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Toward Quantifying Water Pollution Abatement in Response to Installing Buffers on Crop Land

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ABSTRACT / The scientific research literature is reviewed (i) for evidence of how much reduction in nonpoint source pollution can be achieved by installing buffers on crop land, (ii) to summarize important factors that can affect this response, and (iii) to identify remaining major information gaps that limit our ability to make probable estimates. This review is intended to clarify the current scientific foundation of the USDA and similar buffer programs designed in part for water pollution abatement and to highlight important research needs.

At this time, research reports are lacking that quantify a change in pollutant amounts (concentration and/or load) in streams or lakes in response to converting portions of cropped land to buffers. Most evidence that such a change should occur is indirect, coming from site-scale studies of individual functions of buffers that act to retain pollutants from runoff: (1) reduce surface runoff from fields, (2) filter surface runoff from fields, (3) filter groundwater runoff from fields, (4) reduce bank erosion, and (5) filter stream water. The term *filter* is used here to encompass the range of specific processes that act to reduce pollutant amounts in runoff flow.

A consensus of experimental research on functions of buffers clearly shows that they can substantially limit sediment runoff from fields, retain sediment and sediment-bound pollutants from surface runoff, and remove nitrate N from groundwater runoff. Less certain is the magnitude of these functions compared to the cultivated crop condition that buffers would replace within the context of buffer installation programs. Other evidence suggests that buffer installation can substantially reduce bank erosion sources of sediment under certain circumstances. Studies have yet to address the degree to which buffer installation can enhance channel processes that remove pollutants from stream flow.

Mathematical models offer an alternative way to develop estimates for water quality changes in response to buffer installation. Numerous site conditions and buffer design factors have been identified that can determine the magnitude of each buffer function. Accurate models must be able to account for and integrate these functions and factors over whole watersheds. At this time, only pollutant runoff and surface filtration functions have been modeled to this extent. Capability is increasing as research data is produced, models become more comprehensive, and new techniques provide means to describe variable conditions across watersheds.

A great deal of professional judgment is still required to extrapolate current knowledge of buffer functions into broadly accurate estimates of water pollution abatement in response to buffer installation on crop land. Much important research remains to be done to improve this capability. The greatest need is to produce direct quantitative evidence of this response. Such data would confirm the hypothesis and enable direct testing of watershed-scale prediction models as they become available. Further study of individual pollution control functions is also needed, particularly to generate comparative evidence for how much they can be manipulated through buffer installation and management.

Nonpoint-source (NPS) water pollution of streams and lakes is a prominent environmental problem throughout the United States. Major water pollution issues include declining drinking water quality, sedimentation, impaired recreation, and declining health of aquatic ecosystems. Agricultural crop land is a major source of pollutants, including sediment, nutrients (mainly nitrogen and phosphorus), pesticides, and pathogenic microbes. Federal law (Section 319 of the Clean Water Act) mandates government programs to control agricultural NPS pollution.

The National Conservation Buffer Initiative has been established by the U.S. Department of Agriculture (USDA) to reduce agricultural NPS pollution among other conservation objectives by promoting widespread installation of buffers on agricultural lands. Financial and technical assistance is provided to landowners through several USDA programs (e.g., Conservation Reserve Program, Environmental Quality Incentives Program, Conservation Reserve Enhancement Program). Many states also have buffer programs that aug-

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ment federal programs (e.g., Illinois, Iowa, Maryland, Nebraska, Virginia).

Within the context of USDA and related programs, buffers are installed by converting strips of cultivated area within existing fields to permanent vegetation. Buffers may be located in fields and at their margins. Pollution control is provided by natural processes of vegetation and soil that reduce the amount of pollutants that are mobilized and transported to streams and lakes.

Policy makers and land managers are increasingly calling for a clearer estimate of how much reduction in NPS pollution can be expected from buffer installation programs. In this paper, peer-reviewed scientific research is reviewed (i) for evidence of how much reduction in NPS pollution can be achieved by installation of buffers on crop land, (ii) to summarize important factors that can affect this response, and (iii) to identify remaining information gaps that limit our ability to make probable estimates. Both experimental and modeling research results are reviewed.

Several reviews have been published on various aspects of water quality function and design of buffers (e.g., Barling and Moore 1994, Castelle and others 1994, Fennessy and Cronk 1997, Haycock and others 1997, Hill 1996, Lowrance and others 1995a, Muscutt and others 1993, US Dept. of the Army 1991, Vought and others 1994, Wenger 1999). This review extracts information from the larger body of buffer research that provides the most direct evidence of quantitative responses of stream water quality to installation of buffers.

Experimental Studies

Stream and Lake Response

At this time, research reports are lacking that quantify a change in pollutant levels (concentration and/or load) in streams or lakes in response to installation of buffers.

Indirect evidence comes from watershed studies that show an important role that existing buffers can play in maintaining low pollutant levels in streams. Detailed nutrient budgets for agricultural watersheds in southern Georgia indicate that riparian buffers of mature forest that almost completely separate crop land from streams retain large quantities of nutrients from agricultural runoff that, if not intercepted, could substantially increase nutrient levels in those streams (Lowrance and others 1983, 1984b, 1985a, Yates and Sheridan 1983).

Observational studies consistently report a strong positive correlation between nitrogen concentrations in streams and the proportion of watershed area under cultivation (e.g., Hill 1978, Jordan and others 1997, Lowrance and others 1985b, Mason and others 1990, Omernik and others 1981, Schilling and Libra 2000). Accounting for proximity of agricultural land use to streams has been found to improve this correlation (Tufford and others 1998) or have no effect (Omernik and others 1981).

Taken together, the existing data imply that converting land from natural perennial vegetation to cultivated crops can account for elevated nutrient levels in streams and that reverting some of that land back to perennial vegetation, in the form of buffers, should reduce those levels. However, to date, there is no direct quantitative evidence of stream nutrient or other pollutant responses to installation of buffers.

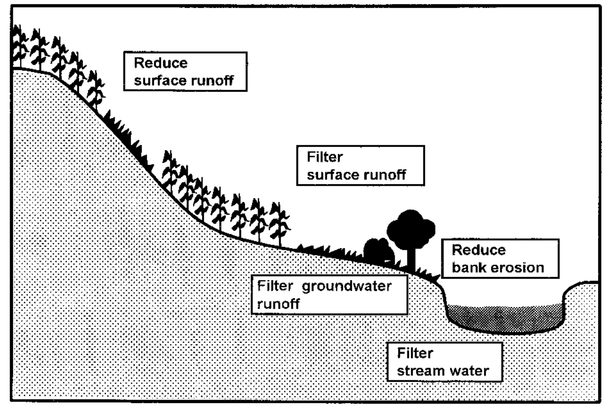

Figure 1. Five general ways that buffers can function to reduce nonpoint source water pollution from agricultural crop-land.

Pollution Control Functions

Most evidence that such a change should occur comes from site-scale studies of individual pollution control functions of buffers. There are five general ways that buffers can function to reduce NPS water pollution from crop land: (1) reduce surface runoff from fields, (2) filter surface runoff from fields, (3) filter groundwater runoff from fields, (4) reduce bank erosion, and (5) filter pollutants from stream water (Figure 1). The term *filter* is used here to encompass the range of specific processes that act to reduce pollutant amounts in runoff flow. Theoretically, if buffer installation improves these functions, then pollutant levels in streams should decrease.

1. Reduce surface runoff from fields. Buffers located within cultivated fields can reduce the amount of pollutants that reach the field margin mainly by inhibiting their mobilization. As runoff proceeds unimpeded down long cultivated slopes, it gathers volume, erosive force, and sediment transport capacity (Flanagan and others 1989). Buffers located in mid-slopes can impede flow and infiltrate some runoff, thereby reducing erosion and transport of soil and its constituents (Renard and others 1997). They also convert part of the erodible cultivated area to more stable vegetative cover. Most studies of in-field buffers are of performance during short-term runoff events.

