# Meta-Analysis of Nitrogen Removal in Riparian Buffers

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## ABSTRACT

**Riparian buffers, the vegetated region adjacent to streams and wetlands, are thought to be effective at intercepting and reducing nitrogen loads entering water bodies. Riparian buffer width is thought to be positively related to nitrogen removal effectiveness by influencing nitrogen retention or removal. We surveyed the scientific literature containing data on riparian buffers and nitrogen concentration in streams and groundwater to identify trends between nitrogen removal effectiveness and buffer width, hydrological flow path, and vegetative cover. Nitrogen removal effectiveness varied widely. Wide buffers (>50 m) more consistently removed significant portions of nitrogen entering a riparian zone than narrow buffers (0–25 m). Buffers of various vegetation types were equally effective at removing nitrogen but buffers composed of herbaceous and forest/herbaceous vegetation were more effective when wider. Subsurface removal of nitrogen was efficient, but did not appear to be related to buffer width, while surface removal of nitrogen was partly related to buffer width. The mass of nitrate nitrogen removed per unit length of buffer did not differ by buffer width, flow path, or buffer vegetation type. Our meta-analysis suggests that buffer width is an important consideration in managing nitrogen in watersheds. However, the inconsistent effects of buffer width and vegetation on nitrogen removal suggest that soil type, subsurface hydrology (e.g., soil saturation, groundwater flow paths), and subsurface biogeochemistry (organic carbon supply, nitrate inputs) also are important factors governing nitrogen removal in buffers.**

The USEPA considers nitrogen one of the primary stressors in aquatic ecosystems (USEPA, 2002a). Though nitrogen is an important nutrient for all organisms, excess nitrogen is a pollutant that causes eutrophication in surface water and contaminates groundwater (Carpenter et al., 1998). Streams receive chronic nitrogen inputs in various chemical forms such as nitrate ($NO_3^-$), ammonia ($NH_3$), and organic N from upland sources such as fertilizers, animal wastes, leaf litter, leaking sewer lines, atmospheric deposition, and highways (Carpenter et al., 1998; Swackhamer et al., 2004). Subsequent eutrophication leads to environmental impacts such as toxic algal blooms, oxygen depletion, fish kills, and loss of biodiversity (Vitousek et al., 1997).

Nitrogen enters aquatic ecosystems in various forms through multiple pathways. For example, nitrous oxides ($NO_X$) enter by atmospheric deposition, whereas $NO_3^-$ often enters through groundwater and particulate nitrogen in the form of plant litter and other detritus follows terrestrial routes. $NO_3^-$ is of particular concern as an environmental stressor because it is biologically reactive, poses a human health risk (i.e., methemoglobinemia; USEPA, 2002b), and often is found in groundwater (Welch, 1991).

Riparian buffers are thought to be an effective, sustainable means of protecting aquatic ecosystems against anthropogenic inputs of nitrogen (Phillips, 1989; Verhoeven et al., 2006) in which nitrogen species may be transformed by various processes including plant uptake, microbial immobilization, soil storage, and groundwater mixing (Lowrance et al., 1997) and denitrification, a microbially mediated transformation of $NO_3^-$ to $N_2$, a gas phase of nitrogen (Korom, 1992). Denitrification removes nitrogen from a system, whereas other biological processes such as uptake by plants eventually return nitrogen to the system through senescence and microbial decay.

Establishing riparian buffers often is considered a best management practice (BMP) by state and federal resource agencies for maintaining water quality (NRCS, 2003; Bernhardt et al., 2005b). Buffer effectiveness depends on buffer ability to intercept and attenuate nitrogen traveling along surface or subsurface pathways. The extent to which riparian buffers attenuate nitrogen and subsequently improve water quality is thought to be a function of buffer width in concert with landscape and hydrogeomorphic characteristics (Vidon and Hill, 2004). By some estimates, the width of a buffer accounts for about 80% of that buffer's nitrogen removal effectiveness (Phillips, 1989). Intuitively, larger and wider riparian buffers should transform and remove more nitrogen from the water. Therefore, numerous State and Federal agencies have guidelines recommending buffers of minimum width to protect stream ecosystems from nutrient inputs (Belt et al., 1992; Christensen, 2000; Lee et al., 2004; Mayer et al., 2005). However, the specific mechanisms responsible for removing nitrogen within buffers are not thoroughly understood. Furthermore, existing information about buffer effectiveness is not synthesized in a practical form and may not be widely distributed to resource managers (Hickey and Doran, 2004). Moreover, managers do not typically have the available resources to assess the effectiveness of site-specific buffers. The purpose of this article is to identify trends in the relations between nitrogen removal capacity and buffer width, as well as hydrological flow path and vegetative cover, extracted from peer-reviewed studies containing empirical data on buffer effectiveness. While we do not provide specific recommendations for buffer width, this meta-analysis of current literature is meant to provide a baseline from which management decisions about riparian buffers can be made in the context of nitrogen attenuation.

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## MATERIALS AND METHODS

Riparian buffers are defined as the zone of vegetation adjacent to streams, rivers, or wetlands (i.e., Lee et al., 2004). For this article, riparian buffer, riparian zone, buffer strip, filter strip, and vegetated filter strip are considered synonyms. We employed database search engines (e.g., Cambridge Abstracts, Google Scholar, etc.) and existing bibliographies (e.g., Correll, 2003) to locate riparian buffer zone literature. We used search terms singly or combination including: riparian, buffer, width, filter strip, vegetated filters, nitrogen, etc. We summarized the results and conclusions from peer-reviewed research papers that contained original data quantifying the effects of riparian buffer width on nitrogen attenuation. Papers that did not relate nitrogen removal to buffer width were not included in the results. Data presented in proceedings and other non-peer-reviewed sources were not included in our meta-analysis.

