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Interactions between flow, periphyton, and nutrients in a heavily impacted urban stream: implications for stream restoration effectiveness

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Abstract

Urban stream restoration is a very complex task due largely to the interactions between the physical, chemical, and biological stream components. Because of these interactions, restoring only a single component to a more natural state could have a negative affect on stream health. We studied pre-restoration interactions between hydrology, nutrients, and periphyton in a stream where wastewater effluent and a highly developed urban watershed dominated stream flow. Floods capable of scouring all visible periphyton from the stream were produced from rainfall events as small as 1.3 cm and created 47 periphyton biomass reset events during our 22-month study period. Despite these disturbances, periphyton biomass rapidly accumulated throughout the stream and reached nuisance levels after 5 days of growth during every season. Floods did, however, severely limit the occurrence of steady-state assemblages, which attained biomass levels 30 times the nuisance level. Although the high frequency of floods did not prevent nuisance levels of periphyton, it did allow more edible early stage periphyton assemblages to become far more common than late-stage, less edible assemblages. In the case of the stream studied, a successful restoration strategy must consider coupled processes relating to hydrology, chemistry, and biota.

Keywords: Stream restoration; Periphyton; Wastewater effluent; Urban watershed; Scouring; Flood frequency

1. Introduction

The restoration of urban streams to their natural form and function is a growing occurrence as cities begin to change their perspective on a stream's recreational, commercial, and intrinsic value. However, the actual process of restoring an impacted stream can be complex because stream improvements must often be tailored to specific site and watershed conditions (Moses et al., 1997). In addition, a diverse group of planners including engineers, biologists, governmental officials, and community residents must be involved to identify the impacts of urbanization on the physical, chemical, and biological components of a stream, and create an effective restoration plan.

Stream hydrology has been the focus of many stream restoration projects as it is a key factor in

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stream ecological recovery (Shields et al., 2003). Stream flow can have a large impact on ecosystem form and function, and influence human use of the stream and the adjacent land. Natural stream hydrology is altered by the construction of impervious surfaces and stormwater drainage systems, which reduce the infiltration of precipitation and increase the frequency, amplitude, and overall severity of floods (Corbett et al., 1997). Additionally, in low order streams, discharge from wastewater treatment plants (WWTP) can significantly alter stream flow producing a relatively constant, nutrient-rich flow that is conducive to high levels of primary production and accrual of algal biomass.

The biological component of a stream is commonly affected by urbanization and a change in this component can be especially visible. Increased primary productivity often accompanies the increased urban nutrient load (Smart et al., 1981), with municipal WWTPs a common urban stream nutrient source. Periphyton biomass generally increases downstream of nutrient point sources (Scrimgeour and Chambers, 2000), and in cases of extreme nutrient input, often reach nuisance levels. Excessive periphyton growth can be detrimental to the stream ecosystem, as well as decrease the stream's capacity to be used by humans. For example, extensive amounts of periphyton can lower dissolved oxygen concentrations (Horne and Goldman, 1994), reduce aesthetic appeal and recreational use, and increase the cost of water extraction (Biggs, 1996). Furthermore, excessive nutrients can alter periphyton assemblage structure, possibly leading to the dominance of cyanobacteria (Cattaneo, 1983; Peterson and Grimm, 1992), which is generally less edible by aquatic grazers (Reinikainen et al., 1994; Roelke et al., 1997, 2004; Ghadiuani et al., 2003). Moreover, cyanobacteria have the potential to produce chemicals toxic to other foodweb components (Hay and Kubanek, 2002; Lehtiniemi et al., 2002).

Stream hydrology and periphyton structure and accumulated biomass are linked through the frequency of scouring events. Typically, periphyton succession begins with an initial colonization of diatoms, followed by filamentous green algae and cyanobacteria (Cattaneo, 1983; Peterson and Grimm, 1992). In eutrophic waters, nuisance levels of periphyton frequently occur with assemblages commonly dominated either by rapidly growing filamentous green algae, or cyanobacteria (Davis et al., 1990). In urbanized watersheds, however, the amplified periphyton accrual may be accompanied by an increased frequency of floods that are capable of scouring the resident periphyton from the stream's substrate and reset periphyton assemblages to a lower biomass and an earlier successional state. So while nutrient availability largely dictates the rate of periphyton growth, the frequency between floods dictates the amount of time available for periphyton accumulation. Therefore, a stream's flow regime can contribute effects equally important to those of nutrient limitation with regard to periphyton accumulation and composition (Biggs and Close, 1989).