Contour buffers. Wide strips of grass installed along topographic contours, often called contour buffers, function to reduce sheet and rill erosion. Contour buffers have long been recommended for erosion control on highly erodible crop lands (e.g., Ayres 1936). Based on recent analysis of several research reports published between 1944 and 1957, installing properly designed contour buffer strips in contour-tilled fields is generally estimated to reduce off-site sediment transport by about one-half (Renard and others 1997). Impact on other pollutants have not been reported.

Vegetative barriers. Vegetative barriers (also called grass hedges) are narrow contour strips for placement in fields where concentrated flows and high sediment loads would inundate a contour buffer. A vegetative barrier is a strip of dense, tall, stiff grass that functions like a porous dam to temporarily pond runoff water, settle its sediment load, and gradually release the water downslope (Dabney and others 1993, 1995, Dewald and others 1996, Kemper and others 1992). Because they impede concentrated flows, they are effective at reducing ephemeral gully erosion (Dabney and others 1997).

In general, water and sediment runoff from tilled plots with barriers has been observed to be similar to that from no-till plots without the barriers (Gilley and others 2000). On tilled row crop plots, 0.7 m-wide vegetative barriers reduced the mass of sediment in runoff by 57% and volume of runoff by 22% compared to plots with no barriers (Gilley and others 2000). At this same site, the barriers also tended to reduce concentrations of various forms of N and P in runoff from fertilized plots (Eghball and others 2000). The percent reduction of mass of sediment and nutrients by vegetative barriers varies with tillage system, residue management, and fertilization practices in the field (Eghball and others 2000, Gilley and others 2000, Raffaele and others 1997).

Sediment-trapping performance depends strongly on particle sizes. In a flume study, good-quality, stiff-grass vegetative barriers 0.14–0.76 m wide were found to remove 90% of sand-sized sediment but only 20% of silt and clay-sized sediments (Meyer and others 1995). Consequently, vegetative barriers are more effective for removing sediments from coarser-textured soils than from finer soils.

Good sediment-trapping performance depends on the ability of the vegetative barrier to stand up to deep ponding and retard flow (Dabney and others 1995, Meyer and others 1995). Tall, stiff grasses with high stem density, such as vetiver and switchgrass, were able to form deeper ponds during high flows (up to 40 cm depth) and trap more sediment than relatively short or limber tall fescue and miscanthus (Meyer and others 1995). At low runoff flows, even short, limber grasses can be effective barriers (Meyer and others 1995, Raffaelle and others 1997). In one plot study, a 0.6 m-wide strip of 20–30 cm-tall bermuda grass reduced sediment in runoff from a tilled area by an average of 63% compared to tilled area without the barrier (Raffaelle and others 1997).

Vegetative barriers are more effective at reducing sediment runoff than contour buffers on sites where gully erosion is a major sediment source (Dabney and others 1997). Performance for reducing erosion may improve with time as terraces form upslope from vegetative barriers (Dabney and others 1995, 1999, Kemper and others 1992). Washout points along vegetative buffers can be a problem that restricts their overall effectiveness (Dewald and others 1996, Kemper and others 1992).

Grassed Waterways. Grassed waterways are strips of buffer installed primarily to convey excess water from fields to the field margin without causing gully erosion. Dense, low-growing grasses are used to stabilize and protect the soil surface against erosion by concentrated runoff flow.

In recent studies at selected sites in 19 states, ephemeral gully erosion has been estimated to account for an additional 21–275% of the amounts of sheet and rill erosion (USDA 1996a). Properly designed grassed waterways may substantially reduce gully erosion, but quantitative estimates were not found.

Grassed waterways can also filter sediment and sediment-attached pollutants from runoff flow. Asmussen and others (1977) and Rohde and others (1980) found 86–98% of sediment from sandy field soils was trapped in their 20–24 m-long grassed waterways. Infiltration was substantial under dry antecedent soil moisture conditions (25–73% of input volume), but much less under wet soil conditions (2–44% of input volume). In these studies, greater than 70% of 2,4-D and trifluralin pesticides in runoff were retained by the waterways.

2. Filter surface runoff from fields. Buffers placed at the lower boundary of cultivated fields can intercept surface runoff flow from fields and retain pollutants that it

Table 1. Pollutant reduction in surface runoff from buffered fields compared to otherwise similar completely cultivated agricultural fields. Pollutant reduction attributable to buffers is expressed as percent of the pollutant amount leaving the completely cultivated field

		Buffer design		Site conditions		Test conditions	
Reference	Location	Vegetation	Width (m)	Slope (%)	Soil texture[a]	Rainfall source	Rain on buffer?
Hall and others (1983)	PA	oats	6	14	SiCL	natural	yes
Schmitt and others (1999)	NE	grass and grass + woody plants	7.5–15.0	6–7	SiL-SiCL	simulated	yes
Clausen and others (2000)	CT	grass + woody plants	30	5	SiL	natural	yes
Uusi-Kämppä and others (2000), Uusi-Kämppä and Yläranta (1996)	Finland	grass and grass + woody plants	10	> 10	C-CL	natural	yes

[a] Soil texture classes: SiCL = silty clay loam; SiL = silt loam; CL = clay loam; C = clay.

[b] Pollutant reduction = [(nonbuffered plot outflow − buffered plot outflow)/nonbuffered plot outflow] × 100%.

carries. Buffers that are designed to function in this way are often called filter strips and riparian buffers.

The major filtering processes include deposition, infiltration, and dilution. Deposition occurs when buffer vegetation retards the velocity of runoff flow, decreasing its capacity to maintain suspended sediment. Crop residue and large soil aggregates can also be sieved out of surface flow by mesh-like plant stems and stable plant debris. Some dissolved pollutants, colloids, and clays may be drawn out of flowing runoff by binding to surfaces of soil, vegetation, and debris in the buffer. Deposition reduces concentration and mass of pollutants in runoff flow, without altering its volume. Infiltration removes dissolved pollutants and very fine particles from surface runoff by diverting some runoff flow into the soil. Infiltration reduces mass of dissolved pollutants by decreasing the volume of surface runoff, but does not affect pollutant concentration. Dilution can reduce pollutant concentration in runoff when rainfall onto a buffer mixes with runoff from the field.

These three major processes also interact with each other. For example, infiltration that substantially reduces runoff volume also improves deposition by reducing the sediment transport capacity of remaining runoff flow (Hayes and others 1984, Lee and others 1989) and improves dilution by reducing the volume that originated from the field (Overcash and others 1981).

Performance. Four reports were found that contain a direct comparison of runoff from a buffered cultivated area to that from an unbuffered cultivated area of otherwise equal size and condition (Table 1). In general, runoff of sediment from buffered plots was substantially lower than that from comparable unbuffered plots (sediment mass 12–82% reduction). Lesser or more variable impact was found for other pollutants and runoff water [e.g., total P mass (−50)–60%; nitrate N mass (−115)−28%; atrazine mass (−120)−91%; dissolved P mass (−245)−14%; water (−163)−66%]. In one study, impact on concentration of pollutants was less variable than impact on mass (Schmitt and others 1999). The impact of recently planted buffers was generally less than that for well-estab-

Pollutant reduction (%)[b]		
Component	Mass	Concentration
sediment	76	
atrazine	65–91	
water	66	
sediment	12–82	40–81
total P	(−50)–60	15–58
bioavailable P	(−110)–39	2–36
dissolved P	(−245)–12	(−38)–(−4)
total N	(−140)–30	3–26
nitrate N	(−115)–28	7–28
atrazine	(−120)–40	(−39)–42
alachlor	(−101)–50	(−20)–52
permethrin	(−83)–80	25–73
dissolved bromide Br	(−134)–25	(−1)–14
water	(−163)–22	
sediment		92
total P		73
total kjeldahl N		70
nitrate N		83
ammonium N		25
chloride Cl		50
total P	27–38	
dissolved P	(−64)–14	
water	0–15	

lished vegetation, and often exhibited higher mass of pollutants in runoff than unbuffered plots (Schmitt and others 1999, Uusi-Kämppä and others 2000).