We calculated nitrogen removal effectiveness in two ways. First, as a percentage based on (i) the percent difference in nitrogen concentration between the influent into and effluent out of the riparian buffer, (ii) percent difference in nitrogen concentration between the terminus of the control buffer and that of the test buffer, or (iii) if recalculation were impossible based on available data, the values presented by the authors were used directly (Appendix 1). We did not distinguish among nitrogen forms when calculating effectiveness as a percentage.

Because $NO_3^-$ was the form of nitrogen most often measured among studies, we also calculated buffer effectiveness as the mean mass of nitrate nitrogen removed in riparian zones per unit distance where authors provided information on influent and effluent concentrations.

Removal effectiveness as a percentage was plotted against buffer width. Linear and nonlinear regression models were fitted to the data to reveal patterns of nitrogen removal based on width. All buffers included in studies for which efficiencies could be calculated were included in the meta-analyses as independent data points.

We grouped studies by vegetation cover type (forest, forested wetland, wetland, herbaceous, herbaceous/forest mix) and by hydrologic flow conditions (e.g., surface vs. subsurface), factors that may influence nutrient attenuation in riparian buffer zones. We then plotted effectiveness against buffer width by these groups.

We also grouped studies by buffer width category (0–25, 26–50, and >50 m, respectively). We chose these categories based on current state recommendations for minimum buffer widths which currently range from 15.5 to 24.2 m (Mayer et al., 2005). Therefore our three width categories include buffers that are as wide as current recommendations (0–25 m), those twice as wide (26–50 m), and buffers much wider than recommended (>50 m). We then analyzed effectiveness (percentage nitrogen removal and nitrate removal per unit length) among buffer factor groups (width category, flow path, and vegetation type) using non-parametric tests because the dependent variables were not normally distributed (Shapiro–Wilk test for normality, $P < 0.001$). All analyses and model fitting were performed with Systat 11.0, Sigma Stat 3.1, and SigmaPlot 9.0 software (SSI, 2004).

## RESULTS

### Buffer Effectiveness

### Overall Patterns

We analyzed data from 89 individual riparian buffers from 45 published studies. Nitrogen removal effectiveness varied widely among studies (Appendix 1). Removal effectiveness at one site was calculated as −258% (Appendix 1), due apparently to very low influent (0.12 mg $L^{-1}$) and effluent (0.43 mg $L^{-1}$) nitrate concentrations, and was removed from further analysis as an outlier. The remaining data showed that overall, buffers were effective at removing large proportions of the nitrogen from water flowing through riparian zones (mean % ± 1 standard error [SE]: 67.5 ± 4.0, $N = 88$; Table 1).

A small but significant proportion of the variance in removal of nitrogen was explained by buffer width ($R^2 = 0.09$, $P = 0.005$, $N = 88$; Fig. 1, Table 1). That is, wider buffers tended to remove more nitrogen, but other factors must also have affected effectiveness. Overall, exponential models ($y = ax^b$) were the simplest models that best fit the effectiveness to buffer relationships. Accordingly, 50, 75, and 90% removal efficiencies were estimated to occur among all buffers approximately 4, 49, and 149 m wide, respectively (Fig. 1, Table 1). These estimates had large variances based on SE of the regression models (Table 1).

**Table 1. Percent effectiveness of riparian buffers at removing nitrogen. Buffer widths necessary to achieve a given percent effectiveness (50, 75, 90%) are approximate values predicted by the nonlinear model, $y = ax^b$. Effectiveness was not predicted (np) for models with $R^2$ Values ≤ 0.2 except for "all studies" model.**

| Buffer variable | $N$ | Mean removal effectiveness | Regression model | $R^2$ | SE† | $P$ | Approximate buffer width by predicted effectiveness<br>50% | <br>75% | <br>90% |
|---|---|---|---|---|---|---|---|---|---|
| | | % ± 1 SE | | | | | | m | |
| **All studies** | 88 | 67.5 ± 4.0 | $y = 39.5x^{0.1644}$ | 0.09 | 36.2 | 0.005 | 4 | 49 | 149 |
| **Width category** | | | | | | | | | |
| 0–25 m | 45 | 57.9 ± 6.0 | $y = 42.1x^{0.1337}$ | 0.01 | 40.7 | 0.5 | np | np | np |
| 26–50 m | 24 | 71.4 ± 7.8 | $y = 50.6x^{0.0964}$ | 0.00 | 39.0 | 0.8 | np | np | np |
| >50 m | 19 | 85.2 ± 4.8 | $y = 56.9x^{0.0883}$ | 0.03 | 21.0 | 0.5 | np | np | np |
| **Flow path** | | | | | | | | | |
| Surface | 23 | 41.6 ± 7.1 | $y = 14.6x^{0.3722}$ | 0.21 | 30.9 | 0.03 | 27 | 81 | 131 |
| Subsurface | 65 | 76.7 ± 4.3 | $y = 62.0x^{0.0631}$ | 0.02 | 34.7 | 0.3 | np | np | np |
| **Vegetation type** | | | | | | | | | |
| Forest | 31 | 72.2 ± 6.9 | $y = 45.7x^{0.1225}$ | 0.04 | 38.4 | 0.3 | np | np | np |
| Forested wetland | 7 | 85.0 ± 5.2 | $y = 85.0x^{0.0809}$ | 0.00 | 15.1 | 1.0 | np | np | np |
| Herbaceous | 32 | 54.0 ± 7.5 | $y = 18.0x^{0.3631}$ | 0.21 | 38.5 | 0.009 | 17 | 51 | 84 |
| Herbaceous/forest | 11 | 79.5 ± 7.3 | $y = 41.5x^{0.2044}$ | 0.39 | 20.1 | 0.04 | 3 | 18 | 44 |
| Wetland | 7 | 72.3 ± 11.9 | $y = 67.8x^{0.0244}$ | 0.01 | 34.2 | 0.9 | np | np | np |

