Wetlands and Aquatic Processes

Effects of Sedimentation on Soil Nutrient Dynamics in Riparian Forests

B. G. Lockaby,* R. Governo, E. Schilling, G. Cavalcanti, and C. Hartsfield

ABSTRACT

The influence of sedimentation rates on biogeochemistry of riparian forests was studied near ephemeral streams at Fort Benning, GA. Upper reaches of seven ephemeral streams had received varying rates of sedimentation stemming from erosion along unpaved roadways at the military installation. Two reference catchments were also included in the study. Decomposition of foliar litter, microbial C and N, N mineralization, and arthropod populations were compared within and among catchments. Rates of sedimentation over the past 25 yr ranged from 0 in references to 4.0 cm yr^{-1}. Decomposition rates declined exponentially with sedimentation rates as low as 0.20 to 0.32 cm yr^{-1} and appeared to reach an equilibrium at a sedimentation rate of 0.5 cm yr^{-1}. Nitrogen mineralization and microbial C and N followed the same trend. Sedimentation had no discernible effect on arthropod populations. These data suggest that biogeochemical cycles may be altered by sedimentation rates that commonly occur in some floodplain forests.

The water filtration function of wetland ecosystems is commonly acknowledged to hold enormous potential benefit for society and, appropriately, has a growing influence on public policy (Novotny, 2003). While a conceptual awareness of water filtration by wetlands is widespread, we know much less about specific aspects such as the capacity of various wetland types to accumulate filtrates. Consequently, our limited understanding significantly restricts the extent to which natural filtration provided by wetlands can be utilized for societal well-being.

A key question is how susceptible the water filtration function may be to ecosystem stresses of anthropogenic origin. The impact of land use on sediment export into streams and elevated deposition in wetlands may represent one type of stress. Already causing a very serious water quality problem throughout the world, expanding human populations and resulting landscape changes will continue to heighten the severity of the sediment export issue (Novotny, 2003). Higher sediment loads in streams imply greater sediment accumulation within riparian wetlands, as those systems respond as filters to the increased loadings. Elevated deposition rates have been observed in riverine forests within watersheds that have significant levels of agriculture or urbanization (Thom et al., 2001; Hupp, 2000; Kleiss, 1996; Howarth et al., 1991).

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Filtration of sediments from floodwaters is a normal function of riparian forests (Meyer, 1990) and, depending on geomorphic position and natural characteristics of associated streams, may result in substantial accumulation of sediment on the floodplain. One of the primary determinants of retention capacity is surface roughness, as it relates to reduction in flow velocity. In riparian forests, the principle component associated with roughness is the vegetation, both standing and dead. While some sediment accumulation reflects a natural process in depositional topographic positions (Hupp and Morris, 1990), there exists little understanding of critical levels beyond which further accumulation becomes a stress in regard to ecosystem integrity and function (Jurik et al., 1994; Adamus and Brandt, 1990).

Although floodplain forests are naturally subject to varying degrees of alluviation, upstream or upslope land use may increase sedimentation rates to the point that the riparian vegetation is severely damaged (Bazemore et al., 1991). Severe damage may affect roughness and result in a reduction of the capacity of the system to trap sediment (Hupp and Osterkamp, 1995; Hupp and Bazemore, 1993). Additionally, excess nutrients, trace elements, and other hydrophobic contaminants are often physically bound to fine sediment (Johnston et al., 1984; Hupp et al., 1993) and forested riparian areas function to remove these environmental contaminants. Consequently, excessive sedimentation has the potential to impair the filtration function and negatively affect downstream water quality (Hupp, 1992; Boto and Patrick, 1979).

At a general perspective, some functional changes may be hypothesized as responses to elevated rates of deposition. If sediment influx exceeds sustainable loadings, tree roots may be subjected to increasingly anoxic conditions resulting in growth declines and, ultimately, mortality (Cavalcanti and Lockaby, 2003; Junk and Piedade, 1997; Broadfoot and Williston, 1973; Kennedy, 1970). In long-term scenarios, a vegetation transition may occur in which species capable of tolerating new site conditions (e.g., developing new roots as alluvium aggrades) increase in dominance.

