed cover watershed.	All 10 streamflow sampling sites had maximum NO ₃ -N and SO ₄ -S concentrations in winter and most reached minima during summer. Loads of organic N, Ca, Mg, K, SO ₄ -S, and Cl were from the mixed cover watershed.

Milburn et al., 1990	New Brunswick	Potato, 10 plots 3–5 ha	Variet soils	Intensive input, 100–200 kg/ha N applied	Variet	5–65 mg/L N	5–70 kg/ha
Drury et al., 1996	Ontario, SW	Subsurface Controlled Surface	16 plots of ryegrass intercrop	Brookston clay loam	677–695 mm 503–633 mm 130–208 mm 265–384 mm	9.8–11.4 N mg/L 7.0–8.7 N mg/L 1.92–3.11 N mg/L 1.62–2.43 N mg/L	Different tillage and drainage treatments
Ng et al., 2002	Ontario, SW	Controlled drainage effects	Corn	Brookston sandy clay loam, 0.05–0.10% slope	650 mm 861 mm (42 yr avg) controlled drainage	11.3 mg/L 19.2 mg/L	Corn from the CDS treatment had 50% greater transpiration rates (47.4 mgm ⁻² s ⁻¹) than the FD treatment (31.7 mgm ⁻² s ⁻¹), 64% average yield increase for corn in CDS and 11% increase water use efficiency
Tan et al., 2002	Ontario, SW	Cropping system	Cont. corn (CC) CBG Rotation Corn (RC) RO Rotation alfalfa (1st yr) RA1 Rotation alfalfa (2nd year) RA2	Brookston sandy clay loam, 0.05–0.10% slope	112 kg/ha NH4 for corn + 16.8 kg/ha fertilizer applied N	650 mm 861 mm (42 yr avg)	15.2 mg/L N 18.0 mg/L N 16.2 mgN ⁻¹ (CO) 99.9 kgNha ⁻¹ (RC) 69.8 kgNha ⁻¹ (RA2)
Madramootoo et al., 1992	Quebec	Potato 5 ha fields	St. Jude sandy loam soil		2–40 mg/L N	14–70 kg/ha	Apr Nov

drainage (e.g., southern Minnesota, Iowa, Illinois, Indiana, and Ohio) (McIsaac & Hu, 2004). Proximity of agricultural basins to large river main-stem influences riverine export, as smaller headwater streams have a higher capacity for instream denitrification than larger rivers and lakes (Alexander et al., 2000). Other important factors influencing riverine export of N include the timing, amount, and annual variability of precipitation. The elimination of nutrient and sediment sinks across much of the landscape that has occurred historically through removal of wetlands, riparian buffers, floodplains, and other depressional areas has also played a role.

Artificial subsurface drainage is the major pathway for nitrate loss from subsurface-drained agricultural lands. All else being equal, installation of subsurface drainage results in greater leaching of nitrate from the soil profile than prior to drainage by shifting the major pathway for excess precipitation (i.e., water not retained by soils or evapotranspired) from surface runoff to subsurface flow. An additional factor contributing to elevated nitrate yields from subsurface-drained lands relative to surface drainage only is that, unlike surface runoff, subsurface drainage waters are often discharged directly to surface waters, bypassing the zones where most denitrification occurs (i.e., at the soil-water boundary primarily in saturated, riparian and wetland soils; see Baker, 2001; Dinges et al., 2002).

Nitrate losses in subsurface drainage flows vary substantially in response to cropping systems, tillage practices, fertilizer application rate and timing, annual rainfall, and other factors. Many studies have shown that small grains (e.g., wheat) and perennial cropping systems such as alfalfa and pasture generally exhibit lower nitrate losses than row crops such as corn and soybeans (Baker & Melvin, 1994; Chung et al., 2001; Lowrance et al., 1984; Randall et al., 1997; Strock et al., 2004; Tan et al., 2002). Economic yields of corn and other row crops generally require nitrate levels in soil water in excess of the amount that can be taken up by the crop (Tan et al., 2002). Studies generally confirm elevated losses with increasing N application rates, and corresponding reductions with reduced application (Gast et al., 1978; Grigg et al., 2003; Hall & Risser, 1993; Jaynes et al., 2001, 2004; Mitchell et al., 2000; Randall & Mulla, 2001). However, high losses of N can occur wherever there is subsurface drainage flow under organic, hydric soils, including fallow drained soils with no N inputs (Randall, 1998). The highest losses tend to occur when a wet spring follows a series of dry years (Randall, 1998, 2004).

Increasingly, studies of drainage suggest that these differences in NO_3^- losses in subsurface flow under varying treatments are primarily driven by the effect of treatments on subsurface flow volumes. Observed reductions in nitrate loads under alfalfa, CRP, or other perennial crops are closely tied to the influence on total drainflow volumes due to differences in crop water use (Bahksh et al., 2002; Chung et al., 2001; Drury et al., 1996; Kanwar et al., 2005; Randall et al., 1997). Likewise, studies examining the effects of

reduced subsurface drain depth or increased drain spacing on nitrate loss in subsurface drainage have attributed the majority of observed effects to differences in subsurface drain flow volumes (Sands et al., 2003; Tan et al., 1998). Bakhsh et al. (2002) and Kanwar et al. (2005) have reported strong linear relationships between annual $\text{NO}_3\text{-N}$ leaching loss in subsurface drainage water with annual subsurface drainage flow volume across a range of cropping systems.

Because subsurface drainage can reduce the amount of water lost as surface runoff, subsurface drainage has been proposed as a strategy for reducing non-point source pollution in areas where sediment and phosphorus are the major concerns (Fausey et al., 2002; Gilliam & Skaggs, 1986; Loudon et al., 1986; Southwick et al., 1990). Sims et al. (1998) reviewed more than 21 studies that summarized P concentrations and export in subsurface drainage waters. Most studies supported the conclusion that losses of phosphorus and sediment are typically lower in subsurface drainage than surface runoff at the field scale. Haygarth and Jarvis (1996) estimated up to a 30% reduction in TP loss after installing subsurface drainage on grazed grassland plots. However, significant P export in either dissolved or particulate forms occurs via subsurface drainage under conditions associated with leaching or elevated sediment/runoff delivery to subsurface drains (see Table 2), and can be as important or more important than surface transport in areas dominated by subsurface drainage (Dils & Heathwaite, 1999; Djodjic et al., 2002; Gachter et al., 1998; Hergert et al., 1981; Laubel et al., 1999; Motoshita et al., 2003; Shirmohammadi et al., 1998; Simard et al., 2000; Stamm et al., 1998; Uusitaloa et al., 2001; Xue et al., 1998). High leaching rates of dissolved P via subsurface drainage are most common in areas characterized by deep sandy soils, high organic matter soils, and soils with very high soil P concentrations (Chapman et al., 2003; Haygarth et al., 1998; Heckrath et al., 1995; Kuo & Baker, 1982; Ryden et al., 1973; Simard et al., 2000; Sims et al., 1998; Skaggs et al., 1994, citing 5 studies).

The intensification of agriculture has led to on-farm accumulation of P imported in fertilizer, animal manures, and animal feed compared with P exported in farm produce (Carpenter et al., 1998; Sims et al., 1998). Many farms now have soil P concentrations that greatly exceed crop and pasture needs (Klatt et al., 2003; Sharpley et al., 2003). Dissolved forms of P have been found to constitute the dominant form of P loss through time in both surface and subsurface drainflow under these conditions (Beauchemin et al., 1996; Gächter et al., 1998; Haygarth & Jarvis, 1998; McDowell & Sharpley, 2001; Motoshita et al., 2003).

Significant mass losses to subsurface drainage waters can also occur via macropores in the soil profile that allow runoff to discharge directly into subsurface drains, a phenomenon termed "preferential flow" (Dils & Heathwaite, 1999; Sims et al., 1998). Macropores may be formed by worms

TABLE 2. Studies reporting phosphorus concentrations and loads in agricultural drainage waters

Study	Study site/drainage	Land use/practices	P export (form)	Notes and conclusions
Baker and Richards, 2002	Illinois Maumee watershed Sandusky watershed	Extensively tile drained corn-soybeans	1.35 kg/ha TP 1.40 kg/ha TP	High export rates relative to most large Midwestern watersheds.
David and Gentry, 2000	Illinois, 15 watersheds average annual water export rates for 1980–1997	Extensively tile drained-6 major watersheds-area weighted average for the state	0.7 to 1.1 kg P ha ⁻¹ yr ⁻¹ 1.0 kg P ha ⁻¹ yr ⁻¹	38% of the TP was in the dissolved form in Illinois rivers
McIsaac et al., 2002	Illinois, east-central	Extensively tile drained	Flow-weighted mean DMRP ranged from –0.05–0.13 mg-P/L in tile effluent –0.12 mg-P/L in the river	There was little apparent relation between Bray P-1 concentration in the surface layers of soil and the concentrations in the tile drain effluent
Xue et al., 1998	Illinois, east-central, 48000 ha watershed	Extensively tile drained corn-soybean	DP export model estimate equation: 3.94×10^6 mg/L/ha × surface water discharge × watershed area	Tiles contribute more than 86% of river flow and 65–69% of river DP export
McDowell and Sharpley, 2001	UK: 9.5 ha catchment, silt loam soils USA: 39.5 ha catchment in Pennsylvania, loam and silt loams	Arable (wheat) and grassland under long-term fertilization Soybeans under long-term fertilization	Concentrations in lysimeters ranging from 0.1–2.5 and 0–1 mg/L Average in surface runoff ranging from 0.1–1 and 0.1–0.3 mg/L	For different soil types, there exists a change point at a certain level of soil test phosphorus (STP) at which the slope of the linear relationship between STP and dissolved reactive phosphorus (DRP) or extractable P increases much more steeply above than below. Because this change point is generally much higher than that required for optimum crop growth, P loss could be reduced if P management were based on STP.

