CHLORIDE AND NITROGEN DYNAMICS IN FORESTED, SUBURBAN, AND URBAN STREAM DEBRIS DAMS

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INTRODUCTION

Human populations are rapidly increasing, and a growing proportion of the global population inhabits urban areas. The global urban population in 2000 was 2.9 billion (47%). In North America, 77% (310 million) of the population lived in urban areas in 2000 (Platt 2003). The Worldwatch Institute projected in 2001 that by 2030 the North American urban population will rise to 372 million (84%) (ibid). As the global urban population increases, urban land use must expand as well. Cities have been dramatically increasing in size; more than 300 U.S. cities have populations of more than a million, and 14 cities are home to more than a billion people (Pickett *et al.* 2001). Metropolitan land use in the contiguous United States has increased from 9% in 1950 to 18% in 2000 (Platt 2003).

This dramatic spread of urban areas offers a new frontier for ecology research, not only concerning effects of urbanization on ecological systems, but also looking at the relationships between ecological and social systems. One model for understanding these relationships is the Human Ecosystem Framework proposed by Machelis *et al.* (1997). This framework begins by recognizing that there are both human and natural sources of spatial heterogeneity and that these sources significantly influence each other. For instance, the spatial distribution of different social groups has a large influence on where critical resources are concentrated due to differential access and control of these resources within and between groups (Pickett *et al.*. 1997, 2001). Urban and suburban areas, because they are so heavily influenced by human systems, are the ideal setting for investigating these relationships (Pickett *et al.*. 1997, 2001). The Baltimore Ecosystem Study (BES) is one of only two urban-focused Long Term Ecological Research sites funded by the National Science Foundation. The project works to bridge social and natural sciences through long term stream monitoring, ecological and social science modeling, and community education and outreach (Groffman et al. 2002). Within BES there has been much study of the effects of urbanization on stream and riparian ecology, with a focus on nitrogen cycling. We looked at urban streams using an ecosystem approach, with a focus on nutrient fluxes and budgets, as well as a watershed approach looking at broader patch dynamics along a rural-urban gradient in Baltimore County, Maryland (McDonnell and Pickett 1990).

The most visible and dramatic change resulting from urbanization is the increase in impervious surface cover. Impervious surface cover in suburban counties within the Chesapeake Bay watershed ranges from 10.6% (for 2-acre lot residential) to 44.4% (multifamily residential). Fifty-five to 75% of this area is car habitat rather than building area (Cappiella and Brown 2001). Watershed 263, a major study site of the Baltimore Ecosystem Study, is 74% impervious.

Impervious surfaces have dramatic effects on the hydrology of urban areas. Increasing impervious surfaces to 10-20% compared to forested watersheds doubles surface runoff, increasing to 35-50% triples runoff, and areas with 75-100% impervious surface have five times more surface runoff than forested watersheds (Paul and Meyer 2001). As a result of this high surface runoff, lag times during storms are significantly shorter and flood peak discharges are higher and occur more rapidly (Paul and Meyer 2001). For example, a relatively small summer storm at Dead Run in Baltimore that precipitated 4-5 inches of rain in 2 hours lead to the equivalent of a 500 year flood with a lag time of only 60 minutes (Miller et al. public lecture). In another study of streams in Washington, Booth and Jackson (1997) found that with 10% impervious surface cover a 2 year urban flood was equivalent to a 10 year forested flood.

The engineering of streams channels with concrete channels, stream stabilization, and storm drainage systems compounds the effects of impervious surface cover by directing surface runoff directly into streams (Paul and Meyer 2001). Dow and DeWalle (2000) found that 43% of precipitation to urban areas leaves through storm sewers.

These increased flows have dramatic effects on channel geomorphology by widening and deepening channels and eroding stream banks. Significant increases in discharges can lead to bank failure (Paul and Meyer 2001). Channel erosion can begin with as little as 2-6% impervious surface cover in a watershed (Morisawa and LaFlure 1979, Dunne and Leopold 1978, Booth and Jackson 1997), and streams are universally unstable in areas with greater than 10% impervious surface area (Booth and Jackson 1997). High flows and peaks can also lead to stream scouring, reducing the presence of large woody debris in urban streams. Finkebine et al. (2000) found that streams in areas with greater than 20% impervious surface generally have little large woody debris.

Because a greater portion of precipitation leaves an urban catchment as surface runoff, groundwater recharge is reduced, which in turn lowers the baseflow discharge of urban streams. In a "typical" forested catchment 50% of precipitation infiltrates the soil. In contrast, with 10-20% impervious surface, precipitation infiltration decreases to 42%, with 35-50% impervious surface, to 35%, and with 75-100% impervious surface, to 15% (Arnold and Gibbens 1996). With decreased groundwater recharge, water tables lower in what is know as hydraulic drought. The lowered water table can results in higher nitrate loading to streams because groundwater no longer passes through the upper soil layers where plants and microbial communities would typically remove pollutants and nutrients such as nitrate (Gold *et al.* 2001). In addition, a second cause of high nitrate concentrations entering streams is higher rates of nitrification in drier, more aerobic riparian soils resulting from lowered water tables (Groffman *et al.*. 2002).

Nitrogen concentrations are typically much higher in urban streams, both as nitrate and ammonium (McConnell 1980, Hoare 1984, Zampella 1994, Wernick *et al.* 1998, all cited in Paul and Meyer 2001). Urban storm water nationally contains from 0.01-12.00 mgL^{-1} nitrate, and 0.32-16.00 mgL^{-1} total nitrogen (organic and inorganic; Makepeace *et al.*. 1995). Nitrate is the most common groundwater pollutant in the United States (E.P.A. 1990). Nitrate loading to receiving waters has been shown to cause a loss of diveristy and changes in community structure (Bowen and Valiela 2001; Howarth *et al.*. 2002). Baltimore streams fall within the Chesapeake Bay watershed, and their water quality can have a large impact on the water qualtiy of the Bay. Nitrate loading to the Chesapeake Bay has resulted in loss of eelgrass (*Zostera marina*) beds which support diversity of animals including the economically important blue crab (*Callinectes sapidus*; CBP 2000).

Sources of nitrogen in stormwater runoff include fertilizers, industrial cleaning operations, feed lots, animal wastes, and fossil fuel combustion (Makepeace *et al.* 1995). Important sources in Baltimore County are illicit discharges, leaky sewer lines (which often run parallel to streams), the use of combined sewer overflow, and septic tanks (Paul and Meyer 2001). In addition to sources of nitrogen to streams, it is important to look at processes which can remove pollutants, especially nitrate, from the groundwater baseflow. Riparian areas have been found to be instrumental in removing excess nitrate and other nutrients and pollutants from groundwater in agricultural areas. As groundwater flows through upper riparian soils, microbial and plant communities immobilize and take up nitrate. In addition, denitrification reesults in lower concentrations of nitrate in riparain soils (Groffman *et al.* 1992; Simmons *et al.* 1992). However, with the lowered water tables associated with urbanization, groundwater does not pass through the upper soil layers and nitrate is not removed.

Chloride can be tremendously high in urban areas, due especially to deicing activities during the winter. Stormwater runoff nationally has been found to contain concentrations of 0.30 to 25000 mgL^{-1}. High concentrations of chloride are associated with winter and spring runoff containing salt from the deicing of roads.

