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STREAMSIDE FOREST BUFFER WIDTH NEEDED TO PROTECT STREAM WATER QUALITY, HABITAT, AND ORGANISMS: A LITERATURE REVIEW[1]

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ABSTRACT: This literature review addresses how wide a streamside forest buffer needs to be to protect water quality, habitat, and biota for small streams (≤~100 km^2 or ~5th order watershed) with a focus on eight functions: (1) *subsurface nitrate removal* varied inversely with subsurface water flux and for sites with water flux >50 l/m/day (~40% avg base flow to Chesapeake Bay) median removal efficiency was 55% (26-64%) for buffers <40 m wide and 89% (27-99%) for buffers >40 m wide; (2) *sediment trapping* was ~65 and ~85% for a 10- and 30-m buffer, respectively, based on streamside field or experimentally loaded sites; (3) *stream channel width* was significantly wider when bordered by ~25-m buffer (relative to no forest) with no additional widening for buffers ≥25 m; (4) *channel meandering and bank erosion* were lower in forest but more studies are needed to determine the effect of buffer width; (5) *temperature* remained within 2°C of levels in a fully forested watershed with a buffer ≥20 m but full protection against thermal change requires buffers ≥30 m; (6) *large woody debris* (LWD) has been poorly studied but we infer a buffer width equal to the height of mature streamside trees (~30 m) can provide natural input levels; (7, 8) *macroinvertebrate and fish communities*, and their instream habitat, remain near a natural or semi-natural state when buffered by ≥30 m of forest. Overall, buffers ≥30 m wide are needed to protect the physical, chemical, and biological integrity of small streams.

(KEY TERMS: riparian ecology; nonpoint source pollution; temperature; nutrients; best management practices; sediment; rivers/streams; macroinvertebrates; fish; streamside forest buffer; nitrate; streambank stability; woody debris.)

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INTRODUCTION

The Environmental Protection Agency (USEPA, 2013) recently reported that 55% of the river and stream length in the United States (U.S.) is in poor condition. Streamside disturbance and poor riparian vegetation cover were the most widespread stressors, reported in 20 and 24%, respectively, of the streams and rivers in the study. Streamside forests have historically formed the natural interface between hillslope and aquatic processes for most watersheds worldwide. This was particularly true in North America, where even streams in grassland prairies were apparently bordered by forest (West and Ruark, 2004). Removal of those natural streamside forests greatly alters the physical, chemical, and biological dynamics of streams, as well as the structure and function of their ecosystems (Hynes, 1975; Gregory *et al.*, 1991; Sweeney, 1993; Naiman and Décamps,

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1997; Sovell *et al.*, 2000; Allan, 2004; Sweeney *et al.*, 2004; see Dosskey *et al.*, 2010 for recent review). Consequently, the impacts of deforestation on streams and rivers have led to a global movement to protect and restore streamside forests to improve stream habitat, water quality, and biodiversity (Bernhardt *et al.*, 2005). Maintaining and/or reestablishing streamside forests can reduce nutrient inputs to streams, contribute terrestrial animals, leaves, and other organic detritus to the food base, affect instream temperatures and algal production through shading, contribute large woody debris (LWD) and root wads as habitat and cover for aquatic life, and affect the physical channel characteristics and stability through bank erosion and sedimentation (see Horwitz *et al.*, 2008 for review).

Afforestation, or the process of restoring forest on cleared and/or cultivated land, is now considered "best management practice" (BMP) in the U.S. and elsewhere for both streamside and highly erodible upland areas (i.e., conservation buffers, *sensu* Bentrup, 2008). The framework for these BMPs arose in North America in the 1970s, with the advocacy of wide zones of streamside vegetation to protect streams from logging activities (Lantz, 1971; FWPCA, 1972), which subsequent research demonstrated to be effective (Newbold *et al.*, 1980). The Food Security Act (1985) (the "Farm Bill") expanded the practice by establishing the conservation reserve program and funding the establishment of "stream borders" of vegetation to reduce erosion. Welsch (1991) provided the first formal prescription in North America for reestablishing a streamside forest as a "buffer" to protect and enhance water resources from land-use impacts in the watershed. The imperative to implement buffers gained traction from research showing that "the quality of streamside forests" was likely the "single most important factor altered by humans that affects the structure and function, and ultimately water quality, of the streams providing water to coastal embayments" (Sweeney, 1992).

Richardson *et al.* (2012) reviewed the evolutionary history of fixed width, streamside forest buffers from both environmental and regulatory perspectives. Past research on buffers has focused primarily on four broad issues: (1) quantifying the level of protection and instream enhancement afforded by a streamside buffer; (2) measuring how those levels vary with grass or forest as the vegetation of choice; (3) debating where and when a streamside buffer of forest or grass is BMP; and (4) determining what buffer width qualifies as BMP.

Our focus here is to review only the literature directly related to "how wide" a streamside forest should be to assure a natural setting for the stream, protect water quality, and enhance stream and river ecosystems and ecosystem services. We emphasize streamside forests because, while they are not necessarily BMP for every reach of stream worldwide, they are BMP in most watersheds where they historically existed in their natural condition (*see* Sweeney and Blaine, 2007 for discussion). For our evaluations of nitrogen and sediment removal, we have added data from grass and shrub buffers to achieve a critical mass of publications to evaluate. However, as Wenger's (1999) review and Wenger and Fowler (2000) point out, although a streamside buffer planted in grass can adequately perform many functions (including trapping sediment and other contaminants), effective performance across all functions requires a buffer covered with forest. Thus, the primary emphasis here is on literature related to the width of streamside land covered with forest to protect streams. Our review considers streams of Strahler (1957) order 1-5, corresponding to watershed areas of ~0.05 to ~100 km^2.

Streamside forests protect and enhance water quality and stream ecosystem health by: (1) blocking the entrance of pollutants into streams and rivers; and (2) enhancing the stream's physical, chemical, and biological characteristics that enable it to provide ecosystem services (*sensu* Daily and Ellison, 2002), such as sequestering carbon, metabolizing organic matter, and degrading and processing of pollutants. As Sweeney and Blaine (2007) pointed out, the ability of streamside forest buffers to enhance instream habitat, chemistry, and biology, as well as ecosystem services (e.g., nitrogen uptake and processing), makes them a BMP for mitigating both point and nonpoint source pollution. Thus, we divide the literature and discussion into two compartments: the upland (see section Streamside Forest Buffers as Barriers to Upland Sources of Nitrogen and Sediments) and instream (see section Streamside Forest Buffers as Promoters of Stable, Healthy, and Functional Stream Ecosystem) characteristics of streamside forest buffers.

STREAMSIDE FOREST BUFFERS AS BARRIERS TO UPLAND SOURCES OF NITROGEN AND SEDIMENTS

The most familiar aspect of streamside forest buffers is their role in creating sufficient space to intercept pollutants created by upland activity before they enter the stream. It is intuitive that wider buffers with more vegetation have greater potential for intercepting, sequestering, degrading, and processing pollutants. What is not intuitive is how efficiently a buffer can abate pollution per unit width and how proximity to the stream and the quality and quantity of vegetation affect that efficiency (discussed below).

While we considered a long list of pollutants for this review, we focus on nitrogen and sediment as the two substances with the strongest supporting literature base. Moreover, widespread hypoxia and anoxia, habitat degradation, alteration of food web structure, and loss of biodiversity in coastal oceans is due to increased inputs of nitrogen from streams and rivers (Howarth, 2008), and sediment has been — and appears still to be — the number one pollutant (by volume) in U.S. waterways (Waters, 1995; Dodds and Whiles, 2004). This limited focus is not meant to diminish the importance of other pollutants.

Nitrogen

The potential for streamside forests to improve water quality by removing nitrogen from upland sources such as agriculture has long been recognized (Asmussen *et al.*, 1979), and because excessive nitrogen concentrations contribute to such problems as anoxia in the Chesapeake Bay (Officer *et al.*, 1984) and the Gulf of Mexico (Diaz and Rosenberg, 2008), this function of streamside buffers has received a great deal of attention.

Asmussen *et al.* (1979) observed that nitrate concentrations in streams draining agricultural areas in Georgia's coastal plain were far lower than those in the fertilizer-enriched groundwaters that fed the streams. They proposed that the forested streamside areas intercepted and removed nitrate from the groundwater as it moved toward and emerged in the streams. Subsequent investigations in Georgia (Lowrance *et al.*, 1984a, b) and elsewhere in the Southeastern Coastal Plain (Peterjohn and Correll, 1984; Jacobs and Gilliam, 1985; Jordan *et al.*, 1993) confirmed and quantified the nitrogen removal, showing that buffers ranging in width from 50 to 90 m could remove 80-95% of the groundwater nitrate as it passed through the buffer.

Based on these early studies, streamside forest buffers became widely recommended as a BMP (e.g., Welsch, 1991). Their effectiveness has been evaluated in a wide range of settings, with the result that the importance of denitrification as the primary mechanism of nitrate removal has been established, and the understanding of the conditions under which streamside buffers are effective has advanced significantly. Because these advances have been well covered by several reviews (e.g., Hill, 1996; Lowrance *et al.*, 1997; Dosskey, 2001; Gold *et al.*, 2001; Vidon and Hill, 2006; Passeport *et al.*, 2013), they are only briefly summarized here.