Changes in vegetation caused by excessive sedimentation may result from and/or drive biogeochemical changes including an alteration of decomposition and mineralization patterns. In some cases, burial of foliar litter by sediment has resulted in subtle changes in mass loss rates (Wang et al., 1994; Mayack et al., 1989). Working near blackwater streams in the Coastal Plain of South Carolina, Mayack et al. (1989) found that sediment burial of sweetgum (*Liquidambar styraciflua* L.) foliage caused alterations in microarthropod dynamics compared with patterns observed in unburied litter. Although microarthropod activity was generally suppressed by litter burial, the activity of shredders was enhanced. Consequently, annual mass loss rates were similar for both categories of litter.

In contrast, Herbst (1980) found substantial decreases in mass loss rates when maple (*Acer* spp.) and cottonwood (*Populus* spp.) leaves were buried. Although Wang et al. (1994) found that initial phases of decomposition were slower for buried leaves, no significant effects of sediment burial were apparent after 17 mo. Herbst (1980) and Mayack et al. (1989) speculated that litter burial decreased organic matter export via streams and dampened the seasonal pulses of nutrient inputs to floodplain soils.

The degree to which sediment deposition can promote temporal uniformity in nitrogen supply may be dependent on many factors, such as the textural composition of sediment. Pinay et al. (1995) demonstrated that floodplain retention of nitrogen was far lower if deposition was dominated by coarse-textured vs. fine-textured sediment. Thus, accumulation of sandy alluvium might be expected to cause reductions in available levels of soil nutrients, such as nitrate (NO_3) and potassium (K), that are easily leached.

Many of the forest sedimentation studies have taken place in the floodplains of higher-order rivers and streams. However, riparian zones of headwaters are crucially important in defining down-basin water quality (Brinson, 1993), but are much less studied (Wardrop and Brooks, 1998). Depending on the magnitude of sediment-driven changes within headwater riparian systems, there is great potential for alterations in the filtration function there and, consequently, greater risk of water quality impairment in the associated, higher-order streams. However, there is no information available regarding how ephemeral riparian forests might respond to anthropogenic stresses and, in particular, how nutrient circulation in any riparian forest might be altered by sedimentation. Consequently, the objectives of this investigation were to examine relationships between key biogeochemical attributes and sediment accumulation in ephemeral riparian forests and to identify levels of sediment accumulation beyond which nutrient cycling attributes change markedly.

MATERIALS AND METHODS

Study Site and Design

The study was installed at the Fort Benning Military Installation near Columbus, Georgia, USA, where disturbance from mechanized vehicle maneuvers has generated significant sediment movement. The installation occupies 73 000 ha in west-central Georgia and east-central Alabama and the portion that lies within the Coastal Plain physiographic province was used for this study. All study sites are associated with riparian forests of ephemeral streams.

The main source of sediment to riparian areas at Fort Benning is unpaved roads and trails that are used by military traffic. The roads are generally established along ridgelines and, consequently, the sandy substrate on road edges and near turn-outs erodes quite readily. Sediment moves downslope primarily in channelized flow through ephemeral streams and accumulates where channels widen and surface roughness increases. Roughness is mainly attributable to woody stems and, to a lesser extent, coarse woody debris. Almost no herbaceous vegetation is present in these erosion features.

Riparian forests of the study site consist of uneven-age, primarily deciduous cover with common occurrences of red maple (*Acer rubrum* L.), tag alder [*Alnus serrulata* (Aiton) Willd.], flowering dogwood (*Cornus florida* L.), sweetgum (*Liquidambar styraciflua* L.), yellow poplar (*Liriodendron tulipifera* L.), sweet bay (*Magnolia virginiana* L.), wax myrtle (*Myrica cerifera* L.), black gum (*Nyssa sylvatica* Marshall), loblolly pine (*Pinus taeda* L.), water oak (*Quercus nigra* L.), red oak (*Quercus rubra* L.), and post oak (*Quercus stellata* Wang), among other species. This is in contrast to the dry uplands dominated by mixed pine forests. Soils consist of poorly drained Bibb (coarse-loamy, siliceous, active, acid, thermic Typic Fluvaquents) and Chastain (fine, mixed, semiactive, acid, thermic Fluvaquentic Endoaquepts) series within riparian areas and well- to excessively drained Troup (loamy, kaolinitic, thermic Grossarenic Kandiudults), Lakeland (thermic, coated Typic Quartzipsamments), and Cowarts (fine-loamy, kaolinitic, thermic Typic Kanhapludults) soils on upper slopes and ridges. Deposition of sandy alluvium in upper reaches of some catchments may have created better-drained conditions than originally existed.