Canadian drinking water guidelines set a maximum concentration of 250 mgL^{-1} for chloride, and the European Economic Community has set a guide level of 25 mgL^{-1} (Makepeace *et al.*. 1995). Chloride concentrations in Baltimore County streams range from below detection limit in forested catchments to 80-2500 mgL^{-1} in suburban streams (Groffman, unpublished data). Sources of chloride in Baltimore County are mostly from road salt, fertilizers and sewage and wastewater treatment effluent. Sources in other areas may additionally include industry, animal wastes, chemical manufacturing, insecticides, and dust control (Herlihy et al. 1998; Makepeace et al. 1995). Herlihy et al. (1998) found that chloride concentrations in Mid-Atlantic streams are highly correlated with land use ($R^2 = 0.48$, $p < 0.05$), and suggested its use as an indicator for human disturbance within a watershed.

Our objectives for this work were to determine possible effects of elevated chloride concentrations on nitrogen processes in streams. There have been several studies on the effects of chloride on nitrification in soils. Hahn *et al.* (1942) found that nitrification is inhibited by adding KCl to soil solutions and that inhibition was caused by the chloride ion. Roseberg *et al.* (1986) found that elevated concentrations of chloride inhibited nitrification but only at pH < 6.

Groffman *et al.* (1995) measured the effects of 100 mg Cl^- L^{-1} and several other groundwater tracers on respiration, denitrification, N-mineralization, and nitrification. They found that Cl^- significantly lowered soil respiration and significantly inhibited N-mineralization and nitrification. Denitrification was inhibited by Cl^- although the results were not significant. The mechanism for chloride inhibition of nitrification is unknown but is independent of cation associations with Cl^- or osmotic potential (Groffman *et al.*. 1995).

We focused this study on debris dams because they are hotspots for nitrogen processes. Organic debris dams are accumulations of organic matter supported by large woody debris that obstruct stream flow (Bilby 1981). Debris dams eventually become water-tight and form small pools upstream. Debris dams play an important role in retaining both sediments flowing down stream and coarse particulate organic matter, which can then be processed into smaller size particles and eventually CO_2 (Bilby and Likens 1980, Valett *et al.* 2002). Bilby and Likens (1980) found that removal of debris dams from a 2nd order stream resulted in a 63% reduction in storage of organic matter within the stream and a 2.5 fold increase in total carbon export. Additionally, the presence of debris dams increases the turnover time of organic matter from 0.31 year to 0.71 year.

Because of their high organic matter content, debris dams are where most nutrient cycling is concentrated in small streams (Valett *et al.* 2002). Bernhardt *et al.* (2003) found that large inputs of woody debris to a stream during an ice storm significantly increased in-stream N-processing. Groffman and Dorsey (2003) found that denitrification rates were higher in debris dams compared to any other stream feature (such as gravel bars, pools, and riffles).

We looked at the effects of elevated chloride concentrations on nitrogen processes in debris dams from streams across a gradient of human development. Chloride was expected to significantly inhibit both nitrification and denitrification. We designed a microcosm lab experiment in which material from debris dams was incubated under various treatments designed to determine the effect of chloride concentration, the effect of nitrate concentrations, and the interaction between chloride and nitrate concentrations.

METHODS

Site Description

Gwynn Falls is a 17 500 ha watershed within the Patapsco River drainage basin in which there is an urban-suburban-rural gradient away from Baltimore City. Since the early 18th century there has been a history of industrial and residential use within the riparian zones of the main channel. Mill industries in the 19th century often engineered stream channels to speed water movement. From the late 1800's into the 1900's there was a history of direct industrial discharge into the stream, especially meat packing waste. In the 1970's a number of hurricanes and resulting flooding lead to the abandonment of industrial and residential use of most of the riparian zone (Groffman et al. unpublished data).

Sampling

Samples were collected from debris dams in three streams in the upper portion of the Gwynn Falls watershed on 3 June 2004. Pond Branch is a forested stream with very low stream N and Cl^- concentrations. The Pond Branch watershed is within the larger watershed of Baisman Run, which is 66% forested and 34% residential. Baisman Run has high concentrations of nitrate due to septic pollution and moderate concentrations of chloride due primarily to road deicing. Glyndon is a suburban area in the upper portion of the Gwynns Falls watershed with 47% residential land use and 4% forested. At Glyndon the stream is highly affected by leaky sewers and road deicing; it has very high stream N and Cl^- concentrations.

Sediment samples from three debris dams were collected in each of the study streams. Stream water samples were taken close to stream gauges. Samples were stored at 4°C between sampling and analysis. Large sticks and insects were removed from soil samples. Sediment was homogenized in a blender with small quantities of respective stream water. 150 g sediment samples were incubated in sealed mason jars with 40 ml of stream solutions. Treatments are detailed in Table 1 and were replicated in triplicate.

Analysis

Soil moisture was determined by drying at 60°C for 24 hours (McInnes et al. 1994). Soil organic matter content was determined by loss on ignition at 450°C for 4 hours (Nelson and Sommers 1996). Ambient stream chloride concentrations were determined by ion chromatography. Gas and sediment samples were taken on days 0, 5, 10, 20 and 30 and were analyzed for CO_2 by thermal conductivity gas chromatography. Microbial respiration was quantified as the amount of CO_2 evolved over a 10 day period. Inorganic N (NH_4^+ and NO_3^-) in sediments was measured by extraction with 2M KCl and colormetric analysis with a flow solution analyzer. Nitrification was calculated as the amount of NO_3^- accumulated over a 10 day period, and N-mineralization as the total inorganic N ($NO_3^- + NH_4^+$).

Denitrification enzyme activity (DEA) was measured using a method described by Groffman et al. (1999). Sediment samples were amended with NO_3^-, dextrose, chloramphenicol and acetylene, and incubated in anaerobic conditions for 90 minutes. Gas samples were taken at 30 and 90 minutes and analyzed for N_20 by electron capture gas chromatography.

Statistical Analysis

T-tests were used to compare sites and treatments. Pearson correlations were used to determine relationships between variables. All statistics were performed using the SAS statistical package.

Debris Dam Density

Five streams were surveyed for the frequency and density of debris dams in December 2004 to determine the effects of urbanization on the presence of debris dams. The streams ranged from forested to suburban to urban and were either small (stream width 1-2 m) or large (stream width 5-10 m). Pond Branch and Baisman Run served as reference streams. Pond Branch is a small forested stream with nitrogen and chloride concentrations below detection limits. Baisman Run is a large suburban stream and has high nitrogen concentrations and low chloride concentrations. Gwynn Falls at Glyndon is a small suburban stream with high concentrations of both chloride and nitrogen. Dead Run is a large urban stream with high concentrations of chloride and nitrogen. Dead Run at Westfield Park is a small tributary of Dead Run and is also urban and highly polluted.

Stream segments of 100 m to 1200 m were surveyed for debris dams. At each debris dam stream width and bank full were measured at the debris dam and 1 m above and below the dam. The supporting structure of the debris dam was noted (i.e. cobbles, large woody debris (LWD), other, or an absence of supporting structure). In the case of LWD, the length and diameter of each piece was measured, as well as whether or not it spanned the stream channel and its position (i.e. perpendicular, at angle or parallel to stream flow).

The dimensions of each debris dam were measured, and the quality of the organic matter was noted (i.e. coarse leaves and twigs or well decomposed) along with the presence or absence of any more urban debris. Debris dam height both in and above the water was measured. Debris dams were defined as any accumulation of debris which obstructed stream flow or in which there was significant decomposition. Pool formation was noted and its mean depth recorded. For each dam a detailed sketch was completed illustrating supporting structures, position of debris dams, and stream flow, as well as any other complexities. Several photographs were also taken of each dam.

For analysis, the rough volume of each debris dam was calculated. Frequency of debris dams was calculated per 100 m and per stream width to separate the effects of stream size from those of urbanization. T-tests were used to compare stream sites.