The streamside area can remove dissolved nitrogen from the subsurface water passing beneath it through two primary processes: denitrification and plant uptake. Denitrification, the reduction in nitrogen oxides (NO_3^- and NO_2^-) to the gases nitric oxide, nitrous oxide and dinitrogen, has been widely recognized as the major removal pathway (Hill, 1996, 2000), although uptake by plants has also been shown to be significant (Peterjohn and Correll, 1984). A third pathway, involving microbial uptake contributing to an accumulation of soil organic matter, has been suggested, but its significance remains to be demonstrated (Gold *et al.*, 2001). Although denitrification transfers nitrogen to the atmosphere, plant and microbial uptake retain the nitrogen within the streamside area where its long-term accumulation may be limited. Denitrification is a respiratory process that requires both a source of organic carbon and local depletion of oxygen concentrations that allow the nitrate to serve as the terminal electron acceptor. Field studies have associated high rates of denitrification with low soil-water oxygen (Sexstone *et al.*, 1985), high concentrations of dissolved organic carbon (Hill *et al.*, 2000), or a high organic content of the soils (Dhondt *et al.*, 2004). These conditions have been shown to occur in hydric soils (Gold *et al.*, 2001) and where soil moisture is high or the water table is near the surface (Groffman *et al.*, 1992; Hefting *et al.*, 2004). Denitrification rates tend to be highest within a few centimeters of the surface (Clément *et al.*, 2002), but can also be significant at depths of up to a few meters (Vidon and Hill, 2004a), where subsurface water may flow through organic carbon in alluvial deposits. It has been difficult to link denitrification rates measured in the field with mass-balance measurements of nitrogen removal in part because rates of denitrification are spatially heterogeneous, occurring as "hot spots" in areas where conditions support the process (Groffman *et al.*, 2009).

Numerous studies have quantified the removal of nitrogen by streamside buffers over a range of environments and buffer widths. Removal is commonly reported as percentage efficiency, or the proportion of nitrogen in subsurface flow removed within the area beneath the buffer. Mayer *et al.* (2007) compiled 65 estimates of subsurface removal efficiency among buffers ranging in width from 1 to 220 m. The majority of these were forested or partially forested buffers, but herbaceous buffers were included as well. Reported removal efficiencies ranged from <0 (i.e., where the buffer appeared to be a nitrogen source) to 100%, with a median removal rate of 91%. When these 65 estimates were included with an additional 23 estimates of nitrogen removal from surface flow in a meta-analysis, Mayer *et al.* (2007) found that buffer effectiveness increased with buffer width, with width explaining a small ($R^2 = 0.09$) but significant ($p < 0.01$) fraction of the variance in removal efficiency. However, when the 65 subsurface studies were considered separately, the effect of buffer width was not significant ($p > 0.3$). The absence of significant explanatory power for width does not necessarily imply that width does not play an important role. As Mayer *et al.* (2007) noted, other factors may obscure a clear influence of width on the collection of studies taken as a whole.

Several studies have provided conceptual or semi-quantitative approaches that link buffer efficiency to physiographic, landscape, edaphic, hydrologic, and geologic factors (e.g., Lowrance *et al.*, 1997; Gold *et al.*, 2001; Rosenblatt *et al.*, 2001; Baker *et al.*, 2006; Vidon and Hill, 2006). Yet, with the notable exception of Vidon and Hill (2006), these analyses have focused on buffer function rather than width and most have been limited either in geographic coverage or quantitative evaluation. Explaining removal distances ranging from a few to nearly 200 m, Vidon and Hill (2006) identified several major factors that influence the width necessary for a buffer to achieve ~90% removal. These included topography and the depth and texture of permeable soils and sediments in both the upland and streamside areas. Their conceptual model suggests that in most cases >90% removal can be achieved within 20 m if slopes are <5%, impermeable layers occur at intermediate depth, and soils and sediments are fine grained. In contrast, they suggested that >30 m is required where the permeable layer is deeper and coarse (sand and gravel) materials dominate.

Vidon and Hill (2006) further suggested that those factors which yield high nitrogen removal in narrow buffers (e.g., <20-m) — small slopes, fine textured soils or sediments, and shallow confining depths — are also likely to restrict water flow through the buffer. Such buffers, despite their high efficiency, may not constitute large nitrogen sinks because they are able to receive and process a limited flux of nitrogen-loaded water. Vidon and Hill's (2006) observations have two implications. First, their conceptual model might be converted to a quantitative model if subsurface water flux is a suitable surrogate for the factors influencing removal efficiency (i.e., if removal efficiency can be shown to vary with water flux). Second, although water flux may vary from site to site, the aggregate of water fluxes in a watershed or region constitute the base flow of that watershed or region. Thus, given a relation between nitrogen removal and water flux, and knowing the average water flux at a watershed or regional scale, it may be possible to identify an effective buffer width appropriate to that watershed or region.

To test whether removal efficiency can be related to water flux, we followed the lead of Vidon and Hill (2006), compiling studies for which estimates of the water flux passing through the streamside area were reported or could be calculated. We included 30 reports of subsurface nitrate removal (Table 1), expanding on the 16 identified by Vidon and Hill (2006) and, in one case (Hoffmann *et al.*, 2006) using a more recent report on a different site at the same stream. For our analysis, water flux, q_L (l/m/day), is here defined as the subsurface flow into the buffer per unit downstream length of buffer. For each study, we present values for the width of the buffer zone and the subsurface nitrogen removal efficiency. For some studies, efficiency was not explicitly reported but could be calculated from reported data. We averaged seasonal differences, weighting them by water flux where possible, and opted for dilution-corrected estimates where available.

Among the 30 studies reporting water flux (Table 1), the median nitrate removal efficiency was 89%. Removal efficiency was not significantly correlated with buffer width (Figure 1, $r = 0.30$, $p = 0.11$), nor was it affected by vegetation type (ANOVA, $p = 0.60$). These results are similar to those of Mayer *et al.* (2007). To compare removal among sites with differing buffer widths, we expressed nitrate removal as a rate per unit distance into the buffer, k_N (m^{-1}), assuming that downslope nitrate flux declines exponentially through the buffer, according to the equation:

$$r(x) = \exp(-k_N \cdot x) \tag{1}$$

in which $r(x)$ is the proportion of the water flux entering the upslope boundary of the buffer that reaches x meters into the buffer. Our assumption of an exponential (or first-order) loss rate may be an oversimplification, as observed patterns of longitudinal declines vary widely. We chose the exponential model in preference to a linear (zero-order) model because several studies have shown steeper nitrate declines near the upslope end of the buffer (e.g., Peterjohn and Correll, 1984; Vidon and Hill, 2006).

In terms of $r(x)$, the removal efficiency can be written as:

$$E_N = 100 \cdot (1 - r(w)) \tag{2}$$

in which w is buffer width. Combining Equations (1) and (2) yields

$$k_N = -\log_e(r(w))/w = -\log_e(1 - 0.01 \cdot E_N)/w \tag{3}$$

We found that k_N varied inversely with water flux (q_L, Figure 2). The equation

$$k_N = \alpha/q_L \tag{4}$$

with $\alpha = 2.72 \pm 0.41$ (SE) l/m^2/day, as estimated by nonlinear regression (NLIN Procedure, SAS/STAT,

TABLE 1. Nitrogen Removal Efficiency and Width of Buffer from Studies That Also Reported Subsurface Water Flux.

Study	Site	Vegetation	Buffer Width (m)	Water Flux (l/m/day)	% Nitrate Removal
Balestrini *et al.* (2011)	Bedollo, Italy	Herb/forest	6.5	14	95
Butturini *et al.* (2003)	Fuirosos, Spain	Forest	18	12	78
Clément *et al.* (2003)	Petite Hermitage, France	Herb/shrub	40	34	75
Clément *et al.* (2003)	Petite Hermitage, France	Forest	25	34	95
Clément *et al.* (2003)	Petite Hermitage, France	Herb	40	34	90
Cooper (1990)	Scotsman Valley, New Zealand	Herb	10	78	53
Correll *et al.* (1997)	German Branch, Maryland	Forest	45	100	38
Correll *et al.* (1997)	German Branch, Maryland	Herb	45	100	44
Hanson *et al.* (1994a, b), Simmons *et al.* (1992)	Sand Hill Brook, Rhode Island	Forest	31	25	95
Hefting *et al.* (2003, 2006)	Hazelbekke, Netherlands	Forest	25	50	38
Hefting *et al.* (2003, 2006)	Ribbert, Netherlands	Herb	23	50	63
Heinen *et al.* (2012)	Beltrum, Netherlands	Herb	5	41	0
Hoffmann *et al.* (2006)	Voldby fen, Denmark	Herb	24	737	64
Jordan *et al.* (1993)	Southeast Creek, Maryland	Forest	60	45	95
Lowrance *et al.* (1984a, b)	Little River, Georgia	Forest	55	50	83
Lowrance (1992a), Bosch *et al.* (1996)	Gibbs Farm, Georgia	Forest	55	70	94
Maître *et al.* (2003)	Morand River, Switzerland	Forest	32	100	55
Messer *et al.* (2012)	Fishing Cr, North Carolina	Forest	60	73	27
Messer *et al.* (2012)	Fishing Cr, North Carolina	Forest	45	68	88
Newbold *et al.* (2010)	Stroud Preserve, Pennsylvania	Herb/forest	37	370	26
Peterjohn and Correll (1984)	Rhode River, Maryland	Forest	50	160	89
Vellidis *et al.* (2003)	Dairy wetland, Georgia	Forest	38	23	73
Vidon and Hill (2004b, c)	Eramosa, Ontario	Forest	220	390	98
Vidon and Hill (2004b, c)	Ganatsekiagon, Ontario	Herb	25	244	60
Vidon and Hill (2004b, c)	Maskinonge, Ontario	Herb/forest	45	72	99
Vidon and Hill (2004b, c)	Speed River, Ontario	Herb/forest	66	66	97
Vidon and Hill (2004b, c)	Road 10, Ontario	Herb	30	44	99
Vidon and Hill (2004b, c)	Hwy 27, Ontario	Herb/forest	33	30	98
Vidon and Hill (2004b, c)	Boyne River, Ontario	Forest	204	320	94
Wigington *et al.* (2003)	Lake Creek, Oregon	Herb	39	17	98

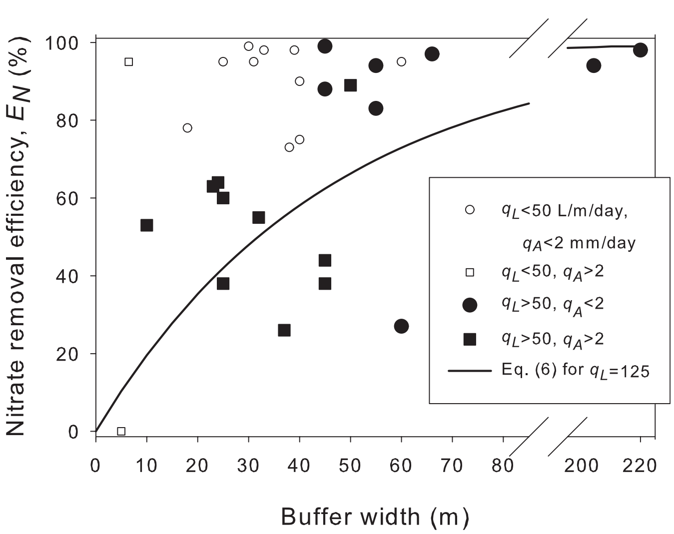

FIGURE 1. Nitrate Removal Efficiency *vs*. Buffer Width. Solid fill designates buffers receiving water flux $q_L \geq 50$ l/m/day. Square symbols designate areal loadings $q_A > 2$ mm/day. The curve is Equation (6) for $\alpha = 2.72$ l/m^2/day and $q_L = 125$ l/m/day.