Nine ephemeral streams were selected to cover a range of disturbance and sedimentation conditions and included two reference catchments and seven that were disturbed in the upper portions. The presence of alluvial fans and buried lower portions of tree stems were used as evidence of sedimentation. In reference catchments, these traits were absent. Paired, circular plots (0.04 ha) were established within each ephemeral catchment with one delineated in the upper portion near the stream origin where sediment was most likely to be received. Another plot was installed lower within each catchment beyond visual evidence of sediment deposition. The latter served as relative (within-catchment) controls for comparisons with upper plots (Fig. 1).

Upper plots on disturbed areas were located within the alluvial fan, and overstory canopies ranged from somewhat open to closed and often had less understory than lower plot counterparts. Lower plots tended to have higher levels of soil moisture due to the lower topographic position. Precipitation during 2002 and early 2003 was roughly similar to long-term averages while precipitation during the summer months of 2003 was often more than the long term for the Fort Benning area (Fig. 2).

Long-term and current sediment deposition (i.e., during the course of the study) have the potential to influence ecological processes singularly or in combination, so each was estimated on all upper plots. There were no visual indications of stem burial or alluvial fans on upper plots in reference catchments or on lower plots in any catchment, so sedimentation rates there were assumed to be nonexistent or to have occurred prior to establishment of the current overstory. Long-term sedimentation rates on upper plots were estimated in July 2003 using the dendrogeomorphic technique of Hupp and Morris (1990). Three to four saplings of one of the following species: tag alder, post oak, red oak, flowering dogwood, loblolly pine, or red maple, were selected at random within each upper plot, excavated to the root collar, cut to remove a disk, and aged. Because no ring width measurements were to be taken and annual rings were relatively distinct, cross-dating was not performed. Sapling ages averaged 25 yr of age and were approximately 8 to 10 cm in diameter. Current sedimentation rates were estimated using erosion pins (Lal, 1994) that were sampled bimonthly.