Klatt et al., 2003	Northern Iowa Loam or silty-clay-loam formed on glacial till. Small areas have slopes up to 14 to 18%. Areas with $\leq 2\%$ slope are artificially drained with subsurface tiles and surface inlets	Lake watershed 59% ag, 27% wetlands Applied P (mainly fertilizer) averaged 15 P kg ha $^{-1}$ yr $^{-1}$, and 40% of the high-testing area (M3P test) was being fertilized.	The mean annual water TP concentration across five basins was 275–474 $\mu\text{g L}^{-1}$. The 2-year mean TP loads for the two gauged basins were 1.50 and 1.51 kg P ha $^{-1}$ yr $^{-1}$.	Water TP concentration increased linearly with increasing STP. Relationships were stronger for M3P and FEP ($R^2 = 0.96$ –0.97 for annual means and 0.77–0.79 for storm-flow events) than for BP or WP ($R^2 = 0.88$ –0.91 and 0.59–0.69, respectively).
Lowrance et al., 1984	GA, tile-drained Agricultural land	Sediment associated P load was 0.09 kg ha $^{-1}$ yr $^{-1}$, accounting for 8.5% of TP	Improving P and soil conservation practices in areas with high test P could reduce P loads to the lake. The two highest flow events for the year carried 19% of the total annual flow and sediment load, but had 30 and 27% of the total annual sediment N and P loads, respectively. This enrichment of sediment-associated N and P indicated that part of this sediment load moved directly from upland areas.	Water TP concentration increased linearly with increasing STP. Relationships were stronger for M3P and FEP ($R^2 = 0.96$ –0.97 for annual means and 0.77–0.79 for storm-flow events) than for BP or WP ($R^2 = 0.88$ –0.91 and 0.59–0.69, respectively).
Berney et al., 1983	Ontario, drained clay soils	Arable fields (permanent bluegrass sod, continuous corn, and rotation corn)	0.88 kg ha TP from corn plots 1.95 kg TP from permanent sod	> 50% of the TP load from level plots lost via SS tile flow. 34% of TP load from subsurface drains was sediment-associated. Crop cover, P fertilization rate, and drain depth significantly affected DP export. DP concentrations were greatest under permanent bluegrass sod, whereas subsurface sediment and sediment-associated P loads were highest from continuous corn.

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TABLE 2. Studies reporting phosphorus concentrations and loads in agricultural drainage waters (*Continued*)

Study	Study site/drainage	Land use/practices	P export (form)	Notes and conclusions
Culley and Bolton, 1983	Ontario, drained clay soils	Arable fields (row crops and cereals)	0.36 kg ha ⁻¹ yr ⁻¹ PP	At least 25% of the TP and 50% of the ortho-phosphorus loads to surface water originated in subsurface drainage
Madramootoo et al., 1992	Quebec, Canada, St. Leonard d'Aston; two tile-drained Sandy loam soils	Two ~5 ha potato fields	0.0002–0.52 mg/L P or 0.02 kg/ha Measured in tile drain flow over two growing seasons (April to November).	
Grant et al., 1996	Denmark	Tile drain output from arable fields	0.04–0.18 kg ha ⁻¹ yr ⁻¹ PP	
Kronvang et al., 1997	Denmark		0.08–0.12 kg ha ⁻¹ yr ⁻¹ PP	
Dils and Heathwaite, 1999	UK: Piestow Hill catchment (120 ha) moderately permeable stony clay loams and glacial till	Grassland/pasture	Mean in drainage water of 0.03–0.96 mg/L P, > 1 mg TP/L during high discharge	Large sediment associated particulate P losses in autumn. Phosphorus concentrations in drain discharge were low (<100 µg/L TP) and stable during base-flow periods (<0.5 µg/L), and generally lower than in the receiving stream.
Foster et al., 2003; Chapman et al., 2005	U.K.: Rosemaund catchment (1.5 km ²): Extensively drained, light to medium loam soils Pister Hill Catchment (90 ha): Wider variety of soils, seasonally waterlogged	Mixed arable in the uplands with grazing in the lower slopes	1.05–1.09 kg ha ⁻¹ yr ⁻¹ PP 0.04–0.95 kg ha ⁻¹ yr ⁻¹ PP 0.51–0.91 kg ha ⁻¹ yr ⁻¹ PP 1.91 kg ha ⁻¹ yr ⁻¹ PP average	Subsurface flow accounted for 60% of PP loss, 85% of suspended sediment (SS) lost in drains was from topsoil. A single tile drain delivered 60% of TP load, attributed to suspended particles through macropores High loads in drain flows attributed to soil characteristics that have high cracking potential during dry periods

Robinson, 1990	U.K.: Rosemaund catchment (1.5 km^2), U.K. (light to medium loam soils)	Extensively drained, mixed arable in the uplands with grazing downstream	1.65 kg ha $^{-1}$ yr $^{-1}$ PP
Haygarth and Jarvis, 1996	UK, southwest Monoliths of clay loam, sandy loam, and sandy soils	Grazed grasslands	1.77 TP in subsurface drains (1.27–5.03 kg/ha yr P in grass and grass clover) 0.38 TP in surface runoff
Haygarth et al., 1998	UK, southwest Monoliths of clay loam, sandy loam, and sandy soils	Grazed grasslands (i) surface plus interflow to 30 cm (undrained) (ii) surface plus interflow to 30 cm on a mole and tile drained soil (iii) mole and tile drains to 85 cm	232 $\mu\text{g l}^{-1}$ TP in the surface 152 $\mu\text{g l}^{-1}$ TP in the surface plus interflow to 30 cm 132 $\mu\text{g l}^{-1}$ TP in subsurface flow to 85 cm
Hooda et al., 1999	Scotland Silty clay loam soils subject to macropore formation	Grass-clover Vs. Grass For 9 yrs; 2–3 cattle slurry applications annually Cut 2–3 times before being grazed by dairy cattle. Mineral fertilizer-P applied to the grass-clover (about 25 kg/ha·yr)	0.5–0.58 kg ha $^{-1}$ yr $^{-1}$ PP 1.68–2.03 MRP and 3.47–5.03 TP kg ha $^{-1}$ yr $^{-1}$ 1.27–1.34 MRP and 2.97–3.58 TP kg ha $^{-1}$ yr $^{-1}$ respectively). 0.16–0.38 mg/L MRP and 0.45–0.79 mg/L TP MRP accounted for 42% of the TP loss and non-MRP form for 41% of the TP loss through field drains. Particulate P represented about 17% of the TP loss.

Extracts from the 0–2 cm layer were 10× greater than below 45 cm, showing substantial enrichment of the Olsen-P extracts from the surface horizons. In all paths, the dissolved P comprised the greatest proportion of the P transferred, with DRP being the dominant form.

Draining land reduced the transfer of TP by about 30% ($\approx 1 \text{ kg l}^{-1} \text{ ha}^{-1} \text{ year}^{-1}$), because it can be sorbed as it flows through soil to drains.

After 9 yr, NaHCO₃ soluble P in the topsoil (0–10 cm) averaged 38 and 47 mg P kg $^{-1}$ for the grass and grass-clover respectively, giving an average increase of 1.0 mg Olsen-P kg $^{-1}$ yr $^{-1}$ in the grass-clover. P losses in subsurface runoff measured in the present study were much larger than previous estimates. Despite the subsoil having much larger P-sorption capacity than the topsoil, significant amounts of P losses could occur through preferential hydrological pathways.