† SE represents the standard error of the regression estimate.

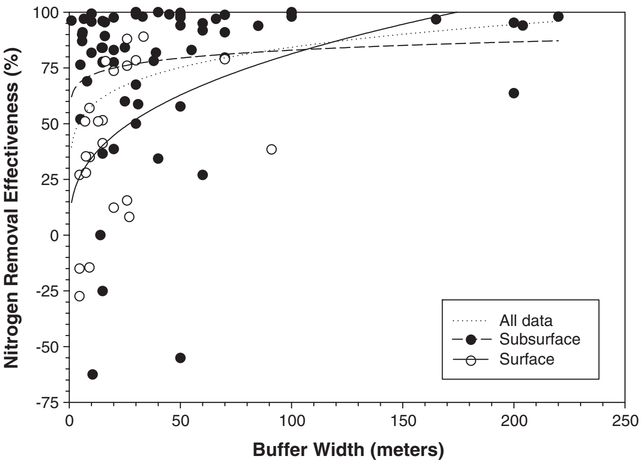

**Fig. 1. Relationships of nitrogen removal effectiveness to riparian buffer width over all studies and analyzed by water flow path. Lines are fitted to model $y = ax^b$.**

## Buffer Width Category

Effectiveness was not related to buffer width when analyzing buffers within width categories ($P > 0.5$, Table 1), suggesting that any effect of buffer width on nitrogen removal occurs only after buffer size reaches a width threshold. This suggestion is supported by the observation that effectiveness differed among buffer width categories (Kruskal–Wallis $H = 10.3$, df = 2, $P = 0.006$; Fig. 2, Table 1). Nitrogen removal effectiveness of buffers >50 m wide was greater than that of buffers 0 to 25 m, whereas effectiveness of buffers 26 to 50 m did not differ from the other categories (Dunn's method of multiple comparisons $Q = 3.0$, $P < 0.05$; Fig. 2, Table 1). Thus, wider buffers are likely to be more efficient zones of nitrogen removal than narrower buffers.

## Surface versus Subsurface Flow

Nitrogen removal effectiveness also differed by flow pattern. Subsurface removal of nitrogen was much more efficient than surface removal (Mann–Whitney $U = 1247.5$, df = 1, $P < 0.001$; Fig. 3, Table 1). Furthermore, subsurface removal of nitrogen did not appear to be related to buffer width ($R^2 = 0.02$, $P = 0.3$; Fig. 1, Table 1), whereas a small but significant proportion of the variance in surface removal of nitrogen was explained by buffer width ($R^2 = 0.21$, $P = 0.03$; Fig. 1, Table 1). That is, wider buffers removed more nitrogen in surface runoff. While some narrow buffers (<15 m) removed significant proportions of nitrogen, six studies (three surface and three subsurface flow) found that narrow buffers actually contributed nitrogen to riparian zones (i.e., had negative effectiveness values; Appendix 1; Fig. 1). Such cases are likely to be short-term events due to nitrification or high rainfall events that lead to rapid inputs of nitrogen (Dillaha et al., 1988; Magette et al., 1989; Sabater et al., 2003). Based on the model $y = ax^b$, 50, 75, and 90% nitrogen removal efficiencies in surface flow were estimated to occur in buffers approximately 27, 81, and 131 m wide, respectively (Fig. 1, Table 1). These models also had large associated variances (SE; Table 1).

## Vegetation Type

Overall nitrogen removal effectiveness did not vary by buffer vegetation type (Kruskal–Wallis $H = 6.9$, df = 4, $P = 0.14$; Fig. 4 and Table 1) suggesting that all buffers were equally effective at removing nitrogen. Forested, forested/wetland, and wetland buffers showed no relationship between buffer width and nitrogen removal effectiveness; however, effectiveness of herbaceous and herbaceous/forested buffers increased with width (Fig. 5, Table 1). Based on the model $y = ax^b$, nitrogen removal efficiencies of 50, 75, and 90% were estimated for herbaceous buffers approximately 17, 51, and 84 m wide and for herbaceous/forest buffers approximately 3, 18, and 44 m wide, respectively (Table 1). Models had large variances (SE; Table 1). Four herbaceous and two forested buffers added to nitrogen loads where buffers were <15 m (Fig. 5). In such cases, nitrification or high