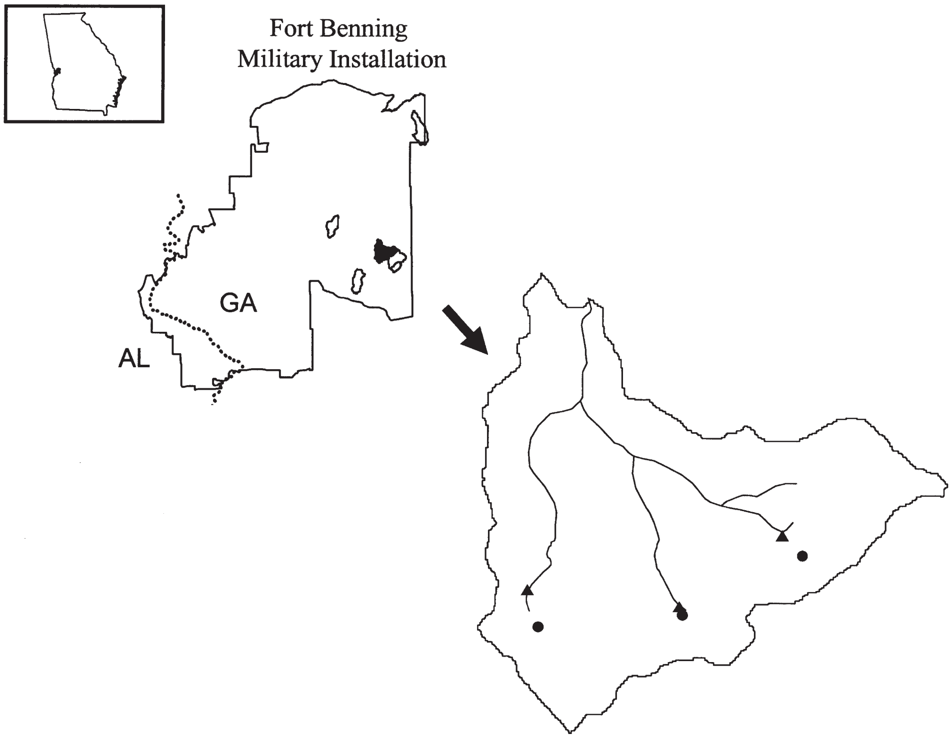

Fig. 1. Location of catchments used in sedimentation study at Fort Benning, GA, with expanded view of three catchments showing orientation of upper (triangle) and lower (circle) plot pairs.

Decomposition was studied using the litter bag approach (Swift et al., 1979). Foliar litter was collected in traps (0.5 m^2, three per plot), air-dried, and sorted by species on an individual plot basis. Eight 13- × 13-cm litterbags composed of nylon mesh (6-mm opening on the upper side and 2-mm opening on the side in contact with the soil) were filled with a mix of litter that reflected the tree species composition of each plot. Litterbags on two of the plots were destroyed by fire and animals. Bags were collected at intervals of 0, 2, 4, 10, 16, 25, 36, and 48 wk between April 2002 and March 2003. The time 0 collection was used to estimate handling loss and starting conditions in terms of C and N content. After removal from the field, bags were placed inside paper sacks for transport to the laboratory. Litter was removed from the bag, gently washed if necessary to remove extraneous material, oven-dried at 70°C for 48 h, and weighed. Total C and N were determined on litter samples using thermal combustion on a PerkinElmer (Wellesley, MA) 2400 Series II CHNS/O analyzer.

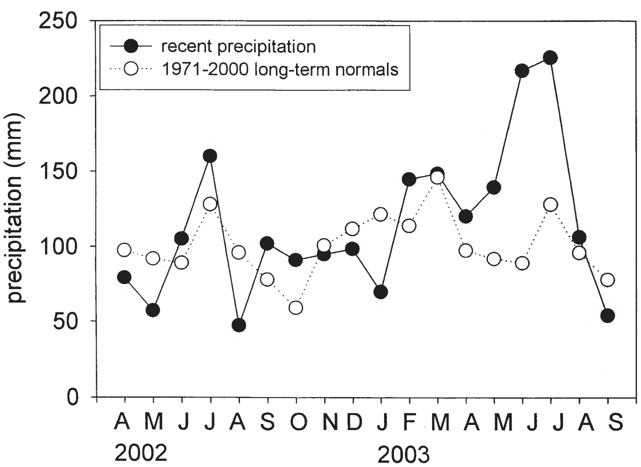

Fig. 2. Precipitation during 2002 and early 2003 and long-term averages (1971–2000) for Fort Benning, GA.

Nitrogen mineralization was estimated using the in situ methodology outlined by Hart et al. (1994). Seven sampling collections (at approximately 3-mo intervals) were made between April 2002 and July 2003. During each sampling, four soil samples (approximately 150 mL each) were collected from each plot to a depth of 7.5 cm and two were bagged and put on ice for transportation to Auburn University laboratories. These were extracted for NH_4–N and NO_3–N to ascertain initial levels of mineral N prior to in situ incubations. The additional two samples were placed in polyethylene bags and returned to the soil where they were buried to the same depth. The latter samples were collected after 30 d and extracted to estimate the amount of N mineralized (sum of NH_4–N and NO_3–N) per day over that time period.

Microbial biomass was determined using the chloroform-fumigation technique (Vance et al., 1987; Brooks et al., 1985). Soil sampling occurred in tandem with N mineralization collections and consisted of two pairs of soil samples (7.5 cm deep, 75 mL each) being gathered with the use of a push probe from each plot. Samples were bagged, put on ice, and transported to laboratories for analyses. One sample within each pair was subjected to chloroform fumigation and then all samples were extracted with K_2SO_4. Extracts were analyzed for total C using a Dohrmann DC-80 total organic carbon analyzer (Rosemount-Dohrmann Analytical, Santa Clara, CA) and total Kjeldahl N (Bremner, 1996).