RESULTS

Chloride

Ambient stream Cl^- concentrations were highest in Glyndon (high density suburban/urban) at 59.01 mg Cl^-L^{-1}, intermediate at Baisman Run (suburban) at 20.44 mg Cl^-L^{-1}, and lowest at Pond Branch (forested) at 2.17 mg Cl^- L^{-1}. Stream chloride concentrations are within a 4.1% error.

Denitrification

Denitrification was much higher in suburban streams than in the forested reference stream due to higher stream N concentrations. Among control treatments at day 0, denitrification was highest in high density suburban/urban (GL) sediments where stream NO_3^- concentrations are highest (2-4 mg NO_3^--N/L) and lowest in forested (PB) sediments where stream NO_3^- concentrations are the lowest (< 0.02 mg NO_3^--N/L) (P < 0.05) (Figure 1). Denitrification was intermediate in lower density suburban (BR) sediments control treatments where stream NO_3^- concentrations were intermediate (Figure1). When forested (PB) sediments were amended with 2 mg NO_3^--N/L, denitrification increased significantly to rates comparable with those in the most urban stream (P < 0.05) (Figure 2).

Chloride amendments at 2500 mg Cl^-/L to forested (PB) stream samples non-significantly inhibited denitrification potential at day 0. Denitrification rates were significantly inhibited by day 15 ($P < 0.001$) and remained significantly lower than control treatments at day 30 ($P < 0.05$).

Chloride additions also showed a non-significant trend to constrain denitrification feedback response to increased NO_3^- concentrations. At day 15 forested (PB) sediment samples amended with both 2 mg NO_3^-/L and 80 mg Cl^-/L had lower denitrification rates than those amended with 2 mg NO_3^-/L only ($P = 0.112$) (Figure 2). However, chloride additions had no effect on denitrification rates in either of the suburban streams, which had both had previous exposure to Cl^- (Figure 2).

Nitrification

Nitrification rates were low across all streams and treatments. Nitrification over days 0-10 ranged from -0.156 to 0.186 mg NO_3-N/kg d with a mean of 0.016 mg NO_3-N/kg d. This may have been caused by an experimental design of the treatment incubations (closed jars) which would favor anaerobic processes.

Chloride additions (80 and 2500 mg Cl^-/L) increased rates of nitrification in all streams over days 0-10. Nitrification rates were higher ($P < 0.1$) in samples with 2500 mg Cl^-/L additions than in control samples across all sample sites. Samples amended with 80 mg Cl^-/L had significantly higher nitrification rates than control samples ($P < 0.001$) and significantly lower rates than samples amended with 2500 mg Cl^-/L ($P < 0.1$) (Figure 3A). From day 10-20, however, 2500 mg Cl^-/L additions decreased nitrification in all stream sites ($P < 0.1$) (Figure 3B).

Total Inorganic Nitrogen

Total Inorganic Nitrogen (TIN) concentrations remained steady over time in the forested (PB) and low density suburban (BR) control treatments, but showed a decline in the suburban/urban (GL) controls ($P < 0.001$) (Figure 5). TIN increased with 2500 mg Cl^-/L additions in the forested stream samples ($P < 0.05$) and the suburban/urban samples ($P = 0.14$), but decreased in the suburban stream samples by day 30 (Figure 4).

Microbial Respiration and Organic Matter Content

Mean microbial respiration from day 0 to 20 was significantly higher in the forested reference stream (PB) than in either of the suburban streams ($P < 0.0005$) across all treatments (Figure 5). This may have been a result of the significantly higher organic matter in debris dams in the forested stream ($P < 0.0001$) compared with the streams in more developed areas (Figure 6). Sediments from the forested stream amended with 80 mg Cl^-/L and 2 mg NO_3^-/L had significantly lower microbial respiration than forested controls ($P < 0.05$), as did forested sediments amended with 2500 mg Cl^-/L ($P < 0.01$). Elevated Cl^- concentrations had no significant effect on microbial respiration in either of the suburban debris dam samples.

Debris Dam Density

Stream Width

A total of five streams were surveyed: three small streams, one of each a forested reference, a suburban/urban, and an urban; and two large streams, one of each suburban and urban. The small streams were between 1 and 2 meters wide, and were statistically the same size, with the exception that Pond Branch (forested reference) was significantly wider than Westfield (small urban) ($P = 0.034$). The large streams were of significantly different width ($P = 0.0007$). Baisman Run (suburban) averaged 5.35 meters wide, while Dead Run (urban) averaged 7.83 meters wide (Table 2).

Support Structure

The type of structure supporting a debris dam varied depending on both the size of the stream and the degree of development in the watershed (forested, suburban, urban). Most debris dams were supported by either large woody debris, large cobbles, or a combination of both. In the three urban streams (Glyndon, Dead Run, Westfield) debris dams were often supported by or included more urban debris, such as plastic school chairs and plastic bags.

Debris dams in large streams were almost exclusively supported by large woody debris (LWD) supporting structure (93% in the large suburban and 86% in the large urban). Most of these dams were supported by a single LWD; only 25% of LWD-supported debris dams were supported by more than one piece of debris.

In small streams, with the exception of the highly urban stream (Westfield), fewer (only 50% in the small forested and suburban streams) debris dams were supported by LWD than in larger streams, however, a much larger percentage of these debris dams were supported by more than one piece of LWD (50% in the small forested and 67% in the small suburban). Debris dams in smaller streams were more often supported by cobbles and rocks than in larger streams (28-73% compared with 7-14%).

Within smaller streams, the percentage of debris dams supported by LWD was similar between the forested reference stream and the suburban stream. However, no debris dams in the urban stream were supported by LWD. A large percentage of debris dams in the small suburban (16%) and urban streams (27%) were supported by something other than LWD or cobbles. In both streams this was often debris accumulation along the side of the stream bank. Although these accumulations did not act as typical debris dams and did not form pools or accumulate sediment, they did account for a significant portion of coarse organic matter in these streams (37%) and therefore may play an important role in nitrogen cycling in urban streams.

Within the larger streams, the suburban reference (Baisman Run) had a higher percentage of debris dams supported by LWD than the urban stream (93% compared with 86%), and a lower percentage of debris dams supported by cobbles or rocks (7% compared with 14%) (Table 2).

Large Woody Debris

Large woody debris from the small forested and small suburban stream were similar in length (P = 0.25) and diameter (P = 0.56). LWD from both small streams were significantly smaller than that from the large suburban stream both in length ($P < 0.001$) and diameter ($P < 0.01$). LWD from both small streams were not significantly smaller in diameter than LWD from the large urban stream, and only LWD from the small suburban stream were significantly smaller in length than the large urban stream (P = 0.037). The orientation of LWD within the stream channel also differed between the reference and suburban streams. Twice as many LWD spanned the stream channel in the suburban stream (60%) compared with the reference stream (33%). LWD in the small reference stream were most freqently perpendicular (58%) rather than at an angle (33%) or parallel (<1%), to the stream flow, whereas LWD in the small suburban stream were most frequently at an angle (60%) rather than perpendicular (40%) to the stream flow (Table 2).

Within the two large streams, LWD from the suburban stream were nearly twice as large both in diameter (P = 0.02) and in length (P = 0.02). Additionally, the majority of LWD from the suburban stream spanned the stream channel (57%), where as none spanned the channel in the large urban stream (Table 2).