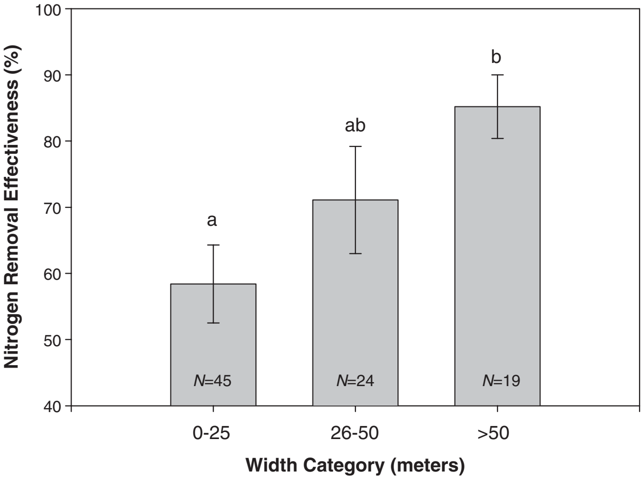

**Fig. 2. Nitrogen removal effectiveness in riparian buffers by buffer width category. Bars represent means ± 1 standard error. Mean ranks of width categories differ if denoted by different letters (Kruskal–Wallis one-way analysis of variance on ranks with Dunn's method of multiple comparisons, $P < 0.05$).**

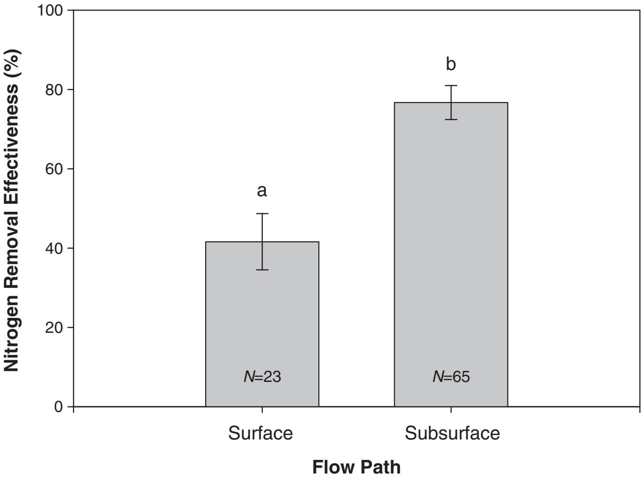

**Fig. 3. Nitrogen removal effectiveness in riparian buffers by water flow path. Bars represent means ± 1 standard error. Mean ranks of flow paths differ if denoted by different letters (Mann–Whitney $U$ test on ranks, $P < 0.001$).**

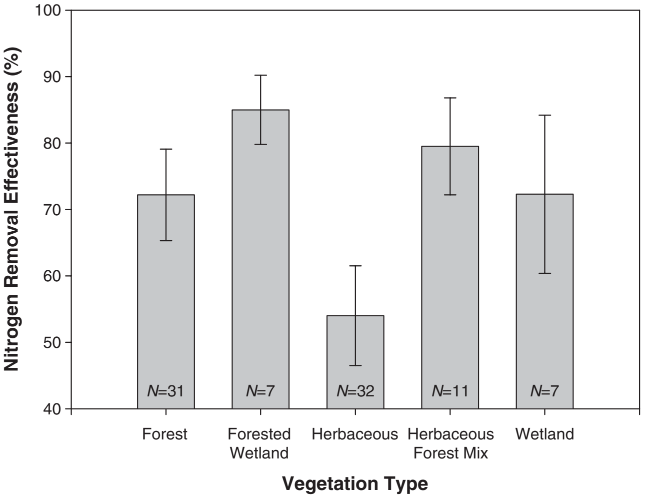

**Fig. 4. Nitrogen removal effectiveness in riparian buffers by buffer vegetation type. Bars represent means ± 1 standard error. Mean ranks of vegetation types do not differ (Kruskal–Wallis one-way analysis of variance on ranks, $P = 0.14$).**

rainfall events may lead to short-term and/or rapid inputs of nitrogen (Sabater et al., 2003).

## Mass Removal of Nitrate Nitrogen

We analyzed data from 60 riparian buffers for which influent and effluent nitrate nitrogen concentrations were available. Similar to percent removal effectiveness, mass removal of nitrate nitrogen per unit length varied widely among studies. Overall, buffers removed nitrate nitrogen at a rate of (mean ± 1 SE) 0.394 ± 0.084 mg $L^{-1}$ $m^{-1}$. Unlike effectiveness, nitrate nitrogen removal did not differ among width categories (Kruskal–Wallis $H = 4.8$, df = 2, $P = 0.09$; Table 2), suggesting that nitrate removal rate remained constant across the entire length of buffers.

Nitrate removal was not related to flow pattern (Mann–Whitney $U = 256.0$, df = 1, $P = 0.11$; Table 2). Nitrate removal also was not related to buffer vegetation type (Kruskal–Wallis, df = 4, $H = 7.3$, $P = 0.12$; Fig. 6, Table 2).

**Table 2. Mass removal of nitrate nitrogen in riparian buffers.**

| Buffer variable | *N* | Mean mass of $NO_3^-$ removed per unit length | 1 SE† |
|---|---|---|---|
| | | mg $L^{-1}$ $m^{-1}$ | |
| **All studies** | **60** | **0.394** | **0.084** |
| **Width category** | | | |
| **0–25 m** | **25** | **0.463** | **0.106** |
| **26–50 m** | **19** | **0.377** | **0.127** |
| **>50 m** | **16** | **0.305** | **0.227** |
| **Flow path** | | | |
| **Surface flow** | **7** | **0.339** | **0.299** |
| **Subsurface flow** | **53** | **0.401** | **0.087** |
| **Vegetative cover** | | | |
| **Forest** | **26** | **0.186** | **0.065** |
| **Forested wetland** | **3** | **0.617** | **0.333** |
| **Herbaceous** | **19** | **0.497** | **0.199** |
| **Herbaceous/forest** | **6** | **0.293** | **0.138** |
| **Wetland** | **6** | **0.957** | **0.359** |