During March and September 2003, 0.1-m^2 samples of unconfined forest floor were collected from upper plots in all catchments and sampled for arthropod populations. The Berlese–Tullgren funnel extraction technique (Seastedt and Crossley, 1980) was used. Litter samples were subjected to constant light and heat for 5 d and all extracted arthropods were stored in 70% ethyl-alcohol, counted, and subsampled for identification to the order level.

Two temperature sensors (HOBO; Onset Computer Corporation, Pocasset, MA) were placed on the soil surface of upper and lower plots within one reference catchment and three disturbed. All sensors were encased in PVC capsules. Temperatures were recorded hourly and downloaded once a month.

Statistical Analyses

Regression analysis was used to estimate decomposition rates (i.e., $y = e^{-kt}$, where y = proportion of original carbon remaining, k = rate coefficient, and t = weeks) separately for each plot across the 48-wk period. Regression relationships between sedimentation rates vs. decomposition rate coefficients, microbial biomass C and N, mineralized nitrogen, and arthropod counts were examined using either the nonlinear model previously listed, simple linear, or natural log transformations. Comparisons of upper vs. lower plots were made within each catchment for N mineralized and microbial C and N using analyses of variance. All analyses were conducted using SAS (SAS Institute, 1985) and all probability levels are reported and considered significant if less than $P < 0.10$.

RESULTS AND DISCUSSION

Sedimentation

Long-term sedimentation rates on upper plots ranged from none in reference catchments to 4.0 cm yr^{-1} on upper plots (Table 1). Similarly, erosion pin estimates of sediment accumulation for the current study period ranged from slightly negative (in the reference catchments R1 and R2) to 3.2 cm yr^{-1}. Average rates for the nonreference catchments (D1–D7) were 1.5 and 1.7 cm yr^{-1} for long-term and current data, respectively, and are higher than some reported elsewhere. Sediment deposition rates within floodplain forests associated with major river systems such as the Coosawhatchie of South Carolina and the Cache of Arkansas have been estimated at 0.02 to 0.20 and 0.20 to 0.36 cm yr^{-1}, respectively (Hupp, 2000). Similarly, Hupp and Bazemore (1993) reported average deposition rates of 0.24 to 0.28 cm yr^{-1} in riverine wetlands of western Tennessee. However, along natural levees of a Wisconsin stream, rates averaged 2.62 cm yr^{-1} (Johnston et al., 1984).

The relationship between historic and current sediment accumulation was pronounced ($P < 0.0001$, $r^2 = 0.59$). This suggested that the extent of sediment movement on the upper plots during the study period was not atypical, at least in a relative sense, compared with that of the preceding 25 yr. Efforts to relate current sedimentation rates to response variables yielded weaker and/or nonsignificant relationships compared with those with long-term sedimentation. Consequently, regressions between response variables and sedimentation are based solely on long-term data. The stronger relationships with long-term rates suggest that sediment accumulation over 25 yr may have a more pronounced effect on biogeochemical processes than that of the current year. This may imply that biogeochemical impacts from sedimentation are manifested through mechanisms that occur over periods of several years to decades such as vegetation transitions.

Temperatures

Average temperatures at 0200 h were identical for upper and lower plots in the D1 through D7 catchments and averaged 21°C annually. The lower plot in the reference catchments (R1 and R2) averaged 19°C at 0200 h vs. 21°C on the upper. Temperatures at 1400 h exhibited more variation among plots with upper locations in disturbed catchments averaging 27°C and corresponding lower plots averaging 25°C. Temperature averages on upper and lower locations within the reference were 25 and 23°C, respectively. The range of temperatures showed greater divergence between upper and lower plots in disturbed catchments (i.e., 7 and 4°C, respectively). Corresponding values for the reference catchments were both 4°C. The wider range of temperatures and slightly higher maximums on upper plots in disturbed catchments may create harsher conditions there for some microbial populations (Swift et al., 1979).

Decomposition

Decomposition rates for foliar litter on upper and lower plots averaged 0.95 and 0.91 yr^{-1}, respectively, and were not significantly different. Rates within this range are near the average of 1.01 noted for wetland forests in the U.S. Southeast (Lockaby and Walbridge, 1998) and well within ranges noted for wetland forests in the temperate zone (Brinson, 1990).

