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Introduction to nonpoint source pollution in the United States and prospects for wetland use

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ABSTRACT

Nonpoint source (NPS) pollution is the major cause of impairment of US surface waters. The dominant source of NPS pollution is agricultural activity, and "traditional" pollutants — nutrients, sediments, and pathogens — are the main detrimental constituents. Erosion from cropland has been declining and is expected to decline further in the 1990s, but it is unclear how this will translate into changes in sediment yields in streams. Pollution by nitrogen is of particular concern in eutrophication of estuaries, as a contaminant of groundwater and as an acidifying agent in atmospheric deposition. Nitrogen fertilizer and emissions of nitrous oxides are major contributors to the problem. The outlook on pesticides is mixed: bans on organochlorine pesticides in the 1970s have resulted in decreasing concentrations in fish tissue; however, herbicides are now a problem for some surface and groundwater sources of drinking water, especially in the Upper Midwest. Metals in NPS pollution are primarily a concern in mining areas and in urban runoff. Declining use of leaded gasoline has resulted in decreased lead in fish tissues, sediments, and surface waters around the nation. New directions in controlling NPS pollution include: (1) a greater emphasis on risk assessment, (2) a move toward regulatory or quasi-regulatory approaches, and (3) a trend toward source reduction. The potential for using wetlands to control agricultural NPS pollution is discussed by contrasting cropland runoff with secondary wastewater effluent.

INTRODUCTION

During the past 20 years, most of our effort to control water pollution has been directed at reducing point source discharges to surface waters. Today, 144 million people in the United States, twice as many as in 1972, are served by municipal wastewater treatment plants that provide treatment at the secondary level or better (US EPA, 1990a). Less than 1% of municipal wastewater is now discharged with no treatment. This upgrading

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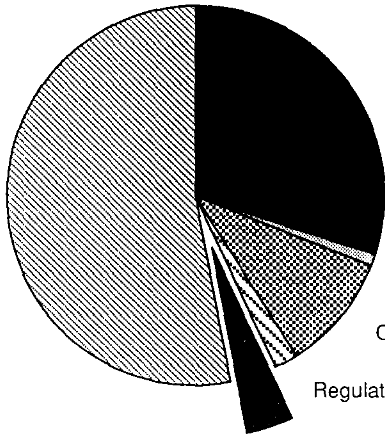

Fig. 1. The annual cost of water pollution control in the United States, 1986. Total expenses in 1986 were US$26.9 billion, calculated in 1982 dollars. Source: Farber and Rutledge (1988).

of sewage treatment plants, generally from primary to secondary treatment, has resulted in a 46% reduction in the discharge of oxygen-consuming pollutants from sewage treatment plants between 1972 and 1982. Had these improvements not occurred, discharges of oxygen-consuming pollutants would have increased by 191% due to population growth (ASIWPCA, 1985). But having made progress in this area, we are left with the problem of nonpoint sources of pollution — contaminated runoff from urban areas, agricultural fields, animal feedlots, roadways, abandoned mines, silviculture, and construction activities. Nonpoint source (NPS) pollution has been identified as the major remaining cause of surface water impairment (ASIWPCA, 1985; US EPA, 1989), but resources allocated for control of NPS pollution accounted for only about 4% of our national water pollution control expenses in 1986 (Farber and Rutledge, 1988; Fig. 1).

This paper presents an overview of the status of NPS pollution in the United States and briefly compares the potential of wetlands for removing pollutants from secondary wastewater with their potential for reducing pollutant loadings from cropland. It begins with a review of several national assessments, including recent reports developed by the US Environmental Protection Agency (EPA) and the states as mandated by sections 305(b) and 319 of the Clean Water Act. This is followed by a closer examination of several types of NPS pollution — sediments, nutrients (particularly nitrates), pesticides, salinity, and metals. Where possible, trends for several pollutants are examined to address the question: is NPS pollution getting better or worse? New directions in the control and monitoring of NPS pollution are discussed briefly. The second part of the paper considers several issues related to the use of constructed or managed wetlands to treat NPS pollution. This section emphasizes some fundamental differences between the use of wetlands for tertiary treatment of municipal wastewater, an area in which we have considerable experience, and their use in mitigating NPS pollution from rural lands, particularly agricultural land, an area in which we have little experience.

NONPOINT SOURCE POLLUTION: STATUS AND TRENDS

About 30% of assessed US surface waters do not "fully support" their designated uses (US EPA, 1990a; Table 1). The EPA has concluded that for roughly two-thirds of impaired waters, the cause of impairment is NPS pollution (US EPA, 1986; Fig. 2). The EPA's most recent biennial report on NPS pollution (US EPA, 1990b), summarized in Fig. 3 and Table 2, reports NPS pollution impacts for 331 804 km (206 179 river miles) of rivers, 2 140 000 ha (5 300 000 acres) of lakes, and 15 000 km^2 (5800 square miles) of estuary. One conclusion reached by these reports is that the traditional pollutants, particularly nutrients and sediments, are the primary causes of surface water impairment.

Although the methodologies used in these reports differ considerably, they concur in the finding that agriculture is the largest single cause of use

TABLE 1

Attainment of designated use for US surface waters. Source: US EPA, 1990a

	Rivers and streams [a]	Lakes [b]	Estuaries and coastal waters [c]
Total national resource	2 900 000	159 500	91 163
Resource within assessed states	1 851 470	90 442	Not given
Assessed resource	835 891	66 023	69 091
Assessed waters meeting use designation (%)			
Fully supporting	69.6	73.7	71.6
Threatened [d]	6.9	17.8	1.3
Partially supporting	20.1	16.6	22.8
Not supporting	10.3	9.8	5.6

[a] Resource estimates are in kilometers. The total resource value is from ASIWPCA (1985). The assessed resource is for 48 reporting states only.

[b] Resource estimate in square kilometers. No primary reference given for the total lake resource. The assessed resource is for 40 reporting states.

[c] Resource estimates in square kilometers; 23 out of 30 estuarine states reporting.

[d] The "threatened" category is a subset of "fully supporting" in the 305(b) report. Thus, percentages sum to more than 100%.