**† 1 SE represents 1 standard error of the means.**

## DISCUSSION

Our meta-analysis suggests that wider buffers tend to be more effective at removing nitrogen. Low $R^2$ values of the overall regression analysis suggest that factors other than buffer width influence buffer effectiveness such as (i) vegetation and depth of the root zone where plants can take up nitrogen (Asmussen et al., 1979; Cooper, 1990), and (ii) hydrological flow paths that favor microbial denitrification (i.e., saturated anaerobic soils, adequate carbon supplies, floodplain connections; Dillaha et al., 1989; Simmons et al., 1992; Hanson et al., 1994; Speiran et al., 1998; Leeds-Harrison et al., 1999; Sloan et al., 1999; Hill et al., 2000, 2004; Steinhart et al., 2001; Schade et al., 2001, 2002; Groffman et al., 2003, 2005; Sabater et al., 2003; Richardson et al., 2004). Furthermore, buffer width was not a factor affecting nitrogen removal effectiveness within buffer width categories, indicating that trends in effectiveness are evident only across a broader range of buffer size. Yet, mean

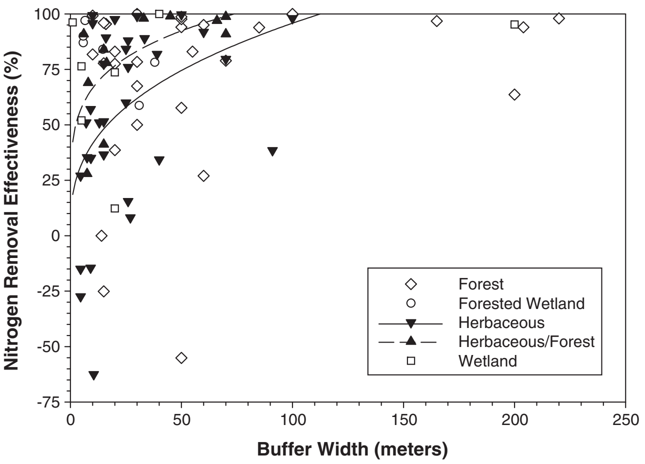

**Fig. 5. Relationships of nitrogen removal effectiveness to riparian buffer width analyzed by vegetation type. Lines are fitted to model $y = ax^b$. Only the regression lines for herbaceous and herbaceous/forest vegetation types are shown because model results for other vegetation types were not significant ($P > 0.3$).**

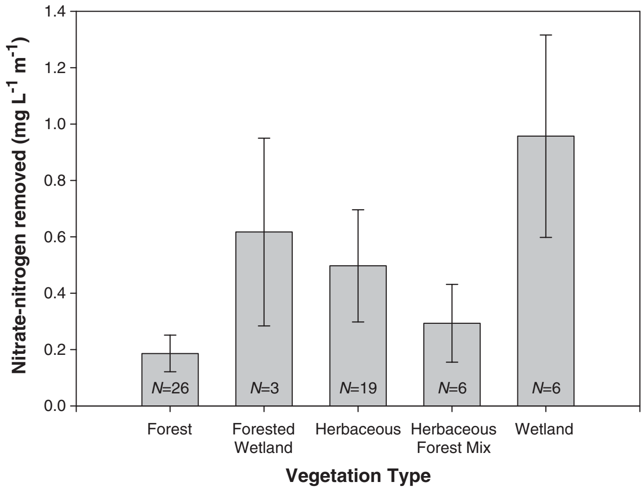

**Fig. 6. Mass of nitrate nitrogen removed in riparian buffers by buffer vegetation type. Bars represent means ± 1 standard error. Mean ranks of vegetation types do not differ (Kruskal–Wallis one-way analysis of variance on ranks, $P = 0.12$).**

nitrogen removal effectiveness in buffers >50 m wide was significantly higher than in narrow buffers (0–25 m), suggesting that buffer width is an important consideration for nitrogen management in watersheds.

Overall, suburface nitrogen removal is more efficient than removal through surface flow. Furthermore, subsurface nitrogen removal may be more directly influenced by soil type, watershed hydrology (e.g., soil saturation, groundwater flow paths, etc.), and subsurface biogeochemistry (organic carbon supply, high $NO_3^-$ inputs) through cumulative effects on microbial denitrification activity than on buffer width per se. Surface flows bypass zones of denitrification, and thus effectively remove nitrogen only when buffers are wide enough and have adequate vegetation cover to control erosion and filter movement of particulate forms of nitrogen. Herbaceous buffers, for example, may be better at intercepting particulate nitrogen in the sediments of surface runoff by reducing channelized flow. Based on a limited data set fitted to a log-linear model, Oberts and Plevan (2001) found that $NO_3^-$ retention in wetland buffers was positively related to buffer width ($R^2$ values ranged from 0.35–0.45). Nitrogen removal efficiencies of 65 to 75% and 80 to 90% were predicted for wetland buffers 15 and 30 m wide, respectively, depending on whether $NO_3^-$ was measured in surface or subsurface flow (Oberts and Plevan, 2001).