The regression relation between decomposition rates and long-term sedimentation rates was significant ($r^2 = 0.62$, $P = 0.04$) (Fig. 3). Apparently, a rapid decrease in decomposition rates occurred even with the smallest sedimentation levels observed (i.e., 0.20–0.32 cm yr^{-1}). Rates of carbon loss appear to approach a reduced equi-

Table 1. Comparisons between upper and lower plots within catchments in terms of net nitrogen mineralized to a 7.5-cm depth. Long-term sediment rates for upper plots and probability levels associated with each comparison are provided.†

	Catchment								
	D1	D2	D3	D4	D5	D6	D7	R1	R2
Sedimentation, cm yr^{-1}	4.0 (0.13)	1.97 (0.45)	1.71 (0.02)	1.47 (0.25)	1.15 (0.02)	0.32 (0.12)	0.20 (0.01)	0	0
Net N mineralized, g ha^{-1} d^{-1}									
Upper	202 (75)	127 (57)	119 (34)	103 (52)	178 (100)	330 (138)	461 (269)	235 (114)	147 (83)
Lower	55 (11)	214 (79)	145 (28)	286 (192)	308 (153)	180 (50)	131 (64)	137 (44)	693 (233)
Probability level	0.29	0.39	0.65	0.47	0.51	0.42	0.17	0.50	0.09‡

† Standard errors are given in parentheses.
‡ Significant at the 0.10 probability level.

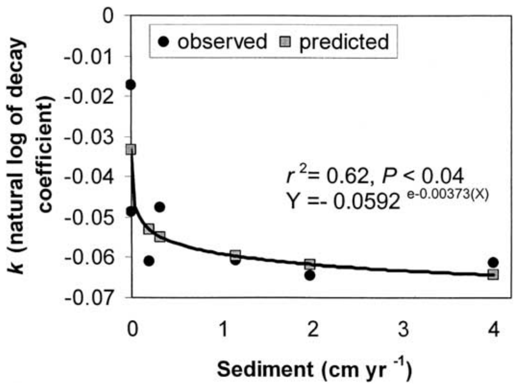

Fig. 3. Regression relation between decomposition rates and long-term sedimentation rates of foliar litter for Fort Benning, GA.

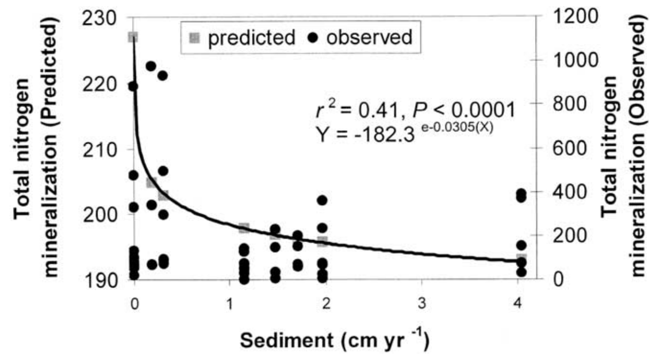

Fig. 4. Relationship between long-term sedimentation and total nitrogen mineralized in upper 7.5 cm of soil on upper plots at Fort Benning, GA.

librium at sediment accumulations above 0.50 cm yr^{-1}. In comparing decomposition among several floodplain forests in the southeastern United States, Baker et al. (2001) observed the lowest rates of foliar litter decomposition on the Cache River floodplain in Arkansas, a system characterized by high rates of sediment accumulation (Hupp, 2000).

Nitrogen Mineralization

Upper plots within disturbed and reference catchments exhibited net mineralized nitrogen levels of 217 and 191 g ha^{-1} d^{-1}, respectively, to a 7.5-cm depth. A number of wetland studies have measured potentially mineralized N (laboratory incubations) (Groffman and Crawford, 2003; White and Reddy, 2003), which are difficult to compare directly to field incubations. Although intact soil cores were used, as opposed to the disturbed soil in bags of the present study, the levels reported by Piatek and Allen (1999) in undisturbed and harvested forests in the North Carolina Piedmont were somewhat higher than those reported here. However, comparisons between the two studies will vary to some extent due to the use of intact cores vs. disturbed soil.