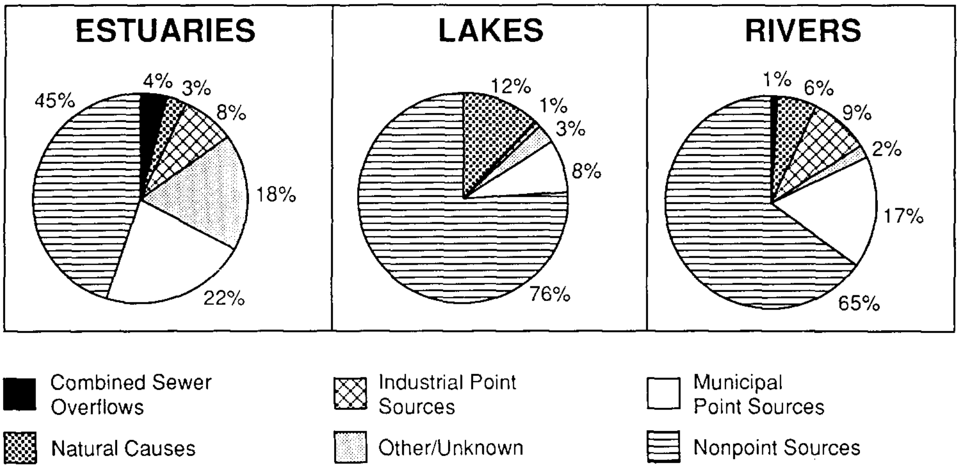

Fig. 2. Sources of pollution causing impairment of US surface waters. Percentages for estuaries are based on area, percentages for lakes are based on numbers of lakes, and percentages for rivers are based on miles. All percentages are calculated on the basis of the assessed resource. Source: US EPA (1986).

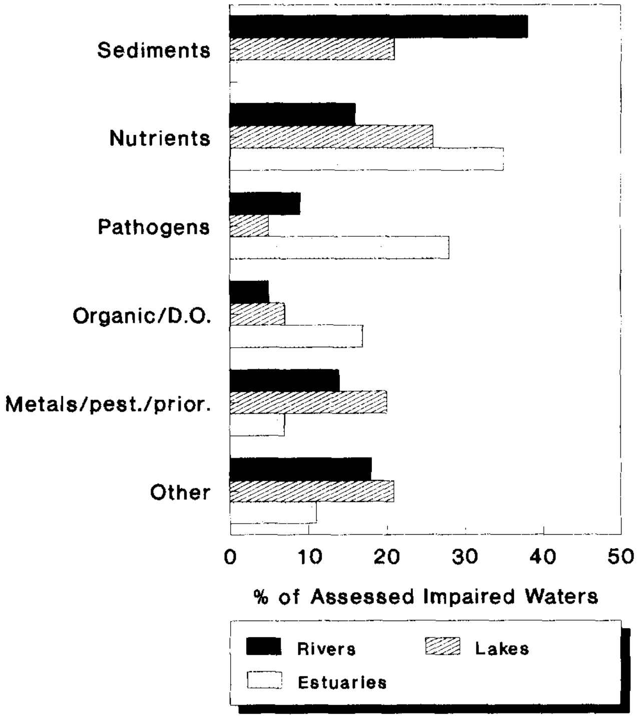

Fig. 3. Causes of surface water impairment by NPS pollution. Source: US EPA (1990b).

TABLE 2

Water quality impairment by nonpoint source pollutants. Source: US EPA, 1990b

	Rivers [a]	Lakes [b]	Estuaries [c]
Nonpoint source impacts reported	331804	21400	15000
Degree of impact (%) [d]			
Nonsupport	52	42	54
Partial support	28	22	36
Threatened	20	36	10
Causes (%)			
Agriculture	41	23	7
Urban	4	6	11
Land disposal	3	4	8
Construction	2	2	
Hydromodification	6	6	
Mining	8	7	16
Natural	8	10	
Others	3	16	43
In-place			16
Unknown	23	21	4

[a] Resource estimate in kilometers. Forty states reporting impacts; 20 reported identified use impairment; 33 identified causes of NPS pollution.

[b] Resource estimates in square kilometers. The number of states reporting impacts is not given; 18 states identified the type of use impairment; 25 states identified causes of NPS pollution.

[c] Resource estimate in square kilometers. Thirteen states reporting impacts.

[d] EPA (1990b) includes "threatened" in the "impaired" category, in contrast to the 305(b) report (US EPA, 1990a).

impairment in assessed rivers and lakes (Table 2). A recent National Academy of Sciences report on alternative agriculture (NRC, 1989) that emphasizes the negative influence of current national agricultural policies on environmental problems has further heightened interest in agricultural pollution.

Other sources of NPS pollution are more important than agricultural sources in several areas of the country, although they contribute to < 10% of the impairment of the assessed national aquatic resources. These sources include urban runoff, land disposal, hydromodification, and mining. A substantial percentage of systems impaired by NPS pollution have an unknown source (Table 2).

The methodologies used in these reports present several problems: (1) only a small portion of the total resource was assessed, (2) there is no consistent methodology for designating water uses (e.g., warmwater fisheries, coldwater fisheries, etc.), (3) there is no uniform methodology for determining use attainment, and (4) because of reporting differences between the 305(b) report (US EPA, 1990a) and the 319 report (US EPA, 1990b), results from the two cannot be directly compared (see US EPA, 1990a). Nevertheless, these reports and the earlier assessment by the ASIWPCA (1985) give a clear impression that NPS pollution, particularly agricultural pollution, is a major source of water quality impairment in the United States.

Several recently completed national studies focus on particular pollutants and resources and give us finer resolution of the scope of the NPS problem. These include two major studies of trends in surface water quality at several hundred stream sites in the US Geological Survey's (USGS) National Stream Water Quality Accounting Network (NASQAN) and National Water Quality Surveillance System (NWQSS), henceforth referred to in aggregate as the USGS network (Smith et al., 1987; Lettenmaier et al., 1991); the National Contaminant Biomonitoring Survey conducted by the US Fish and Wildlife Service (USFWS) (Schmitt et al., 1990; Schmitt and Brumbaugh, 1990); and the EPA's National Groundwater Pesticide Survey (US EPA, 1990c).