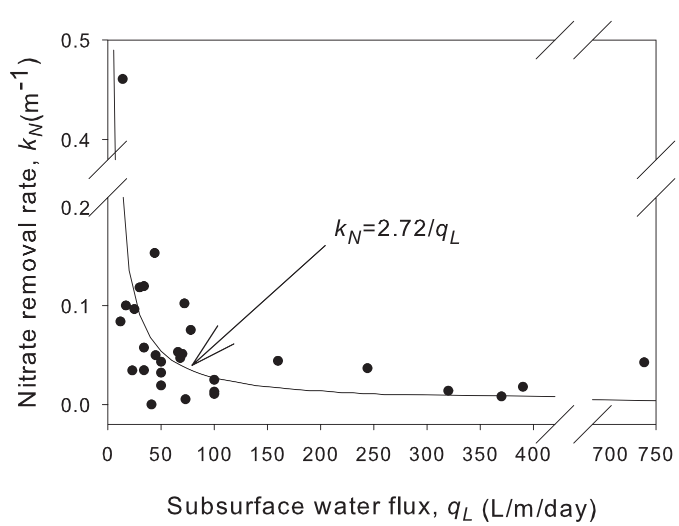

FIGURE 2. Nitrate Removal Per Unit Distance *vs*. Subsurface Water Flux. The curve is the least squares fit of Equation (4).

version 9; SAS Institute, Cary, North Carolina) explained 36% of the variance in k_N ($p < 0.01$).

To address whether buffer width explained significant variation in removal efficiency once water flux was taken into account, we combined Equations (1) and (4) to obtain $\log_e(r(x)) = -\alpha x/q_L$ which, after taking logs a second time yields the equation:

$$r_{LL} = \log_e(\alpha) + \log_e(w) - \log_e(q_L) \tag{5}$$

in which $r_{LL} \equiv \log_e\ [-\log_e\ (r(x)]$. Multiple regression of r_{LL} against $\log_e(x)$ and $\log_e(q_L)$ (GLM Procedure, SAS/STAT, version 9; SAS Institute) yielded a significant model ($R^2 = 0.24$, $p = 0.03$), with significant ($p < 0.05$) regression coefficients for both width and water flux. In a Type I analysis, with water flux entered first, $\log_e(w)$ explained 14% of the total variance and $\log_e(q_L)$ explained 10% of the total variance in r_{LL}. Thus, it appears that the failure of univariate models or simple correlations to detect a significant influence of buffer width on removal efficiency (Mayer *et al.*, 2007; and Figure 1) can be explained by the large obscuring influence of water flux.

Combining Equations (1, 2, and 4) yields a simple model for removal efficiency:

$$E_N = 100(1 - \exp(-\alpha w/q_L)) \tag{6}$$

E_N calculated from Equation (6) explained 37% of the variance in measured E_N (Table 1), although it should be noted that this result is essentially the same as obtained from the nonlinear regression for α in Equation (4).

We caution that Equation (6) does not include many of the factors known to influence nitrate removal, as discussed above, and leaves 63% of the variance unexplained. It does demonstrate a large influence of subsurface water flux, which may act as a surrogate for other factors such as the texture, organic content, and depth of riparian soils. In this sense, Equation (6) provides a quantitative interpretation of Vidon and Hill's (2006) conceptual model. It also supports their inference that sites with low water flux require narrower buffers to achieve a given efficiency than sites where water flux is high.

Subsurface water flux is not easily measured and, as per Table 1, it may vary greatly from site to site. However, on the watershed (catchment) and regional scale, the average water flux is determined by base flow and drainage density, and is more easily obtained. Base flow in the eastern U.S. ranges roughly from 0.1 to 0.3 $m^3/yr/m^2$ of watershed area (or m/yr), and is near 0.2 m/yr for most of the land area (Santhi *et al.*, 2007). Drainage density has been estimated at 1.2 km per km^2 (referred to simply as 1.2 km^{-1}) (Woodruff and Hewlett, 1970; Baker *et al.*, 2007) based on 1:24,000-scale mapping. Using higher resolution maps, Baker *et al.* (2007) estimated an average drainage density of 2.2 km^{-1} for the Chesapeake Bay basin. A runoff of 0.2 m/yr together with a drainage density of 2.2 km^{-1} yields a water flux of 125 l/m/day as a reasonable estimate of the daily flux of groundwater reaching each side of a meter length of stream.

From Equation (4), an average water flux of $q_L = 125$ l/m/day corresponds to a removal rate, k_N, of 0.022 m^{-1} and, from Equation (6), to efficiencies of 35% for a 20-m wide buffer, 48% for a 30-m buffer, and 90% for a 100-m buffer (Figure 1). These efficiencies are broadly consistent with those reported by Weller *et al.* (2011), who analyzed the relation between stream water nitrate concentrations and land use in 321 watersheds in the Chesapeake Bay watershed. They found that buffers in the Mid-Atlantic Piedmont portion of the Bay watershed had a median width of 34 m and an inferred mean nitrate removal efficiency of 35%. This is less than the 52% which Equation (6) predicts for $q_L = 125$ l/m/day. However, Weller *et al.* (2011) identified only buffers that intercepted flow paths from croplands and so it is likely that their buffers represented higher-than-average water flux sites. Their observed 35% efficiency would correspond to a q_L of 215 l/m/day, or 1.7 times the average. Weller *et al.* (2011) also found that buffers in the Mid-Atlantic Coastal Plain of the Bay watershed had a median width of 109 m and a removal efficiency of 95%. This agrees well with Equation (6) which, for $q_L = 125$ l/m/day, estimates 91% efficiency.

Using the Chesapeake Bay watershed average q_L of 125 l/m/day, Equation (6) plots a curve (Figure 1) that falls below most of the efficiency measurements in Table 1 (and Figure 1). Of the 30 studies, 24 were conducted on sites with $q_L < 125$ l/m/day and thus are predicted to be higher than the $q_L = 125$ curve. The median q_L for the 30 studies was 58 l/m/day, or only half of our admittedly rough estimate for the Chesapeake Bay watershed as a whole. We offer three possible explanations for the relatively low water flux of the majority of studies. First, many studies investigated only shallow groundwater movement and some may have missed water flux that reached the stream via deeper pathways. Flow through deeper pathways may bypass zones of intensive nitrogen removal thus diminishing the overall effectiveness of the buffer (Böhlke and Denver, 1995; Burt *et al.*, 1999; Puckett and Cowdery, 2002; Puckett *et al.*, 2002; Spruill, 2004; Heinen *et al.*, 2012). Second, studies of specific catchment area (Bren, 1998; Burkart *et al.*, 2004) suggest that groundwater flow to streams may be largest in small headwater streams and areas of convergent slopes, conditions that may be underrepresented among reported removal estimates. Third, there may have been a bias toward establishing studies at sites rich in organic matter and prone to anoxia and thus having a high denitrification capacity.

We used a threshold of 50 l/m/day to identify sites where water fluxes were low enough to make their potential impact on watershed-scale water quality relatively less important. Twelve such low-flux sites were identified (Figure 1) with a median q_L of 32 l/m/day, and a median removal efficiency of 95%. With the low-flux sites excluded, nitrate removal efficiency among the remaining 18 sites ($q_L \geq 50$ l/m/day) correlated with buffer width ($r = 0.49$, $p = 0.046$). Among these higher flux sites, the highest efficiency reported from buffers <40 m wide was 64%, whereas the median efficiency of wider buffers was 89%. Efficiencies of the wide-buffered sites, however, varied widely ranging from 27 to 99%. Taken together, these results suggest that among sites that supply most of the stream water, high efficiencies (>65%) cannot be expected for buffers <40 m wide but are likely to be attained for buffers wider than 40 m.

We caution that our inference that narrow buffers yield high removal efficiency only where water flux is low is based on the relatively limited number of cases for which water flux is known. High removal rates have been reported for a number of narrow buffers where the water flux is not known (e.g., Brüsch and Nilsson, 1993; Osborne and Kovacic, 1993; Cey *et al.*, 1999; Borin and Bigon, 2002; Dukes *et al.*, 2002; Schoonover and Williard, 2003) and more information from cases such as these could substantially affect the analysis.