Given the challenging nature of stream restoration, the diverse stakeholder interests, and likely funding constraints, it is difficult to undertake a restoration project that simultaneously addresses the hydrological, biological, and chemical aspects of a stream. However, because stream systems are complex, i.e., components are inter-linked, implementing a restoration plan that focuses on one component at a time may not yield a progressive gain towards achieving restoration goals. This paper presents pre-restoration data from a heavily impacted urban stream that represents physical (flow), biological (periphyton), and chemical (nutrients) components of the system. Our purpose is to characterize the interactions between highly visible stream characteristics that are commonly affected by urban changes. In addition, this study provides an example of a system where restoration of a single component (in this case stream hydrology) to a more natural state may be ineffective at improving stream health, and may possibly even reduce the current stream quality and ecologic function.

2. Materials and methods

2.1. Study site

The study was conducted in Carter Creek, a third-order semitropical stream located adjacent to an urban area of approximately 150,000 (Bryan/College Station, Texas, 30°38′N, 96°29′W). Our sampling period spanned May 2000 to February 2002. During this time, the stream received secondary-level treated wastewater effluent from two municipal wastewater

treatment plants, and effluent from a small community wastewater plant, which used a waste-stabilization pond treatment process (Fig. 1). The upper WWTP released 15,000–19,000 m^3 day^{-1} of wastewater effluent into Burton Creek, just upstream of its confluence with Carter Creek. The middle WWTP was approximately 3.5 km downstream of the upper discharge and released 19,000–23,000 m^3 day^{-1}. The lower WWTP produced a maximum effluent volume of 34 m^3 day^{-1}, which entered the stream approximately 4 km downstream of the middle discharge. Visible flow in Carter Creek above the Burton Creek confluence was never observed during the study period, and wastewater effluent was the only visible source of water entering the stream between rain events.

Approximately 70% of the stream's 14 ha watershed was developed and the urbanized area was located entirely within the upper portion (above station 3) of the watershed. Undeveloped pasture and woodlands surrounded the lower section of the stream. The stream's substrate was made up almost entirely of sand. Periphyton mats were commonly visible in areas of low to moderate flow throughout the entire stream length. A layer of periphyton covered surfaces along stream edges and in backwaters throughout our study. Visually, these mats commonly appeared to be dominated by diatoms with very little filamentous algae present.

We chose six sampling stations along the creek (Fig. 1). Station 1 was approximately 0.8 km upstream of the middle WWTP discharge, and station 2 was approximately 0.1 km downstream of the middle WWTP discharge area. The remaining four stations were distributed along the lower reaches of the stream to approximately 0.5 km upstream of its confluence with the receiving river. The channel length between the first and last stations was approximately 13 km.

2.2. Hydrology

Base flow discharge patterns were determined by weekly measurements of stream velocity (m s^{-1}) at station 3, where a culvert created a hydrologic control point. Flow was measured with a General Oceanic mechanical flowmeter. Here we also measured the cross-sectional area of the stream in order to calculate stream discharge (m^3 s^{-1}). The proportion of the stream flow that originated from the municipal WWTPs was calculated using stream discharge measurements at station 3, and the daily records (provided by WWTPs) of the volume of effluent released.