Sediments

Sedimentation has been identified as a major source of NPS impairment of US rivers and lakes (Fig. 3). Excessive sedimentation results in destruction of fish habitat, decreased recreational use, and loss of water storage capacity. The US Department of Agriculture (USDA) has estimated that annual offsite costs of sediment derived from cropland erosion alone are US$2–6 billion, with an additional US$1 billion arising from loss in cropland productivity (USDA, 1987). Erosion from agricultural lands has declined since the dust bowl years, from more than 3.6 billion metric tons/year in 1938 to 3.0–3.1 billion metric tons/year in the 1980s (USDA, 1990). Lee (1990) reported that cropland erosion declined by 12% between 1982 and 1987. This occurred primarily as a result of decreased erosion from land that was continuously cropped throughout the study period. In comparison, conversion of land (cropped land to noncropped land) was relatively unimportant in the overall decline in erosion during 1982–1987. The use of conservation tillage increased during the 1980s, from around 16 million hectares (40 million acres) in 1980 to 36 million hectares (88 million acres) in 1988, when it accounted for 30% of all cropland. Conservation tillage is expected to increase in the future (USDA, 1990) and when combined with expanded implementation of other conservation compliance programs should result in further reductions in cropland erosion during the 1990s (USDA, 1990).

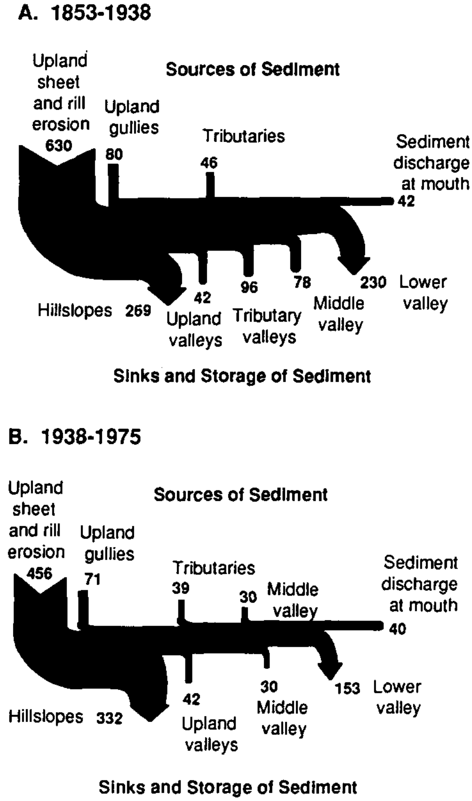

Fig. 4. Sediment budgets for Coon Creek, Wisconsin. The watershed has an area of 360 km^2. Numbers are annual averages in 10^3 tons/year. Source: Trimble (1981), as modified in Meade and Parker (1985). 1 ton = 1.016 metric ton. Copyright 1981 AAAS, reprinted with permission.

Does declining erosion from agricultural land result in improved conditions in rivers and lakes? From a water quality standpoint, the relationship between gross erosion (defined as a loss of soil from a parcel of land) and sediment yield (defined as the loss of suspended solids from the watershed) is complex; changes in erosion do not necessarily translate simply into changes in sediment yield. This discrepancy is best illustrated by a classic historical analysis of erosion and sediment yield in the agricultural watershed of Coon Creek, Wisconsin (Trimble, 1981). During the period 1853–1938, annual sheet and rill erosion was 640 000 metric tons. This amount declined to 463 000 metric tons as a result of improved soil conservation, a decrease of 28% (Fig. 4). However, sediment yield remained virtually constant (41–43 $\times 10^3$ metric tons/year) throughout the period of record because stream bank and channel erosion increased when sheet and rill erosion declined (Fig. 4). From a broader perspective, it has been estimated that nationally 25% of sediment yield occurs from stream bank erosion (Van der Leeden et al., 1990).

Stream modifications, such as channelization and reservoir construction, also affect sediment yield. For example, the completion of reservoirs on the Missouri River during the 1950s and 1960s was probably a major contributing factor in an observed 50% decline in sediment discharges by the Mississippi River to the Gulf of Mexico (Meade and Parker, 1985). Finally, other land uses account for about half the overall sediment loading to US surface waters, placing an upper limit on the potential for reducing sediment yields by controlling cropland erosion (Van der Leeden et al., 1990). The USDA has estimated that a reduction of one billion tons in gross erosion from the Nation's cropland would cause a 13–18% decline in the production of sediments, with concomitant reductions in total phosphorus and total organic nitrogen of 5–7% and 7–9%, respectively (USDA, 1990).

With this perspective, it is perhaps not surprising that concentrations of suspended solids (SS) have not changed at most USGS network stations since the 1970s (Table 3), despite declining cropland erosion. Smith et al. (1987) and Lettenmaier et al. (1991) both showed that concentrations of suspended solids did not change significantly at most of the USGS network sites. Among remaining sites, roughly the same number have shown an increase in suspended solids as have shown a decrease (Table 3). Smith et al. (1987) observed that increasing SS concentrations tend to occur in areas with high rates of soil erosion.

Nutrients

Phosphorus

Nutrients have been identified as the dominant cause of impairment by NPS pollution in lakes and estuaries (Fig. 3). In the majority of freshwater lakes, phosphorus (P) is the limiting nutrient for algal growth (Chiaudani and Vighi, 1974; Miller et al., 1974), an exception being lakes highly enriched in municipal wastewater, which typically have low N:P ratios (Miller et al., 1974; Baker et al., 1985b). There is no national data base on trends in lake eutrophication. In the USGS stream trend studies of Smith et al. (1987) and Lettenmaier et al. (1991), total phosphorus (TP) concentrations paralleled SS concentrations: the majority of stations (75–80%) exhibited no change; of those that changed, more decreased than increased (Table 3). However, Smith et al. (1987) concluded that reductions in TP concentrations were probably associated with declines in point sources. For the Great Lakes, municipal point sources of phosphorus were reduced by

TABLE 3

Trends of selected water quality variables in major rivers of the United States. Sources: Smith et al., 1987 (for 1974–1981); Lettenmaier et al., 1991 (for 1978–1987)

Constituent	Median [a]	1974–1981			1978–1987		
		Number of stations			Number of stations		
		Increasing	Decreasing	No Change	Increasing	Decreasing	No Change
Chloride	15	101	34	154	65	32	295
Suspended solids	67	43	39	194	12	19	121
Total P	0.13	43	50	288	12	69	308
Nitrate [b]	0.41	116	27	240	82	24	284
Lead	4	2	23	219	2	23	325
Cadmium	< 2	16	2	231	1	16	298
Arsenic	< 1	21	4	220	1	25	286

[a] Median concentrations are in mg/l for major common constituents and μg/l for trace metals (lead, cadmium, and arsenic).
[b] 1978–1987 data are for total nitrogen.