Our meta-analysis suggests that vegetation type has a limited impact on buffer effectiveness (Table 1). Only buffers with herbaceous vegetation were more effective when wider (Table 1). However, buffer width may indirectly affect factors promoting denitrification. For example, narrow buffers that produce little vegetative biomass may not provide sufficient stocks of organic material for microbial denitrifiers.

Regardless of width, buffer integrity should be protected against (i) soil compaction (e.g., vehicles, livestock, and construction of impervious surfaces) that might inhibit infiltration or disrupt water flow patterns (Dillaha et al., 1989; NRC, 2002), (ii) excessive leaf litter removal or alteration of the natural plant community (e.g., raking, tree thinning, introduction of invasive species) that might reduce carbon-rich organic matter from reaching the stream, and (iii) practices that might disconnect the stream channel from the flood plain (i.e., urbanization, channelization, bank erosion, stream incision, hard drainage surfaces, and drain tiles) and thereby reduce the spatial and temporal extent of soil saturation (Paul and Meyer, 2001; Groffman et al., 2003, 2005).

## CONCLUSIONS

Based on our meta-analysis, riparian buffers of various types are effective at reducing nitrogen in riparian zones, especially nitrogen flowing in the subsurface. Our study shows that, while some narrow buffers (0–25 m) remove nitrogen, wider buffers (>50 m) more consistently removed significant portions of nitrogen probably by providing more area for root uptake of nitrogen (Asmussen et al., 1979; Cooper, 1990) or more sites where groundwater conditions favor denitrification (Hanson et al., 1994; Leeds-Harrison et al., 1999; Sloan et al., 1999; Hill et al., 2000, 2004; Schade et al., 2001, 2002; Steinhart et al., 2001; Sabater et al., 2003; Richardson et al., 2004). Maintaining buffers around stream headwaters (Peterson et al., 2001; Richardson et al., 2004; Bernhardt et al., 2005a; Bernot and Dodds, 2005) will likely be most effective at maintaining overall watershed water quality while restoring degraded riparian zones, and stream channels may improve nitrogen removal capacity (Groffman et al., 2005). However, because streams and riparian zones have limited capacity to process nitrogen, watershed nutrient management efforts also must include control and reduction of point and nonpoint sources of nitrogen from atmospheric, terrestrial, and aquatic inputs. Furthermore, overtaxing the nutrient removal capacity of riparian zones and floodplain wetlands may lead to losses of biodiversity and production of nitrous oxides (Verhoeven et al., 2006). Establishing a network of buffers adequate to maintain watershed water quality will be dependent on local and centralized conservation activities as well as government regulations and standards (Mayer et al., 2005; Verhoeven et al., 2006).

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**Appendix 1. Summary table of riparian buffer effectiveness at removing nitrogen (N) by vegetation type, hydrologic flow path, buffer width, and soil type.**