Most surface runoff studies have involved passing runoff from cultivated area through buffers of different size or design (Table 2). Studies having this general experimental design provide information on filtration processes within buffers and factors that determine level of performance. Pollutant reduction is quantified relative to the amount entering the buffer.

Among these studies, typical conditions include sites throughout the eastern United States, predominantly on grass strips, 3–20 m wide, on 2–16% slopes, under shallow dispersed (sheet flow) conditions. Sediment and sediment-attached nutrients have received the most study. Pesticides, dissolved nutrients, and microbes have received less attention to date. Most studies have focused on mass of pollutants in runoff flow. Impacts of buffers on pollutant concentrations have been underreported.

General performance trends indicated in Table 2 include: (1) Buffers can retain 40–100% of the sediment mass that enters them from a cultivated field. (2) Sediment-attached pollutants (e.g., total P, lindane, permethrin) are reduced to a lesser degree than sediment. This is probably because they are attached to finer particles that settle less-easily from runoff flow (Alberts and others 1981, Schmitt and others 1999). Masses of mainly dissolved pollutants (e.g., nitrate N, dissolved P, atrazine) are reduced the least. (3) Dissolved pollutant masses are reduced by similar percentages as water volume. (4) Concentrations of pollutants are reduced by similar or smaller percentages than masses. (5) In some situations mass and concentration of a pollutant may actually increase. This probably occurs when pollutants trapped within filter strips are remobilized in subsequent runoff flows (Coyne and others 1998, Magette and others 1989).

Studies conducted using animal wastes directly or in runoff from uncultivated land application areas are not included in Table 2 because of major differences in the amounts and types of pollutants (e.g., low content of mineral sediment; relatively enriched in organic matter and organic forms of nutrients) that could potentially produce performance results different than for crop land runoff.

Numerous factors have been identified that can exert substantial influence on filter performance and can explain the variability within and between studies reported in Table 2. These factors encompass field runoff characteristics, site conditions, buffer design, and buffer management.

Field runoff factors. Very high runoff amounts can reduce effectiveness of buffers. Submergence of buffer vegetation and inundation with deposited sediment can greatly reduce filtering capability (Dillaha and others 1988, 1989, Ree 1949, Wilson 1967). Hilly crop fields were observed to concentrate runoff flow in natural drainageways and inundate buffers at those locations with sediment (Dillaha and others 1989). Concentrated runoff did not appear to be a problem on flatter landscapes with more uniform slopes. Larger water runoff events caused lower relative retention of dissolved pollutants by buffers in one study (Lee and others 2000), but not in others (Arora and others 1996, Misra and others 1996). Retention of sediment and sediment-bound pollutants were not significantly affected by size of runoff event (Lee and others 2000).

Sediment-filtering performance depends strongly on particle size. In general, particles larger than silt settle first, such as sands and soil aggregates, and are effectively trapped within filter strips (Hayes and others 1984). Performance declines greatly for dispersed silt and clay that can remain in suspension. Consequently, filter strips are more effective where sediments in runoff are from coarse-textured or well-aggregated field soils.

Table 2. Pollutant reduction in surface runoff passing through buffers from cultivated agricultural fields. Reduction is expressed as percent of the amount entering the buffer

		Buffer design		Site conditions	
Reference	Location	Vegetation	Width (m)	Slope (%)	Soil texture[d]
Dillaha and others (1989)	VA	grass	4.6–9.1	11–16	SiL
Magette and others (1989)	MD	grass	4.6–9.2	≈2–4	SL
Coyne and others (1995)	KY	grass	9.0	9	SiL
Arora and others (1996)	IA	grass	20.1	2	SiCL
Robinson and others (1996)	IA	grass	3.0–9.1	12	SiL
Lowrance and others (1997a)	GA	grass	8	2–3	LS
		grass + forest	≈50	2–3	LS
Patty and others (1997)	France	grass	6–18	7–15	SiL
Barfield and others (1998)	KY	grass	4.6–13.7	9	SiL
Coyne and others (1998)	KY	grass	4.5–9.0	9	SiL
Tingle and others (1998)	MS	grass	0.5–4.0	3	SiCl
Sheridan and others (1999)	GA	grass	8.0	3.5	LS

[a] Reduction values for concentration calculated from values reported in Table 4 of this reference.

[b] Reduction values calculated from values reported in Table 3 of this reference.

[c] Concentration reduction values calculated from values reported in Table 1 of this reference.

[d] Soil texture classes: SiCL = silty clay loam; SiL = silt loam; SL = sandy loam; LS = loamy sand; C = clay; CL = clay loam; L = Loam.

[e] Pollutant reduction = [(buffer inflow − buffer outflow)/buffer outflow] × 100%.

Site condition factors. Buffer performance is greatest when runoff flows across a buffer in shallow uniform (sheet) flow. Uneven land that concentrates runoff flow within a buffer can substantially limit buffer effective-

Test conditions		Pollutant reduction (%)[e]		
Rainfall source	Rain on buffer?	Component	Mass	Concentration
simulated	not indicated	sediment	53–98	62–94[a]
		total P	49–93	59–80
		dissolved P	(−47)–55	(−27)–16
		total N	43–91	54–77
		ammonium N	9–89	27–71
		nitrate N	7–78	15–45
		water	(−42)–62	
simulated	not indicated	sediment	66–82[b]	
		total P	27–46	
		total N	(−6)–48	
simulated	no	sediment	99	
		E. coli	43–74	
		water	88	
natural	yes	sediment	40–100	
		atrazine	11–100	
		metolachlor	16–100	
		cyanazine	8–100	
		water	9–98	
natural	yes	sediment		70–85
natural	yes	alachlor	82	62
		atrazine	79	40
		alachlor	96	91
		atrazine	97	97
natural	yes	sediment	87–100	
		nitrate N	47–100	
		dissolved P	22–89	
		atrazine	44–100	
		lindane	72–100	
		isoproturon	99	
		diflufenican	97	
		water	43–100	
simulated	no	sediment	> 90	
		dissolved phosphate P	> 90	
		nitrate N	> 90	
		dissolved ammonium N	> 90	
		dissolved atrazine	> 90	
		water	90	
simulated	not indicated	sediment	96–98	79–87[c]
		E. coli	75–91	(−48)–8
		E. streptococci	68–74	(−106)–(−46)
		water	76–85	
simulated	not indicated	sediment	88–98	
		metolachlor	91–98	48–69
		metribuzin	91–98	48–68
		water	83–93	
natural	yes	sediment	78–83	63
		water	56–72	

ness (Daniels and Gilliam 1996, Dickey and Vanderholm 1981, Dillaha and others 1988, 1989).

Steeper slopes reduce performance (Dillaha and others 1988, 1989, Muñoz-Carpena and others 1993, Robinson and others 1996). Faster velocity of runoff down steeper slopes maintains a higher sediment transport capacity as well as allowing less time for infiltration than on lesser slopes (Dickey and Vanderholm 1981, Phillips 1989a).