For the higher flux sites ($q_L > 50$ l/m/day), Figure 1 displays a potentially puzzling divergence of E_N into two clusters, one with $E_N > 80\%$, the other with $E_N < 65\%$. We can account for much of this divergence by considering the ratio of water flux to buffer width, given by q_A (mm/day) = q_L/w. The ratio q_A can be thought of as an areal loading, representing the subsurface water flux that enters the buffer from upslope as though it were applied uniformly over the buffer land surface. As shown in Figure 1, the high efficiency cluster ($E_N > 80\%$) was characterized by low areal loadings ($q_A < 2$ mm/day) and the low efficiency cluster by high loadings ($q_A > 2$ mm/day), with one exception in each cluster. Thus, the wide variation in E_N among buffers in the width range of 30-50 m reflects, in part, variation in water flux. For our "average" water flux of 125 l/m/day, a q_A of 2 mm/day would be attained with a 63-m buffer, suggesting that >80% removal efficiency at the watershed level may be difficult to achieve.

We further caution that if lateral subsurface fluxes are highly variable, then for a given buffer width, the aggregate catchment or regional-scale efficiency may be lower than Equation (6) would suggest. For, example, if 25% of stream length in a watershed contributes only 5% of the total streamflow, the aggregate efficiency of a 100-m buffer would drop from 89 to 83%. Given that specific catchment area tends to be highest for headwater streams (Bren, 1998; Burkart *et al.*, 2004), greater regional-scale reduction in nitrogen removal might be achieved with wider buffers on headwater streams.

This review has focused on the removal of nitrogen from subsurface flow. Buffers also remove nitrogen from surface runoff. In agricultural cropland settings surface runoff may account for <10% (Lowrance, 1992b; Clausen *et al.*, 2000; Newbold *et al.*, 2010) to 25% (Peterjohn and Correll, 1984) of the total annual nitrogen load to the buffer. The primary removal mechanisms appear to be infiltration (e.g., Borin *et al.*, 2005; Lowrance and Sheridan, 2005) and deposition of the particulate fractions (e.g., Dillaha *et al.*, 1988; Lee *et al.*, 2003). Mayer *et al.*'s (2007) meta-analysis yielded surface removal estimates of 45, 52, and 81% for buffer widths of 20, 30, and 100 m, respectively. However, another meta-analysis (Zhang *et al.*, 2010) arrived at much higher removal efficiencies of 73% for a 10-m buffer increasing to 88% at 20 m and reaching a plateau of 92% at 35 m. The relatively large difference between these studies is not easily explained, but taken together they suggest that a width of 20-30 m should be reasonably effective for removal of nitrogen from surface runoff.

In summary, we found that that nitrogen removal per unit width of buffer varied inversely with subsurface water flux. Where water flux is low, narrow buffers can provide high removal efficiencies, but such sites account for relatively little of the watershed- and regional-scale base flows in streams and, therefore, can have relatively little effect on overall water quality. Among sites with water fluxes sufficient to contribute substantially to streamflow, the median nitrate removal efficiency was 55% (range: 26-64%) for buffer widths <40 m, and 89% (range: 27-99%) for buffer widths >40 m. Our simple model developed from these observations, when applied to a site with average water flux, predicts removal efficiency of 48% for a 30-m buffer, increasing to 90% for a 100-m buffer. Given the wide variation among sites, however, we suggest that the best interpretation of this model is that effective nitrogen removal at the watershed scale probably requires buffers that are at least 30 m wide and that the likelihood of high removal efficiencies continues to increase in buffers wider than 30 m.

Sediments

Vegetation, from grass to forest, can protect water quality by intercepting sediments flowing overland from upslope land disturbances such as forestry and agriculture (Trimble and Sartz, 1957; Haupt, 1959; Haupt and Kidd, 1965; Wilson, 1967; Asmussen *et al.*, 1977). A streamside area is variously referred to as a vegetated filter strip, a buffer strip, a riparian buffer, or a streamside forest buffer, depending in part upon setting and vegetation type. In this section we use "buffer" as an inclusive term, qualifying it as streamside to clarify location, and as grass, shrub, or forest, as appropriate.

In his review of early literature on grass buffers in agricultural settings, Dosskey (2001) concluded that they removed 40-100% of the sediments that entered them from cultivated fields. Buffer widths ranged from 0.5 to 20 m, and while Dosskey made no inferences about the relative merits of wider buffers, he did discuss several factors that influence buffer effectiveness and contribute to their highly variable performance. These included soil type, vegetation, slope, sediment load, rainfall intensity, and microtopography.

Liu *et al.* (2008) conducted a meta-analysis of 85 estimates of sediment removal by vegetated buffers. All were from agricultural settings and the majority involved relatively small experimental plots, many employing simulated rainfall or simulated additions of sediments. Relatively few of the studies were conducted in streamside settings and nearly all the buffers consisted of grass rather than forest. They found that sediment removal efficiency (E_S, the percentage of inflowing sediment trapped within the buffer) increased with buffer width according to the relationship:

$$E_S = 13.4\log_e(w) + 56.9 \tag{7}$$

in which w (m) is buffer width. This equation, which explained 34% of the variance, predicts that E_S increases from 78% for a 5-m wide buffer to 88 and 97% at widths of 10 and 20 m, respectively. Liu *et al.* (2008) further found that removal efficiency was maximized at a slope of 9%. Two subsequent meta-analyses, by Yuan *et al.* (2009; 93 estimates) and Zhang *et al.* (2010; 81 estimates), were similar in scope to Liu *et al.* (2008) and included most of the studies reviewed by Liu *et al.* (2008). Their results were similar to those of Liu *et al.* (2008), with average efficiencies for a 10-m buffer of 84% (Yuan *et al.*, 2009) and 90% (Zhang *et al.*, 2010). Both concluded that the additional benefits of a buffer wider than 10-m may be limited. Both Yuan *et al.* (2009) and Zhang *et al.* (2010) included enough results to compare forested buffers to grass buffers, and both found that forested buffers were about as effective as grassed buffers.

The majority of studies reviewed by Liu *et al.* (2008), Yuan *et al.* (2009), and Zhang *et al.* (2010) were conducted on rectangular, physically confined plots that allowed for replication and uniformly distributed overland flow, and in many cases, they employed simulated applications of rain, sediments, or both. While replicated plots allow strong experimental designs for comparisons of such factors as buffer width, slope, and vegetation, they also create several constraints that limit their applicability to streamside buffer settings. First, the delivery of overland flow to the plots is typically limited by small upslope source areas (e.g., <25 m upslope distance) or, in the case of simulated applications, relatively small design storms (e.g., rainfall less than the 10-year return interval). Larger storms may reduce E_S by transporting sediment at greater depths and velocities (Dillaha *et al.*, 1989a; Schellinger and Clausen, 1992; Arora *et al.*, 1996; Daniels and Gilliam, 1996; Robinson *et al.*, 1996; Blanco-Canqui *et al.*, 2006; Gharabaghi *et al.*, 2006) and it is the larger storms (e.g., those with return intervals of 10 years or more) that deliver most of the sediments to waterways in the long run (Edwards and Owens, 1991; Langdale *et al.*, 1992; Larson *et al.*, 1997). The second constraint is that most experimental plots were designed to physically confine and assure uniform distribution of the overland flow (Dosskey, 2001), whereas in normal field situations overland flow often concentrates in preferred pathways, and such concentrated flow can reduce buffer effectiveness (Dillaha *et al.*, 1989a, b; Dosskey *et al.*, 2002; Helmers *et al.*, 2005). Third, experimental plot studies typically do not account for long-term accumulation of sediment within the buffer, which may reduce buffer effectiveness and require wider buffers to accommodate the accumulated sediment (Dillaha *et al.*, 1989b; Magette *et al.*, 1989). Finally, in many of the studies cited by Liu *et al.* (2008) and Yuan *et al.* (2009) infiltration played a large role in sediment removal (e.g., Arora *et al.*, 2003; Abu-Zreig *et al.*, 2004; Mankin *et al.*, 2007). Infiltration may be less important in actual streamside buffers, which are often characterized by moist conditions and hydric soils.

We have attempted to compensate for some of these constraints by compiling studies (Table 2) that were conducted either on streamside buffers receiving flow from an unconfined upslope area, or on plots in which the peak hydraulic loading onto the buffer exceeded 1.0 l/s/m measured transverse to flow. This loading is roughly what is expected from 250 m (measured upslope) of cropped fields for a 30-min storm with a two-year return interval in eastern North America (USDA-NRCS, 1986; Bonnin *et al.*, 2004). Of the studies reviewed by Liu *et al.* (2008) and by Yuan *et al.* (2009), 13 E_S measurements (10 published reports) met our criteria. On the basis of a supplementary literature search, we added another nine measurements (seven reports). The sediment removal efficiency from these studies correlated with width

TABLE 2. Sediment Removal Efficiencies Measured in Unconfined Natural Settings or on Plots Using Hydraulic Loadings >1.0 l/s/m.