51–67% between 1975 and 1985 (CEQ, 1990), resulting in major reductions in lake phosphorus levels. Further reductions will depend upon reductions in NPS phosphorus loadings, which in 1986 comprised 59–88% of total phosphorus loadings in the Great Lakes (CEQ, 1990).

Where increases in phosphorus occurred, Smith et al. (1987) found statistical associations between TP increases and measures of fertilized acreage and cattle population. Thus, there is some suggestion that phosphorus loadings from agricultural areas are increasing.

Nitrogen

NPS nitrogen pollution is important in at least three arenas: (1) eutrophication of surface waters, (2) groundwater contamination in agricultural areas, and (3) acidification of forested watersheds. Nitrogen is particularly important in regard to the eutrophication of estuaries, since algal growth in estuaries is usually nitrogen limited (reviewed in Stoddard, 1991). An interesting aspect of this problem in Chesapeake Bay is that atmospheric deposition appears to be a major source of nitrogen (Stoddard, 1991). Direct deposition of nitrogen (NH_4^+ and NO_3^-) to the water surface of the Bay accounts for about 12% of the total N loading; additional atmospheric nitrogen loading comes from deposition to the watershed. Although about 90% of the nitrogen deposition to the Chesapeake Bay watershed is retained in plants and soils, the remainder passes through the watershed and into the Bay. This accounts for an additional 22% of the Bay's nitrogen budget, which together with direct deposition to the Bay surface means that 35% of the Bay's total nitrogen loading is derived from atmospheric deposition (Stoddard, 1991; Table 4). Other nonpoint sources (e.g., animal waste, fertilizer fluxes, etc.) account for 16% of total nitrogen inputs. These

TABLE 4

Nitrogen Budget for the Chesapeake Bay [a]. Source: Stoddard (1991), modified from Tyler (1988) and Fisher et al. (1988)

	Inputs to the watershed	Flux to the Bay
Fertilizer	11.3	0.4
Animal waste	13.9	0.4
Nitrogen deposition to land	11.4	1.1
Total for watershed:	36.6	1.9
Point sources		2.4
Nitrogen deposition to Bay surface		0.6
Total nitrogen input:		4.9

[a] Inputs and fluxes are in 10^9 mol/year.

findings suggest that reductions of nitrous oxide emissions, which are generated primarily by automobiles and other vehicles, would ameliorate the eutrophication problem in Chesapeake Bay.

One of the greatest NPS pollution concerns in the Midwest and other agricultural areas is nitrate contamination of groundwater. Nielsen and Lee (1987) used USGS well water records in conjunction with an analysis of sensitivity factors to estimate the potential for nitrate contamination in groundwater. In 474 counties out of 1663 agricultural counties with adequate data, measured nitrate levels were > 3 mg/l in more than 25% of the wells; 87 of these counties had measured nitrate levels greater than the EPA's maximum contaminant level (MCL) of 10 mg/l in more than 25% of the wells. These counties are located primarily in the Great Plains, the Corn Belt, the Southwest, and the Northwest. More recently, the National Pesticide Survey, a statistically designed survey of pesticides and nitrate in drinking water wells of the United States, showed that more than half had detectable levels of nitrate; in 2.4% of domestic rural wells (254 000 wells) and 1.2% of community supply wells (1130 wells), nitrate exceeded 10 mg N/l (US EPA, 1990c; Table 5).

Finally, elevated atmospheric nitrogen inputs may also result in nitrogen saturation of forested watersheds. Nitrate levels are elevated in atmospheric deposition throughout the eastern United States, occurring in a $SO_4^{2-}:NO_3^-$ ratio of about 2:1 (reviewed in Baker et al., 1991). Nitrate is generally thought to be efficiently retained in watersheds, neutralizing inputs of HNO_3 (Baker et al., 1991). However, recent evidence suggests that high inputs of nitrate may exceed the demand by watershed plants and microbes, resulting in nitrate saturation and subsequent breakthrough of HNO_3 into streams and lakes (Stoddard, 1991). This process is potentially important because the additional inputs of HNO_3 would exacerbate lake and stream acidification. There is conclusive evidence of nitrogen saturation in parts of northern Europe. However, in the United States, where atmospheric nitrate inputs are generally lower, the evidence for nitrogen saturation is sketchy. Some evidence that nitrogen saturation may be occurring has been found in the Catskill and Adirondack Mountains of New York and in the Mid-Appalachian and Smoky Mountains in the Southeast (Stoddard, 1991).

Is nitrogen pollution getting better or worse? Several lines of evidence suggest that it is getting worse, at least for surface waters. First, surface water concentrations increased in the USGS network stations: far more stations exhibited increased nitrate concentrations during 1974–1981 than decreases, and far more stations exhibited increases in total nitrogen during 1978–1984 than declines (Table 3). It is perhaps even more important that rivers discharging into estuaries on the East Coast and the Gulf of

TABLE 5

Pesticides and nitrate in community well systems and rural domestic wells. Source: US EPA, 1990c [a]

Constituent [b]	Use	% detected		HAL/ MCL	% above HAL/MCL [c]	
		CWS	RDW		CWS	RDW
Nitrate	Fertilizer	52.1	57.0	10	1.2	2.4
Pesticides	Various	10.4	4.2	Various	0	0.6
Atrazine	Herbicide	1.7	0.7	3	–	X
DBCP	Nematode	0.4	0.4	0.2	–	X
Alachlor	Herbicide	0	0.03	2	–	X
EDB	Insecticide	0	0.2	0.05	–	X
Lindane	Insecticide	0	0.1	0.2	–	X
DCPA	Herbicide	6.4	2.5	4000	–	–
Prometon	?	0.5	2.6	100	–	–
Simazine	?	1.1	0.2	1.0	–	–
Hexachlorobenzene	Fungicide	0.5	0	1	–	–
Dinoseb	Herb./Fung.	0.03	0	7	–	–
Bentazon	Herbicide	0	0.1	20	–	–

[a] This was a statistically based survey of community water supply (CWS) wells and rural domestic wells (RDW). During 1988–1990, 1300 wells were sampled, representing an estimated population of 94600 community water supply wells and 10.5 million rural domestic wells.