(Continued on next page)

TABLE 2. Studies reporting phosphorus concentrations and loads in agricultural drainage waters (*Continued*)

Study	Study site/Drainage	Land use/practices	P Export (form)	Notes and Conclusions
Ullen & Persson, 1999	Central Sweden clay soils	Small grains	0.02–0.09 kg ha ⁻¹ yr ⁻¹ PP 0.29 kg ha ⁻¹ yr ⁻¹ TP 63% was in particulate form (PP) while the remainder was either in phosphate form (PO ₄ -P) or in other dissolved or colloidal forms.	< 1% of the soil surface contributed via macropores evenly distributed sampled areas. Transport from the field was mainly episodic. 50% of annual P transport occurred in an average of 140 hours. Compared with flow-proportional and frequent sampling, manual and fortnightly sampling underestimated the transport of Tot-P and suspended solids (SS) by 59 and 42%, respectively.
Uusitalo et al., 2001	SW Finland clayey soils	Wheat and barley	47–79 mg kg sediment PPi 45–82 mg/kg Ppi Concentrations were comparable in SSD to surface runoff	47% of potentially bioavailable P (AER-P) consisted of PPi. Concluded desorbable PP derived from topsoil may be as important a contributor to potentially algal-available P as DRP in both surface and subsurface runoff from clayey soils.
Stamm et al., 1998	Switzerland	Intensively managed grassland	0.23 kg ha ⁻¹ yr ⁻¹ SRP at site I 1.129 kg ha ⁻¹ yr ⁻¹ SRP at site II	P strongly increase with increasing flow rates, indicating preferential flow paths

Gächter et al., 1998	Switzerland on a loam formed by glacial till	78% ag, intensive dairy and pigs on grassland; 45% artificially drained; Manure applied at average of 47 kg/ha P	1.15 kg ha ⁻¹ yr ⁻¹ SRP SRP concentrations of water percolating out of the topsoil were in the range of 400–4000 mg m ⁻³ and agreed well with the range of concentrations measured in drainage systems during elevated water discharge. Extrapolating to the entire watershed, drainage effluents may contribute 58–100% to the total SRP load (793 kg/yr) of the river.	Soil macropores and artificial drainage systems were most important pathways for the vertical and lateral transport of SRP from P-enriched soil surfaces to surface waters Counter to conventional wisdom, leaching and not surface runoff was the most important mechanism for P transfer from soils to surface waters in the investigated watershed.
Turtola and Paajanen, 1995	Finland on drained clay soil	Before vs. after improvement of soil with subsurface drainage	1408 kg/ha sediment vs. 1168 after 0.69 kg/ha PP vs. 0.58 after 0.12 kg/ha DP vs 0.09 after	Where wood chips were used as backfill, soil erosion and particulate P losses were not reduced. N leaching during barley cultivation was much higher after IMP (14 vs. 7 kg ha ⁻¹ a ⁻¹) due to increased drain discharge

or other burrowing animals, or by soil shrinkage or “cracking,” as is typical of clay soils under dry conditions. Where conditions give rise to significant preferential flow, mass losses of sediment, particulate P, pesticides, and other contaminants in subsurface drainage can be comparable to losses in surface runoff (Chapman et al., 2003; Haria et al., 1994; Heathwaite & Dils, 2000; Hooda et al., 1999; Laubel et al., 1999; Simard et al., 2000). Contaminant transport via macropore flow is most significant during storms, and may shift the dominant form and pathway for P loss to sediment-associated P in drainflow during periods when storm flow events dominate total flows (Grant et al., 1996; Kronvang et al., 1997; Laubel et al., 1999; Ulen et al., 1998). Mass losses in drainflows are driven by high rainfall intensity that accelerates breakthrough time for macropore formation (Heppell et al., 2002).

Many on-farm agricultural practices influence the effects of subsurface drainage on hydrology, nutrient, and sediment losses. Important variables include tillage systems as well as rate and timing of fertilizer and chemical applications. For example, implementing no-till or reduced tillage on drained cropland, or using subsurface drainage with untilled pasture, may increase the likelihood of preferential flow to subsurface drains, increasing potential contaminant delivery to surface waters (Granovsky et al., 1993; Moroizumi & Horino, 2004). Furthermore, although no-till and reduced tillage may decrease runoff through effects on crop ET and increased soil water infiltration and holding capacity, increased subsurface flow also elevates nitrate losses.

Adding subsurface drainage to cropped lands already drained with surface drainage has the potential to generate water quality benefits where sediment in surface runoff and turbidity are the major source of water quality impairments, and wherever an improved capacity of soils to absorb precipitation events leads to greater infiltration and reduced rapid overland flow. The overall ecosystem benefits arising from reductions in surface runoff with subsurface drainage are, however, uncertain, given the corresponding increase in nitrate loads associated with rapid subsurface flow. Furthermore, in many cases, the bulk of sediment and P contribution to surface water loading occurs episodically (during events that may overwhelm the capacity of drainage systems) and from a small proportion of the landscape (Kronvang et al., 1997). In such cases, any change in loading due to subsurface drainage would have a relatively small effect (e.g., 60% reduction of 20% of the load translates to a 12% reduction). Finally, in many P-impaired watersheds, lag times in response to long-term accumulation of P in watershed sediments will overwhelm short-term improvements in P management (McDowell & Sharpley, 2001; McDowell et al., 2004).

The application of anhydrous ammonia and liquid manure as fertilizer has additional implications for water quality and aquatic ecosystems. Application of liquid animal manures to subsurface drained land, and the subsequent transport of wastes into surface water systems via subsurface drainage

waters, has been identified as a major transport pathway for pathogens including *E. coli*, *Salmonella*, *Campylobacter*, and *Shigella* (Jamieson et al., 2002). Storm events that lead to high subsurface drain flows following application of anhydrous ammonia fertilizer or liquid manure to drained fields, particularly via preferential flow, may infrequently contribute high levels of pathogens and/or ammonia to streams and surface waters at levels high enough to cause toxicity to fish and aquatic organisms or to impair surface waters for human uses (Dean & Foran, 1990; Fleming & Bradshaw, 1992; Geohring et al., 2001; Jamieson et al., 2002; McLellan et al., 1993). Under current manure application practices, leachate reaching subsurface tile drains frequently exceeds drinking water supply and recreational use standards (Geohring et al., 1998; Warnemuende & Kanwar, 2000). Ammonia is toxic to aquatic life at elevated concentrations, but the relevant criteria values vary by stream pH and temperature (U.S. EPA, 1999). Breakdown of ammonia and organic wastes also creates biochemical oxygen demand (BOD) in streams, rivers, and lakes that can reduce levels of dissolved oxygen (DO) below those necessary to support a full range of aquatic life.

Aquatic ecosystem response to nutrient loading. The effects of nutrient enrichment on aquatic ecosystems has been the subject of decades of research and environmental monitoring. An expanding body of work focuses on assessing the implications of changes in nutrient budgets for regional and global ecosystems (Bennett et al., 2001; Hecky & Kilham, 1988; Mallin et al., 2001; Nixon, 1995; Schindler, 1978; Smetacek et al., 1991; Vitousek et al., 1997; Vollenweider et al., 1992).

In both marine and freshwater ecosystems, nutrients regulate the biomass and species composition of algal and diatom assemblages that form the basis of the diet for macroinvertebrates (Blomqvist et al., 2004; Hagarthey et al., 2002; Shieh et al., 2002). Major changes in nutrient levels and ratios primarily affect ecosystem functioning by altering food webs. Eutrophication (i.e., the complex chain of effects stimulated by nutrient enrichment) is often cited as the most widespread water quality problem in both freshwater (Bennett et al., 2001; Carpenter et al., 1998; Moss et al., 1989; NRC, 2000; Smith et al., 1999) and coastal/marine ecosystems (Nixon, 1995; Rabalais, 2002; Seitzinger et al., 2002). The most observable effect of anthropogenic eutrophication is the decline of water suitability for human uses due to the increased growth of undesirable (sometimes even toxic) algae and aquatic weeds, and hypoxia (low oxygen concentration) caused by the decomposition of plant matter stimulated by the increase in available organic carbon. Dense algal mats can also reduce light penetration, reducing habitat quality for macroinvertebrates, periphyton, and fish spawning and rearing (Correll, 1998). Toxic algal blooms contribute to fish kills, contamination of drinking water, and the formation of carcinogens during water chlorination, and have been tentatively linked to neurological impairment in humans (Burkholder & Glasgow, 1997; Kotak et al., 1993).

Recent reviews have summarized the effects of increased inputs of nitrogen (Rabalais, 2002) and phosphorus (Bennett et al., 2001; Smith et al., 1999) on aquatic ecosystems. P is the primary limiting nutrient in most streams, lakes, and reservoirs (Hecky & Kilham, 1988; Schindler, 1977; Vollenweider, 1968), and therefore plays the dominant role in eutrophication of surface, estuarine, and some coastal waters (Correll, 1998; Daniel et al., 1998; Sharpley & Menzel, 1987; Sharpley et al., 1994; Sims et al., 1998, Smith et al., 1999). Likewise, excess nitrogen is the primary nutrient threatening coastal fisheries and ecosystems around the world (Doering et al., 1999; Goolsby et al., 1999; Howarth & Marino, 2006; Mitsch et al., 1999, Rabalais, 2002), although it can also play a role in freshwater eutrophication (Dodds & Welch, 2000). Although the phenomenon of hypoxia in the Gulf of Mexico is perhaps the best known example, excess N has repeatedly been identified as the element controlling algal blooms and generating water quality problems in estuaries along the U.S. Atlantic coast as well (Doering et al., 1999), where agriculture also contributes the major portion of total N loading (Skaggs et al., 2003).