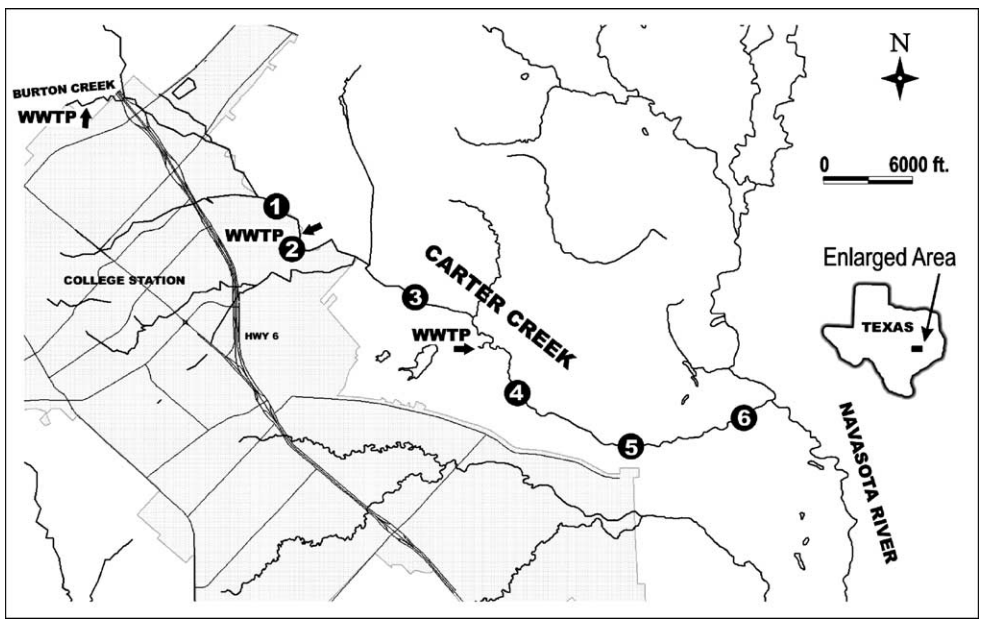

Fig. 1. Location of Carter Creek sampling sites and wastewater treatment plants. Arrows denote locations of wastewater discharge sites.

We estimated the influence of watershed runoff on stream flow by measuring stream discharge at station 3 at 30-min intervals during, and following a rain event. The amount of rain required to substantially disturb the stream's sand substrate, and thus significantly scour or bury periphyton assemblages was determined through numerous visual observations during the 2-year study period. The occurrence of scouring events in the stream was then estimated for the study period through analyses of daily rainfall records collected at four locations in the watershed.

2.3. Nutrients

Nutrient concentrations were collected at each station once per month during the study period. Nutrients analyzed included nitrate (NO_3) + nitrite (NO_2), ammonia (NH_3), and soluble reactive phosphorus (SRP) following standard methods (AWWA, 1998). During months when periphyton accumulation was measured (see Section 2.4), these data were recorded on the first day that periphyton were sampled.

2.4. Periphyton

2.4.1. Accumulation

Early stage colonization and accumulation processes were measured seasonally across 6-day periods in May, August, and November of 2001, and February of 2002. Periphytometers were constructed from 1.27 cm wide rings cut from a 3.175 cm diameter (10.13 cm^2 total surface area) Schedule 40 PVC pipe (see Murdock, 2002). During each of the seasonal periods, 10 rings were slid over a 1.5 m long pole of 2.54 cm diameter Schedule 40 PVC pipe and secured with plastic cable ties. The PVC pole was then driven into the streambed until the top ring was 2.5 cm below the water surface, thus standardizing depth distribution of samples across stations. PVC has frequently been used as a colonization substrate for periphyton (Goldsborough et al., 1986; Lemmens, 2003). Additionally, the PVC samplers should be adequate for this study because periphyton habitat preferences may become eroded in eutrophic waters, thereby reducing the importance of substrate type (Moss, 1981; Danilov and Ekelund, 2001).

Five samplers were placed at each station, taking care to closely match their placement across stations with regard to instream light regimes and water velocities. Station current velocities were measured with a mechanical flowmeter and all poles put into similar low current conditions. The first samples (T1) at each station were taken 48-h after the initial deployment. Subsequent samples (T2–T5) were taken at four consecutive 24-h intervals resulting in a total accumulation time of 6 days (day 0 to day 5).