[b] Pesticide acronyms are as follows: EDB = ethylene dibromide; DBCP = 1,2-dibromo-3-chloropropane; DCPA = dimethyl tetrachloroterephthalate.

[c] For individual pesticides, percentages above HAL/MCL are not given. An "X" indicates that some wells exceeded the HAL/MCL and a "–" indicates that none did. HAL = health advisory limits; MCL = maximum contaminant levels.

Mexico exhibited increases in nitrate concentrations of 20–46% (Smith et al., 1987). The probable cause of this increase is the increased use of nitrogen fertilizers, from around 6.5 million tons in 1970 to 9–12 million tons in the 1980s (USDA, 1990). Smith et al. (1987) found strong associations between increased nitrate and measures of agricultural activity, supporting the contention that fertilizer nitrogen is a major cause of the uptrend.

Salinity

Excessive salinity is a problem primarily in arid regions, where it can lower crop yields, add to water treatment costs, and increase the maintenance cost of water supply systems. The USGS network studies (Table 2) show that 2–3 times more stations have experienced increases in chloride (a good surrogate for salinity) than have experienced declines. In the western United States, salinity problems are caused largely by irrigated agriculture, which both adds salt load and causes reduction in flow through evaporation. In the Colorado River Basin, damages due to salinity are conservatively estimated at US$311 million/year (USDI, 1989). Salinity in the Colorado Basin declined during the early 1980s due to unusually high flows, but salinity levels are expected to increase over the next 20 years as a result of continued development and a return to more normal hydrologic conditions. Salinity control has reduced the salt load by 158 000 metric tons/year, but an additional million-ton reduction will be needed by the year 2010 to keep salinity at Imperial Dam (near the US–Mexico border) below the established criterion of 879 mg/l.

Increased chloride concentrations in many eastern US streams are probably associated with increased use of road salt (Smith et al., 1987). In several regions of the National Stream Survey (a randomized sampling of streams in the eastern United States), 15–30% of the lower stream reaches had chloride concentrations more than 10 times higher than would be expected from natural sources (A.T. Herlihy, Utah State University, personal communication, 1991), indicating widespread chloride contamination in this part of the country.

Pesticides

As with nitrate, there is widespread public concern about contamination of groundwater by pesticides in agricultural areas. Of particular interest in recent years has been the potential for herbicide contamination, because herbicide use has increased fourfold since 1966. However, concerns about increased use of herbicides in conjunction with expanded use of conservation tillage may be overstated, as several studies indicate that total herbicide use may not change appreciably with the acceptance of conservation measures (Fawcett, 1987; Logan, 1987). A compilation of existing data by Williams et al. (1988), summarized in NRC (1989), shows that 46 pesticides have been detected in groundwater from 26 states; atrazine, aldicarb, and alachlor were the most commonly detected. Nielsen and Lee (1987) concluded that the potential for pesticide contamination was greatest in the Eastern Seaboard, the Gulf States, and the Upper Midwest. In the EPA's National Pesticide Survey (US EPA, 1990c), 12 of the 126 pesticides and pesticide metabolites analyzed were found in detectable quantities. One or more of these pesticides were detected in 10.4% of the nation's community water supply wells and in 4.2% of rural domestic wells. None of the community water supply wells and only 0.6% of the rural domestic wells exceeded maximum contaminant levels (MCL) or health advisory limits (HAL) for pesticides. Five pesticides exceeded the MCL/HAL limits in

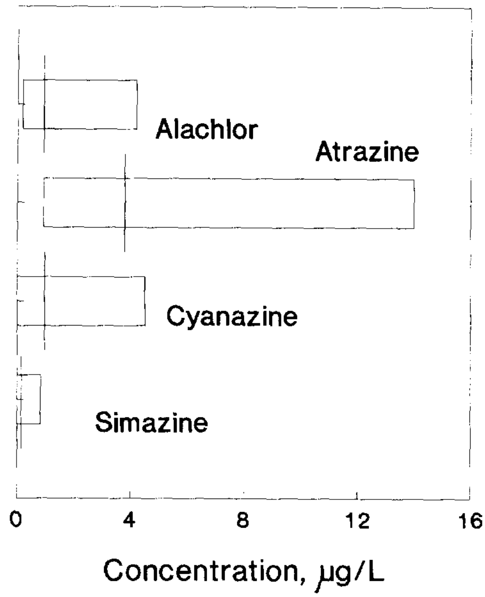

Fig. 5. Herbicide concentrations during the post-application period in 150 Midwest rivers. Median and quartiles shown. Source: Goolsby and Thurman (1990).

rural domestic wells: atrazine, alachlor, DCPA acid metabolites, lindane, and EDB (Table 5). Thus, from a national perspective, a small number of pesticides has contributed significantly to contamination of groundwater. Two of these pesticides, DBCP and EDB, are no longer in use in the United States.

In the Midwest, surface water concentrations of herbicides commonly exceed proposed standards, at least during part of the year. Baker et al. (1985a) reported seasonally elevated concentrations of atrazine in two Ohio rivers; peak concentrations of several pesticides in finished tap water derived from surface water exceeded what are now proposed MCL/HAL limits. In an ongoing study, the USGS measured herbicide concentrations in 150 rivers in the Midwest, an area that accounts for about 60% of the pesticides (mostly herbicides) used in the nation (Goolsby and Thurman, 1990). Concentrations of all measured herbicides were low during the pre-application period, but increased following herbicide application (May–June). During this period, nearly half the samples had atrazine concentrations above the HAL limit of 3 $\mu g/l$ (median = 3.8 $\mu g/l$); alachlor, cyanazine, and simazine also exceeded HAL or proposed MCL limits (Fig. 5). Although runoff from farmland is undoubtedly the major mode of transport for pesticides, herbicides are also distributed by aerosol drift and by volatilization, which typically accounts for losses of 10–30% of