Debris Dam Size

Small streams in general had smaller debris dams than the large streams (P = 0.035). Within the small streams, debris dam size varied according to the degree of development within the watershed. Debris dams in the small suburban stream were the same height and volume as those in the small forested reference stream. Debris dams in the small urban stream were higher than those in both the reference stream ($P < 0.01$) and small suburban stream ($P < 0.01$), and had a smaller volume than both the reference stream (P = 0.06) and the small suburban stream (P = 0.08). Pools were present at a similar frequency in the small reference and suburban streams (~50%); however, the pools were deeper in the small suburban stream than in the forested reference stream (P = 0.08). The small urban stream had a much lower frequency of pool formation (13%), although the pool depth was not significantly different than in either the suburban or the reference streams (Table 2).

Within the larger streams, debris dams in the suburban stream were higher (*P* = 0.073) and nonsignificantly larger (*P* = 0.24) than debris dams in the large urban stream. Pool formation was much more frequent in the suburban stream (67%) than in the urban stream (29%), but pool size was not significantly different.

In all streams, as development in the watershed increased, the proportion of very small dams in a stream increased, the percent of larger dams decreased, and the diversity of dam sizes decreased (Figure 7). This is especially apparent when comparing the two large streams. The suburban stream has debris dams falling into all eight size classes, and distribution is fairly even. The large urban stream, however, has debris dams predominantly in the smallest size class. Additionally it should be noted that because there were so few debris dams in the large urban stream, the portions in the two larger size classes represent only one debris dam each (Figure 7).

Debris Dam Frequency

The frequency of debris dams was calculated several ways: volume of debris per stream width, volume of debris per 100 m of stream, and number of dams per 100 m of stream. The large suburban stream had the highest volume of debris per stream width, with 0.82 m^3/stream width. All other streams had less than 0.05 m^3 debris/stream width. Both urban streams had very low volume/width (0.01m^3/width) compared with the small reference stream (0.03m3/width) and the large suburban stream (0.82m^3/width).

The larger streams had the most volume of debris per 100m, and the large suburban stream had more than twice the volume of debris per 100m than the large urban stream. Within the smaller streams, the suburban stream had the most volume per 100m, nearly three times that of the urban stream and twice that of the reference stream. The urban stream had the lowest volume per 100m (Table 2).

Although the large suburban stream had the highest volume of debris per 100m and per stream width, it had the lowest number of dams per 100m, only 1.25 dams/100m, less than half the frequency of debris dams in the large urban stream. The small streams had a much higher frequency of dams than the larger streams. The suburban stream had the highest frequency of dams/100m (18 dams/100m), dams in the urban stream were the next most frequent (15 dams/100m), and dams were the least frequent in the small reference stream (11.5 dams/100m) (Table2).

DISCUSSION

Ambient stream chloride concentrations were as expected for these streams. Concentrations were highest at the high density suburban site, lower at the suburban site, and very low at the forested reference site.

Denitrification

As expected, high stream nitrate concentrations stimulated denitrification. Denitrification was highest in debris dams from the urban stream with the highest nitrate concentrations and was lowest in the forested reference stream which had very low nitrate concentrations. This is expected because denitrification is nitrate limited and therefore will increase with increasing nitrate concentrations (Holmes et al. 1996; Martin et al. 2001, Kemp and Dodds 2002). This was confirmed by the increase in denitrification activity in debris dam samples from the forested reference stream that were amended with 2 mg $NO_3^- L^{-1}$. This suggests that the high organic matter debris dams play an important role in maintaining water quality.

The high concentrations of chloride inhibited denitrification significantly in all streams and therefore may be instrumental in limiting stream response to nitrogen pollution. Unfortunately, watersheds which have high concentrations of NO_3^- also tend to have high concentrations of chloride because both result from urbanization and human development. Therefore, because of chloride's effects on denitrification, we should not rely on in-stream processing in debris dams to significantly lower stream nitrate concentrations in heavily developed watersheds where high peaks of Cl^- are seasonally common.

Contrary to expectations, low concentrations of chloride had no effect on denitrification in suburban and urban streams that had prior exposure to it. This may be due to microbial adaptation to salt pollution. It is important to note, however, that high concentrations of chloride did have significant effects on denitrification in these streams.

Low concentration of Cl^- had no significant effect on denitrification in debris dam samples from forested samples, but non-significantly increased it. This may have been a result of the higher nitrate concentrations in these samples caused by the salt effect. The salt effect of NaCl is the lysing of cells and release of organic nitrogen. This influx of organic nitrogen stimulates nitrification, increasing nitrate concentrations. It is possible that this increase in nitrate concentrations stimulated denitrification, masking any inhibitory effect of the chloride on denitrification rates. This effect was not seen in high chloride treatments because although the effect on nitrification was similar, the higher chloride concentrations inhibited denitrification to a greater extent that could not be masked by increased nitrate concentrations.

Nitrification

Nitrification was low in debris dams from all streams and under all treatments. This was most likely caused by an experimental design which favored anaerobic processes. However, chloride additions had some significant effects on nitrification rates in debris dam samples from all three streams. Over the first ten days of the incubation, all debris dam samples amended with chloride experienced a net increase in nitrate concentrations, whereas all debris dam samples not amended with chloride experienced a net decrease in nitrate concentrations. This is a result of a salt effect, through which the high salt concentrations lyse cells, releasing organic nitrogen which is then nitrified to nitrate. In incubations not amended with chloride, nitrate concentrations decreased as denitrification occurred.

Over the second ten days nitrification rates were negative in incubations with high chloride concentrations. This is because the influx of organic nitrogen resulting from the initial salt effect was exhausted, and the inhibitory effect of the chloride is the dominant one.

Total Inorganic Nitrogen

The total inorganic nitrogen (TIN) concentrations were steady over time in debris dam samples from forested and suburban streams in control incubations, but declined in debris dam samples from urban streams. This is consistant with the higher rates of denitrification in the urban stream.

TIN was significantly higher in debris dams from the forested reference and urban streams treated with high chloride concentrations. Again, this due to the salt effect which stimulates nitrogen mineralization and is consistent with nitrification results. Over time, TIN in forested and urban debris dam samples treated with high chloride remained steady. Following the initial increase due to the salt effect and mineralization, TIN concentrations did not decrease because denitrification was inhibited by the high chloride concentrations.

TIN in debris dams from both the forest and urban control treatments was consistently lower than that of the high chloride treatments because there was no salt effect. Additionally, TIN declined over time because denitrification was occurring and removing nitrogen from the system. There were no inputs of nitrogen because the anaerobic environment does not favor nitrification, an aerobic process.

Microbial Respiration and Organic Matter Content

Microbial respiration was significantly higher in debris dams from forested streams than urban and suburban streams. This is due to the significantly higher organic matter contents of the forested reference stream. It is very typical for urban streams to have less organic matter.

Debris dam from the forest reference stream amended with chloride and nitrate and with high chloride concentrations had significantly lower microbial respiration. This confirms the salt effect on microbial populations.

There was no effect of chloride on microbial respiration in debris dams in suburban and urban streams. Respiration in these streams was already lower than respiration in the forest stream. This may be more evidence to suggest adaptation to chloride pollution.

Debris Dams

As stream size increased, the frequency of debris dams decreased. This is consistent with the findings of Bilby and Likens (1980) that debris dam frequency decreased from 1^{st} to 3^{rd} order streams in the Hubbard Brook watershed in New Hampshire. The higher frequency of debris dams in small streams is primarily due to lower flows which make it easier for smaller woody debris to become lodged in the stream channel and form the supporting structure of the dam. Larger woody debris are also less likely to be washed away during high flows in a smaller stream.