| | | | | $NO_3^-$ Concentration | | | | |
|---|---|---|---|---|---|---|---|---|
| Vegetation type | Flow path | Buffer width | N form | Mean influent | Mean effluent | Nitrogen removal effectiveness | $NO_3^-$ removed | Study |
| | | m | | mg $L^{-1}$ | | % | mg $L^{-1}$ $m^{-1}$ | |
| Herbaceous | surface | 4.6 | Total N | – | – | −15 | – | Magette et al. (1989) |
| Herbaceous | surface | 9.2 | Total N | – | – | 35 | – | Magette et al. (1989) |
| Herbaceous | surface | 7.5 | Total N | 68 | 44 | 35 | – | Schmitt et al. (1999) |
| Herbaceous | surface | 15 | Total N | 68 | 33 | 51 | – | Schmitt et al. (1999) |
| Herbaceous | surface | 4.6 | $NO_3^-$ | 1.86 | 2.37 | −27 | −0.11 | Dillaha et al. (1988) |
| Herbaceous | surface | 9.1 | $NO_3^-$ | 1.86 | 2.13 | −15 | −0.03 | Dillaha et al. (1988) |
| Herbaceous | surface | 4.6 | $NO_3^-$ | – | – | 27 | – | Dillaha et al. (1989) |
| Herbaceous | surface | 9.1 | $NO_3^-$ | – | – | 57 | – | Dillaha et al. (1989) |
| Herbaceous | surface | 91 | Total N | 21.6 | 13.3 | 38 | – | Zirschky et al. (1989) |
| Herbaceous | surface | 27 | $NO_3^-$ | 0.37 | 0.34 | 8 | <0.01 | Young et al. (1980) |
| Herbaceous | surface | 26 | $NH_3$ | 3.61 | 3.05 | 16 | – | Schwer and Clausen (1989) |
| Herbaceous | surface | 26 | TKN | 48.9 | 11.76 | 76 | – | Schwer and Clausen (1989) |
| Herbaceous | surface | 7.1 | $NO_3^-$ | – | – | 51 | – | Lee et al. (2003) |
| Herbaceous | surface | 13 | $NO_3^-$ | – | – | 51 | – | Bingham et al. (1980) |
| Herbaceous | surface | 33.4 | $NO_3^-$ | – | – | 89 | – | Bingham et al. (1980) |
| Herbaceous | surface | 26 | $NO_3^-$ | – | – | 88 | – | Bingham et al. (1980) |
| Herbaceous | subsurface | 40 | $NO_3^-$ | 0.35 | 0.23 | 34 | <0.01 | Sabater et al. (2003) |
| Herbaceous | subsurface | 60 | $NO_3^-$ | 1.7 | 0.14 | 92 | 0.03 | Sabater et al. (2003) |
| Herbaceous | subsurface | 20 | $NO_3^-$ | 12.42 | 0.30 | 98 | 0.61 | Sabater et al. (2003) |
| Herbaceous | subsurface | 10.5 | $NO_3^-$ | 0.08 | 0.13 | −63 | −0.01 | Sabater et al. (2003) |
| Herbaceous | subsurface | 15 | $NO_3^-$ | 11.56 | 7.34 | 37 | 0.28 | Sabater et al. (2003) |
| Herbaceous | subsurface | 15 | $NO_3^-$ | 12.35 | 2.79 | 77 | 0.64 | Sabater et al. (2003) |
| Herbaceous | subsurface | 25 | $NO_3^-$ | 15.5 | 6.2 | 60 | 0.37 | Vidon and Hill (2004) |
| Herbaceous | subsurface | 70 | $NO_3^-$ | 1.55 | 0.32 | 80 | 0.02 | Martin et al. (1999) |
| Herbaceous | subsurface | 39 | $NO_3^-$ | 16.5 | 3 | 82 | 0.35 | Osborne and Kovacic (1993) |
| Herbaceous | subsurface | 25 | $NO_3^-$ | 12.15 | 1.92 | 84 | 0.41 | Hefting and de Klein (1998) |
| Herbaceous | subsurface | 16 | $NO_3^-$ | 2.8 | 0.3 | 89 | 0.16 | Haycock and Burt (1993) |
| Herbaceous | subsurface | 10 | $NO_3^-$ | 7 | 0.3 | 96 | 0.67 | Hefting et al. (2003) |
| Herbaceous | subsurface | 100 | $NO_3^-$ | 375 | < 5 | 98 | 3.70 | Prach and Rauch (1992) |
| Herbaceous | subsurface | 10 | $NO_3^-$ | 7.54 | 0.05 | 99 | 0.75 | Schoonover and Williard (2003) |
| Herbaceous | subsurface | 30 | $NO_3^-$ | 44.7 | 0.45 | 99 | 1.48 | Vidon and Hill (2004) |
| Herbaceous | subsurface | 50 | $NO_3^-$ | 6.6 | 0.02 | 100 | 0.13 | Martin et al. (1999) |
| Herbaceous/forest | surface | 7.5 | Total N | 68 | 49 | 28 | – | Schmitt et al. (1999) |
| Herbaceous/forest | surface | 15 | Total N | 68 | 40 | 41 | – | Schmitt et al. (1999) |
| Herbaceous/forest | surface | 16.3 | $NO_3^-$ | – | – | 78 | – | Lee et al. (2003) |
| Herbaceous/forest | subsurface | 8 | $NO_3^-$ | – | – | 69 | – | Dukes et al. (2002)† |
| Herbaceous/forest | subsurface | 15 | $NO_3^-$ | – | – | 84 | – | Dukes et al. (2002)† |
| Herbaceous/forest | subsurface | 6 | $NO_3^-$ | 6.17 | 0.56 | 91 | 0.94 | Borin and Bigon (2002) |
| Herbaceous/forest | subsurface | 70 | $NO_3^-$ | 11.98 | 1.09 | 91 | 0.16 | Hubbard and Lowrance (1997) |
| Herbaceous/forest | subsurface | 66 | $NO_3^-$ | 5.8 | 0.17 | 97 | 0.09 | Vidon and Hill (2004) |
| Herbaceous/forest | subsurface | 33 | $NO_3^-$ | 5.7 | 0.11 | 98 | 0.17 | Vidon and Hill (2004) |
| Herbaceous/forest | subsurface | 45 | $NO_3^-$ | 17.8 | 0.18 | 99 | 0.39 | Vidon and Hill (2004) |
| Herbaceous/forest | subsurface | 70 | $NO_3^-$ | 1.65 | 0.02 | 99 | 0.02 | Martin et al. (1999) |
| Forest | surface | 30 | $NO_3^-$ | 0.37 | 0.08 | 78 | 0.01 | Lynch et al. (1985) |
| Forest | surface | 70 | $NO_3^-$ | 4.45 | 0.94 | 79 | 0.05 | Peterjohn and Correll (1984) |
| Forest | subsurface | 50 | $NO_3^-$ | 26 | 11 | 58 | 0.30 | Hefting et al. (2003) |
| Forest | subsurface | 200 | $NO_3^-$ | 11 | 4 | 64 | 0.04 | Spruill (2004) |
| Forest | subsurface | 10 | $NO_3^-$ | 6.29 | 1.15 | 82 | 0.51 | Schoonover and Williard (2003) |
| Forest | subsurface | 14 | $NO_3^-$ | 0.02 | 0.02 | 0 | 0.00 | Sabater et al. (2003) |
| Forest | subsurface | 30 | $NO_3^-$ | 0.02 | 0.01 | 50 | <0.01 | Sabater et al. (2003) |
| Forest | subsurface | 50 | $NO_3^-$ | 0.49 | 0.76 | −55 | −0.01 | Sabater et al. (2003) |
| Forest | subsurface | 15 | $NO_3^-$ | 28.64 | 35.84 | −25 | −0.48 | Sabater et al. (2003) |
| Forest | subsurface | 20 | $NO_3^-$ | 1.14 | 0.70 | 39 | 0.02 | Sabater et al. (2003) |
| Forest | subsurface | 20 | $NO_3^-$ | 0.12 | 0.43 | −258 | −0.02 | Sabater et al. (2003) |
| Forest | subsurface | 15 | $NO_3^-$ | 3.23 | 0.72 | 78 | 0.17 | Sabater et al. (2003) |
| Forest | subsurface | 20 | $NO_3^-$ | 6.40 | 1.44 | 78 | 0.25 | Sabater et al. (2003) |
| Forest | subsurface | 55 | $NO_3^-$ | – | – | 83 | – | Lowrance et al. (1984) |
| Forest | subsurface | 20 | $NO_3^-$ | – | – | 83 | – | Schultz et al. (1995) |
| Forest | subsurface | 85 | $NO_3^-$ | 7.08 | 0.43 | 94 | 0.08 | Peterjohn and Correll (1984) |
| Forest | subsurface | 204 | $NO_3^-$ | 29.4 | 1.76 | 94 | 0.14 | Vidon and Hill (2004) |
| Forest | subsurface | 50 | $NO_3^-$ | 13.52 | 0.81 | 94 | 0.25 | Lowrance (1992) |
| Forest | subsurface | 60 | $NO_3^-$ | 8 | 0.4 | 95 | 0.13 | Jordan et al. (1993) |
| Forest | subsurface | 16 | $NO_3^-$ | 16.5 | 0.75 | 95 | 0.98 | Osborne and Kovacic (1993) |
| Forest | subsurface | 16 | $NO_3^-$ | 6.6 | 0.3 | 95 | 0.39 | Haycock and Pinay (1993) |
| Forest | subsurface | 15 | $NO_3^-$ | – | – | 96 | – | Hubbard and Sheridan (1989) |
| Forest | subsurface | 165 | $NO_3^-$ | 30.8 | 1 | 97 | 0.18 | Hill et al. (2000) |
| Forest | subsurface | 50 | $NO_3^-$ | 6.26 | 0.15 | 98 | 0.12 | Hefting and de Klein (1998) |
| Forest | subsurface | 220 | $NO_3^-$ | 10.8 | 0.22 | 98 | 0.05 | Vidon and Hill (2004) |
| Forest | subsurface | 50 | $NO_3^-$ | 7.45 | 0.1 | 99 | 0.15 | Jacobs and Gilliam (1985) |
| Forest | subsurface | 10 | $NO_3^-$ | 13 | 0.1 | 99 | 1.29 | Cey et al. (1999) |
| Forest | subsurface | 100 | $NO_3^-$ | 5.6 | 0.02 | 100 | 0.06 | Spruill (2004) |
| Forest | subsurface | 30 | $NO_3^-$ | 1.32 | nd | 100 | 0.04 | Pinay and Decamps (1988) |
| Forest | subsurface | 100 | $NO_3^-$ | 12 | nd | 100 | 0.12 | Spruill (2004) |