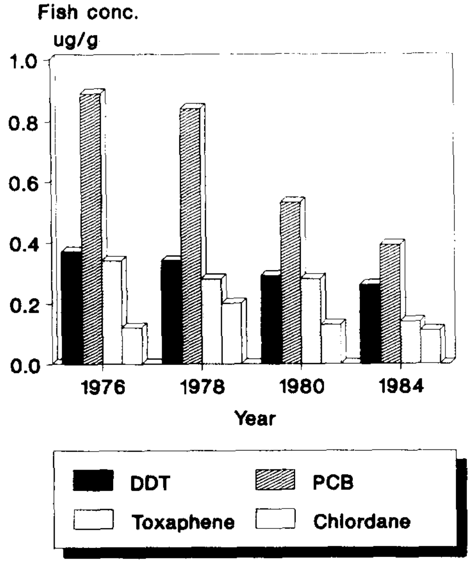

Fig. 6. Trends in several organochlorine compounds in fish tissue in US surface waters, determined in the National Contaminant Biomonitoring Program. Fish tissue concentrations are in $\mu g/g$ wet weight for whole fish. Source: Schmitt et al. (1990).

field application for many pesticides (see review by Grover, 1991). The USGS recently reported that the average concentration of triazine herbicides in Upper Midwest precipitation was 0.5 $\mu g/l$ (Capel, 1991).

The situation for many organochlorine compounds whose use was discontinued during the 1970s and early 1980s is encouraging. Results from the National Contaminant Biomonitoring Program (Schmitt et al., 1990), a periodic survey of fish tissue contaminants from 112 stations located throughout the nation, show that concentrations of organochlorine pesticides in fish have generally declined. Among those that declined significantly between 1976 and 1984 were DDT, PCBs, toxaphene, chlordane, and endrin (Fig. 6). Schmitt et al. (1990) noted that the presence of *p,p'*-DDT in some of their fish samples suggested recent inputs. Rappaport et al. (1984) postulated that continuing input of DDT to the United States occurs by atmospheric transport from Central America, where it is still used.

Metals

In the National Water Quality Inventory, the States generally considered metal contamination to be a relatively minor cause of surface water impairment; metals accounted for around 7–8% of NPS impairment of rivers and lakes (US EPA, 1990a). However, NPS metal contamination is a significant issue with respect to urban runoff and runoff from mining sites. In a comprehensive study of urban runoff in 22 cities, the EPA concluded that copper, lead, and zinc in urban runoff posed a significant threat to aquatic life. Median concentrations were 34, 144, and 160 $\mu g/l$ for copper, lead, and zinc, respectively. Each of these metals exceeded criteria for the protection of aquatic life in more than half the collected runoff samples (US EPA, 1983). Probably of greater significance in terms of the total number of miles of impaired streams and rivers is the leaching of metals from abandoned coal mines in the Appalachian region and abandoned metals mines in the West (Moore and Luoma, 1990). Studies by the US Fish and Wildlife Survey and the Appalachian Regional Commission (reviewed in Herlihy et al., 1990) indicate that there are approximately 10 000 km of acidic mine drainage streams in the Appalachian region. Pollution by metals, particularly selenium, is a problem in some wetlands receiving irrigation return flow, but the problem appears to be limited to closed-basin systems in the West (Deason, 1989).

Undoubtedly, the most important NPS problem of metals has been the atmospheric deposition of lead resulting from combustion of leaded gasoline. Perhaps one of the most successful efforts to control NPS pollution was the removal of lead from gasoline during the early 1970s. The decline in lead use resulted in decreased concentrations in surface waters at the USGS network stations (Smith et al., 1987; Lettenmaier et al., 1991), in Mississippi River delta sediments (Treffry et al., 1985), and in fish tissues in the National Contaminant Biomonitoring Program (Fig. 7).

NEW DIRECTIONS

In the nearly 20 years since passage of the Clean Water Act, there has been uneven progress in controlling NPS pollution. However, past experiences have led to a recent refocusing of efforts in several areas, including (1) development of a risk assessment approach to pollution control, (2) a trend towards source reduction as a key to reducing pollution, and (3) a shift from voluntary efforts to regulation and monetary incentives.

Risk assessment framework

EPA is gradually developing a risk assessment approach for dealing with pollution problems, so that the effort to control a particular type of pollution will have some relation to the risk posed to humans or ecosystems (SAB, 1990). As a starting point, there is a need to move beyond chemical monitoring to measures of ecological condition (Hughes and Larsen, 1988; Larsen et al., 1988; US EPA, 1989). This is a high priority if we are to

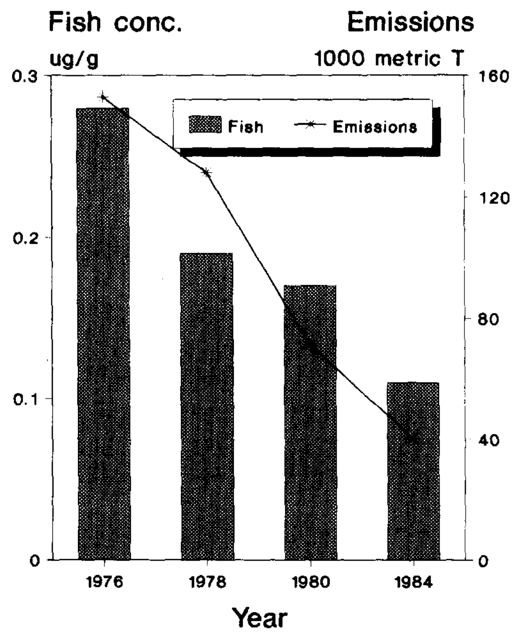

Fig. 7. Median lead levels in fish collected in the National Contaminant Biomonitoring Program (source: Schmitt and Brumbaugh, 1990), compared with US emissions of lead (source: CEQ, 1990). Fish tissue concentrations are in $\mu g/g$ wet weight for whole fish.

determine impacts from diffuse sources, which are often caused by multiple pollutants, commonly in conjunction with habitat alteration. States are currently required to develop biological criteria ("biocriteria") under Sections 303 and 304 of the Clean Water Act, although research to support this effort has been limited (GAO, 1990).