The relationship between net mineralized N and sedimentation rates on upper plots was significant (r^2 = 0.41, P < 0.0001) (Fig. 4). Comparisons of upper vs. lower plots within catchments revealed only one significant difference (Table 1). Net mineralized N levels on the lower plot within catchment R2 were significantly higher than those of the upper. The lack of differences between upper and lower plots may be in part due to generally high levels of variation associated with mean estimates (Table 1).

Microbial Carbon and Nitrogen

Microbial C averaged 81 and 394 μg g^{-1} on upper plots of disturbed and reference catchments, respectively. These values are lower than ranges reported by White and Reddy (2003) for Everglades soils, Groffman et al. (1996) across a soil wetness gradient in multiple wetland types of New York, and Schilling et al. (1999) on the Pearl River floodplain in Mississippi. However, values of the present study are closer to those of Stoeckel and Miller-Goodman (2001) from a floodplain forest in South Carolina. Microbial N levels were 45 and 21 μg g^{-1}, respectively, on upper plots of the reference and disturbed catchments. Again, these values are somewhat lower than those reported by Schilling et al. (1999) and Groffman et al. (1996).

Comparisons between upper and lower plots within catchments suggested that, at higher sedimentation rates (i.e., above 0.32 cm yr^{-1}), microbial C and N were reduced on upper plots (Table 2). In terms of microbial C, there were significant differences between upper and lower plot values on all catchments where upper plots had received sediment at rates higher than 0.32 cm yr^{-1}. There were no significant differences within the reference catchments or D6 and D7, the two having the lowest sediment accumulation rates apart from the references. In all cases where there were differences, the

Table 2. Comparisons between upper and lower plots within catchments in terms of microbial C and N in upper 7.5 cm of soil. Long-term sediment rates for upper plots and probability levels for each comparison are provided.†

	Catchment								
	D1	D2	D3	D4	D5	D6	D7	R1	R2
Sedimentation, cm yr^{-1}	4.0 (0.13)	1.97 (0.45)	1.71 (0.02)	1.47 (0.25)	1.15 (0.02)	0.32 (0.12)	0.20 (0.01)	0	0
Microbial C, μg g^{-1}									
Upper	56 (14.0)	27 (8.7)	18 (4.8)	108 (19.0)	21 (6.8)	231 (26.8)	449 (63.7)	420 (60.0)	367 (50.1)
Lower	374 (12.0)	411 (40.0)	84 (15.7)	258 (57.7)	412 (30.0)	252 (22.3)	452 (41.1)	605 (100.6)	395 (67.1)
Probability level	0.0001‡	0.0001‡	0.001‡	0.02‡	0.0001‡	0.55	0.96	0.13	0.74
Microbial N, μg g^{-1}									
Upper	8 (3.1)	10 (3.9)	4 (2.6)	10 (4.2)	4 (2.0)	45 (9.4)	65 (13.2)	36 (8.0)	54 (8.1)
Lower	35 (11.7)	34 (5.4)	12 (5.5)	27 (6.8)	41 (8.0)	37 (4.2)	37 (4.2)	53 (7.0)	47 (8.5)
Probability level	0.01‡	0.002‡	0.22	0.04‡	0.0001‡	0.47	0.35	0.14	0.55

† Standard errors are given in parentheses.
‡ Significant at the 0.10 probability level.

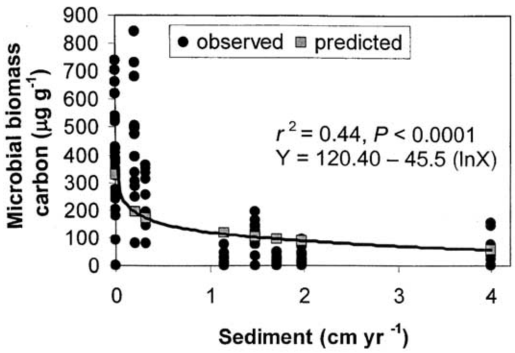

Fig. 5. Relationship between long-term sedimentation and microbial biomass carbon in upper 7.5 cm of soil at Fort Benning, GA.

lower plot values were higher. Similarly, catchments with sedimentation rates above 0.32 (with the exception of D3) displayed significant differences between upper and lower plots in terms of microbial N (Table 2). As was the case with microbial C, where significant differences did occur, lower plot values were higher.