High concentrations of N also threaten drinking water supplies (Nolan et al., 1997). The majority of studies of flow-weighted nitrate concentrations in drainage effluent report levels that exceed U.S., Canadian, and European drinking water standards (see Table 1). A comprehensive survey of water quality in the U.S. found that about 20% of the groundwater wells and about 10% of stream sites had concentrations of nitrate that exceed the U.S. Federal Drinking Water Standard (10 mg/L). Groundwater samples from agricultural areas have higher concentrations of nitrate than either urban or forested areas. Almost half the stream sites and 55% of groundwater wells sampled in areas where agriculture is the primary land use had concentrations of nitrate above 2 mg/L. The risk of nitrate contamination in groundwater varies with soil properties, depth to groundwater, land use, and groundwater-surface water connections (Nolan et al., 1997). Some nitrate in groundwater may reflect past contamination from agricultural practices prior to nutrient management improvements (Eidem et al., 1999). In some cases, subsurface drainage may actually reduce the risk of contamination of groundwater wells by nitrate (as well as atrazine and other chemicals common in agricultural runoff) by diverting subsurface flow to surface waters (Eidem et al., 1999, Gilliom et al., 2006).

In addition to human health effects, nitrate can be directly toxic to aquatic organisms, primarily due to the conversion of oxygen-carrying pigments to forms that are incapable of carrying oxygen. Camargo and others (2005) recently reviewed published data on toxicity of nitrates to freshwater and marine animals. Adverse effects were reported for a range of freshwater invertebrates, fishes, and amphibians at long-term exposures to nitrate concentrations at 10 mg/L. Nitrate toxicity to aquatic animals increases with increasing concentrations and exposure times, and generally decreases

with increasing body size, water salinity, and environmental adaptation. The maximum NO_3 concentration recommended for full protection of sensitive organisms was 2 mg/L for freshwater and 20 mg/L for marine environments.

High NO_3 levels and anoxic conditions in response to anthropogenic inputs of dissolved inorganic nitrogen (DIN) have increased the production of nitrous oxide (N_2O), a powerful greenhouse gas and catalyst in the destruction of stratospheric ozone (Seitzinger & Kroeze, 1998). Production of N_2O in streams, rivers, and estuaries in response to nitrate loading from subsurface drainage waters has been implicated as a potentially important source of indirect agricultural emissions of N_2O . However, there is currently great uncertainty about the magnitude of this effect relative to other sources and its implications (Reay et al., 2003; Well et al., 2005).

Freshwater ecosystems. The severity of nutrient enrichment effects on freshwater ecosystems varies in response to the degree to which agriculture alters natural biogeochemical cycles from pre-agricultural conditions. All else being equal, increased nutrient loading tends to have greater impacts in historically oligotrophic or mesotrophic systems, where native biological communities are adapted to relatively lower levels of nutrients (e.g. northern lakes), than in systems that are somewhat eutrophic under natural conditions (Dodds et al., 1998). Altered nutrient cycling tends to manifest more irreversibly in lake and shallow wetland (lentic) systems than in lotic systems, where nutrients and contaminants are flushed from the system and/or cycled downstream; see Carpenter et al., 1999).

Freshwater systems that are poorly buffered (e.g., softwater or low-alkalinity lakes) may also be vulnerable to acidification from deposition of nitrate and ammonium. Surface water acidification causes direct mortality to acid-sensitive fish and aquatic organisms and increases the toxicity of other naturally occurring elements such as aluminum. It also enhances mercury accumulation in fish and aquatic organisms, decreasing survival, size, and density of aquatic biota, and altering trophic structure in lakes and streams (Rabalais, 2002).

In watersheds characterized by long-term import of P in excess of exports, the net accumulation of P stored in agricultural soils and bottom sediments of lakes, wetlands, stream channels and ditches means that sediments can become a very long-term source of P, with P continually being taken up and remobilized in response to seasonal and climatic dynamics. Such pools of P mean that there can be a long lag time in surface water quality response even if there are significant reductions in surface water loading due to improvements in on-farm nutrient management (McDowell & Sharpley, 2001; McDowell et al., 2004).

Lakes and Wetlands (Lentic habitats). The eutrophication of lakes, reservoirs, and estuaries generally follows a sequence of cascading trophic effects. Most lakes and reservoirs respond to nutrient enrichment (mostly P) with a strong increase in algal biomass (Dillon & Rigler, 1974; Jones &

Bachmann, 1976; OECD, 1982). The production of zooplankton and benthos also tends to peak, followed by sharp reductions in secondary production. As producers die and sink to the bottom, they create increased BOD, resulting in bottom hypoxia (0–2 mg/L dissolved oxygen) and anoxia (0 mg/L). Under eutrophic conditions, green algae (*chlorophyta*) also tend to decline relative to N-fixing cyanobacteria (Krupa & Czernas, 2003; Romanowska-Duda et al., 2002). Increased dominance of cyanobacteria is often associated with shifts in the zooplankton community (e.g., from larger to smaller sizes; see Dawidowicz et al., 2002; Pattinson et al., 2003). Cyanobacteria may also lower the diversity of consumers directly due to their toxicity (Aboal et al., 2002).

Increased turbidity and algal growth also results in a loss of submerged aquatic vegetation, exacerbating oxygen deficiency and altering habitat (Rabalais, 2002). Shifts in zooplankton and macroinvertebrate communities can influence higher trophic levels (Hall & Rudstam, 1999; Kling, 1998; Ludsin et al., 2001). For example, altered nutrient balances cause trophic shifts in fish assemblages from specialized insectivores to generalized insectivores and herbivores, as the consumer community responds to the change in relative abundance of primary producers (Rashleigh, 2004). Threshold effects are possible in which nutrient enrichment drives highly irreversible ecological shifts. For example, phosphorus accumulation in bottom sediments and their resuspension have been shown to drive permanent shifts to eutrophic conditions in freshwater lakes (Carpenter et al., 1999). Effects on food chains include major, persistent, and irreversible shifts in fish and aquatic communities to dominance by species tolerant of low water clarity, low oxygen, and poor water quality. Jeppesen et al., (2005) reviewed long-term studies of lake response to reduced nutrient loading for a range of lake systems degraded by anthropogenic eutrophication. Although many lakes did respond positively to reductions in P inputs, ecological recovery was delayed by internal loading (i.e., resuspension of P accumulated in bottom sediments or plant biomass), and often confounded by complex internal and external dynamics. Reduced loading led to many lakes reaching new equilibria after 10–15 years, rather than returning to pristine conditions.

Some paleolimnological evidence suggests that many prairie wetland and lake systems were naturally mildly eutrophic (Allan et al., 1980; Menzel et al., 1984). Wet prairies, wetlands, streams, and riparian zones probably acted as nutrient sinks, removing biologically available N and P from river waters before they reached coastal ecosystems (Timmons & Holt, 1977). Such systems may be relatively less impacted by nutrient enrichment than systems that evolved under conditions of greater nutrient limitation.

Both natural and restored wetlands are often noted for their role as sinks for nutrients, sediment, and other contaminants present in agricultural runoff and drainage waters. The effectiveness of wetlands as sinks for nitrate and phosphorus varies depending on size, ratio of wetland size to contributing

area, contaminant loading rates, and position of the wetland in relation to surface waters and nonpoint sources (Baker et al., 2004; Brueske & Barrett, 1994; Phipps & Crumpton, 1994; Tomer et al., 2003). Wetlands with high nutrient or sediment loading rates may become "saturated" and cease to function as effective filters (Baker et al., 2004; Fennessy et al., 1994). The contribution of subsurface drainage to nutrient alteration of wetland ecosystems will depend on the type of wetland and its position in relation to upstream drainage systems. Except by design for nutrient removal, subsurface drainage systems generally discharge directly to surface drainage ditches or surface waters.

In turn, agricultural runoff has been shown to alter vegetation composition (Freeland et al., 1999; Kantrud, 1986) and reduce invertebrate abundance (Dunn & Agro, 1995; Euliss & Mushet, 1999) in wetlands. Nutrient enrichment or hydrologic changes may additionally impair unique wetland plant and animal communities. Many wetland plant communities are adapted to specific hydrologic and soil conditions. Under conditions of N enrichment, weedy or invasive species such as cattails (*Typha* spp), reed canary grass (*Phalaris arundinacea*), purple loosestrife (*Lythrum salicaria*), and common reed (*Phragmites australis*) may outcompete less aggressive native wetland plants (Galatowitsch et al., 1999; Green & Galatowitsch 2003; Herr-Turoff & Zedler, 2005; Rickey & Anderson, 2004; Werner & Zedler, 2002; Woo & Zedler, 2002). Several types of wetlands in 12 Midwestern states are now listed as "rare" by the Nature Conservancy, with altered structure and function contributing as much as direct habitat loss to their endangerment (Grossman et al., 1994).