A second component of risk assessment is comparative risk analysis. On a national scale, the 305(b) and 319 program reports are intended to provide a comparative risk assessment of water quality problems, although as noted earlier, there are severe deficiencies in the methodologies used in these reports and in most other monitoring programs used for assessment of pollution (Hren et al., 1990). EPA's Ecological Monitoring and Assessment Program (EMAP; US EPA, 1991), now under development, will provide a statistically based representation of ecological conditions in the Nation's waters. The goal of this national-scale sampling effort is to evaluate status and trends in ecological conditions, but EMAP is also designed to address specific issues, such as causes and sources of impairment by NPS pollution, through more detailed diagnostic studies. These studies should be useful in developing policies that would provide the greatest improvement in ecological condition per dollar invested.

At the watershed scale, a key component of risk assessment is targeting major sources of pollution. This trend has emerged from a consensus that many past watershed efforts have taken a fragmented and inefficient approach to controlling NPS pollution (Humenick et al., 1987; NWQEP, 1988; US EPA, 1989; Water Quality 2000, 1990). Efforts to geographically target NPS pollution reduction are based on a growing recognition that small parts of watersheds often contribute a disproportionately large share of NPS pollutants and that these areas need to be targeted to reduce overall watershed pollutant loadings (NWQEP, 1988; CBP, 1990). A good example is the USDA's targeting of highly erodible lands for inclusion in its Conservation Reserve Program. Efforts to improve watershed NPS models should greatly enhance prospects for geographic targeting.

Emphasis on source reduction

Increasingly, we are finding that the best way to reduce pollution is not to produce it. As previously discussed, source reduction has proven effective in reducing NPS inputs of lead and organochlorine pesticides. Bans on detergents containing phosphorus have proven to be a cost-effective method for reducing phosphorus levels by 50% in wastewater effluent. In farming practices, reduction of fertilizer inputs has been identified as a cost-effective approach for reducing nutrients in runoff (Magleby et al., 1990). Approaches to prevent groundwater pollution, such as wellhead protection programs, are vastly less expensive than remediation efforts.

Shift from voluntary approaches to regulatory and monetary incentives

Low voluntary participation rates have been a problem in many watershed NPS pollution reduction efforts; voluntary efforts alone are now thought to be insufficient to control major NPS pollution (NWQEP, 1988; US EPA, 1989; CBP, 1990; GAO, 1990). Controls on local land use, animal waste management, and other measures have been suggested as regulatory complements to conventional NPS abatement methods (CBP, 1990; GAO, 1990). Between regulatory control and purely voluntary efforts are quasi-regulatory efforts and cost-share approaches, such as the conservation compliance measures (the "sodbuster" and "swampbuster" provisions) in the 1990 Farm Bill.

There has also been a gradual move from the "command-control" strategy of pollution reduction, which dictates specific pollution reduction methods, toward the use of market incentives. Some market incentive tools, such as the trading of pollution credits, may not be feasible for control of

NPS pollution. More realistic may be "pollution taxes" on fertilizers (now in use in Iowa) and pesticides, which would tend to increase the efficiency of application and thereby decrease the likelihood of these chemicals moving offsite as pollutants. There has also been considerable interest in eliminating disincentives for reducing agricultural pollution caused by the current crop price support system used in the United States, which tends to maximize crop production at the expense of environmental concerns (NRC, 1989).

USING WETLANDS TO CONTROL NONPOINT SOURCE POLLUTION

The goal of the papers in this volume is to evaluate the prospects for using constructed or natural wetlands to ameliorate the effects of NPS pollution in rural areas. There is relatively little published information on the efficacy of using wetlands to control rural NPS pollution, but considerable experience in using wetlands as tertiary treatment for municipal wastewater (Richardson, 1985; Kadlec and Alford, 1989; Knight, 1990). Therefore, in starting this exercise it is useful to address the questions: how is cropland runoff different from the effluent of a secondary wastewater treatment plant, and what are the implications of these differences in considering the potential of wetlands to remove pollutants from cropland runoff? For this comparison, it is useful to employ data from streams in agricultural areas of the "Corn Belt and Dairy Region", which were compiled by Omernik (1976). This region, having both extensive agricultural areas and numerous wetlands and sites suitable for wetland restoration and creation, has a high potential for wetlands control of NPS impacts. Data on secondary treatment plant effluent is based on a nationwide compilation of nutrient levels in wastewater treatment plant effluent (Gakstatter et al., 1978).

As a first consideration, note that the chemical composition of secondary effluent from wastewater treatment plants is relatively constant, with standard deviations around 10% of the mean (Table 6). Although data are not available for making a similar statistical calculation of uncertainty for cropland runoff, it appears that annual pollutant loadings from cropland may vary by one order of magnitude (reviewed in Novotny and Chesters, 1981). Thus, one could design a wetlands system for tertiary treatment of municipal wastewater from published effluent data, but site-specific loading data would be needed to design an efficient wetland treatment system to treat cropland runoff. Second, whereas total nitrogen concentrations in secondary effluent and cropland runoff are similar, total phosphorus concentrations in cropland runoff are only 1/20 of the concentrations in secondary effluent. Therefore, wastewater effluent is strongly nitrogen

TABLE 6

Characteristics of secondary effluent from municipal wastewater treatment plant, and cropland runoff

	Secondary effluent [a] concentration (mg/l)	Cornbelt cropland [b]	
		Concentration (mg/l)	Loading (kg ha^{-1} $year^{-1}$)
Suspended solids [c]	10–30	50–1000	
Total P	6.8 ± 0.4	0.14	0.31
Soluble P	5.3 ± 0.4	0.06	0.13
Total N	15.8 ± 1.2	4.4	9.5
Sol. inorg. N	8.4 ± 0.45	3.4	7.3
N:P	2.4	31.4	

[a] From Gakstatter et al., 1978. Means and standard deviations represent effluent from 244 activated sludge treatment plants located throughout the country.
[b] Source: Omernik, 1976. Total P and N loadings are means for "mostly agricultural" land (> 90% cropland) within the "Corn Belt and Dairy" region, based upon 80 sampled streams. Soluble P and N are calculated as the product of the ratio of soluble/total nutrients, calculated for "mostly agricultural" lands throughout the country ($n = 91$), and the total nutrient concentrations in the Corn Belt and Dairy region.
[c] The range for suspended solids is from Tables 1–3 of Novotny and Chesters (1981). The estimate for cropland is a range for stream yields in the agricultural areas of Wisconsin and Minnesota (Hindall, 1975; Otterby and Onstad, 1981).