Soil types that have higher infiltration capacity can reduce runoff to a greater degree than soils having lower infiltration conditions (Muñoz-Carpena and

Table 2. (Continued)

		Buffer design		Site conditions	
Reference	Location	Vegetation	Width (m)	Slope (%)	Soil texture[d]
Schmitt and others (1999)	NE	grass and grass + woody plants	7.5–15.0	6–7	SiL-SiCL
Lee and others (2000)	IA	grass and grass + woody plants	7.1–16.3	5	L-SiCL
Uusi-Kämppä and others (2000)	Finland	grass and grass + woody plants	10	10	C-Cl

others 1993, Overcash and others 1981, Verchot and others 1997a). Wet antecedent soil conditions reduce infiltration and reduce performance (Asmussen and others 1977, Rohde and others 1980, Verchot and others 1997a, Young and others 1980).

Buffer design factors. Wider buffers (greater downslope distance through the buffer) generally perform better than narrower ones (Barfield and others 1998, Coyne and others 1998, Dillaha and others 1989, Magette and others 1989, Patty and others 1997, Pearce and others 1997, Robinson and others 1996, Schmitt and others 1999, Vought and others 1991). For sediment, deposition diminishes rapidly with increasing distance into a buffer, such that most deposition occurs within a few meters of the field edge. Infiltration and dilution appear to maintain substantial influence on runoff across greater buffer widths (Phillips 1989a, Schmitt and others 1999).

Dense, stiff grass is the preferred vegetation, based on its flow-retarding structure (Barfield and others 1979, Kao and Barfield 1978, Muñoz-Carpena and others 1993, Tollner and others 1976, Williams and Nicks 1988, Wilson 1967). Because submergence by runoff reduces performance (Dickey and Vanderholm 1981, Wilson 1967), taller grasses may function better than shorter ones under high runoff conditions. All surface runoff experiments to date have involved some kind of grass, sometimes in combination with shrubs and/or trees. The inclusion of woody plants had no effect (Schmitt and others 1999) or lessened (Uusi-Kämppä and others 2000, Uusi-Kämppä and Yläranta 1996) the pollutant retention function of buffers on surface runoff.

Buffer management factors. Longer-term performance of filter strips can decline if not managed properly. Sediment accumulation at the field edge of buffers has been observed to create dikes that divert subsequent field runoff to low points along buffers where it flows across as concentrated flow (Dillaha and others 1989). Maintenance of sheet flow conditions may require periodical removal of accumulated sediment or other modification of surface topography.

Accumulated nutrients, sediment, and fecal bacteria appear to be released during subsequent runoff flows and reduce net retention of pollutants (Coyne and others 1998, Magette and others 1989, Young and others 1980). Maintenance of vigorous plant growth to take up nutrients and stabilize sediments and periodic harvest of vegetation to prevent recycling of nutrients are thought to be important management activities to maintain long-term performance of buffers as net sinks for pollutants. Long-term studies of this issue, however, have yet to be reported.

3. Filter groundwater runoff from fields. Buffers placed at the lower boundary of cultivated fields may also intercept shallow groundwater flow and, through various soil processes, remove pollutants transported in it. Groundwater flows from under agricultural fields toward streams, carrying with it dissolved and colloidal pollutants gathered from infiltrated surface water.

Test conditions		Pollutant reduction (%)[e]		
Rainfall source	Rain on buffer?	Component	Mass	Concentration
simulated	yes	sediment	84–98	76–93
		total P	71–96	55–79
		bioavailable P	61–94	39–65
		dissolved P	50–90	19–43
		total N	57–91	27–52
		nitrate N	53–90	24–48
		atrazine	33–90	(−5)–43
		alachlor	42–92	10–61
		permethrin	54–95	27–83
		dissolved bromide Br	44–88	13–31
		water	36–82	
simulated	yes	sediment	70–94	
		total P	46–93	
		dissolved phosphate P	28–85	
		total N	50–90	
		nitrate N	41–88	
		water	25–80	
natural	yes	total P	27–38	
		dissolved P	(−64)–14	
		water	0–15	

Groundwater that flows through or near the root zone of buffer vegetation appears most affected. Because groundwater is often closest to the root zone in riparian areas, this location represents a good opportunity for using buffers to intercept groundwater and remove pollutants.

Several mechanisms have been identified that immobilize and transform pollutants from groundwater flow, including plant and microbial uptake; chemical reactions, such as precipitation and sorption; and microbe-mediated oxidation/reduction and pesticide degradation. These processes change pollutant concentrations without much affect on volume of runoff. Dilution or concentration of pollutants in subsurface flow may also occur depending on whether infiltrated surface water within the buffer is less or more concentrated, respectively, than the groundwater.

Groundwater processes are longer-term than the storm-event time frame that typifies most surface runoff processes. Groundwater typically flows much more slowly than surface runoff, sometimes taking years to flow the width of a riparian buffer (Bosch and others 1994, Groffman and others 1996, Hubbard and Lowrance 1996). The biological processes that are important for filtering pollutants from groundwater also require longer time periods than the physicochemical processes that dominate buffer impacts on surface runoff.

Performance. Only one report directly compares groundwater runoff from a buffered field area to that from an unbuffered cultivated area of otherwise equal size and condition. Clausen and others (2000) found that installation of a grass buffer reduced nitrate concentration in groundwater runoff by 35%. Reductions of other pollutants, total kjeldahl N (0%), ammonium N (17%), chloride Cl (4%), and total P (−122%), were not statistically significant.

Because substantial N removal from groundwater can also occur under cultivated crops (Clausen and others 2000, Gilliam and others 1979) it is important that direct comparisons are made between cultivated crops and buffers for a proper assessment of change in this function from one condition to the other.

Many studies have reported on change in concentrations of various pollutants in groundwater runoff flowing through buffers 25–125 m wide (Table 3). An overview of these reports indicates that nitrate concentration is commonly reduced by >90% from that entering the buffer to concentrations often <1 mg L^{-1}. Two notable exceptions, however, are reports of outflow that measured as high as 17 mg-N L^{-1} even after a large reduction from input levels (Correll and others 1997, Snyder and others 1998). In contrast, P concentration tends to increase in buffers. Results for other components in runoff have been variable or have received relatively little study.

Several factors have been identified that can determine the level of groundwater filtration by buffers and explain the variability between reports in Table 3.

Table 3. Pollutant reduction in shallow groundwater passing through buffers from cultivated agricultural fields. Reduction is expressed as percent of amount entering the buffer