Average debris dam volume and the diversity of dam volumes increased as stream width increased. Greater stream width allowed larger woody debris to interact with the stream. Often LWD with diameters greater than 15 cm crossed the small forested stream channel but were up to 1 m above the stream surface due to the high slope of the stream bank. Debris dams and LWD in larger streams also must be large to have any longevity. The higher flows of large streams will wash away most small debris dams that would be well established in a smaller stream, therefore small dams make up only a small percentage of dams in the large suburban stream (Table 2).

Debris dams in large streams were primarily supported by LWD rather than cobbles or rocks, whereas smaller streams had debris dams supported with cobbles as frequently as those supported by LWD. Again, this is due to the higher flows in large streams in which more support is needed to create and maintain a dam. Small streams are shallow enough and have low enough flow to allow debris to accumulate on cobbles that rise out of the water.

As the degree of development increases in a watershed, there is an increase in the percentage of small debris dams and a decrease in the average dam volume (Figure 7). However, in both large and small streams, dam frequency (debris dams/100m) increased as development increased. This can be largely attributed to the differences in flow regimes between streams in forested watersheds and those in progressively more urban areas. As a watershed becomes more developed the resulting lowered water table leads to decreased stream base flows and increased peak flows, so that the flow regime of an urban stream is much more extreme than that of a forested stream. Debris accumulations that would be too small or unsupported to weather the higher base flows of a forested stream are able to persist in more urban streams. Therefore, more developed streams have a high frequency of small debris accumulations. Even very small accumulations of debris in the small urban stream were well decomposed because there was so little stream flow to wash them away.

Although the number of dams/100m was greater in more urban streams, the volume of debris was much greater in less developed streams than in the large and small urban streams. Among the large streams, this is a result of both debris supply and different flow regimes. The large suburban stream contained large amounts of debris both in dams and as loose debris, whereas the large urban stream had very little organic debris in the stream channel. The lack of more significant debris dams as development increases can also be attributed to altered flow regimes. Just as the lowered base flows allow for a greater number of smaller accumulations, the increased peak flows may prevent any significant long term accumulations of debris that would be able to withstand more mild peak flows in a forested watershed, therefore the vast majority of debris dams and accumulations in urban streams are small.

Pool formation and depth also decreased with increasing development in a watershed. This is probably a result of the orientation of LWD in more urban streams which were more often at an angle or parallel to the stream flow than the forested stream in which they were most often perpendicular to stream flow. Over all streams, 70% of debris dams supported by a LWD perpendicular to stream flow formed pools, 52% of debris dams with LWD at an angle to stream flow formed pools, and only 20% of dams supported by LWD that was parallel to stream flow (one dam) formed a pool. Pools were formed by 40% of debris dams supported by cobbles. Therefore streams in more developed watersheds tend to have debris dams which are less likely to form pools. Although there was a higher frequency of dams in the small urban and suburban streams than in the forested reference, the lower frequency of pools suggests that debris accumulations in more developed streams alter stream flow less than those in the forested reference stream and therefore may have less ability to alter water quality.

In both the urban and the small suburban streams, debris dams were less frequently formed by woody debris and cobbles and were often formed by more urban structures: chairs, tires, and the gabion substrate used to stabilize the stream bank. Additionally, in the small urban and suburban streams, several debris accumulations were supported by nothing more than the stream bank. Especially in the large urban stream and the small suburban stream, debris dams often contained plastic bags and other garbage that made up a portion of the dam volume.

Decomposition was best in very large dams and in the small suburban and urban streams where flow was very low. Debris dams in these streams seemed to have a fairly low turnover rate because there was such a high degree of decomposition, especially in the small suburban stream. This is probably a result of the very low flows in the small suburban and urban streams or in addition because of sewer leakage in the small suburban stream. The low flows also may be the reason that many dams in the more developed streams were less dense and occasionally consisted solely of accumulations of debris along stream banks.

CONCLUSIONS

Nitrate pollution is an important issue across the country. Debris dams have been shown to be a hotspot for denitrification and therefore important in lowering the nitrate concentrations of polluted streams (Groffman and Dorsey submitted). Groffman and Dorsey (submitted) have introduced the idea of using debris dams and other in-stream processing mechanisms to improve water quality. However, we have shown that high chloride concentrations have the immediate effect of increasing nitrification through the salt effect and the continued effect of decreasing denitrification through inhibiting denitrifying bacteria. Therefore fostering the formation of debris dams in polluted streams will not be effective in developed areas if high chloride concentrations constrain any positive effects that the debris dams have on nitrate concentrations.

In order to maintain the importance of debris dams as hot spots for denitrification, chloride concentrations in urban streams should be kept as low as possible. Because there are no efficient ways to remove chloride from a stream system, maintaining low chloride concentrations mean limiting chloride inputs to the watershed. In urban areas, the primary source of chloride is from the use of road salt. Alternatives to road salt such as calcium magnesium acetate have been researched (Road Transport Research 1989; Dunn and Schenk 1979; Ihs and Gustafson 1996; Fritzsche 1992; Tanner and Wood 2000) and their uses are difficult to introduce because of higher costs and lower effectiveness. Other research has gone into more efficient methods for applying traditional deicing agents, such as applying salt in solution and using better monitoring and information systems to avoid over application (Road Transport Research 1989, Moore et al. 1995, Kallberg 1996; Kasinskas 1979, Matsuzawa, et al. 1996) More research is needed to better understand the effects of chloride on biogeochemical processes in urban streams, how these effects might be mitigated, and how to improve current technologies for snow and ice removal to be both environmentally benign and efficient.

The volume of organic debris in debris dams and in general decreases as streams become more urbanized. This is largely a result of lower standing stocks of organic debris in urban streams and the modified hydrology that occurs with urbanization. Because debris dams and woody debris are important not only for nutrient cycling, but for organic matter processes and biological habitat, there has been a fair amount of research into the effects of adding woody debris to streams for habitat improvement (Hildebrand et al. 1997, Larson et al. 2001). These projects have had only moderate success. Hildebrand et al. (1997) found that adding LWD both systematically and randomly increased pool formation but did not signficantly effect total benthic macroinvertabrate abundances. Larson et al. (2001) evaluated the effectiveness of LWD additions in six urban in-stream rehabilitation projects by measuring LWD and pool frequency and benthic macroinvertebrate abundances. They also found that LWD additions increased pool frequency; however pool spacing was lower in rehabilitated urban streams than in forested streams with the same quantity of large woody debris. In addition, they found no improvement in biological habitat over 2-10 years, and only limited success in controling downstream sedimentation. These studies are important illustrations of the need for watershed approaches to restoration. LWD additions can do little to capture sediment or organic matter if there is little organic matter in the stream system or if heavily modified hydrology and flow regimes do not allow significant debris dams to accumulate. In fact, in our large urban stream there was a large log about 15 cm in diameter and perpendicular to the stream flow which had accumulated no organic debris. The same log at our large suburban stream site would have accumulated a significant amount of debris quite rapidly because there is a great deal of organic debris in the stream system. In-stream habitat improvement such as LWD additions will not be successful in urban streams because human damage to and alteration of the ecosystem is so deep. Restoration has to occur at the watershed scale and address issues such as restoring hydrological function and increasing standing stocks of organic matter in urban streams, otherwise in-stream solutions can only be temporarily and moderately successful.