Rivers and Streams (Lotic systems). In lotic systems (streams and rivers), nutrient cycling varies along a longitudinal continuum (Vannote et al., 1980). In natural streams, headwater streams are often structured by high allochthonous (terrestrial) inputs of organic material, leaves, and other terrestrial detritus. Macroinvertebrate communities in these reaches are typically dominated by insect shredders adapted to processing coarse organic material into a fine particulate form that is then available to collector invertebrates (Schlosser, 1990; Vannote et al., 1980). As stream size increases, instream production gradually shifts to autotrophy, with a greater proportion of primary production instream.

Nutrient enrichment of streams can shift the dynamics of heterotrophy and autotrophy along the longitudinal continuum, increasing production of periphyton, benthic (bottom), and suspended algae in headwaters (Gregory, 1980; Grimm & Fisher, 1986; Hill & Knight, 1988; Kohler & Gelbrecht, 1998; Lohman & Priscu, 1992; Lohman et al., 1991; Triska et al., 1983). Higher algal production and reduced organic detrital inputs leads to cascading effects on stream consumers, shifting macroinvertebrate community dominance from shredders (i.e., species that process coarse organic material inputs) to collectors and scrapers (i.e., species that feed primarily on producer

phytoplankton) (Correll 1998; Hershey et al., 1988; Karr & Dudley, 1981; Sharpley & Menzel, 1987). Studies of macroinvertebrate communities in streams draining agricultural areas support this prediction (MacFarlane, 1983; Menzel et al., 1984; Porter et al., 1999; Zimmer & Bachmann, 1978). Numerous studies and reviews have documented linkages between nutrient enrichment and fish community change in both streams and lakes as well (Carpenter et al., 1996; Jeppesen et al., 2005; Ludsin et al., 2001; Taft & Jones, 1999; Wolter et al., 2000). Miltner and Rankin (1998) found a negative correlation between biotic integrity in streams and high levels of nutrient enrichment ($TP > 0.06 \text{ mg/L}$ and $TN > 0.61 \text{ mg/L}$). Yuan and Norton (2004) found that increasing nitrate-nitrite concentration was one of three stress indicators that best explained local measures of biological integrity in an Ohio cornbelt watershed (the other two being substrate quality and riparian quality).

Coastal, Marine, and Estuarine systems. Nutrients limit algal production in many estuarine and marine waters (Downing, 1997; Hecky & Kilham, 1988; Howarth, 1988; Lapointe & Clark, 1992; Vitousek & Howarth 1991), with some cyclical or seasonal patterns in N versus P limitation (Anderson et al., 2002; Howarth & Marino, 2006). Biologically available N is generally the key constraint on primary productivity in marine systems (Conley, 2000; Vitousek & Howarth, 1991), but many estuaries also show seasonal variation in nutrient limitation with signs of P limitation common during the spring flush (Conley, 2000; Rabalais, 2002). Many systems show threshold effects in response to nutrient loading, with productivity initially increasing but falling dramatically once hypoxic conditions develop (Caddy et al., 1993; Rabalais, 2002).

Anthropogenic additions of N to coastal and estuarine ecosystems are responsible for the increasing size and extent of seasonal cycles of hypoxia and benthic anoxia throughout the world (Diaz & Rosenberg, 1995; Goolsby et al., 2001; National Research Council, 2000; Rabalais et al., 2001). Stratification caused by fresh river water flowing over saltier marine water during the spring and summer prevents mixing of oxygen-rich waters into deeper waters. Organic matter produced in marine waters in response to fertilization by nutrients from river basins settles to the bottom, using up available oxygen as it decomposes. The loss of benthic organisms ultimately threatens important and economically valuable marine fisheries (Diaz & Rosenberg, 1995). Gulf of Mexico hypoxia affects an area from which 25% of U.S. commercial fisheries capture occurs, valued at roughly \$3 billion. Although fishery production in the Gulf has been maintained for past few decades, reductions in production, abundance, and diversity of species in bottom habitats and benthos have been shown (Mitsch et al., 1999; Rabalais, 2002).

Changing nutrient ratios also have implications for marine and estuarine food webs (Turner & Rabalais, 2003). The changing balance of N, Silicon (Si), and P can affect diatoms (which require silica), a key food source for

zooplankton (Conley et al., 1993; Turner & Rabalaism, 2003; Turner et al., 1998). Cascading effects on phytoplankton communities have been observed in response as the dominance of diatoms is reduced relative to cyanobacteria, including noxious forms responsible for nuisance algal blooms (Rabalais, 2002). Toxic algal blooms have caused fish kills and disrupted ecosystems in estuaries from the Baltic Sea to the Atlantic coast of the United States (Anderson & Garrison, 1997; Burkholder & Glasgow, 1997; Burkholder et al., 1992; Glibert et al., 2001; Rosenberg et al., 1988; Shumway, 1990).

PESTICIDES AND HERBICIDES IN AGRICULTURAL DRAINAGE WATERS

Many agricultural pesticides and herbicides are now regularly detected at low levels in rivers, streams, and groundwater (Coupe et al., 1995; Hatfield, 1998). A 2002 assessment of the nation's ecosystems included an indicator of chemical contamination of surface waters, using data from the 1992–1998 USGS National Water Quality Assessment (NAWQA) study as well as sediment and fish contaminant data from EPA's Environmental Monitoring and Assessment Program (EMAP) (Heinz Center, 2002). The suite of compounds selected for inclusion in the indicator accounted for 75% of currently used agricultural pesticide applications by amounts used, and 90% of the nation's historical use of organochlorine pesticides (most of which are now banned in the United States), as well as a number of pesticide degradation products. Eighty-three percent (83%) of monitored streams in farmland areas had at least one pesticide for which the concentration exceeded aquatic life guidelines promulgated by the EPA (USEPA, 1986). All streams had at least one pesticide at detectable levels throughout the year, and 75% had an average of five or more.

The most recent NAWQA report that summarized data from an additional 15 basins sampled in 1998–2001 (for a total of 51 basins), found that 57 percent of 83 streams in agricultural areas had concentrations of at least one pesticide that exceeded one or more aquatic-life benchmarks at least once during the year. In agricultural areas, the samples that most frequently exceeded a benchmark involved chlorpyrifos (21 percent of sites), azinphos-methyl (19 percent), atrazine (18 percent), *p, p'*-DDE (16 percent), and alachlor (15 percent) (Gilliom et al., 2006). Scribner et al. (1996) found trace levels of herbicides—a few of which exceeded drinking water standards—in the majority of 76 reservoirs studied in the Midwest. Most had between 5–8 detected herbicides of the 14 tested. Another USGS study found atrazine above EPA's drinking water standard in one third of the samples of the Platte River (Stamer & Wieczorek, 1996).

Kladivko and colleagues (2001) examined more than 40 peer-reviewed studies from more than 20 different research program sites in North America for results of runoff and pesticide losses in subsurface drainage water. Overall mass losses in subsurface drainage were typically less than 0.5% of applied herbicides, whereas mass losses of 1–5% of applied pesticide were

not uncommon in surface runoff (Kladivko et al., 2001). Pesticides were commonly detected in drainage flows during the first drainage events after application, usually during the spring, and there was high year-to-year variability in losses due to timing of precipitation events in relation to application. Peak concentrations could be very high, but were usually short-lived. This review and others underscored the importance of flow-proportional sampling schemes in subsurface drainage study design, given that the bulk of mass losses occurs during flow events that could be missed by infrequent grab sampling (Kronvang et al., 1997; Wang et al., 2003).

Although most studies have found concentrations and loads of pesticides to be significantly lower in subsurface drainage waters than in surface runoff (Kladivko et al., 2001; Southwick et al., 1990, 1997), significant mass losses have been reported in subsurface drainage, particularly association with preferential flow (Elliott et al., 1998; Flury, 1996; Fortin et al., 2002; Jaynes et al., 2001; Köhne & Gerke, 2005). Haria et al. (1994), in a study not reviewed by Kladivko et al. (2001), found that macropore flow was the dominant path for pesticide losses in a drained, heavy clay soil in southern England.