		Buffer design		Pollutant reduction				
Reference	Location	Vegetation	Width (m)	Component	Buffer inflow (mg L^{-1})	Buffer outflow (mg L^{-1})	Concentration reduction (%)[a]	Note
Lowrance and others (1984a)	GA	forest and pasture	not indicated	nitrate N	2.1–6.3	0.1–0.3	95–98	calculated from Fig. 3, seasonal averages
				total N	2.3–7.5	0.9–1.1	63–87	
				Ca	6.0–10.3	2.1–2.3	63–77	
				Mg	2.6–3.7	1.5–1.9	46–69	
				K	2.2–4.2	1.1–1.9	29–55	
				sulfate S	6.4–11.3	5.8–9.9	0–43	
				chloride Cl	11.1–11.9	10.1–13.0	(−20)–20	
				ammonium N	0.04–0.28	0.02–0.32	(−700)–91	
				total P	not indicated	not indicated	not significant	
				dissolved P	not indicated	not indicated	not significant	
				dissolved organic N	not indicated	not indicated	not significant	
Peterjohn and Correll (1984)	MD	forest	50–75	nitrate N	6.8–7.4	0.1–0.8	90–99	calculated from Table 3, yearly averages.
				total P	0.02–0.13	0.06–0.25	(−313)–(−90)	
				ammonium N	0.07–0.08	0.27–0.44	(−496)–(−265)	
				dissolved organic N	0.15–0.21	0.24–0.27	(−66)–(−29)	
Jacobs and Gilliam (1985)	NC	forest	≈ 50	nitrate N	8.1	0.1	99	calculated from Fig. 3.
				chloride Cl	13.5	7.9	42	
Lowrance (1992)	GA	forest	55	nitrate N	13.5	0.8	94	calculated from Table 1.
				chloride Cl	16.0	7.5	53	
Haycock and Pinay (1993)	United Kingdom	grass	25	nitrate N	3.0–11.5	0.1–0.7	92–97	calculated from Fig. 3.
		forest	38	nitrate N	2.6–9.0	0.1–0.3	92–(>99)	
Jordan and others (1993)	MD	forest	≈ 55	nitrate N	8.5	0.1	99	calculated from Fig. 3.
				sulfate S	2.2	6.8	(−209)	
				phosphate P	0.002	0.075	(−3650)	
				dissolved organic P	0.004	0.030	(−650)	
				ammonium N	0.01	0.06	(−500)	
				dissolved organic N	0.08	0.13	(−63)	
				chloride Cl	23	12	48	
Correll and others (1997)	MD	grass	37	nitrate N	24	13	46	calculated from Fig. 2.
				ammonium N	0.08	0.21	(−163)	
				dissolved organic N	0.14	0.24	(−71)	
				chloride Cl	30	36	(−20)	
		forest	48	nitrate N	24	17	29	
				ammonium N	0.37	0.08	78	
				dissolved organic N	0.14	0.27	(−93)	
				chloride Cl	30	23	23	
Hubbard and Lowrance (1997)	GA	grass + forest	70	nitrate N	9.7–14.2	0.69–1.56	84–94	calculated from Tables 1–3, mature forest.
				ammonium N	0.20–0.29	0.06–0.24	(−15)–74	
				chloride Cl	12.4–14.5	4.3–6.3	57–68	
Lowrance and others (1997a)	GA	grass + forest	68	atrazine	< 0.05–1.51 μg L^{-1}	< 0.05–0.23 μg L^{-1}		derived from Tables 5 and 6.
				alachlor	< 0.05–1.86 μg L^{-1}	< 0.05 μg L^{-1}		
Verchot and others (1997b)	NC	grass + forest	≈ 75	nitrate N	6.2–9.7	0.0–1.0	89–100	calculated from Fig. 4.
				chloride Cl	5.0–9.0	4.0–63.0	(−688)–20	
Snyder and others (1998)	VA	forest	125	nitrate N	9.1	4.7	48	
Clausen and others (2000)	CT	grass	35	nitrate N	4.85	1.44	70	
				ammonium N	0.02	0.02	0	
				total kjeldahl N	0.22	0.26	(−18)	
				total P	0.128	0.099	23	
				chloride Cl	18.77	10.52	44	

[a] Concentration reduction = [(buffer inflow − buffer outflow)/buffer outflow] × 100%.

Field runoff factors. Among groundwater pollutants, concentration of nitrate appears to be most greatly affected within riparian buffers. There are indications that dissolved P is not effectively filtered by buffers (Jordan and others 1993, Lowrance and others 1984a, Osborne and Kovacic 1993, Peterjohn and Correll 1984). In the only study of pesticides to date, concentrations of alachlor and atrazine in groundwater were often too low at the buffer edge to enable an accurate assessment of their removal from groundwater by the buffer (Lowrance and others 1997a).

Long-term interception of field runoff by buffers may lead to a decline in effectiveness. Nutrients accumulated by plant and microbial uptake eventually recycle into mobile forms when those organisms die and decompose, adding to the influx from runoff and potentially saturating nutrient immobilization processes within buffers. Subsequent increase in amounts of P in output from buffers has been indicated (Osborne and Kovacic 1993). An increase of N leaving buffers in groundwater, however, has not been reported, probably because at the sites studied thus far the rate of denitrification to gaseous N responded effectively enough to remove the additional N inputs from mineralization (Groffman and others 1992, Hanson and others 1994).

Spatial patterns of lateral flow of groundwater may be important to the effectiveness of buffers in some locations. At one research site (Hubbard and Lowrance 1997, Lowrance 1992, Lowrance and others 1997a), shallow groundwater flow was well dispersed across the buffer zone (Bosch and others 1994, 1996). At another, the majority of groundwater flowed to the stream through only 12% of the length of the stream border (Cooper 1990). Other spatial heterogeneities of lateral groundwater flow have also been reported (Haycock and Burt 1993). The extent and potential consequences of spatial variability of groundwater flow on buffer performance remains largely unstudied.

Site condition factors. The greatest reduction in nitrate occurs in groundwater that flows slowly through or near the root zone of a buffer. At this location, plant root densities and microbe populations are greatest and conditions are best for promoting immobilization and denitrification.

As soil depth to groundwater increases, capacity for denitrification becomes increasingly limited by low availability of organic matter (Ambus and Lowrance 1991, Jordan and others 1993, Lowrance 1992, Parkin and Meisinger 1989, Schnabel and others 1996). Organic matter decomposition is important to produce anoxic conditions in groundwater that favor activity of denitrifying bacteria. Most organic matter is available near the soil surface. In deeper groundwater, denitrification is similarly limited by low availability of organic matter (Nelson and others 1995). Thus, groundwater flowing below the root zone will probably be less-affected by a buffer (Bohlke and Denver 1995).

Groundwater flow through the root zone typifies most sites reported in Table 3. However, these conditions may not accurately reflect those that would occur under buffers converted from crop land because shallow groundwater typically prevents successful cultivation. Sites that represent candidates for buffer installation are more likely to have somewhat deeper groundwater than many of the sites in Table 3 and, consequently, exhibit somewhat smaller N removal rates. The two studies in Table 3 that reported relatively high N concentration in groundwater outflow from buffers attributed it in part to relatively deeper groundwater flow.

Slower-moving groundwater theoretically should experience a greater degree of nutrient removal within buffers (Warwick and Hill 1988, Phillips 1989a). Pulses of faster-moving groundwater has been reported to transport nitrate farther into a buffer before being attenuated (Haycock and Pinay 1993). However, high overall nitrate removal has been measured over a broad range of flow-through rates, ranging from a few days (Haycock and Pinay 1993) to a few years (Groffman and others 1996, Hubbard and Lowrance 1996).

Seasonal fluctuations in plant growth and microbial activities, water table depth, and input amounts, among other relevant site conditions have been reported (Groffman and others 1992, Lowrance 1992, Osborne and Kovacic 1993, Schnabel and Stout 1994), but their overall impact on filtering by buffers appears to be small or uncertain, at least on the U.S. Eastern Coastal Plain (Lowrance and others 1984a, Peterjohn and Correll 1984, Snyder and others 1998).

Buffer design factors. Location of a buffer is important to bring a root zone into contact with pollutants in groundwater. Riparian zones are potentially good locations, but not all riparian areas will be effective. Ephemeral streams formed by surface runoff and perennial streams with high banks represent riparian situations where root zones may not intercept groundwater. In some landscapes, shallow groundwater may be present, but most pollutant flow to the stream occurs in deeper groundwater under the root zone (Bohlke and Denver 1995).