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APPENDIX

TABLE 1: Sediment incubations and treatments

	Incubation	Sed	Water	Additions
1	Pond Branch Control	PB	PB	None
2	Baisman Run Control	BR	BR	None
3	Glyndon Control	GL	GL	None
4	Pond Branch + Cl	PB	PB	80 mg/L Cl
5	Baisman + Cl	BR	BR	80 mg/L Cl
6	Pond Branch + NO3	PB	PB	2 mg/L NO3
7	Pond Branch + NO3 + Cl	PB	PB	2 mg/NO3 and 80 mg/L Cl
8	Pond Branch plus Gly	PB	GL	None
9	Baisman plus Gly	BR	GL	None
10	Pond Branch plus BR	PB	BR	None
11	Pond Branch high Cl	PB	PB	2500 mg/L Cl
12	Baisman high Cl	BR	BR	2500 mg/L Cl
13	Glyndon high Cl	GL	GL	2500 mg/L Cl

TABLE 2. Frequency and denity of debris dams in five streams in Baltimore County, MD.

	Pond Branch	Baisman Run	Glyndon	Dead Run	Westfield
stream width (m)	1.86[a]	5.35[b]	1.69[ac]	7.83[c]	1.19[d]
stream width- dam (m)	2.13	5.11	1.69	8.62	1.19
bank full (m)	>10	7.72	-	20.4	-
Support structure (%)					
LWD	52%	93%	56%	86%	0%
> 1 LWD	67%	23%	50%	17%	0%
LWD and cobbles	13%	7%	17%	14%	0%
cobbles/rocks	48%	7%	28%	14%	73%
LWD					
length (m)	1.99[a]	5.59[b]	1.71[a]	2.70[a]	-
Diameter (cm)	9.67[a]	25[b]	7.7[a]	10.5[a]	-
span channel?	33%	57%	60%	0%	-
perpendicular	58%	54%	40%	33%	-
at angle	33%	46%	60%	17%	-
parallel	<1%	<1%	0%	50%	-
Debris dam					
height (cm)	9.30[a]	29.44[b]	8.60[a]	23.14[b]	16.87[c]
Volume (m^3)	0.14[a]	13.17[bc]	0.16[a]	2.04[b]	0.07[c]
Pool formation					
yes	57%	67%	50%	29%	13%
depth (cm)	13.53[a]	42.21[b]	14.70[a]	36.00[b]	10.50[ab]
Frequency					
total volume (m^3)	3.25	197.48	2.86	14.25	0.99
vol (m^3)/stream width (m)	0.03	0.82	0.05	0.01	0.01
vol (m^3)/100m	1.63	16.46	2.86	7.13	0.99
dams/100m	11.50	1.25	18.00	3.50	15.00

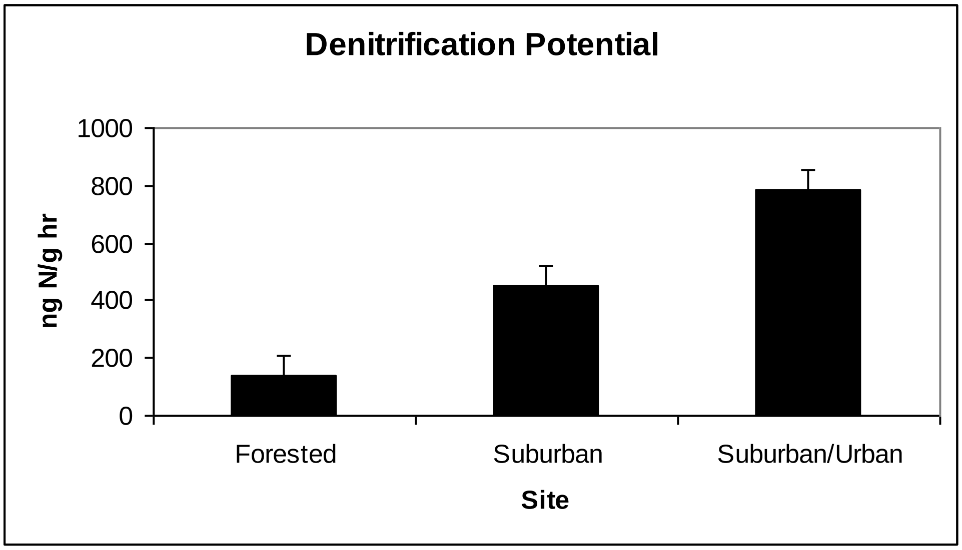


FIGURE 1. Denitrification potential in forested, suburban, and suburban/urban stream debris dams.

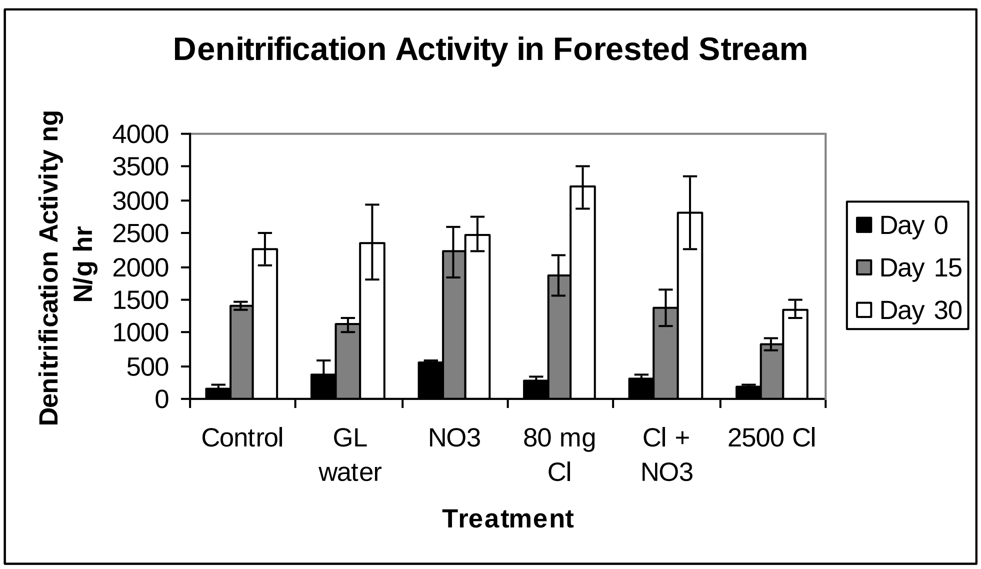
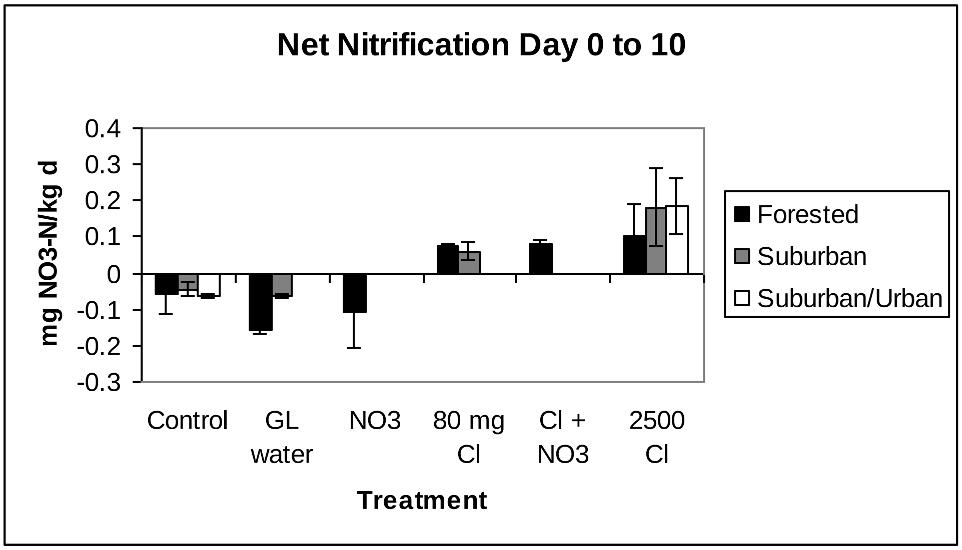
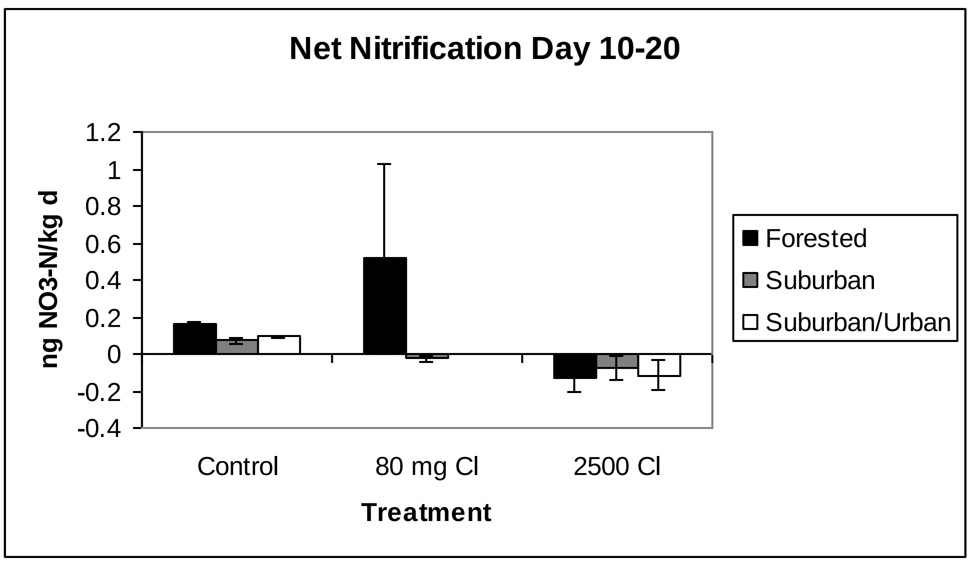