As with P, many herbicides and pesticides are strongly adsorbed to soil constituents. The amount of pesticide adsorbed by soil is dependent both on properties of the pesticide and the soil, with sorption generally increasing in proportion to organic matter or clay content of soil. Thus, conditions that give rise to concern are those associated with sediment losses (e.g., runoff or preferential flow) or immediately post-application, before chemicals adsorbed to soil constituents (e.g., large or prominent subsurface drainage flow events following application). Especially in loam soils, even strongly adsorbed chemicals can move along preferential flow pathways (Flury, 1996; Schulz, 2004). Schulz (2004) concluded that the relative importance of subsurface versus surface runoff for pesticide transport depended primarily on the soil adsorption properties of the pesticide. Other key factors governing variability in pesticide/herbicide losses include topography, macropore structure of soils, and the location of subsurface drainage systems (Leu et al., 2005).

Ecological effects of pesticides and herbicides. Pesticides in aquatic environments may impair both structure and function of microbial, macroinvertebrate, and aquatic plant communities, effects of which can significantly alter food webs (DeLorenzo et al., 2001). Many pesticides are highly toxic to fish and other aquatic organisms at normal rates of application (Grist, 1986). Five percent of 1454 fish kills reported in 32 states 1992–1993 were attributed to pesticides (Padgett et al., 2000). Pesticide-related fish kills were estimated to have killed 6–14 million fish per year from 1977 to 1987 (Pimentel & Greiner, 1997; USEPA, 1990). However, accurate documentation of fish kills due to pesticides is difficult; most pesticide-related fish kills go unreported

and, even when reported, the number of fish killed is often underestimated (USEPA, 1990).

Sublethal and ecosystem effects of pesticides in aquatic environments are even more difficult to quantify. Mechanisms and acuity of toxicity vary greatly between pesticides and among individual organisms, and are largely unknown. Some chemicals may be easily metabolized, whereas others bioaccumulate (Solomon et al., 1996). In many cases metabolites (or breakdown products) may also be of concern (David et al., 2003; DeLorenzo et al., 1999). A summary of aquatic ecosystem effects of commonly used agricultural pesticides is presented in Table 3.

Levels of pesticides known to induce lethal behavioral effects in aquatic macroinvertebrates and fish are frequently detected in surface waters, particularly during spring flushes (Gilliom et al., 2006). Kronvang et al. (2004) monitored concentrations of two herbicides (bentazone and MCPA), two fungicides (fenpropimorph and propiconazole) and two insecticides (dimethoate and pirimicarb) in drainage water in response to a 16 mm simulated rainfall on a grass field. Although mass losses represented much less than 0.01% of applied pesticide and the duration of exposure was just 7 hours, significant mortality/inactivity was observed for the macroinvertebrate *Gammarus pulex*, an important food source for stream fishes.

Both atrazine and glyphosate have been found in experimental mesocosms to have significant lethal effects on aquatic plants, aquatic insects, fish, and amphibians, particularly tadpoles (Britson & Threkold, 2000; Relyea, 2005). Effects have been observed in laboratory studies and experimental mesocosms as well as in aquatic ecosystems (Cox, 2001). Concentrations of 0.1 and 1.0 ppb of atrazine caused declines in the population of water fleas (*Daphnia spp.*) in experiments conducted in a lake in northern Germany. Low concentrations of atrazine have been shown to cause a variety of adverse effects in fish, including reduced sperm production, disruptions of normal behavior, kidney damage, and decreased ability to withstand warm temperatures (see Table 3). Alachlor significantly reduced algal biomass and community composition in streams in Nebraska, with effects lasting from 7 days to permanent shifts in dominant algae (Spawn et al., 1997).

The herbicide atrazine is the most commonly detected agricultural herbicide or pesticide in surface waters, and is consistently detected in samples from rivers, streams, groundwater, lakes, and reservoirs (Cox, 2001; Gilliom et al., 2006). Most ecological studies have found few community-level effects from concentrations of atrazine routinely detected in surface waters fed by subsurface drainage waters. Although locally toxic to planktonic drift, duckweed, and algae, concentrations downstream from subsurface drainage outlets are rarely high enough to cause direct toxicity to fish or adult frogs (Battaglin & Fairchild, 2002; Lakshminarayana et al., 1992).

TABLE 3. Selected studies reporting effects on aquatic biota or ecosystems of commonly used agricultural herbicide/pesticides or their metabolites

Contaminant	Concentration	Non target organism	Effect	Study
Herbicides				
Atrazine	0.04 ug/L	Atlantic salmon <i>Salmo salar</i>	Reduced release of a sex hormone from testes; reduced milt production by 50%	Moore and Waring, 1998
Atrazine	0.5 ug/L	Goldfish <i>Carassius auratus</i>	Induced "burst swimming," sudden spurt of nondirected movement, followed by immobilization	Saglio and Triasse, 1998
Atrazine	1 ug/L	Salmon <i>Salmo salar</i>	Reduced behavioral response of male salmon to urine of females	Moore and Lower, 2001
Atrazine	3 ug/L	Rainbow trout <i>Oncorhynchus mykiss</i>	Increased in blood protein indicating stress	Davies et al., 1994
	5 ug/L	Goldfish <i>Carassius auratus</i>	Decreased grouping behavior	
Atrazine	5 ug/L	Zebrafish <i>Brachydanio rerio</i>	Altered swimming behavior	Steinberg et al., 1995
	5 ug/L	Rainbow trout <i>Oncorhynchus mykiss</i>	Kidney damage	
Atrazine	10 ug/L	Shiners <i>Cyprinid spp.</i>	Reduced tolerance to warm temperatures	Messaad et al., 2000.
Atrazine	10 ug/L	Trout <i>Salmo trutta</i>	Kidney damage	Zoulmi et al., 1995
Atrazine	20 ug/L treatment of pond	Bluegill, <i>Lepomis macrochirus</i>	Reduced offspring by 90%; probably related to food base	Kettle et al., 1987
Atrazine		Aquatic plants invertebrates	Drastically reduced Reduced mayflies, dragonflies, beetles, and other insects	DeNoyelles et al., 1989.
Atrazine	15 ug/L	Freshwater snails	Altered and increased foraging behavior	Roses et al., 1999
Atrazine		Amphibians (a species of frog <i>Hyla chrysolecis</i>)	Increased mortality of tadpoles	Britson and Threkold, 2000
		Tiger salamander <i>Ambystoma spp</i>	Increased growth hormone (thyroxine) and decreased (corticosterone); slowing metamorphosis	Larson et al., 1998

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TABLE 3. Selected studies reporting effects on aquatic biota or ecosystems of commonly used agricultural herbicide/pesticides or their metabolites (*Continued*)

Contaminant	Concentration	Non target organism	Effect	Study
Atrazine		Alligators <i>Alligator mississippiensis</i>	Inhibited activity of estrogen and progesterone in female oviducts; increased activity of an enzyme that "feminizes" hatchlings converting male hormones into female hormones	Vonier et al., 1996
Atrazine	0.1 and 1 ug/L	Water fleas <i>Polypheus spp</i>	>90% population decline	Lampert et al., 1989
Atrazine	0.12 ug/L	Marine algae <i>Chlorophyta</i>	Decline in photosynthesis and productivity	Dewey, 1986
	2 ug/L	Algae <i>Chlorophyta</i>	Changed abundance and community dominance in experimental ponds	
	2 ug/L	Water moss <i>Fontinalis spp.</i>	Depressed photosynthesis 90%	
	3 ug/L	Algae <i>Chlorophyta</i>	Reduced chlorophyll	
	5 ug/L	Water fleas <i>Polypheus spp</i>	Skewed sex ratio	
	10 ug/L	Blue-green algae cyanobacteria	Inhibited growth in spring; stimulated in summer	
	10 ug/L	Seagrass beds	Decreased photosynthesis	
	11 ug/L	Algae (multiple spp.)	Reduced biomass	
	12 ug/L	Freshwater green algae <i>Chlorophyta</i>	25% reduction in photosynthesis	
	12 ug/L	Wild celery <i>Vallisneria</i>	50% mortality and reduced reproduction	
Atrazine	12 ug/L	Freshwater algae <i>Chlorophyta</i>	Reduced abundance in experimental streams	
	15 ug/L	Algae (multiple spp)	Reduced productivity and nutrient uptake in wetlands	
	20 ug/L	Plant-eating insects	Reduced abundance in experimental ponds	
	20 ug/L	Aquatic plants	Reduced growth and abundance	
Atrazine	Various	Aquatic ecosystems	Food chain affects; Reductions in DO commonly observed with Atrazine due to reduced primary productivity	DeLorenzo et al., 2001
Atrazine	25 ug/L, 75 ug/L	Periphyton, <i>Zizania</i> and <i>Daphnia</i>	Negative impacts; effects on periphyton composition varied with the N:P supply ratio	Detenbeck et al., 1996
	15 ug/L	<i>Daphnia</i>	Depressed survival	

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TABLE 3. Selected studies reporting effects on aquatic biota or ecosystems of commonly used agricultural herbicide/pesticides or their metabolites (*Continued*)