At each station, a sample consisted of six rings, comprising the upper two rings from each of three randomly selected poles. Of each pair of rings, one ring was used for chlorophyll *a* analyses, and the other for microscopic examination. The rings used for chlorophyll *a* determination were individually wrapped in aluminum foil, placed on ice, and taken back to the laboratory and frozen until analyses could be performed. A Turner Designs 10-AU fluorometer was used to measure chlorophyll *a* ($\mu g\,l^{-1}$), corrected for phaeophytin content (US EPA, 1992) and converted to $mg\,m^{-2}$. The rings used for microscopic examination of assemblage structure were scraped with a razor blade and algae were placed into 30 ml scintillation vials containing 1 ml of glutaraldehyde. Cells were identified using Cox (1996), and Prescott (1962, 1978). For the purposes of this research, taxa were placed into functional groups. The 12 algal functional groups used were single-cell pennate diatoms, chain-forming pennate diatoms, single-cell centric diatoms, chain-forming centric diatoms, single-cell green algae, colonial green algae, filamentous green algae, coccoid cyanobacteria, sheathed filamentous cyanobacteria, unsheathed filamentous cyanobacteria, red algae, and flagellates. Cell dimensions were measured with a stage micrometer to calculate the biovolume of each cell by comparing its shape to geometric shapes of known volume (Sicko-Goad et al., 1977). The cell biovolume percentage of each functional group was calculated for each sample.

2.4.2. Standing biomass

From April 2001 to February 2002, 30-day standing biomass samples were collected and analyzed for chlorophyll *a* concentration to quantify late-successional stages of periphyton assemblages. Two additional samplers were installed at each station and six rings were removed at each station 30 days after installation. Late-successional samples were processed as described for early successional samples.

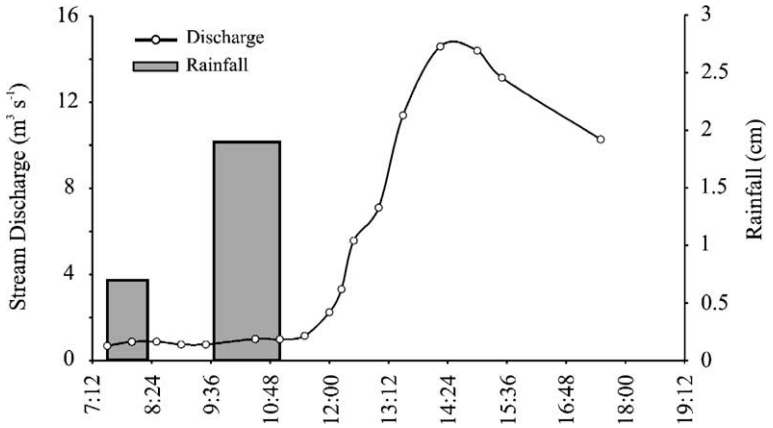

Fig. 2. Change in stream discharge after a 2.6 cm rainfall event. Bars show time of rainfall and amount of precipitation. Base flow was $0.68\,m^3\,s^{-1}$ and the maximum discharge measured was $14.4\,m^3\,s^{-1}$.

Due to the magnitude and frequency of floods in the stream, most periphyton samplers were lost before 30-day sampling could occur.

2.5. Data analyses

Principle Component Analyses (PCAs) were performed, using the last sample in each season to detect multivariate relationships between (1) early stage periphyton biomass and (2) early stage periphyton composition, and the abiotic parameters measured in each season. PCA data were standardized by subtracting the mean and dividing by the standard deviation.

3. Results

3.1. Hydrology

With a combined WWTP discharge of 30,000–75,000 $m^3\,day^{-1}$, wastewater effluent averaged 70% of stream discharge during base flow and often reached 100%, independent of season. Low effluent percentages (<40%) always coincided with a rain event. Effluent volume from the most downstream WWTP was negligible, typically making up less than 0.0009% of the stream discharge. Baseline discharge volumes averaged $0.7\,m^3\,s^{-1}$. However, stream discharge increased substantially after even minimal rain events. For example, a 20-fold change in stream flow occurred following a rain event of only 2.6 cm (Fig. 2). The maximum discharge volume measured was $61\,m^3\,s^{-1}$, which was prior to the stream overflowing the channel during an 8.1 cm rain event.

Approximately 1.3 cm of rainfall was sufficient to mobilize the sand substrate and completely scour or bury all previously visible periphyton. Applying this knowledge to the number of rainfall events that exceeded 1.3 cm, the number and frequency of reset floods that occurred during this 22-month study was 47 (Fig. 3). Consecutive days having greater than 1.3 cm of rainfall were counted as part of the same reset flood. The average number of days between reset floods was 11, with maximum periods between scouring events (34, 28, and 70 days) all occurring during the summer months.