Study	Location	Vegetation	Setting	Buffer Width (m)	E_S (%)
Arora *et al.* (1996)	Iowa	Grass	Field plots, natural runoff	20	65
Arora *et al.* (2003)	Iowa	Grass	Field plots, simulated runoff	20	86
Clausen *et al.* (2000)	Connecticut	Grass	Streamside buffer	30	92
Daniels and Gilliam (1996)	North Carolina	Grass	Streamside buffer	5	43
Daniels and Gilliam (1996)	North Carolina	Grass/forest	Streamside buffer	16	45
Deletic and Fletcher (2006)	Australia	Grass	Field plot	65	83
Dunn *et al.* (2011)	Prince Edward Island	Grass	Streamside buffer	10	64
Dunn *et al.* (2011)	Prince Edward Island	Grass	Streamside buffer	20	82
Fiener and Auerswald (2003)	Munich	Grass	Grassed waterway	13	77
Fiener and Auerswald (2003)	Munich	Grass	Grassed waterway	25	97
Gharabaghi *et al.* (2006)	Ontario	Grass	Streamside plots, simulated runoff	3	55
Gharabaghi *et al.* (2006)	Ontario	Grass	Streamside plots, simulated runoff	20	90
Helmers *et al.* (2005)	Nebraska	Grass	Streamside buffer	13	80
McKergow *et al.* (2006)	Western Australia	Forest	Streamside buffer	10	21
McKergow *et al.* (2006)	Western Australia	Grass	Streamside buffer	10	64
Newbold *et al.* (2010)	Pennsylvania	Grass/forest	Streamside buffer	27	43
Peterjohn and Correll (1984)	Maryland	Forest	Streamside buffer	19	90
Peterjohn and Correll (1984)	Maryland	Forest	Streamside buffer	60	94
Sheridan *et al.* (1999)	Georgia	Grass	Streamside buffer	8	78
Sheridan *et al.* (1999)	Georgia	Grass/forest	Streamside buffer	35	83
Sheridan *et al.* (1999)	Georgia	Grass/forest	Streamside buffer	59	95
Ziegler *et al.* (2006)	Thailand	Sedge	Streamside buffer	30	80

($r = 0.51$, $p = 0.015$, $n = 22$), but was unaffected by vegetation type ($p = 0.85$, ANOVA). The relation between efficiency and width was described reasonably well (28% of variance explained) by the equation:

$$E_S = w/(k_{50} + w) \tag{8}$$

in which the parameter k_{50} minimized the error sums of squares at a value of 5.8 m (Figure 3). The form of the equation (a rectangular hyperbola) was chosen, in part because it realistically predicts no removal at zero meters and 100% removal at infinite buffer width, and in part for simplicity. The curve is described by only one estimated parameter, k_{50}, which represents the buffer width that would be expected to remove 50% of the sediments. Equation (8) predicts 64% removal for a 10-m buffer increasing to 84% for a 30-m buffer. These removal efficiencies are 20-40% lower than those estimated by Liu *et al.* (2008), as given by Equation (7) above. Moreover, the 87% removal that Liu *et al.* (2008) predict for a 10-m buffer would, under this reanalysis, require a buffer 40 m wide.

Figure 3 suggests that much of the potential removal can be achieved in a width of 20 m, which according to Equation (8), would be 78% efficient. The gains beyond 20 m appear modest, but the predicted increase to 84% efficiency at 30 m, represents

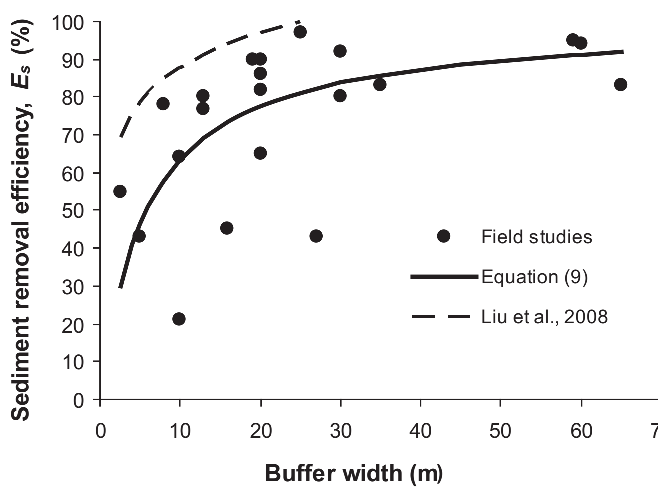

FIGURE 3. Sediment Removal Efficiency *vs*. Buffer Width. Studies were conducted in unconfined field settings or in plots where hydraulic loading was >1.0 l/m/s. Also shown are the least-squares fit of Equation (8) and the equation obtained by Liu *et al.* (2008), as fitted to 79 studies conducted under a broad range of conditions.

a 28% reduction in the quantity of sediments delivered to the stream. It is useful to consider this reduction from the standpoint of water quality. From the studies cited by Liu *et al.* (2008), we tabulated the concentrations of suspended sediments in buffer outflows, where they were either published or could be calculated. The median outflow concentration was

1,000 mg/l, with a range of 75-11,000 mg/l. Among the studies listed in Table 2, and considering only the 15 studies conducted in natural settings and without experimental sediment additions, the median outflow sediment concentration was 60 mg/l. This is in the range of concentrations that have been documented to have sublethal effects on fish (Newcombe and Jensen, 1996), and it is substantially above the median (38.5 mg/l) of flow-weighted sediment concentrations of the nine major rivers feeding the Chesapeake Bay (Gellis *et al.*, 2004).

It is likely that much of the sediment escaping narrow (e.g., <20 m) buffers consists of fine silts and clays and that the potential to remove these fractions would continue to increase with increasing buffer width. Sediment is delivered to a buffer as a mix of particle sizes. Typically, only a few meters are needed to allow deposition of sands (>50 μm) and some of the larger silt fractions (Dabney *et al.*, 1995). Buffers are less effective in removing smaller particles (i.e., fine silts [2-10 μm] and clays [<2 μm]) (Meyer *et al.*, 1995; Daniels and Gilliam, 1996; Lee *et al.*, 2000, 2003; Syversen and Borch, 2005; Gharabaghi *et al.*, 2006; White *et al.*, 2007), which are less easily deposited and travel much longer distances through the buffer. Although gravitational deposition is limited by low settling velocities, fine particles may also be removed by infiltration (Dillaha *et al.*, 1989a; Muñoz-Carpena *et al.*, 1999), exchange with soil water (Fiener and Auerswald, 2003), and aggregation and adhesion to vegetative surfaces (Dabney *et al.*, 1995; Tromp-van Meerveld *et al.*, 2008). Further evidence that fine particles can be effectively removed by wider buffers comes from two studies showing that eroded clay particles labeled with ^{137}Cs (from atom bomb testing) were transported average distances of 80-100 m into streamside forest buffers (Cooper *et al.*, 1987; Lowrance *et al.*, 1988).

The studies reviewed in the previous paragraphs measured the trapping of sediments that might otherwise reach a stream, but none (except Ziegler *et al.*, 2006) measured sediment concentrations in the receiving stream. Because the stream integrates upstream as well as lateral inputs, it is difficult to assess stream impacts from plot- and field-scale experiments. Davies and Nelson (1994) found that visually estimated silt cover in streams draining logged *Eucalyptus* forests was higher than in matched control reaches where forest buffers were 10-30 m wide, but not where buffers were >30 m wide. Their results were consistent over slopes ranging from 5 to 70%. Jones *et al.* (2006) found higher fine sediments in the riffles of streams where the average upstream buffer width was 15 m than where buffer width was 30 m.

In summary, studies of the ability of streamside buffers to trap sediment, when limited to streamside studies or comparable field conditions, show that buffers 10 m wide can be expected to trap about 65% of sediments delivered by overland flow, while 30-m buffers can be expected to trap about 85% of sediments. The increased removal attained by wider buffers represents a small fraction of the total sediments (by mass), but probably a large fraction of the finer silts and clays, which are typically released from narrow buffers in concentrations high enough to impair water quality.

STREAMSIDE FOREST BUFFERS AS PROMOTERS OF STABLE, HEALTHY, AND FUNCTIONAL STREAM ECOSYSTEM

Here, we attempt to assess how habitat or biological variables (channel width, channel meandering and bank stability, temperature, inputs of LWD, and macroinvertebrate and fish communities) respond to variation in the width of the streamside forest.

Channel Width

Because stream ecosystems and ecosystem processes are largely associated with the streambed (Allan and Castillo, 2007), an increase in channel width and/or complexity of the bed in a given reach increases the amount of ecosystem per unit length of stream and the potential for delivery of ecosystem services (Sweeney *et al.*, 2004). Numerous studies have found that for low-order rural streams, channels are significantly wider when the banks are forested (Zimmerman *et al.*, 1967; Sweeney, 1992; Davies-Colley, 1997; Trimble, 1997; Sweeney *et al.*, 2004; McBride *et al.*, 2008). The widening may be due to a reduction in bank armoring related to suppression of grasses by overstory shading (Davies-Colley, 1997; Stott, 1997) in combination with increased near bank turbulence during over bank flows in forested reaches (McBride *et al.*, 2007). Hession *et al.* (2003) reported a similar phenomenon in urban watersheds and showed that the increased width of forested channels was independent of the level of urbanization. McBride *et al.* (2008) provided a conceptual model describing the channel widening process associated with afforestation of the riparian zone. In one eastern North America study, the channel width of forested reaches was consistently (and significantly) greater than for contiguous deforested reaches across 16 study streams (Sweeney *et al.*, 2004). In this study, the width of streamside forest buffer ranged from 25 to 233 m across study sites, suggesting that significant stream widening occurred with as little as 25 m of streamside forest. Further analysis of that study shows no correlation between the ratio of forested to deforested channel width and the width of streamside forest (Sweeney *et al.*, unpublished data).

Although the process of stream widening in forested reaches may reverse itself in large stream and river channels (Anderson *et al.*, 2004) or under unusual circumstances (Murgatroyd and Ternan, 1983; Andrews, 1984; Hey and Thorne, 1986; Rosgen, 1996), current literature suggests that forested streams associated with watersheds $\leq$100 km^2, or about fifth order or smaller in size, are significantly wider than their deforested counterparts when all other factors are equal. This means that >90% of total stream length in the continental U.S. might be expected to share that response (Leopold *et al.*, 1964; USEPA, 2013).

In conclusion, a streamside forest of 25 m can maximize the width of small streams but it appears that little or no additional widening occurs in response to forest buffers $\geq$25 m. More research is needed to determine the minimum width of forest buffer needed to significantly increase the width of a deforested stream channel.