In relation to long-term sedimentation, microbial C and N exhibited significant regression relationships (i.e., both $P < 0.0001$) with an r^2 value of 0.44 and 0.21, respectively (Fig. 5 and 6, respectively). A long-term sedimentation rate between 0.32 and 0.5 cm yr^{-1} seemed to be an approximate threshold beyond which major reductions in both microbial C and N became apparent. In addition, microbial C and N on upper plots were significantly correlated (r^2=0.73, $P < 0.0001$).

Arthropods

There was no relationship between numbers of arthropods and long-term sedimentation rates. Total counts from upper reference and disturbed plots averaged 139 and 156 per 100 g of forest floor in March and 255 and 173 in September, respectively. These values are somewhat higher than those recorded in coarse woody debris of the Atchafalaya River basin by Lockaby et al. (2002), although this disparity might be expected between coarse woody debris and foliar litter.

Regardless of sedimentation level, samples were dominated by orders Acarina, Collembola, Coleoptera, and Diptera. Acarina often dominates in decomposition studies because they are the most abundant in nature (Seastedt et al., 1989). Collembola also was a dominant order in the Atchafalaya.

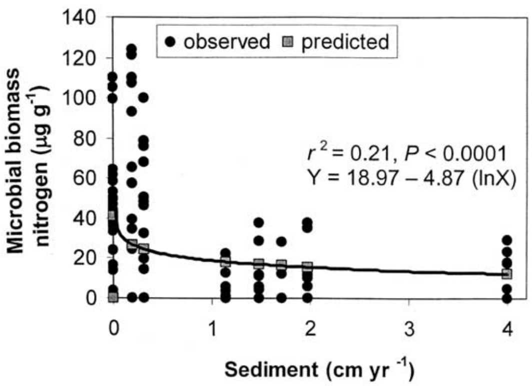

Fig. 6. Relationship between long-term sedimentation and microbial biomass nitrogen in upper 7.5 cm of soil at Fort Benning, GA.

CONCLUSIONS

Sedimentation levels near 0.20 cm yr^{-1} were associated with marked declines in decomposition rates, N mineralization, and microbial biomass C and N. However, impacts of sedimentation on arthropod numbers and composition were not apparent. These data suggest that reduced decomposition rates could be associated with declines in microbial populations rather than changes in arthropod populations. The higher maximum daily temperatures and wider temperature ranges on upper disturbed plots could be causal factors in the shifts in decomposition rates, N mineralization, and microbial biomass. Decomposition rates, N mineralization, and microbial biomass appeared to reach a low equilibrium somewhere between long-term sedimentation rates of 0.32 and 0.5 cm yr^{-1}.

Relationships between long-term (25-yr average) sedimentation rates and biogeochemical indices were stronger than for current year rates. This suggests that some biogeochemical alterations are driven by processes that occur over longer time periods such as changes in litter quality reflecting vegetation transitions. Clarification of the nature of the long-term influence would be a worthy goal of future studies.

Sedimentation rates as low as 0.20 cm yr^{-1} are not uncommon in floodplain forests (Hupp, 2000). This is particularly true in catchments with significant amounts of disturbance such as urbanization and agriculture. These results imply that proper road maintenance should receive a very high priority on military installations and other areas where highly erodible soils occur. Given the projected increases in human populations, a greater proportion of river basins throughout the world will likely be subjected to significant levels of disturbance. Since even relatively low levels of sediment accumulation can alter some aspects of biogeochemical cycling in riparian forests, key functions such as water filtration might be vulnerable as well.

Although we did not measure surface roughness, our field observations on impacted sites suggest that higher levels of sediment may physically bury coarse woody debris, forest floors, and perhaps induce mortality in small vegetation. This would result in reduced roughness and, theoretically, less effective filtration since surface runoff velocities would be slowed to a lesser degree. Consequently, surface runoff containing sediment would be more likely to reach streams.

During restoration of riparian forests that are likely to receive high sediment inputs, this possibility should be taken into account and consideration should be given to selection of vegetation species that are tolerant of sediment accumulation.

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