3.2. Nutrients

Between rain events, nutrient concentrations were frequently elevated throughout the stream with maximum concentrations almost always occurring at Station 2, just below the middle WWTP discharge. Concentrations of NO_3 combined with NO_2 ranged from 1.10 to 13.30 $mg\,l^{-1}$ (mean of 7.32 $mg\,l^{-1}$). Concentrations of SRP ranged from 2.72 to 11.25 $mg\,l^{-1}$ (mean 6.40 $mg\,l^{-1}$). NH_3 had a mean concentration of 0.25 $mg\,l^{-1}$, with two separate spike events (3.50 and 2.90 $mg\,l^{-1}$) observed at Stations 1 and 2, respectively. All nutrients consistently exhibited a spatial trend of decreasing concentration with distance downstream,

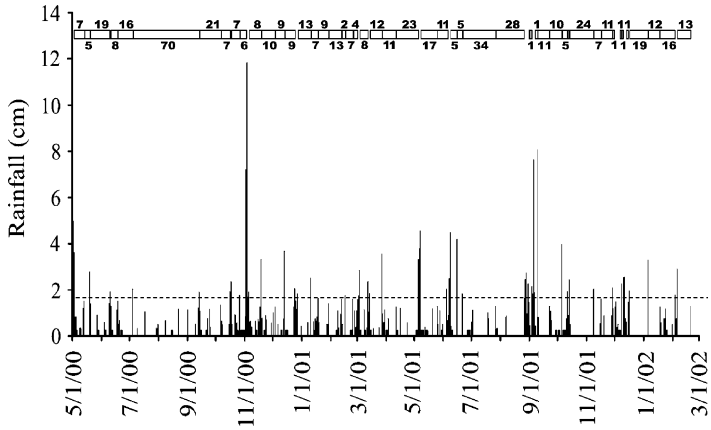

Fig. 3. Periphyton assemblage reset frequency. Vertical lines represent daily rainfall averaged over four recording stations across the watershed. The dashed horizontal line represents the minimum rainfall needed to completely scour all visible periphyton from the substrate. The upper bar displays the number of days between rain events that exceeded that minimum threshold.

although the decreases were not pronounced. While temporal differences were pronounced, there was no consistent seasonal trend (see Fig. 4, for example, with combined NO_3 and NO_2).

Although instream processes played a role in reducing nutrient concentrations, rainfall appeared to have a much greater, and more unpredictable, effect on their reduction. For example, $NO_3 + NO_2$ and SRP were reduced 97 and 82%, respectively, from their mean concentrations after a 4.1-cm runoff event, presumably through dilution (see Fig. 5). Yet, by the time stream discharge returned to levels amenable for

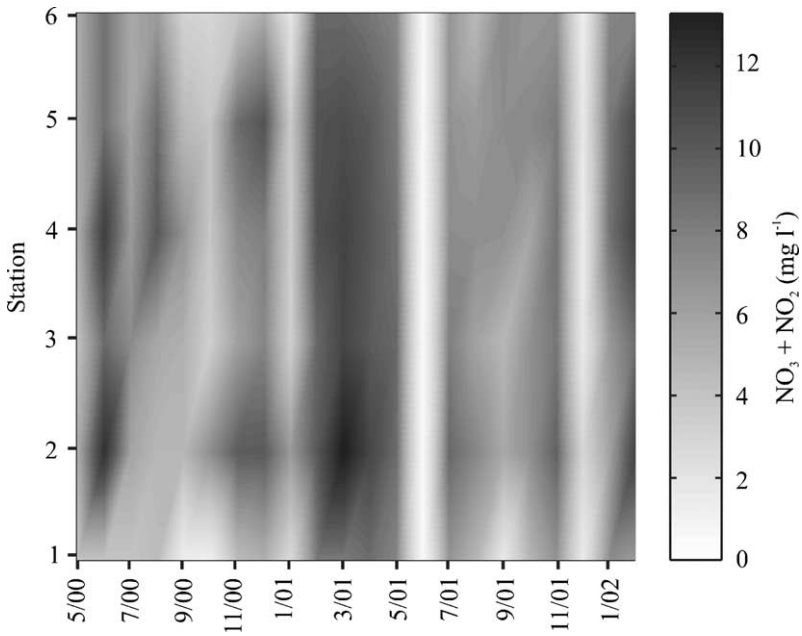

Fig. 4. Combined $NO_3 + NO_2$ concentrations collected monthly at each station. The *x*-axis represents time on a monthly interval, and the *y*-axis is station number. Shades of gray correspond to nutrient concentrations with darker shades equal to higher concentrations.