Channel Meandering and Bank Erosion

The meandering of a stream or river and movement across its floodplain are natural processes (Leopold *et al.*, 1964) to which aquatic organisms can and do adapt (Allan and Castillo, 2007). However, the release of excessive sediments and associated nutrients caused by high rates and movement of channels can be detrimental, due to the instability of habitat for aquatic biota and the degradation of habitat for biotic communities (Laubel *et al.*, 2003). Channel meandering and migration are significantly affected by the stability of streambank soil, which in turn is affected by the abundance and type of streamside vegetation and associated factors. However, the influence of vegetation appears to be greatest when the ratio of rooting depth to channel depth is such that roots extend to the toe of banks (Thorne, 1990; Anderson *et al.*, 2004). Otherwise, erodibility of bank sediments and hydraulics of flow will contribute most to bank instability and erosion (Pizzuto, 1984; Allmendinger *et al.*, 2005).

Whether streams are more stable with or without forest is complicated. An interlocking network of tree roots can increase bank strength and, therefore, resist erosion; however, trees that fall into the river can divert flow and trigger scour and local bank erosion (Montgomery, 1997). In general, forested streams exhibit greater channel stability than deforested streams. Beeson and Doyle (1996) studied 748 river bends in four streams following major floods in 1990 and showed that major bank erosion was 30 times more prevalent on bends without forest. Other studies have also shown that forested stream reaches exhibit slower channel migration and lower floodplain accretion rates of sediment and thereby provide more stability than deforested channels (Hession *et al.*, 2003; Allmendinger *et al.*, 2005). In the Sacramento River in California, Micheli *et al.* (2004) analyzed the effect of floodplain vegetation removal on river channel migration between 1949 and 1997 and found that deforested agricultural floodplains were 80-150% more erodible than floodplains with a streamside forest. They reported these results as consistent with earlier studies of Johannesson and Parker (1989) and Odgaard (1987), where channels bordered by streamside forest tended to migrate roughly half as fast as deforested channels. Finally, Laubel *et al.* (2003) showed that establishment of streamside forest buffers along highly modified, channelized streams can reduce bank erosion rates.

Few, if any, studies have measured channel or bank stability in response to variation in the widths of streamside forests. However, Burckhardt and Todd (1998) looked at nine pairs of stream bends in seven streams in Missouri in which each pair consisted of one forested concave bank and one unforested concave bank. Stream bends with unforested banks had an average local migration rate three times greater than those with forested banks, but there was no apparent correlation between rate of channel migration and width of the streamside forest (which ranged from 10 to >61 m in the study). These findings suggest, albeit indirectly, that streamside forest widths of around 10 m provide some protection (but buffers <10 m were not assessed in the literature). Zaimes *et al.* (2006) provided direct evidence from a heavily altered Iowa landscape where bank erosion was lowered significantly by the presence of a streamside forest 10 m wide along reaches bordered by row crop agriculture and actively grazed cow and horse pastures. They estimate that establishment of streamside forest buffers could reduce streambank soil loss and sediment release by 77-97%.

In conclusion, it is clear that streamside forests help reduce bank erosion and channel meandering. Because the data are limited, more studies of this phenomenon are needed for forest buffers of various widths.

Temperature

Unforested streams, particularly small streams, experience higher summer maximum water temperatures than those under the full shading of a forest canopy (e.g., Brown and Krygier, 1970; Lee and Samuel, 1976; Lynch *et al.*, 1985; Sweeney, 1993). The clearing of a streamside forest can also reduce winter temperatures and increase diel thermal variation (Rishel *et al.*, 1982). Elevated temperatures may reduce the habitat available to fishes (Barton *et al.*, 1985; Jones *et al.*, 2006; Whitledge *et al.*, 2006), alter the life histories and reproductive success of aquatic insects (Vannote and Sweeney, 1980; Sweeney, 1993), and alter stream ecosystem metabolism (Bott *et al.*, 1985; Sinsabaugh, 1997; Uehlinger *et al.*, 2000). Streamside forest buffers can reduce or eliminate the thermal effects of forest clearing (reviewed by Moore *et al.*, 2005), by reducing the solar radiation reaching the stream (Brown, 1969; Groom *et al.*, 2011). Reductions in water temperature due to streamside forest restoration have been directly linked to recovery of benthic macroinvertebrate communities (Parkyn *et al.*, 2003).

Because light passes obliquely through the canopy to the stream, the shading and temperature control that a riparian buffer provides depend in part on the width of the buffer. In narrow buffers, light may pass to the stream entirely through the understory where light attenuation is reduced (Sridhar *et al.*, 2004; Groom *et al.*, 2011). Brazier and Brown (1973) measured stream shading among 13 buffers and concluded that a 17-m buffer provided 90% of full-forest shading, while a 24-m buffer provided shading equivalent to a full forest. However, among 33 sites, Groom *et al.* (2011) found that 31 m of buffer provided 92% of the shade provided by 52 m of buffer. DeWalle (2010) modeled the exposure of a stream to solar energy based on considerations of the daily solar track, the orientation and width of the stream, the geometry of the forest canopy, and the attenuation of light within the canopy. He concluded that a width of 12 m should provide about 80% of full-forest shade under most conditions, but that for a stream with a north–south orientation, shade increased incrementally with width, even beyond 30 m. Thus, it seems clear narrow buffers (e.g., 15 m) can provide most of the shading of a full forest, but there is less agreement on how much incremental shading is provided by wider buffers. And, of course, shade is only indicative of the degree of actual temperature control.

Figure 4 summarizes studies that have directly examined the role of buffer width in regulating stream temperature. Most of the studies are site specific, involving one or more experimental watersheds matched to nearby controls, although three of the studies involved 17-33 sites dispersed throughout a region. Nearly all involve small streams (<5 km^2 in drainage area). The temperature response to input of solar energy varies inversely with streamflow (Brown, 1969; Moore *et al.*, 2005), making small streams more vulnerable to thermal extremes, as well as more amenable to experimental study. However, the cumulative downstream effects of deforestation throughout a stream network may be significant (Beschta and Taylor, 1988).

Temperature increases were consistently observed in streams without buffers. Figure 4 lists a few of such reports. For buffer widths between 4 and 20 m, four site-specific studies reported temperature increases >2°C, while two reported no increase. The minimum width reporting no increase, implying a fully effective buffer, was 10 m (Burton and Likens, 1973). For buffers wider than 20 m, two site-specific studies found temperature increases, but in only one

Buffer Width (meters)	0 5 10 15 20 25 30 50
Temperature increases (≥2 °C except as noted)	●Burton and Likens (1973) ●Kiffney et al. (2003)* ●Harris (1977) ●Rishel et al. (1982) ●Hewlett and Fortson (1982) ●Kiffney et al. (2003) (1.6 °C)* ●Sweeney (1993) ▲Davies and Nelson (1994) (1.2 °C) ▲Groom et al. (2011) (0.7 °C) ●Johnson and Jones (2000) ▲Jones et al. (2006) ●Wilkerson et al. (2006) ●Brown and Krygier (1970)
Mixed temperature responses	●Janisch et al. (2012) ●Jackson et al. (2001)
No temperature increase	●Burton and Likens (1973) ●Wilkerson et al. (2006) ●Lee and Samuel (1976) ▲Jones et al. (2006) ▲Groom et al. (2011) ■Sridhar et al. (2004) ●Rishel et al. (1982) ■Chen et al. (1998 a,b) ▲Davies and Nelson (1994)
● Site specific (1-3 streams per treatment) ▲ Regional (6-17 streams) ■ Modeling	

*Data also reported and analyzed by Gomi et al. (2006) with similar results

FIGURE 4. Summary of Studies Relating Temperature Increases to Buffer Width. Temperature increases refer to summer daily maxima relative to a fully forested condition.

case (Harris, 1977; 30 m) was the increase >2°C. Two other site-specific studies, at widths of 23 and 30 m, reported no increase. Among the regional studies, Davies and Nelson (1994) reported that maximum daily temperature increased 1.2°C among six streams with buffers 0-10 m wide, but they observed no temperature increase in 12 streams with buffers ranging from 10-30 m wide. Studying 17 streams of varying buffer width, Jones *et al.* (2006) concluded that that a 15-m buffer raised summer maximum stream temperatures by 2°C relative to that expected of a 30-m buffer. Although Jones *et al.* (2006) selected the streams to eliminate confounding by the percent forest cover of the surrounding watershed, they did not address the potential influence of gaps in the buffers. Thus, they may have underestimated the temperature protection provided by an uninterrupted 15-m buffer. In a study of 33 sites, Groom *et al.* (2011) observed that summer maximum daily temperatures were 0.7°C higher among sites with buffer widths averaging 31 m, but that there was no increase in temperature among sites with an average of 52 m of buffer. Chen *et al.* (1998a, b) developed a simulation model to estimate potential temperature control by riparian buffers and concluded that widening buffers beyond 30 m provided no additional temperature control. However, an alternative model (Sridhar *et al.*, 2004) simulated near-maximal temperature control with as little as 15 m of buffer (Lanini *et al.*, 2004).

In short, it appears that buffers with widths of 10-30 m are often, but not always, fully effective in preventing temperature increases. All buffers wider than 10 m were partially effective, holding increases to ≤3°C as opposed to increases >5°C typical of unbuffered streams (Lee and Samuel, 1976; Harris, 1977; Johnson and Jones, 2000). For buffers wider than 20 m, reported increases did not exceed 2°C. And where multiple buffer widths were considered within the same study, the wider buffers were more effective (Davies and Nelson, 1994; Kiffney *et al.*, 2003; Jones *et al.*, 2006; Groom *et al.*, 2011).