**Continued on next page.**

**Appendix 1. Continued.**

| | | | | $NO_3^-$ Concentration | | | | |
|---|---|---|---|---|---|---|---|---|
| **Vegetation type** | **Flow path** | **Buffer width** | **N form** | **Mean influent** | **Mean effluent** | **Nitrogen removal effectiveness** | **$NO_3^-$ removed** | **Study** |
| | | m | | mg $L^{-1}$ | | % | mg $L^{-1}$ $m^{-1}$ | |
| Forest | subsurface | 60 | $NO_3^-$ | – | – | 27 | – | Groffman et al. (1996) |
| Forest | subsurface | 30 | $NO_3^-$ | – | – | 68 | – | Spruill (2000)‡ |
| Forested wetland | subsurface | 31 | $NO_3^-$ | 62.7 | 25.9 | 59 | 1.19 | Hanson et al. (1994) |
| Forested wetland | subsurface | 38 | $NO_3^-$ | 30.6 | 6.7 | 78 | 0.63 | Vellidis et al. (2003) |
| Forested wetland | subsurface | 14.6 | $NO_3^-$ | – | – | 84 | – | Simmons et al. (1992) |
| Forested wetland | subsurface | 5.8 | $NO_3^-$ | – | – | 87 | – | Simmons et al. (1992) |
| Forested wetland | subsurface | 5.8 | $NO_3^-$ | – | – | 90 | – | Simmons et al. (1992) |
| Forested wetland | subsurface | 6.6 | $NO_3^-$ | – | – | 97 | – | Simmons et al. (1992) |
| Forested wetland | subsurface | 30 | $NO_3^-$ | 1.06 | nd | 100 | 0.04 | Pinay et al. (1993) |
| Wetland | surface | 20 | $NO_3^-$ | 57 | 50 | 12 | 0.35 | Brüsch and Nilsson (1993) |
| Wetland | surface | 20 | $NO_3^-$ | 57 | 15 | 74 | 2.10 | Brüsch and Nilsson (1993) |
| Wetland | subsurface | 5 | $NO_3^-$ | 6.56 | 1.55 | 76 | 1.00 | Clausen et al. (2000) |
| Wetland | subsurface | 5 | $NO_3^-$ | 3 | 1.44 | 52 | 0.31 | Clausen et al. (2000) |
| Wetland | subsurface | 1 | $NO_3^-$ | 1 | – | 96 | – | Burns and Nguyen (2002) |
| Wetland | subsurface | 200 | $NO_3^-$ | 10.5 | 0.5 | 95 | 0.05 | Fustec et al. (1991) |
| Wetland | subsurface | 40 | $NO_3^-$ | 77.48 | 0.31 | 100 | 1.93 | Puckett et al. (2002) |

† Values represent the average of 16 buffers.

‡ Values represent the average of 14 buffers.