Contaminant	Concentration	Non target organism	Effect	Study
Atrazine	20 ug/L	Aquatic biota	Threshold for no-observed effects	Huber et al., 1996
Atrazine	50 ug/L	Aquatic ecosystems	Threshold for ecologically important effects	Solomon et al., 1996
Not clear, but Atrazine likely played a role	River water from 11 river basins; Atrazine ranged from 1–20 ug/L	Fish, various spp.	Altered ratios of male and female sex hormones.	Goodbred et al., 1997
De-ethylatrazine (the most toxic metabolite of Atrazine)		Marine algae	Inhibited photosynthesis	DeLorenzo et al., 2001
Alachlor	10 ug/L	Algal community	Reduced chlorophyll	Carder and Hoagland, 1998
	30 ug/L		Reduced total algal cell densities	
Glyphosate (roundup)	90 ug/L		Biovolume reduced	
	2.3 ug/L	<i>Gammarus</i> (scuds, or amphipods)	Lethal after 48 hrs.	Folmar et al., 1979
	2.3 ug/L	Fathead minnow <i>Pimephales notatus</i>	Lethal after 48 hrs.	
	55 ug/L	Midge larvae <i>Chironomidae</i>	Higher temperatures and pH increased the toxicity.	
	140 ug/L	Rainbow trout <i>Oncorhynchus mykiss</i>		
Glyphosate	0.55–2.5 mg/L	Tadpoles (<i>Hyla versicolor</i> , <i>Rana pipiens</i> , <i>Rana sylvatica</i>)	Lethal effects; eliminated two species of tadpoles and nearly exterminated a third species, resulting in a 70% decline in tadpole species richness in experimental ponds	Relyea, 2005
Fungicides				
Manab, Dineb		Blue-green algae Cyanobacteria	Toxic at all levels tested	DeLorenzo et al., 2001
Nabam	60 ug/L	Marine phytoplankton spp.	Toxic	

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TABLE 3. Selected studies reporting effects on aquatic biota or ecosystems of commonly used agricultural herbicide/pesticides or their metabolites (*Continued*)

Contaminant	Concentration	Non target organism	Effect	Study
Organophosphate insecticides			In general, suppression of zooplankton predation can stimulate increase in algae	DeLorenzo et al., 2001
2,4-D	100 mg/L	Marine phytoplankton spp.	EC50	DeLorenzo et al., 2001
2,4-dichlorophenol (2,4-D metabolite)	5 mg/L		EC50	
2,4,5-T	52 mg/L	Marine phytoplankton spp.	EC50	
2,4,5-trichlorophenol	1.8 mg/L	Marine phytoplankton spp.	EC50	
2,4-D, Sevin, malathion, and glyphosate		Aquatic communities (algae and 25 spp of animals)	<ul style="list-style-type: none"> - Reduced zooplankton diversity by eliminating cladocerans; copepods increased in abundance - Reduced the diversity and biomass of predatory insects and had an apparent indirect positive effect on several species of tadpoles. 	Relyea, 2005
Parathion	7.86 g/L	Green algae <i>Chlorophyta</i>	Reproductive inhibition	Faust et al., 1994
Fenitrothion	10.1 mg/L	Freshwater bivalve <i>C. fluminea</i>	LC50; inhibited cholinesterase activity	Oneto et al., 2005
Temephos	10 and 100 mg/L		Variable effects on growth	
Chlorpyrifos	Simulated drainage ditches	Phytoplankton	Greater increase in abundance in presence of nutrient enrichment than with nutrient enrichment alone	DeLorenzo et al., 2001
	1.2 ug/L	Phytoplankton	Persistent reduction in growth	
	10 ug/L	Bacteria	Increased abundance and productivity	

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TABLE 3. Selected studies reporting effects on aquatic biota or ecosystems of commonly used agricultural herbicide/pesticides or their metabolites (*Continued*)

Contaminant	Concentration	Non target organism	Effect	Study
	4-1000 ug/L	Freshwater ponds	Reduced zooplankton abundance, reduced DO and increased CO ₂ , leading to algae and cyanobacteria blooms	Mani and Konar, 1988
Carbamate insecticides				
Carbaryl & its metabolite	5 mg/L and 3.7 mg/L 3.7 mg/L	Marine bacterium <i>V. fisheri</i> Green alga, diatoms, and cyanobacteria	EC50 level Inhibition of C uptake in 90% of algae tested	Somasundaram et al., 1990 Peterson et al., 1994
Carbofuran	0.67 mg/L 20.5 mg/L	Green alga, diatoms, and cyanobacteria Marine bacterium <i>V. fisheri</i>	Relatively low toxicity to most species tested EC50 level	Somasundaram et al., 1990
Carbofuran metabolites phenol and methylamine	60.9 and 34.6 mg/L	Marine bacterium <i>V. fisheri</i>	EC50 level	
Chlorinated hydrocarbons				
Aldrin, dieldrin and endrin	<1 mg/L	Green and blue-green algae	No significant effects on respiration	Vance and Drummond, 1969
DDT	<10 ug/L	Marine planktonic algae	Photosynthesis inhibited	Lal and Lal, 1988
P,p'-DDT	3.6-36 ug/L	Green algae	Photosynthesis inhibited	
Chlorophenols	0.42-150 mg/L	Green algae	Toxicity increased with number of substituted Cl-atoms	Shigeoka et al., 1988
Endosulfan	47 and 130 ug/L 360-600 ug/L 10 mg/L 100 mg/L	Marine red algae Green algae	Chronic impairment of female growth Chronic impairment of reproduction Delayed and interfered with reproduction; reduced growth rates 15% Completely inhibited growth	Thursby et al., 1985 Netrawali et al., 1986

In the 1990s, at the behest of the U.S. EPA and Ciba-Geigy, the Institute of Wildlife and Environmental Toxicology (TIWET) at Clemson University convened a panel of experts to conduct a comprehensive review of ecological effects of atrazine in North American surface waters (Solomon et al., 1996). The panel concluded that in most places, atrazine does not pose a significant risk to the aquatic environment (Solomon et al., 1996). Although effects on algae, phytoplankton, and macrophyte production may occur in small streams affected by agricultural runoff, recovery from any effects is generally rapid. However, site-specific risk assessments were recommended for areas where intensive use is a concern, or where exposures exceed an ecologically significant threshold of ~50 ug/L.

Some pesticides can travel long distances from application sites in air or rain and/or may bioaccumulate (Cox, 2001). For example, rain deposits an estimated 110,000 Kg of atrazine in the Mississippi River basin every year, and in some cases, concentrations in rainwater have exceeded drinking water standards (Clark & Goolsby, 2000). Majewski and Capel (2000) found atrazine in rainfall at nearly every site where rainfall was collected.

Many pesticides and herbicides, even those that are metabolized rapidly in soils, can persist in aquatic environments. Elevated concentrations of atrazine following spring flushes can persist over several months (Larson et al., 1995). Despite declines in atrazine use in the Mississippi River Basin between 1975 and 1997 (from 3.45 Mg to 2.27 Mg), atrazine loads in the river did not decrease (Clark & Goolsby, 2000). Concentrations in the Mississippi River typically range from 1–5 $\mu\text{g}/\text{L}$, but pulses exceed the drinking water standard set by EPA (USGS, 1999).

Concerns that mixtures of chemicals may be more toxic to aquatic organisms than predictions based on traditional risk assessments for single chemicals also have some basis in empirical research (DeLorenzo et al., 2001; Vonier et al., 1996). Atrazine has been shown to synergistically increase the toxicity of organophosphate insecticides to an aquatic midge (Pape-Lindstrom & Lydy, 1997). Carder and Hoagland (1998) found that toxicity from atrazine and metolachlor appeared to be additive rather than synergistic. Fairchild et al. (1994) did not find evidence of synergistic effects with herbicide (atrazine) and insecticide (esfenvalerate) mixtures. Atrazine altered species composition of macrophytes but did not alter plant biomass, total system metabolism, or the bioavailability of the insecticide to zooplankton or fish.

Recently, the endocrine and hormonally active properties of agricultural chemicals have generated increasing concern and research (Sumpter, 2005). More than 60% of all agricultural herbicides (by mass) used in the United States are capable of disrupting the endocrine or reproductive systems of a wide range of taxa, including humans (Short & Colborn, 1999). Both active and inert ingredients have been identified as having potential endocrine

disrupting affects. Although pesticides and herbicides tend to be present at very low levels, often below detection limits, endocrine effects are also possible at very low levels. For example, atrazine and alachlor at relatively low concentrations commonly detected in surface waters have been shown to increase mortality or disrupt the hormone systems of both amphibians and alligators (Vonier et al., 1996, Crain et al., 1997). A USGS study of fish from 11 river basins nationwide found that river water contaminated with pesticide residues was associated with altered ratios of "female" to "male" sex hormones in fish (Goodbred et al., 1997).