That the width of buffer needed to prevent temperature increase may vary between 10 and 30 m from site to site reflects the many factors that influence the relation between stream temperature and streamside vegetation. These include the length of exposed stream, stream size and orientation, latitude, local topography, and the type, height, and density of streamside vegetation (Moore *et al.*, 2005; DeWalle, 2008). Models that incorporate these factors can potentially resolve much of this variability. When properly calibrated, models can predict temperatures closely (e.g., Sinokrot and Stefan, 1993; Rutherford *et al.*, 1997; Chen *et al.*, 1998a, b; Westhoff *et al.*, 2007). Nonetheless, there remain substantial gaps in the theoretical understanding of stream temperature dynamics (Johnson, 2003). Interception of solar radiation cannot yet be reliably predicted from simple measurements of canopy cover. Factors, such as microclimate beyond the buffer (Hewlett and Fortson, 1982; Weatherley and Ormerod, 1990), water exchanges with hyporheic zones (Constantz, 1998, 2008), and inputs from groundwater and lakes (Mellina *et al.*, 2002) can influence stream temperatures in ways that have not yet been incorporated into models. Finally, accurate modeling may require data and effort that exceed the resources normally available for site-specific application.

In summary, it appears that buffer widths of ≥20 m will keep stream temperatures within 2°C of those that would occur in a fully forested watershed but that full protection from measureable temperature increases is assured only by a buffer width of ≥30 m.

Large Woody Debris

Maser and Sedell (1994) thoroughly reviewed the LWD literature and showed that: (1) streamside forests are the primary source of LWD (stems, branches, and rootwads >1 m in length and >10 cm in diameter) for both streams and large rivers; and (2) LWD provides nutrients and food for aquatic organisms, increases the diversity of instream habitats by forming dams and attendant pools, and helps dissipate the energy of water and keep its sediments from moving downstream. Reviews by Bragg (2000) and Diez *et al.* (2001) called attention to the role of LWD in channel development, oxygenation, and turbulent mixing of water, organic carbon and nutrient cycling, species habitat, and other important aspects of stream and river ecosystems. More specifically, LWD in natural streams impacts important factors, such as the quantity and quality of sediments (Montgomery *et al.*, 1996), levels of organic carbon (Bilby and Likens, 1980; Bilby, 1981) and nutrients (Webster *et al.*, 2000; Ensign and Doyle, 2005), the instream flow patterns of water (Gippel, 1995; Shields and Gippel, 1995; Wilcox *et al.*, 2006; Wilcox and Wohl, 2006), and channel heterogeneity for macroinvertebrates and fish (Angermeier and Karr, 1984; Wallace *et al.*, 1995; Abbe and Montgomery, 1996; Wright and Flecker, 2004). In addition, it appears that streamside forest buffers and their LWD also play a role in significantly increasing the level of certain ecosystem services (e.g., water filtration and treatment) provided by per unit length of stream channel. The services result from greater contact of stream water with benthic sediments (due to channel widening associated with forested banks) containing microorganisms capable of processing, degrading, or sequestering organic matter and inorganic nutrients

(Sweeney, 1992, 1993; Sweeney *et al.*, 2004) and increased downwelling of stream water into hyporheic sediments caused by flow deflected by LWD (Sawyer *et al.*, 2011).

Although LWD is important to streams and rivers, its density has declined due to historic channel clearing and streamside logging practices (Montgomery and Piégay, 2003; Shields *et al.*, 2006). The lack of LWD causes increased channel instability and bank erosion in streams and a decrease in the level of complexity of instream habitat (Montgomery, 1997). Consequently, reintroducing LWD is a common practice used to restore streams and rivers to their natural state or for restoring trout habitat (Gippel, 1995; Braudrick and Grant, 2000; see Lehane *et al.*, 2002 for review).

LWD recruitment to a stream depends on the presence of a streamside forest. For example, a small stream flowing through second growth forest in Pennsylvania had 7.5 time the number of pieces and 27 times the volume of LWD than a downstream contiguous reach that was deforested (Sweeney, 1993). In Oregon, a stream draining an old-growth wilderness area had more than 10 times the amount of LWD per unit length than a stream with an adjacent forest that had been logged during the previous 30 years (Maser and Sedell, 1994).

We were not able to find any publications addressing the issue of streamside forest width and the input of LWD to streams. However, unlike leaves, which blow across the forest floor and get entrapped in streams, natural recruitment and entrainment of LWD require that trees fall directly into the stream channel because of their size and mass. Thus, for a streamside forest buffer to mimic the natural inputs of LWD to streams, its average width would have to be equal to the average height at maturity of the dominant streamside trees in the region so that input of LWD from trees dying and falling toward the stream can, in fact, make it to the channel. In Europe, Diez *et al.* (2001) suggested that, given the size of streamside trees, "mature strips at least 20 m wide, including large, senescent trees and standing snags, would be necessary to ensure the necessary input of logs." In the Mid-Atlantic region of the U.S., this would be between 20 and 30 m.

In conclusion, in lieu of direct studies bearing on this issue, we infer at this time that a streamside forest can best provide a natural level of LWD to streams if its width is generally ~30 m or equal to the height at maturity of the dominant streamside trees.

Macroinvertebrates

On the basis of an extensive literature review, Sweeney (1993) concluded that "the presence or absence of trees adjacent to stream channels may be the single most important factor altered by humans that affect the structure and function of stream macroinvertebrate communities." The review produced a conceptual model of the multiple pathways that streamside forests affect the relative abundance, growth, and reproduction of stream macroinvertebrates. In the intervening 20 years, numerous studies have expanded this model and documented that riparian vegetation and land use in a watershed greatly influence the structure and function of stream macroinvertebrate communities (Townsend *et al.*, 1997; Allan, 2004; Kratzer *et al.*, 2006; Death and Collier, 2010). Here we focus only on studies designed to specifically test, evaluate, or model how much streamside forest is needed, and in what condition, to protect, maintain, and/or otherwise support natural levels and life histories of macroinvertebrates in streams. These studies, in turn, add an important perspective to the use of macroinvertebrate community structure as an index of water quality and ecosystem health (Lenat, 1993; Resh *et al.*, 1995).

Several studies bear on the width of streamside forest needed to protect macroinvertebrates. Many demonstrate how forest buffers of varying widths can help mitigate the effects of logging activity on macroinvertebrate communities. Although Kreutzweiser *et al.* (2005) reported that only a 3 m-wide piece of undisturbed forest was needed to prevent significant changes in the abundance and structure of macroinvertebrate communities in response to selective harvesting of trees (up to 42% tree removal), other studies showed that a wider streamside forest is needed to buffer the effects of more extensive logging. For example, Newbold *et al.* (1980) and Davies and Nelson (1994) both found that buffer widths of ≥30 m were needed to prevent significant changes in macroinvertebrate communities when the forest beyond 30 m was clear cut. Similarly, Kiffney *et al.* (2003) showed that pollution-tolerant insects (chironomids) increased in abundance with decreasing buffer width, but pollution-sensitive insects (mayflies) did not. In this case, a buffer of at least 30 m was needed to keep chironomid levels the same as in controls, while keeping key macroinvertebrate habitat requirements (i.e., levels of light, water temperature, and periphyton) from escalating significantly above normal conditions due to logging.

The protection afforded by a streamside forest to macroinvertebrate populations is related to both the interception of pollutants (e.g., sediment) from the upland logging activity and the maintenance of instream natural habitat. Thus, surrounding a stream with conditions (light, temperature, humidity, leaf and woody debris fall, and other factors) typical of a mature forest provides a setting conducive to providing natural instream habitat for macroinvertebrates. In this regard, Rykken *et al.* (2007) studied forest microclimate at various distances (1, 10, 20, and 70 m) from streams bordered by >70 m of old-growth forest, no forest, or 30 m of old-growth forest (clear cut beyond) and showed that a full 30 m of forest was needed adjacent to the stream to keep microclimate from deviating significantly from natural, old-growth forest conditions for the terrestrial life stages of aquatic macroinvertebrates (e.g., adult insects). For low-order (first-second) tropical streams in Costa Rica, Lorion and Kennedy (2009a) observed that Ephemeroptera-Plecoptera-Trichoptera (EPT) taxon richness was statistically higher in reaches flowing through natural forest than in sparsely forested pasture, while EPT richness in reaches bordered by a 15-m forest buffer were intermediate and statistically the same as for reaches in natural forest or pasture. Both biomass and density of macroinvertebrates were higher in forested reaches than pasture, but the data were highly variable and not statistically significant. These data suggest that 15 m was not wide enough to maintain all aspects of macroinvertebrate ecology equal to those in a naturally forested state.

In one of the few replicated field studies involving forest buffers >25 m but not involving logging, Sweeney *et al.* (2004) quantified macroinvertebrate abundance and production in riffles and pools over a one-year period for paired reaches for 16 (first-fifth order) piedmont streams, which were completely deforested but without agriculture impacts, *vs.* mature forest buffers averaging 72 m wide (range 25-223 m). They reported significantly more macroinvertebrates per unit stream length in forested reaches than in deforested reaches when averaged across all habitats and seasons of study.

Regardless of buffer width, several studies show that the amount of protection and support for macroinvertebrate communities increases with increased proportion of trees in the streamside forest. For example, Moore and Palmer (2005) investigated the pattern of macroinvertebrate community structure along a gradient of agriculture and urban development and showed that macroinvertebrate taxon richness increased significantly with increased percentage of forest in the 30 m-wide streamside buffer. However, their result likely reflects greater buffering (protection) capacity of denser streamside forest as well as greater preservation of instream habitat. For example, England and Rosemond (2004), who studied various degrees of deforestation within a 30-m buffer in seven suburban/rural streams, showed that an intact, contiguous streamside forest is needed to assure that species of macroinvertebrates adapted to living in forested streams receive an adequate amount and quality of organic matter as food. For heavily farmed watersheds (~87% agriculture), Zum-Berge *et al.* (2003) showed that macroinvertebrate communities in streams bordered by 100 m-wide streamside forest (>28% trees) were significantly less altered (higher macroinvertebrate community index scores, lower percentage chironomid levels, and lower Hillsenhoff Biotic Index scores) than communities in streams bordered by 100 m of deforested land (<10% trees). In contrast, Roy *et al.* (2005) found no significant differences in macroinvertebrate assemblage integrity (richness, densities) when they compared paired 200-m long stream reaches in five suburban catchments, each differing in the amount of forest in a 30-m buffer (open [~32%] *vs*. closed [~77%] canopy). However, this lack of significance appears to be due to the overriding influence of catchment-scale impacts upstream of the study reaches. Thus, they concluded that restoring streamside forest at the reach scale, although necessary for restoring macroinvertebrate community structure and function, may in some situations be insufficient by itself to mitigate major impacts generated upstream at the catchment scale.