Vegetation is important for filtering nutrients by taking up nutrients and by producing organic matter that supports microbial activities that immobilize and transform pollutants. Comparisons of different vegetation types has yielded variable results. For nitrate removal, forest has been reported to be less effective (Correll and others 1997, Groffman and others 1991, Schnabel and others 1996), more effective (Haycock and Pinay 1993; Osborne and Kovacic 1993), and no different (Addy and others 1999) than grass vegetation. Perennial grass may be more effective for promoting denitrification than cultivated crops (Sotomayor and Rice 1996). For P removal, forest has been reported to be less effective than grass on an annual basis (Osborne and Kovacic 1993). For transformations and plant uptake of atrazine and alachlor, poplar trees and cultivated corn differed little during the growing season (Peterson and Schnoor 1992).

The importance of reforestation to improve nutrient removal from groundwater flow is uncertain. Trees appear capable of substantial removal of nitrate from the saturated zone (O'Neill and Gordon 1994). Mature forests can take up a large portion of nutrients in runoff (Lowrance and others 1984b, Peterjohn and Correll 1984). However, reforestation of a formerly forested riparian pasture did not improve nitrate removal (Lowrance and others 1995b). Most research on subsurface processes has focused on riparian buffers containing mature forest.

Buffer width has not been studied directly for comparative impacts on subsurface runoff. However, measurements at intermediate distances across several of the riparian buffers identified in Table 3 indicate that most nitrate concentration reduction occurs within 10–30 m of entering the buffers (Haycock and Pinay 1993, Hubbard and Lowrance 1997, Jacobs and Gilliam 1985, Jordan and others 1993, Lowrance 1992, Peterjohn and Correll 1984).

Buffer management factors. Periodic harvest of vegetation may be necessary to maintain nutrient removal over the long term. Plant uptake and storage in above-ground biomass can amount to a large portion of nutrient removal from runoff (Lowrance and others 1984b, Peterjohn and Correll 1984). But, evidence suggests that this storage is only temporary until that vegetation dies and decomposes (Vanek 1991). Harvest can remove these nutrients from a buffer before they can be released. Limited harvest of trees has been performed without causing short-term detriment to filtering functions of a riparian forest buffer (Hubbard and Lowrance 1997, Sheridan and others 1999).

Groundwater drainage patterns may be manipulated to improve buffer performance. Drainage improvements, such as tiles and ditches, allow groundwater runoff to pass through buffer zones without the influence of filtering processes (Jacobs and Gilliam 1985, Osborne and Kovacik 1993, Warwick and Hill 1988). Controlling drainage to reduce bypass flow and to raise the water table into the root zone can improve nitrate removal (Gilliam and others 1979).

4. Reduce bank erosion. A large portion of stream sediment loads in some agricultural regions is derived from bank and channel erosion. Particularly severe channel and bank erosion occurs extensively in central U.S. crop lands. In this region, increased runoff resulting from extensive land conversion to cultivation combined with channel straightening have induced rapid channel incision through deep loess and alluvium leading to high unstable banks and widening by mass failures (Harvey and Watson 1986, Piest and others 1977, Schields and others 1995a, 1995b, Schumm and others 1984, Simon 1989, Smith and Patrick 1991). Sediment may also come from nonincising channels that are widening through bank failure and through surface erosion of banks caused by rainfall, runoff, and storm flows. In the central United States and nearby Ontario, Canada, channel and bank erosion have been estimated to contribute up to 60% of total sediment discharge of streams (Culley and Bolton 1983, Odgaard 1987, USDA 1998). Sediments from eroding banks can also contribute a major portion of the total P load carried by agricultural streams (Svendsen and others 1995).

Vegetation can promote stability of stream banks in several ways. These processes are reviewed by Thorne (1990) and briefly summarized here. In general, surface erosion of banks can be reduced by vegetation in the same manner that contour buffers protect field slopes: Plant shoots protect the surface from raindrop impact and surface runoff flow and retard surface runoff flow, roots bind soil together to increase cohesive resistance to erosion, and permanent vegetation promotes infiltration that reduces the erosive force of surface runoff. Vegetation can provide some resistance to mass failure of banks. Roots reinforce banks by increasing cohesion of bank soil, but only to the depth of most roots, typically less than 1 m. Vegetation and tree debris accumulations on the toeslope can encourage deposition and buttressing of the bank. Rainfall interception and transpiration reduce the water content of the bank soil, thus reducing weight of the bank. By promoting more open soil structure, vegetated soils drain (and therefore lose weight) more rapidly as floods recede. Thorne (1990) considers better drainage to be a major contribution of vegetation to bank stability.

Performance. Quantitative data on sediment reductions in streams attributable to bank stabilization have not been reported.

The absence of natural vegetation on stream banks has been strongly correlated with accelerated erosion (e.g., Beeson and Doyle 1995, Whipple and others 1981). However, field observations do not confirm an across-the-board benefit of vegetation for controlling bank erosion (Thorne 1990). In many situations, the forces causing bank erosion are much greater than the protective capabilities of vegetation (Harvey and Watson 1986, Schumm and others 1984, Shields and others 1995a, 1995b, Thorne 1990).

Site factors. Buffer installation is not likely to retard incision and rapid widening-induced mass failure, especially on larger streams. Based on examination of numerous observations in the United States, a necessary precondition to stabilizing a stream bank with buffer vegetation appears to be stabilization of the toe of the bank (Grissinger and Bowie 1984, Henderson 1986, Shields and others 1995b, Thorne 1990). Once the toe is stabilized, buffer vegetation can markedly aid stability by protecting and strengthening the upper slope of banks (Henderson 1986, Shields and others 1995a, 1995b). Consequently, on incising and rapidly widening streams, buffers are not likely to reduce sediment loads from bank erosion sources (Henderson 1986, Shields and others 1995a, 1995b). In these situations, erosion is dominantly controlled by gravitational slope failure and fluvial processes.

Along relatively stable streams, however, buffers may substantially retard further mass failure and bank surface erosion (Shields and others 1995a, Thorne 1990). Localized rapid erosion that may prevent establishment of a buffer can be stabilized by more specialized vegetative techniques prior to installation of a buffer (e.g., Gray and Leiser 1982, USDA 1996b).

Buffer design factors. In general, herbaceous plants with fibrous root systems are better able to protect banks from surface erosion (Thorne 1990). Grasses on lower banks may also encourage deposition of stream sediment and buttressing (Thorne 1990, Trimble 1997).

Trees with deeper woody roots appear better than herbaceous species for increasing soil shear strength (Waldron and Dakenssion 1982, Waldron and others 1983) that retards mass slope failures. The advantage of trees may be limited, however, to banks with shallower slopes. On high, steep banks, large trees may increase mass failure by adding weight to the bank and creating toppling leverage (Thorne 1990). Other minor, but perhaps locally important problems with tree roots include piping erosion through dead root channels and root penetration that breaks up large cohesive blocks of soil (Thorne 1990).

5. Filter stream water. After agricultural runoff enters stream flow, various processes may further reduce the amount of pollutants that a stream carries. Some of these processes may be enhanced by the installation and management of riparian buffers.

Plant debris produced in riparian areas contributes to accumulations of organic matter in stream bed sediments where it supports microbial processes that denitrify nitrate (e.g., Meyer and others 1988, Hill 1983a) and degrade pesticides (e.g., Isensee 1991, Pionke and Chesters 1973, Stucki and others 1995) that are carried in stream water. Sediments can also chemically adsorb dissolved P from stream water, but the relative importance of organic constituents of sediments in this process is probably negligible (Logan 1982). Pollutants transported in deep groundwater that discharges up through the stream bed might also be filtered within bed sediments before reaching the channel (Hill 1997).

Large woody debris in stream channels can retard stream flow and promote deposition of sediment (Dudley and others 1998, Thorne 1990, Wallerstein and others 1997) and associated nutrients. The dominant proportion of P transported in stream flow is typically associated with sediment (e.g., Logan 1982, Svendsen and others 1995).