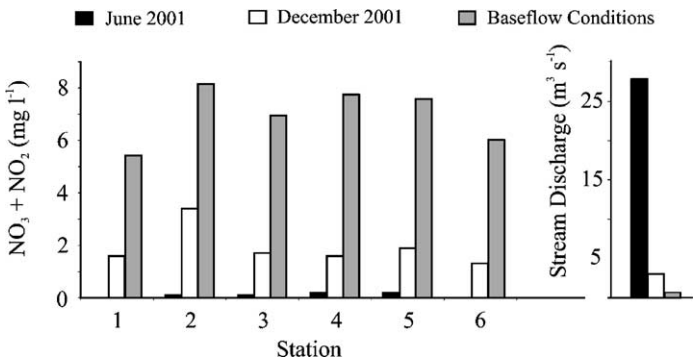

Fig. 5. Flood-related changes in stream $NO_3 + NO_2$. Values at two different levels of rainfall (June 4.1 cm, and December 2.67 cm), and at base flow. Base flow values shown are the mean concentration at each station during the study collected during base flow conditions. SRP concentrations (not shown) had similar trends.

periphyton colonization, nutrient concentrations were again within their normal ranges.

3.3. Periphyton

3.3.1. Early accumulation

Early stage periphyton colonization biomass reached nuisance levels (>100 mg chl *a* m^{-2}) regardless of season within the 6-day growth period (Fig. 6). Peak biomass occurred just downstream of the middle WWTP discharge (Station 2) in all seasons except November, when peak biomass occurred at Station 1. Early stage periphyton colonization showed a spatial trend with a general decrease in biomass with distance downstream during each seasonal experiment. A seasonal trend was also evident with the highest biomass

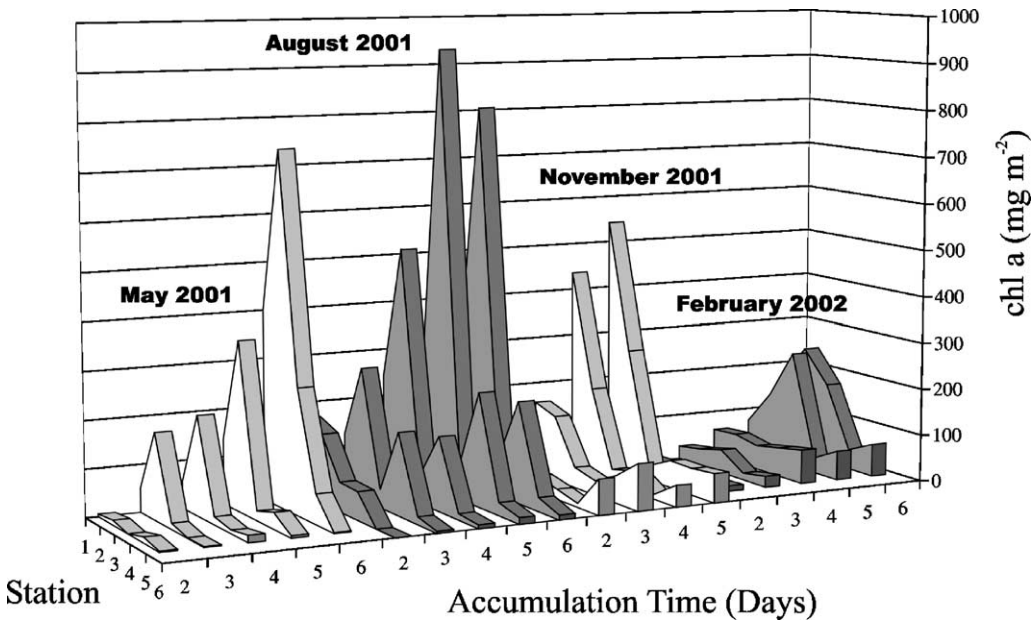

Fig. 6. Average early periphyton colonization measured as chlorophyll *a* concentration (mg m^{-2}). Stations 1–6 (upstream to downstream) are on the *z*-axis and chlorophyll *a* is on the *y*-axis. The *x*-axis shows seasonal differences in accumulation patterns for periphyton across the five consecutive days of sampling. The first sample was taken on day 2 and the last sample on day 6 for each season, except in November where a flood washed away samplers before the day 6 sample could be taken.