FIGURE 2. Denitrification Activity in Pond Branch (forested) stream debris dam samples incubated in ambient stream water, suburban stream water, or amended with NO_3^-, 80 mg Cl^- L^{-1}, NO_3 and Cl^-, and 2500 mg Cl^- L^{-1} at 0, 15, and 30 days.

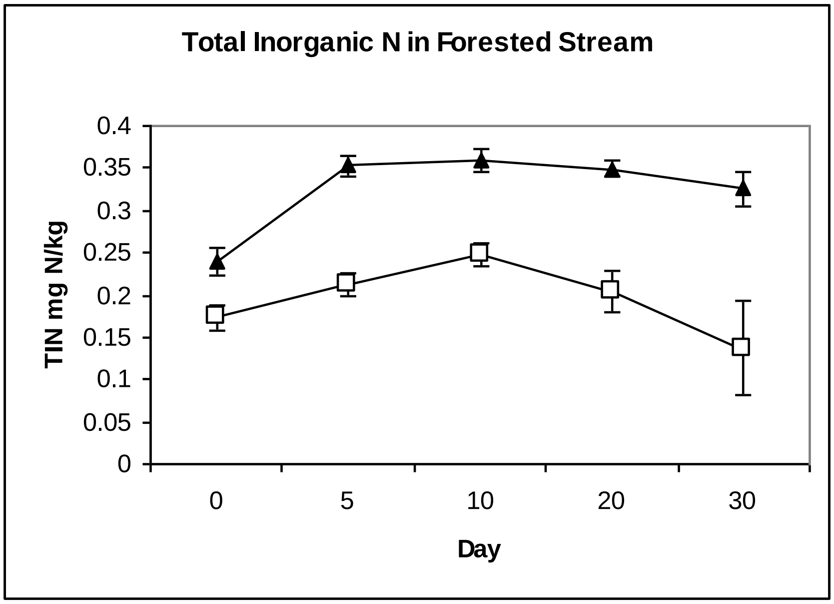


A



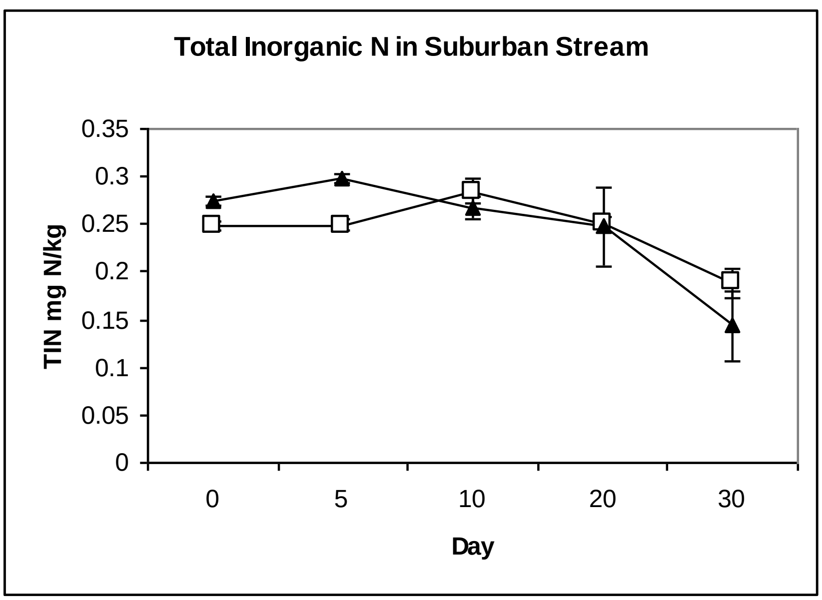
B

FIGURE 3. Net nitrification activity in forested, suburban, and suburban/urban stream debris dams treated with urban stream water, or amended with NO3, 80mg Cl L-1, NO3 and Cl-, and 2500 mg Cl L-1 over the first 10 days of the incubation (A) and over the second ten days (B).



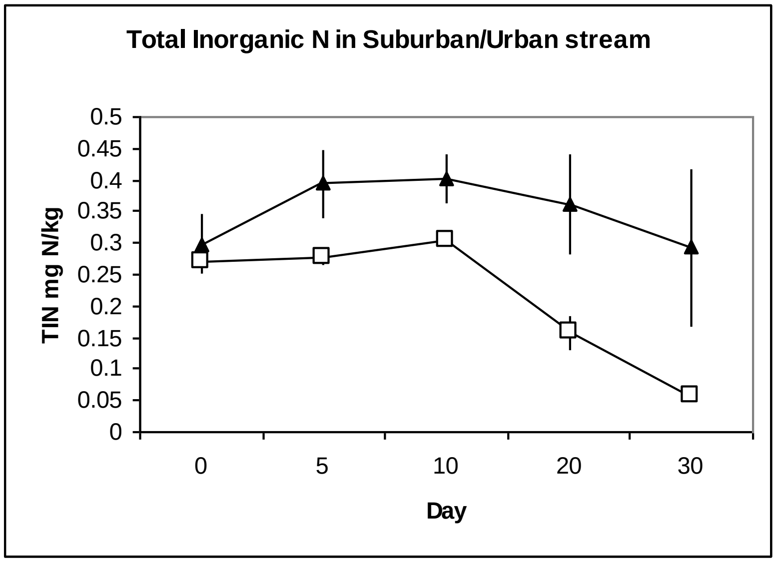
Total Inorganic N in Forested Stream
TIN mg N/kg
0.4
0.35
0.3
0.25
0.2
0.15
0.1
0.05
0
0
5
10
20
30
Day

A



Total Inorganic N in Suburban Stream
TIN mg N/kg
0.35
0.3
0.25
0.2
0.15
0.1
0.05
0
0
5
10
20
30
Day

B



C

FIGURE 4. Total inorganic nitrogen in Control (☐) and 2500 mg Cl- (▲) treatments at 0, 5, 10, 20, and 30 days in debris dams sampled from (A) Pond Branch (forested); (B) Baisman Run (suburban); and (C) Glyndon (suburban).