limited, with a mean N:P ratio of 2.4:1, whereas cropland runoff is generally phosphorus limited, with a mean N:P ratio of 31:1 (Table 6). Third, both N and P occur primarily in the soluble form in wastewater treatment plant effluent; this is also true for nitrogen in cropland runoff (the predominance of soluble N increases with increasing fertilizer nitrogen use by agriculture; see Omernik, 1977), but not for phosphorus, which occurs primarily in association with particulate matter. Fourth, concentrations of suspended solids are usually < 30 mg/l in secondary treatment plant effluent, compared with values of 100–1000 mg/l for cropland. Perhaps the most critical difference, however, is that pollutant loading from croplands is largely event-driven, with extreme variations in both flow and pollutant concentrations, whereas the flow and composition of effluent from a wastewater treatment plant is relatively stable. Concentrations of suspended solids can vary by 2–3 orders of magnitude in cropland runoff within a year, with peak flows carrying the great majority of sediment loading (Fig. 8).

Because of these differences, the relative importance of pollution removal processes would be different in wetlands receiving cropland runoff than in wetlands receiving tertiary wastewater. First, sedimentation of particles would be a major process for removing suspended solids and

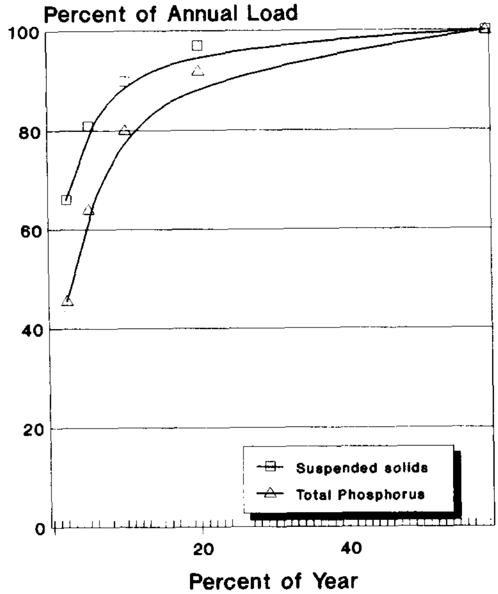

Fig. 8. Suspended solids and phosphorus loadings as a function of time for Honey Creek, Ohio. Infrequent peak flow periods contribute disproportionately to the total annual loading: > 90% of the suspended solids and phosphorus loading occurs in < 10% of the time. Source: Baker et al. (1985a).

phosphorus in wetlands receiving cropland runoff, since phosphorus is present largely in the particulate form. The high N : P ratios in cropland runoff suggest that plant uptake would be relatively more important as a long-term phosphorus retention process in agricultural wetlands than in wastewater systems, where phosphorus is grossly oversupplied relative to plant nutrient requirements. In contrast, adsorption of phosphorus to soils is a major mechanism of phosphorus removal in wetlands receiving wastewater effluent (Richardson, 1985); sedimentation is unimportant because phosphorus occurs as a soluble species. Denitrification would be an important mechanism in wetlands receiving cropland runoff, perhaps even more efficient than in wastewater wetlands (Kadlec and Alvord, 1989; Knight, 1990), since inorganic nitrogen is oversupplied relative to phosphorus in cropland runoff. A compilation of nutrient balances (Nixon and Lee, 1986) indicates that typically 20–80% of the phosphorus and 10–90% of the nitrogen are retained in natural wetlands.

These observations suggest that the design limitation for maximum pollutant removal would be based upon the need to retain sediments during peak flows. In this regard, the design of constructed wetlands for cropland runoff would have more in common with wetlands designed for treatment of urban runoff (Weidenbacher and Willenbring, 1984; Barten, 1986; Meiorin, 1989; also see Loucks, 1990) than with those designed for treatment of secondary wastewater.

In exploring the feasibility of using wetlands for efficient removal of sediment-bound pollutants, the first consideration is the size of wetland needed to control sediment movement. Very limited information from urban systems suggests that the ratio of wetlands to drainage area needed to achieve a reasonable reduction of suspended solids is on the order of 1:20 (Weidenbacher and Willenbring, 1984; Barten, 1986; Meiorin, 1989). A second consideration is the design lifetime. With sedimentation rates in wetlands on the order of 1 cm/year (McIntyre and Naney, 1991), it appears that the effective lifetime of a wetlands system designed for efficient sediment removal may be on the order of a few decades. Van der Valk and Jolly (1992) and Ethridge and Olson (1992) discuss the additional research needed to develop design criteria for rural wetlands treatment systems.

Pesticides in cropland runoff are also a concern for cropland wetland systems. Of particular interest is atrazine, which is the most heavily used herbicide in the United States. Goolsby and Thurman (1990) showed that, for medium-sized watersheds in the Midwest (800–2000 km^2), the median post-application concentration of atrazine in streams was 10 $\mu g/l$, with a 75% quartile value of 16 $\mu g/l$. This has two implications. First, atrazine is phytotoxic and is probably toxic to algae at concentrations of 1–10 $\mu g/l$ (deNoyelles et al., 1982; Kosinski and Merkle, 1984; Johnson, 1986), although it has a fairly short half life (ca. weeks; Huckins et al., 1986). It is therefore reasonable to suspect that atrazine would inhibit growth of algae in some wetlands, at least during part of the year. Second, if a constructed wetland were to promote flow to the groundwater system, there would be a possibility of causing groundwater contamination.

In the preceding few paragraphs, wetlands are regarded as engineered systems designed to remove pollutants. There are other benefits to constructing wetlands in rural areas, such as habitat for wildlife. As engineers and wetlands scientists move from using wetlands for tertiary wastewater treatment into the area of rural NPS control, the ancillary benefits of wetland treatment systems may equal or exceed the benefits of the pollutant removal function. Thus, the optimal design of a rural constructed wetland may not necessarily be the one that provides the maximum pollutant removal or even the most cost-effective pollutant removal, but the one that balances the pollutant removal function of wetlands with other ecological functions.

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