Flooding events can transport large volumes of pollutant-laden stream water across floodplains where it is subject to surface and subsurface filtering processes described in previous sections. Stream water can also intermix with shallow groundwater and be filtered in riparian zones during low flow periods (Grimm and Fisher 1984, Komor and Magner 1996, Triska and others 1993).

Pollutant transformation in stream sediments. At this time, there are no research reports available that quantify the impact of buffers on pollutant removal in stream sediments.

Several studies have reported on overall N removal within temperate, agricultural streams (Cooke and White 1987, Hill 1979, 1983b, Hoare 1979, Jansson and others 1994, Kaushik and Robinson 1976). Hill (1997) recently reviewed these studies and concluded that eutrophic streams frequently remove <10% of annual N and P inputs. Though much higher removal (20–80%) is typically observed during low flow periods in summer, low annual rates probably result from short residence times associated with winter high flow periods when most N and P is transported. Nitrogen removal also declines rapidly as streams become larger (Alexander and others 2000, Hill 1988). Among existing reports, the highest annual N removals (17–68%) have been reported in the smallest agricultural streams (Hoare 1979, Kaushik and Robinson 1976).

The degree to which pollutant removal rate can be modified by riparian management, including riparian buffer installation, is probably small compared to overall removal. Detritus from riparian buffer vegetation is only one of several possible sources of organic matter to streams that include crop debris, soil organic matter in eroded sediments, sewage, and autochthonous production in streams. Organic matter–facilitated denitrification is only one of several N removal mechanisms that include uptake by aquatic macrophytes and algae. Some evidence also suggests that denitrification in bed sediments is improved by organic matter additions only where bed sediments contain <1–2% organic matter (Hill 1997). Thus, the capacity to improve nutrient removals by increasing detritus inputs to streams may be limited.

The best opportunity for enhancing stream denitrification may be on smaller streams with finer-textured sediments. For small streams, streamside vegetation is commonly the dominant source of stream organic matter. Riparian contributions to total inputs generally decreases as streams become larger (Cummins and others 1983, Vannote and others 1980). However, even in one seventh-order river, debris input from fairly continuous riparian forest was reasoned to represent the majority of particulate organic matter in river flow (Chauvet and Decamps 1989). Larger amounts of organic matter in streams has been correlated with forested riparian areas than with grass or deforested reaches (Gurtz and others 1988, Sweeney 1993). Capacity for channel denitrification may be greater in streams having silty sediments than those having coarse sands and gravels (Hill 1983a).

Sediment storage in stream channels. The impact of buffers on sediment storage depends on the degree to which they lead to aggradation of channels. Observational studies indicate that large woody debris can act as a significant sediment trap along banks of rivers (Thorne 1990) and promote channel aggradation in some streams (Wallerstein and others 1997). The capacity of smaller channels to accumulate sediment, however, is limited both by their small size and by the need of farmers to maintain adequate drainage. If channels are incising, sediment stored in this way will likely be temporary (Thorne 1990).

Sediment trapping also affects P transport. Sediment-bound fractions represents the dominant proportion of P transported by streams. High total P removal (20–65% of stream load) has been observed during low flow depositional periods, but annual removal can be much smaller (<5%) due to sediment flushing during high flows (Dorioz and others 1989, Svendsen and Kronvang 1993, Svendsen and others 1995).

Riparian filtering of stream water. During floods, one 16-km^2 forested floodplain was reported to trap 10–20% of the suspended sediment load from a large river (Brunet and others 1994). The narrow riparian zone, 10–50 m wide and occupying only 1.1 km^2 of this floodplain, trapped nearly all of this amount (Brunet and others 1994). Along a small agricultural stream, half of the sediment accumulated in the riparian zone over many years was determined to have originated higher up in the watershed (Lowrance and others 1998).

During low-flow periods, stream water that intermixes with groundwater under riparian zones probably will be subject to filtering processes in a similar manner as groundwater field runoff. Streams can vary greatly in capacity for intermixing with groundwater, depending to a large extent on the permeability of stream bed and riparian substrates and on the spatial and temporal patterns of groundwater flow toward streams (Komor and Magner 1996, Kalkhoff 1995, Triska and others 1993). Estimates of how large an impact this function can have on amounts of pollutants in eutrophic streams, however, were not found.

Modeling Studies

Mathematical models offer an alternative way to estimate water quality changes in streams in response to installation of buffers. Accuracy of modeled estimates depends on the degree to which models accurately account for and integrate important functions and performance-governing factors. To date, few modeling studies have directly addressed water pollution response to installation of buffers.

Stream Response

Tim and Jolly (1994) used the AGNPS model to estimate sediment outflow reduction from a watershed in response to surface runoff control by buffers. By their estimate, installation of contour buffers or filter strips throughout their study watershed would reduce sediment yield by 47% and 41%, respectively, and by 71% if both types were installed, compared to existing conditions in their watershed.

Site-Scale Functions

Hamlett and Epp (1994) used the CREAMS model to compare surface runoff from a noncontour tilled field to that from similar fields having a strip crop system (that functions similarly to a contour buffer), contour tillage with grass waterway, or filter strip. When buffer systems were employed, surface runoff volume and transported masses of sediment and total P were reduced by about 25%, 70%, and 80%, respectively. For total N, the strip crop performed best (52% reduction), followed by contour tillage with grass waterway (25%) and filter strip (16%).

Williams and Nicks (1988) used CREAMS to estimate sediment runoff from a cultivated field with and without filter strips. Among their estimations, installation of 15 m-wide filter strips with a good grass stand reduced sediment runoff by 29–46% compared to the completely cultivated field. Other results demonstrated how the modeled estimate of sediment reduction depended on field runoff factors (storm intensity), field conditions (slope and slope configuration), and filter strip design (strip width and density of vegetation).

Buffer impacts on surface runoff depend on what other field management practices are employed. A smaller impact of buffer installation was predicted when installed in contour tilled fields than in noncontour tilled fields (Hamlett and Epp 1994). Installing a strip cropping system had greater impact on total N runoff under an excessive fertilization regime than when strict nutrient management practices were employed (Hamlett and Epp 1994). Interaction among field management and buffer practices appear to be important factors for determining a water quality response to buffer installation.

Information Gaps

Major information gaps remain that limit our capability to estimate a probable impact of buffer installation on NPS pollution in streams and lakes. Gaps in experimental evidence also reflect on deficiencies in modeling approaches because validation and calibration depend on the availability of experimental data.

Gaps in Experimental Evidence

Stream and lake response. There remains a clear need to quantify NPS pollution reduction in streams in response to buffer installation. Despite an abundance of indirect evidence suggesting that this response should occur, the concept remains to be experimentally confirmed and quantified.

Watershed-scale response data are both technically and administratively difficult to produce, which probably accounts for a dearth of such data (Gale and others 1993, Sutton and others 1996). Among other problems, urban areas and livestock production operations in agricultural watersheds can also be major sources of pollutants and mask the impacts that crop land buffers have on overall stream water quality (Garrison and Asplund 1993, Osborne and Wiley 1988).

The value of stream response data, however, is particularly high, because it would fill a critical need to confirm the concept and provide data for validating and calibrating models that, then, could be used to estimate performance at other locations in lieu of further experimental data.

Reduce surface runoff from fields. The sediment runoff control functions of contour buffers appears substantially researched and modeled. Much less has been published regarding the quantitative impacts of grassed waterways and vegetative barriers. For any of these buffer designs, little experimental data exist for field runoff control of nutrients, pesticides, or microbes.

The evidence suggests that there may be optimum combinations of cropping practices and buffer designs for controlling pollutant runoff. As yet, few studies have investigated the potential for capitalizing on synergy between buffer types and other field practices.