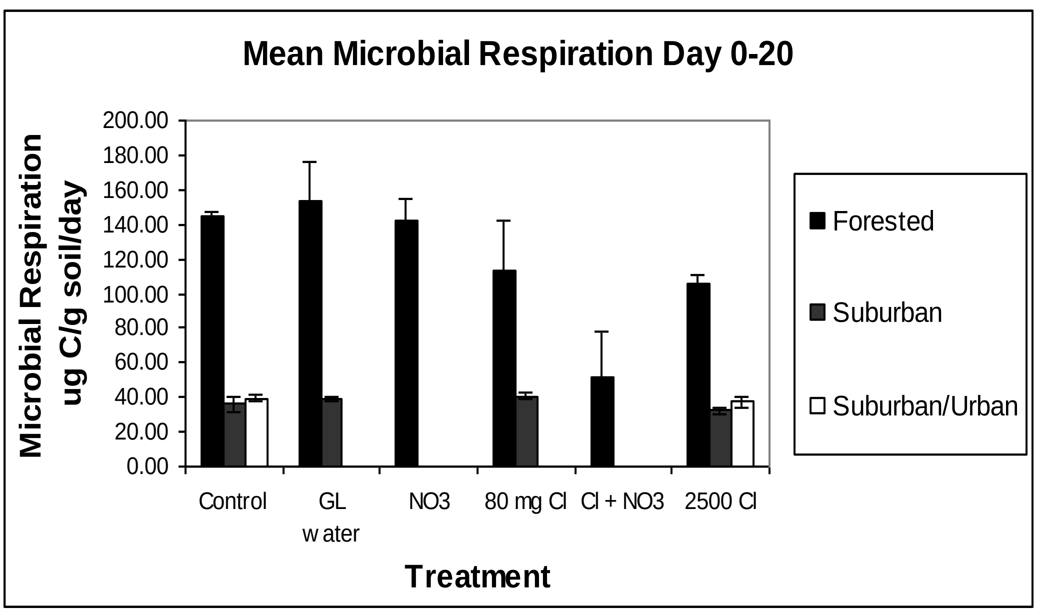


FIGURE 5. Mean microbial respiration over days 0-20 at Pond Branch (forested), Baisman Run (suburban) and Gwynn Falls at Glyndon (suburban/urban) across all treatments.

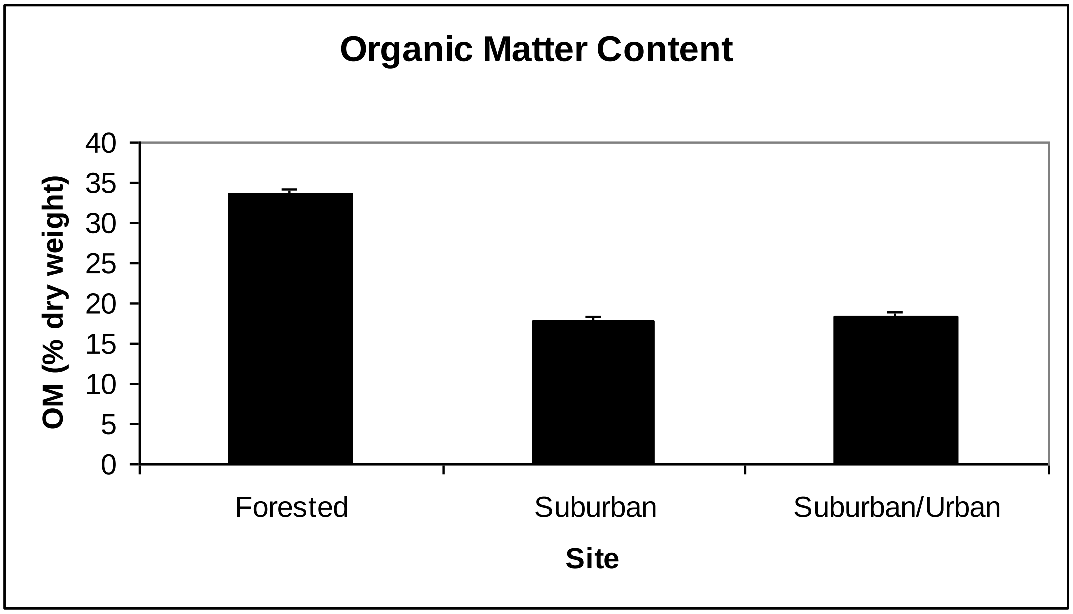


FIGURE 6. Organic matter content of debris dams at Pond Branch (forested), Baisman Run (suburban), and Gwynn Falls at Glyndon (suburban).

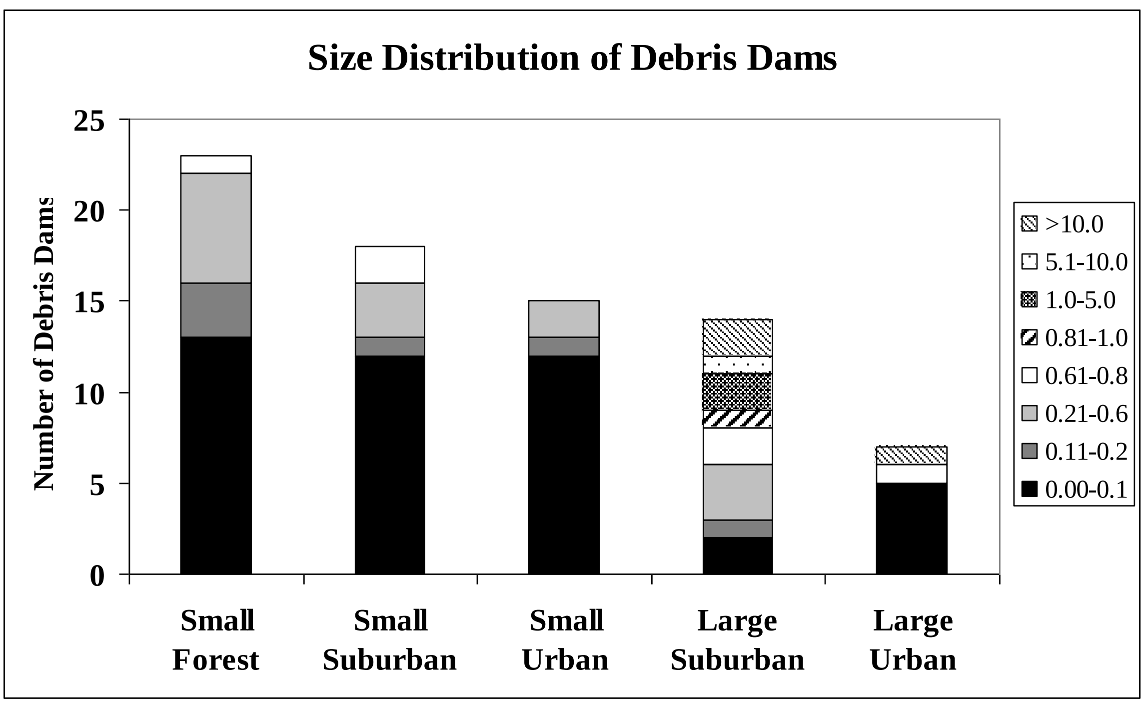


FIGURE 7. Size distribution of debris dams in five streams in Baltimore County, MD.