In conclusion, it appears that a ≥30 wide buffer of dense streamside forest is needed to protect and support natural levels of macroinvertebrates as well as macroinvertebrate activity in small streams. For stream reaches without a dense streamside forest, Sweeney *et al.* (2002) showed it is possible to go from a deforested condition to canopy crown closure over the stream within 15 years if seedlings are protected from herbivory and competing vegetation. Moreover, once a streamside forest is completely restored, it appears that macroinvertebrate densities and biomass can return to undisturbed levels within about 25 years if there are nearby sources of colonizers (Fuchs *et al.*, 2003). This relatively long (15 + 25 = 40 years) restoration time indicates the need to conserve and protect existing streamside forests at a minimum of 30 m and to undertake proactive streamside afforestation sooner rather than later.

Fish

The importance of streamside area to the conservation and management of freshwater fish was reviewed by Pusey and Arthington (2003), but they did not address minimum buffer width. However, other papers have confirmed the widespread importance of streamside areas to the maintenance and restoration of diverse fish communities (Meador and Goldstein, 2003) and have shown that manipulating forest cover in the first 30 m of the streamside area can have a greater impact on fish assemblage integrity than changes to instream habitat (Brazner *et al.*, 2005). There are several well-replicated, controlled studies that bear directly on how wide the streamside forest needs to be to protect and conserve fish communities. Horwitz *et al.* (2000) quantified fish abundance over a one-year period for paired reaches (forested *vs*. deforested) in 15 streams (≤5th order), with the average buffer width of forested reaches being 72 m. Their results were equivocal because, although fish densities per unit area were greater in deforested reaches, no significant difference was observed for total fish abundance per unit length between deforested and forested reaches. Moreover, some species were more abundant in one or the other reaches. This may be related to the high mobility of fish and the fact that the paired reaches were contiguous. In contrast, Jones *et al.* (2006) established and quantified the relationships in rural watersheds (in Georgia) among streamside forests, aquatic habitat (stream temperature, fine sediment load), and trout reproductive success (biomass of young) to assess the impact of reducing forest buffers from 30 to 15 m. They found that reducing streamside forest to 15 m resulted in higher peak temperatures (~2°C higher) and more fine sediments (~25% higher). Their linear regression models and Monte Carlo uncertainty analyses predicted an 87% reduction in young trout biomass. They concluded that a 30-m buffer would enable ~63% of Georgia's second- to fifth-order trout streams to support trout, whereas reducing the buffer to 15 m would likely decrease the number of trout supporting streams to ~9%. In contrast, Lorion and Kennedy (2009b) showed that a 15-m wide forested buffer on first- and second-order tropical stream reaches in Costa Rica was sufficient to maintain all aspects of fish community structure (species richness, density, biomass) that, while statistically different from in unforested pasture reaches, were statistically the same as in adjacent or nearby reaches flowing through natural forest.

Jones *et al.* (1999) also demonstrated the importance of streamside forest to fish communities with an unusual approach of documenting the impact of removing patches of streamside forest of different lengths and widths in otherwise completely forested watersheds. They showed that deforesting >30 m of the streamside zone for only ~1 km of length caused significant reduction in density, abundance, and structure of fish communities, with the impact intensifying with longer patch lengths of deforestation. Since the areas adjacent to the deforested streamside patches remained as intact forest, this study demonstrates the importance of streamside forest to the quality of instream habitat for fish.

For agricultural watersheds, Lee *et al.* (2001) studied 18 streams (≤5th order) to assess the significance of a 200-m wide buffer zone relative to streamside forest in the upstream portion of a stream's watershed. They concluded that, although having streamside forest in both the local and upstream areas significantly improved both the Fish Index of Biotic Integrity and species richness levels, local streamside forest was more influential. In contrast, Fischer *et al.* (2010) studied fish responses to vegetated buffered (30 m wide) and unbuffered reaches of 41 stream reaches in agricultural watersheds in Iowa and found little to no relationship between the presence of buffers and fish assemblage structure and instream habitat characteristics. However, although active agriculture had been excluded from the buffered reaches for 20 years, the vegetation along those reaches was dominated by grass (~80%) rather than forest and so the lack of differences in fish communities and habitat between buffered and unbuffered reaches likely reflects the fact that both the buffered and non-buffered reaches were largely deforested.

For urban streams, Roy *et al.* (2005) studied 30 reaches of stream with a 30-m wide buffer but with vegetative conditions ranging from 39 to 100% forest cover in the buffers. They showed that percentage streamside forest in a 30-m buffer zone is the best predictor of the abundance and richness of sensitive fish, but streamside forest alone in urban streams is insufficient to maintain healthy fish communities at that width (due to other, catchment-scale factors such as sediment load).

In conclusion, most available data based on replicated, controlled studies suggest that a streamside forest of ≥30 m is needed to protect and maintain fish communities in a natural or near-natural state.

SUMMARY, PERSPECTIVE, AND CONCLUSION

We reviewed the scientific literature that addresses the question: "how wide does a streamside forest buffer need to be to protect water quality, assure natural stream habitat, and maintain the natural structure of important stream communities?" The review underscored the role that streamside forests play in protecting and enhancing the water quality of downstream rivers and estuaries by keeping pollutants out of stream and river channels. Streamside forests also enhance the quality and health of instream physical, chemical, and biological characteristics, which enable the stream and its ecosystem to provide important services, such as sequestering carbon, metabolizing organic matter, and degrading and processing of pollutants. To that end, the review included both upland- and instream-specific measures of the influence of streamside forest buffers on specific and important stream or streamside ecosystem properties, components, or functions.

The upland side focused on two variables: subsurface nitrate and overland sediment movement through the buffer. For subsurface nitrate, removal efficiencies were strongly influenced by subsurface water flux. Among sites where water flux was measured, we found that high removal rates (>80%) in narrow (<30 m) buffers occurred only where water flux was low (<50 l/m/day) and therefore contributed minimally to stream water quality. For sites with higher, more meaningful water flux (i.e., >50 l/m/day), the median nitrate removal efficiency was 55% (range: 26-64%) for buffer widths <40 m, and 89% (range: 27-99%) for buffer widths >40 m. Our simple model developed from these observations, when applied to a site with average water flux, predicts removal efficiency of 48% for a 30-m buffer, increasing to 90% for a 100-m buffer. We conclude that forest buffer widths of ≥30 m or more are needed to achieve significant nitrate removal at the watershed scale. For sediment, studies restricted to field settings, or comparable experimental loadings, showed that 10-m and 30-m buffers can be expected to trap ~65 and ~85% of sediments, respectively. Although increased sediment removal by wider buffers (i.e., 30 m) is a small fraction of total sediments, it represents a substantially larger fraction of fine silts and clays that typically impair water quality. Hence, we interpret the 20% increase in sediment trapping observed by increasing buffer width from 10 to 30 m in width to have ecological significance.

The review of the instream impact of buffers focused on six functions. The key insights associated with each of those were: (1) stream width: small streams (i.e., those associated with watersheds ≤120 km^2 or about fifth order or smaller in size) bordered by ~25 m of forest generally had significantly wider channels than those with no buffers but little or no additional widening occurs in response to forest buffers ≥25 m; (2) channel meandering and bank erosion: the presence of a streamside forest reduces bank erosion and channel meandering, but more studies are needed to determine the general response to variation in forest width; (3) temperature: buffer widths of 20 m or wider keep stream temperatures within 2°C of a fully forested watershed, but full protection from measureable temperature increases is not assured unless buffer width exceeds 30 m; (4) LWD: no studies bear directly on this issue, but we infer from the literature that a forest width of ~30 m, or a width equal to the height at maturity of the dominant streamside trees in the region, can provide natural levels of debris for a stream; (5 and 6) macroinvertebrates and fish: a streamside forest of at least 30 m provides instream habitat of sufficient quality to maintain these communities in a natural or near-natural state.

We recognize that the optimal width for a buffer may vary from site to site and that an ideal buffer policy might call for variable buffer widths that accommodate site-specific factors and are possibly scaled or based on stream size. Richardson *et al.* (2012) reviewed the potential for variable-width buffers, noting that fixed-width buffers have been the norm, largely because they "are administratively simple to implement and assess." Yet they also pointed out that a fully effective buffer policy "will require carefully designed, large-scale field experiments, coupled with long-term monitoring of spatial…and catchment scales." On the basis of this review of the literature, we conclude that, although we currently have a relatively advanced scientific understanding of buffer function in some areas, the available field data are only sufficient to describe broad relationships between buffer width and function and remain inadequate for developing quantitative recommendations for defensible, variable-width buffers.

A high level of uncertainty still permeates all aspects of the primary question addressed by this review: "How wide does a streamside forest buffer need to be to protect water quality, assure natural stream habitat, and maintain the natural structure of important stream communities?" The studies varied widely with regard to the ecosystem properties, components, or functions they measured, as well as what they actually measured, and how, where and at what frequency they measured it. Despite this uncertainty, we conclude based on the literature on eight major stream or streamside ecosystem factors (properties, components, or functions), that streamside forest buffers ≥30 m wide are needed to protect water quality, habitat, and biotic features of streams associated with watersheds ≤100 km^2, or about fifth order or smaller in size. This conclusion is similar to Wenger's (1999) report which remains a timely complement to the present review in several respects. Here we have emphasized that streamside forests not only protect water quality but promote stable, healthy, and functional stream ecosystems as well.

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