Considerably more research is needed to detect ambient levels of pesticides and herbicides in the environment and evaluate the impacts of pesticides on aquatic microorganisms and ecosystems (DeLorenzo et al., 2001; Gormley et al., 2005). The majority of pesticides have not been thoroughly evaluated for health or aquatic ecosystem effects, and there is enormous uncertainty surrounding the health and environmental effects of many of these chemicals. For many pesticides, production and use data are not accessible, and reliable tests are not available. For example, the Heinz Center (2002) assessment noted that there are no drinking water benchmarks for 33 of the 76 common pesticides analyzed, and no aquatic life benchmarks for 48 of the 76. Furthermore, current water quality standards do not account for mixtures of chemicals or seasonal pulses of high concentrations. In addition, potential effects on the reproductive, nervous, and immune systems, as well as on sensitive individuals, are poorly understood.

Finally, the long-term community level effects of pesticides and herbicides are still largely unknown. Heckman (1981) studied aquatic communities in drainage ditches of orchards in Germany, comparing contemporary species structure and abundance to results from an earlier study prior to widespread chemical usage. He concluded that herbicides had had little effect on floral species diversity, but that insecticides had drastically reduced predatory species and essentially eliminated a species of water mite. Many insect species had developed resistance to the agricultural chemicals used in the orchards, resulting in shifts in community dominance.

SUMMARY AND CONCLUSION

Aquatic ecosystems have been substantially modified and impacted by the land use, hydrologic, and water quality changes associated with the extensive development of agricultural surface and subsurface drainage. The cumulative effects of changes occurring over the past century has been as follows:

- widespread declines in many intolerant species;
- dramatic shifts in the composition of aquatic communities; and
- homogenization of aquatic faunal assemblages toward more tolerant, generalist species.

The effective conservation of aquatic biodiversity will require substantial reductions in delivery of sediment, nutrients, and agrochemicals to surface waters and ecosystems. Conservation and restoration of aquatic ecosystem structure and function will also require reestablishing more natural hydrologic regimes and fluvial habitats at multiple spatial and temporal scales. Coordinated protection of functional wetlands, riparian zones, and other remnant natural habitats should remain a priority.

Although many of the effects of agriculture on aquatic ecosystems cannot be attributed to specific practices in isolation, the use of artificial drainage and, in particular, subsurface drainage on agricultural croplands does have some specific implications. Relative to surface drainage only, subsurface drainage shifts the volume, timing, and pathway by which waters intercepted by agricultural lands enter surface waters. By shifting the major pathway for excess precipitation (i.e., water not retained by soils, groundwater, or evapotranspired) from surface runoff to subsurface flow, subsurface drainage typically reduces concentrations of sediment, phosphorus, and pesticides in drainage waters relative to surface runoff. Together, these contaminants represent the most widespread cause of water quality impairments to freshwater and coastal ecosystems. However, subsurface drainage waters are more likely to bypass riparian buffer strips and other "sinks" where contaminant loads in surface runoff may be trapped and filtered. Furthermore, subsurface drainage typically carries much greater loads of nitrate-nitrogen than surface runoff, particularly under some cropping systems such as corn-soybeans.

The magnitude and direction of hydrologic and water quality effects of subsurface drainage relative to surface drainage varies widely with local soil characteristics, topography, land use, and drainage system design. Recent reviews have covered a range of proposed strategies that could reduce the water quality effects, hydrologic effects, and subsequent aquatic ecosystem impacts caused by subsurface drainage (Baker, 2001; Dinges, 2004; Keeney, 2002). Although the question of which strategies or practices are most promising for restoring or sustaining aquatic biodiversity in drained agricultural landscapes is not the subject of this review, the vast majority of studies underscore the importance of integrated, landscape scale approaches and/or tailoring best practices and management systems to local conditions (see Alexander & Allan, 2006; Dinges, 2002; Environment Canada, 2004; Mitsch et al., 2001; Schnepf & Cox, 2006, 2007). Conservation drainage designs such as controlled drainage that allow slower and more gradual drawdown of water tables below the crop root zone show potential under some scenarios to significantly reduce nitrate losses, provide additional soil water storage, and improve water quality generally (Dinges, 2004; Drury et al., 1996; Fausey, 2004; Kladivko et al., 1999). Drainage system designs that optimize depth and spacing to balance multiple criteria such as capacity to absorb rainfall-runoff events against crop yields may also be an important

tool (Wiskow & van der Ploeg, 2003). Combining efforts to protect and restore strategically located wetlands, riparian zones, stream and river channels and other fluvial features; while simultaneously maintaining and improving highly productive agricultural lands with on-farm best practices is considered by many to be a holistic and viable solution (Crumpton & Helmers, 2004; Dinnes, 2004; Zucker & Brown, 1998).

Because subsurface drainage significantly enhances crop yields and economic performance, maintaining and improving existing drainage and associated yields on wet agricultural soils presently in production may make it economically feasible for landowners to achieve the same yield on less land (Zucker & Brown, 1998), perhaps simultaneously increasing the feasibility of setting more land aside primarily for the production and maintenance of ecological services. (Note that economic incentives to individual landowners under any given agronomic scenario, rather than the results of aggregate analyses, tend to drive management decision-making on the landscape (Keeney, 2002). As a result, careful attention needs to be given not just to analysis of on-farm management, but to policy strategies that can provide the appropriate incentives.) However, it is important to understand the direct and indirect ecological implications of subsurface drainage to mitigate for and/or minimize downstream/off-site impacts.

Clearly, effective conservation of aquatic biodiversity will require substantial reductions in yields of sediment, nutrients, and agrochemicals to surface waters and ecosystems. For example, a series of reports issued by the Mississippi River/Gulf of Mexico Watershed Nutrient Task Force ("Task Force"), advised by a panel of experts under the EPA's Science Advisory Board, have recommended a minimum of 20–30% reductions in Mississippi riverine nitrogen loads to effectively reduce the size of hypoxic zone in the Gulf (Doering et al., 1999; Mitsch et al., 1999). The Task Force's most recent draft 2008 Action Plan advises a dual nutrient strategy targeting at least a 45% reduction in riverine flux of both total nitrogen and total phosphorus, measured against the 1980–1996 time period (Mississippi River/Gulf of Mexico Watershed Nutrient Task Force, 2007).

Strategies proposed to reduce the effects of subsurface drainage on aquatic ecosystems are acknowledged to vary widely in their scope and effectiveness as well as their contemporary economic and political feasibility (Mitsch et al., 2001; Petrolia & Gowda, 2006). Most studies of on-farm changes in fertilizer management, tillage practices, controlled drainage, and other methods that can be implemented economically have shown maximum reductions of 15–30% in N losses when evaluated on an individual basis (Baker, 2001; Dinnes, 2004; Keeney, 2002). Furthermore, the benefits of combined use of improved practices are not completely additive (e.g., a 15% reduction from nitrogen crediting plus 20% reduction from controlled drainage does

not automatically sum to a 35% reduction). Even with significant reductions, it is unclear how quickly nitrogen fluxes in large rivers and estuaries, such as the Mississippi River, will respond to reductions in nutrient losses from agricultural lands upstream (Baker, 2001; Mitsch et al., 1999). Conservation and restoration of aquatic ecosystem structure and function will therefore require reestablishing more natural hydrologic and nutrient regimes at multiple spatial and temporal scales, requiring some attention to landscape-scale design (Fausch et al., 2002; Poff et al., 1997).

Maintaining and restoring important landscape features such as small, geographically distributed headwater wetlands, riparian areas, and flood-plains could mitigate for the water quality, hydrological, and ecological impacts of drainage simultaneously, serving multiple beneficial functions by providing distributed water storage and flood protection, wildlife/aquatic habitat, uptake, breakdown, and removal of nonpoint source contaminants in surface waters (Dinnes et al., 2002; Hey, 2001; Hey et al., 2004). Distributed buffers combined with ecologically based drainage designs might be more socially efficient in the long run by reducing maintenance costs and some kinds of disaster and environmental spending, maintaining economically valuable ecological services, and sustaining biodiversity. For example, current ditch and drainage designs result in unstable ditches that contribute to downstream water quality problems as they work to re-establish fluvial features and stability, requiring periodic costly investment in ditch maintenance. By designing drainage systems with fluvial processes in mind, maintenance expenditures might be reduced or avoided while improving the habitat value and water quality performance of ditches (Powell et al., 2007; Ward et al., 2004). Attention to both landscape-level and site-level design (e.g., position, extent, distribution) and monitoring will be important to ensure that restored channels, wetlands, and riparian buffers perform as expected for a range of desired ecological services.

Ultimately, a combination of strategies will be needed to effectively and jointly address the undesirable effects of surface and subsurface drainage on aquatic ecosystems. Both on-farm changes in cropping systems and nutrient management, as well as off-site wetland and riparian habitat protection and restoration in critical areas distributed across the landscape, are likely to be necessary (Baker, 2001; Boody et al., 2005).

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