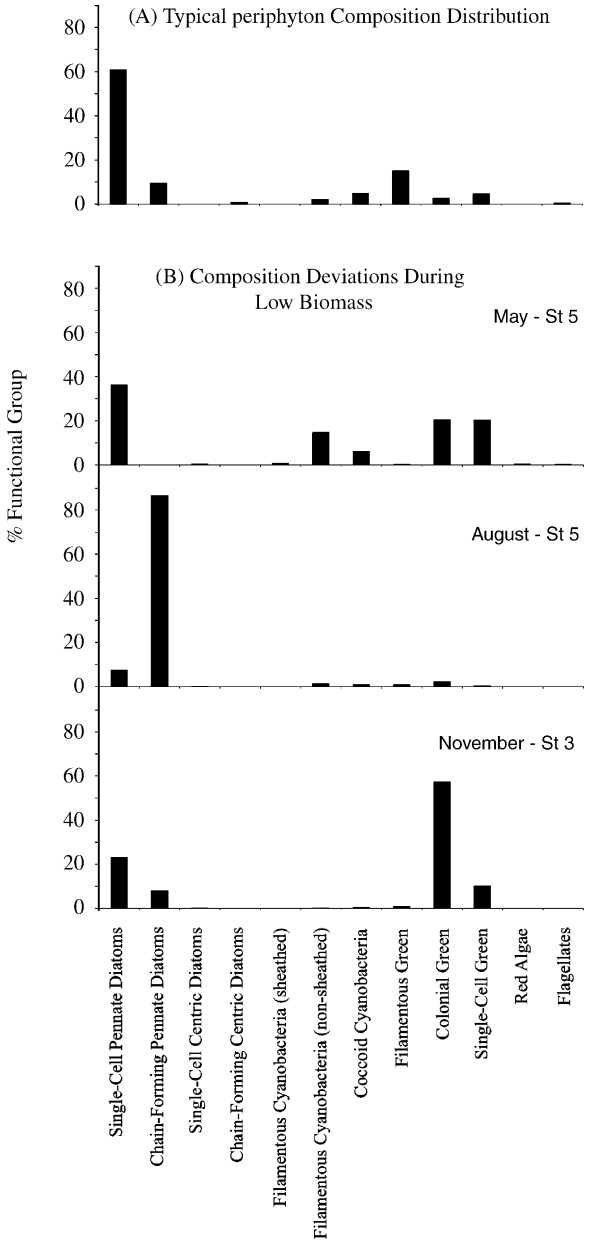

Fig. 7. Early periphyton colonization assemblage composition. (A) Typical periphyton composition distribution seen during all seasons and at all stations. (B) Compositional changes in periphyton at stations that obtained low biomass in May, August, and November.

occurring in the summer, lowest in the winter, and intermediate levels in the spring and fall.

In order to get a better description of the effects of interactions between nutrients and periphyton growth, a principal component analysis was performed incorporating periphyton colonization biomass, nutrient concentrations, and water quality. The PCA did not show a relationship between periphyton biomass and nutrient concentration (see Murdock, 2002).

3.3.2. Assemblage composition

In general, diatoms dominated early succession periphyton assemblages during all seasons with single-cell pennate diatoms being the dominant group (see Fig. 7). During the spring, summer, and fall, the stations that attained the lowest biomass deviated from this pattern. These low-biomass assemblages comprised a greater percentage of cyanobacteria and green algae, or were predominately chain-forming pennate diatoms. The PCA of water quality parameters, nutrient concentrations, and assemblage composition did not show any recognizable trend (see Murdock, 2002). Very low levels of cyanobacteria were found during the colonization process. However, cyanobacteria assemblages were routinely observed on the rocks surrounding the middle WWTP outfall, and these rocks were rarely, if ever, subjected to scouring because the outfall construction protected them from scouring during floods.