Filter surface runoff from fields. Few studies have employed a proper experimental design for directly quantifying how much NPS runoff is reduced by converting cultivated land at field margins to buffers. Most of a relatively large number of studies have been designed to evaluate various factors that can affect pollutant retention properties within buffers themselves, such as width or slope. This emphasis has led to a relative dearth of direct information on response to installation of buffers that would come from a comparison of pollutant runoff from fields containing buffers with that from fields that are unbuffered.

Sediment retention by grass buffers has received far more attention than other pollutants, yielding a relative deficiency of data on alternative vegetation designs and on other important pollutants and the specific processes that affect them.

The extent of nonideal filtering conditions and the magnitude of their impact on buffer performance remains to be investigated. Experimental results reported in Tables 1 and 2 were collected under conditions that should yield a relatively high level of pollutant retention: low to moderate field runoff amounts; shallow sheet flow passing through dense, young grass filters; mostly under high infiltration conditions. Ideal filtering conditions, such as these, may not occur in many field situations (e.g., Dillaha and others 1989). However, the extent to which most fields depart from these ideal conditions and the amount of effect this may have on buffer performance remains largely unstudied, as does design and management modifications that might enhance performance on such sites.

Filter groundwater runoff from fields. Only one study was found that directly compared pollutant attenuation in groundwater under a buffer area to that under a similar cultivated area. Most studies have been designed to evaluate pollutant attenuation and various controlling factors within buffers themselves. Nitrogen removal has been the predominant focus of these studies. Because nitrogen removal from groundwater can be substantial under cropped conditions, direct comparisons between buffers and crops are needed to determine just how much better buffers perform in this regard than the cultivated crops they would replace.

More study is needed of sites that would typify buffer areas converted from cultivation. Water tables within the root zone appear to account for very high N removal observed in many existing studies. However, such sites may not normally be cultivated. Further investigation is needed of pollutant attenuation in groundwater in appropriate environmental settings.

Attenuation of pollutant mass in groundwater flow is underreported. This is probably because it requires additional information on groundwater flow paths and rates that is more difficult to obtain. Such hydrologic information, however, is critical for determining the extent that reduced concentrations in groundwater reflects reduced mass of pollutants entering a stream from groundwater as a whole (Lowrance and others 1997b). Furthermore, the extent of uneven lateral flow to and through buffers to streams and its consequences for pollutant retention remain to be addressed.

Reduce bank erosion. Existing reports suggest that in locations where channels are not incising below the root zone or rapidly widening by mass failure, the prospects appear good for buffers to provide substantial resistance to bank erosion.

Translating bank stabilization to a reduction of NPS sediment loads in streams is limited mainly by the lack of quantitative data on the relative contributions of bank erosion to total stream sediment loads, and on the proportion of sediment contributed by sites that could be stabilized by buffers. Consequences of erosion reduction on stream loads of nutrients, such as phosphorus, also remain to be investigated.

Filter stream water. The potential for buffer installation to promote filtering of pollutants within perennial streams appears to be limited, except seasonally or under local circumstances. Contributions of organic matter to streams can be manipulated through riparian and floodplain management, particularly for large woody debris, which depends on production of forest vegetation. However, research has yet to directly address responses of streamwater filtering processes to establishment and management of buffers and the quantitative impact of these responses on pollutant amounts in agricultural streams. Furthermore, the potential for pollutant removal in intermittent or ephemeral channels remains largely unexplored.

Status of Model Development

To accurately predict a response in stream water quality to installation of buffers, a model must be capable of accurately (i) accounting for all of the important functions of buffers, corresponding crop land, and interactions, and (ii) describing the spatial heterogeneity of NPS runoff and buffer performance-governing conditions throughout a watershed (Schlosser and Karr 1981, Tim 1996).

Site-scale models have been developed that can evaluate field runoff reduction and filtering of surface runoff by buffers. For example, empirical models, such as the USDA Universal Soil Loss Equation and CREAMS, have been used to develop optimum designs for contour buffers (Renard and others 1997; Wischmeier and Smith 1965, 1978) and filter strips (Flanagan and others 1989). Process-based models have been developed to describe sediment trapping by filter strips (e.g., Barfield and others 1979, Hayes and others 1984, Muñoz-Carpena and others 1993, 1999, Tollner and others 1976). A recently developed process-based model, REMM (Lowrance and others 2000), combines groundwater filtering with surface runoff filtering within buffers, but this model remains to be widely validated with experimental data. Other component functions of buffers, such as bank stabilization and filtering stream water, have yet to be modeled in ways that enable estimation of NPS pollution response to installation of buffers.

Watershed-scale water quality models typically are extensions of site-scale models. Consequently, they are also presently limited to describing surface runoff abatement functions and factors of buffers.

Additional performance-governing factors emerge at the watershed scale. Uneven spatial distribution of pollutant sources, flow paths, and site conditions implies the importance of locating buffers in the right places to maximize their interaction with potential NPS pollutants (e.g., Bren 1998, Cooper 1990, Endreny and Wood 1999, Phillips 1989b). Furthermore, the impact that crop land buffers can have on total pollutant amounts in a stream will depend on the proportion of pollutant load that is contributed by crop land. Geographic information systems have been coupled with some water quality models to provide a means for describing important spatial and temporal heterogeneitics across watersheds (e.g., Prato and Shi 1990, Tim 1996, Tim and Jolly 1994). Most watershed models, however, are currently limited in their ability to handle the large amounts of data that are needed to adequately describe such conditions (Tim and Jolly 1994).

Substantial development work remains to produce a model that is capable of accurately estimating the impact of buffer installation on NPS pollution in streams and lakes. Model capabilities are improving as they become more comprehensive and new techniques provide better means to account for variable conditions across watersheds.

Despite the distinct advantages that models can provide, their accuracy still must be tested by comparison to experimental measurements. Consequently, watershed models cannot completely substitute for gaps in experimental data.

Summary and Conclusions

Though U.S. federal and many state programs have been established that encourage farmers to convert some of their cultivated land to buffers, it remains unclear what degree of pollution reduction to expect from this action. In conducting this review of the scientific literature, no experimental study was found that reported on the impact of riparian buffer installation on pollutant levels in streams or lakes.

Indirect experimental evidence that a pollution reduction should occur comes from research on individual functions of buffers. Abundant evidence clearly indicates that buffers can retain pollutants from surface runoff from fields, filter surface and groundwater runoff at field margins, stabilize eroding banks, and contribute to processes that remove pollutants from stream water flow. Much less certain, however, is the degree to which buffers function in these ways compared to the cultivated crop conditions that they would replace. Few existing studies have been designed to make this comparison. Numerous factors, including the nature of the cropping practices in the field, have been identified that can influence the magnitude of this difference.

Mathematical models offer an alternative way to predict water pollution responses to installation of buffers. However, a comprehensive model has yet to be developed that can accurately account for all major functions and performance-governing factors in buffers and corresponding crop land, and their spatial distribution throughout a watershed. At this time, only surface runoff reduction and filtration functions of buffers have been modeled on a watershed basis. A site-scale model has been recently developed that couples both groundwater and surface runoff filtering, but it remains to be widely validated and extended to the watershed scale. Other pollutant control functions of buffers have yet to be modeled.

A great deal of professional judgment is still required to extrapolate our current knowledge of buffer functions into broadly accurate estimates of water pollution abatement in response to buffer installation on crop land. Major experimental data and model development needs remain to be addressed. The greatest need is to produce direct experimental evidence of this response. Such data would confirm the hypothesis, and enable direct testing of watershed-scale prediction models as they become available. Further study of individual pollution control functions is also needed, particularly to generate comparative quantitative evidence for how much they can be manipulated through buffer installation and management.

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