3.3.3. Standing biomass

From April 2001 to February 2002, there were only 7 months in which all the PVC pipe samplers were not washed away due to floods. Of those 7 months, there were only three (July, August, and December 2001) where scouring did not appear to influence periphyton on the PVC pipe samplers. Assemblages in these months were assumed to have reached a steady state after the 30-day incubation period, and reached very high chlorophyll *a* values. Across all stations, maximum monthly values were 3764 mg chl *a* m^{-2} in July, 3358 mg chl *a* m^{-2} in August, and 3340 mg chl *a* m^{-2} in December. Steady-state accumulated biomass appeared similar in the summer and winter, and generally decreased with increasing distance downstream from wastewater outfalls. In December, chain-forming centric diatoms dominated the assemblage structure, whereas during summer a red alga (*Compsopogon* sp.) was dominant (see Murdock, 2002).

4. Discussion

4.1. Nutrients

Because the stream had very little natural base flow, the WWTP effluent strongly influenced in-stream processes. Effluent dominated nutrient-loading created consistently elevated nutrient concentrations throughout the stream. But stream nutrient processing capabilities did not appear overwhelmed because nutrients consistently showed a moderate decrease downstream of the WWTP discharges. Low nutrient concentrations only occurred during precipitation events through dilution by runoff. This was not an issue for periphyton growth dynamics though, because runoff events strong enough to significantly reduce nutrient levels scoured away the periphyton.

Under base flow conditions, nutrient concentrations did not show a visible relationship to periphyton biomass, and statistical analysis suggested that nutrients did not regulate periphyton accumulation. So nutrients were not a limiting resource in the rapid colonization and accumulation of periphyton, nor did they appear limiting in high biomass, late-stage assemblages. As a result, the current nutrient load can potentially support a higher, more rapidly accumulating periphyton biomass than what is currently present.

4.2. Periphyton biomass

The watershed's ability to produce substantial amounts of runoff in combination with the stream's easily displaced sand substrate created a volatile flow regime and unstable stream habitat. As a result, even relatively small amounts of rainfall (approximately 1.3 cm) were able to decimate periphyton assemblages. The average growth period of only 11 days prohibited maximum periphyton levels to occur in all but 4 months during the 22-month study. However, rapid colonization and accumulation rates almost always allowed nuisance levels of periphyton to become established before the next scouring event.

Horner et al. (1983) and Welch et al. (1988) suggested that a nuisance level of periphyton has a chlorophyll *a* content >100–150 mg m^{-2}. Similar critical values were reported by Nordin (1985) who suggested that recreational use of a system will be affected by algal chlorophyll *a* values >50 mg m^{-2}, and aquatic life will be affected by chlorophyll *a* values >100 mg m^{-2}. During early colonization in this stream, chlorophyll *a* measured on introduced substrates reached levels higher than peak values commonly reported for eutrophic systems (Morin and Cattaneo, 1992; Biggs, 1996). Periphyton chlorophyll *a* levels in Carter Creek exceeded 100 mg m^{-2} by day 2 in August, by day 3 in May and November, and by day 5 in February. While these early assemblages reached extremely high biomass levels, when given a greater time interval to accumulate, as in the 30-day samples, chlorophyll *a* levels continued to increase substantially, reaching greater than 30 times the nuisance value regardless of season.

4.3. Periphyton composition

Despite the extreme accumulation of periphyton in this stream, our observations still agree with current theory for periphyton succession in that single-cell pennate diatoms dominated early assemblages. Based on small cell size and baring toxin production, these assemblages would have been highly edible for grazers. During each colonization experiment, however, other functional groups became dominant at the station with the lowest biomass. But again due to predominantly smaller cells, these functional groups still appeared to be edible.

Steady-state community composition in Carter Creek did not fit well with typical successional theory in eutrophic waters. Instead of cyanobacteria and filamentous green algae as the capstone assemblage, we observed chain-forming centric diatoms in the winter and filamentous red algae in the summer. Yet, these assemblages were still likely to be resistant to many invertebrate grazers due to their large size. The less edible capstone assemblages observed in our 30-day samples were rarely seen on the sandy surface of the natural streambed. This was due to the frequency of disturbances (see Roelke et al., 1999; Roelke, 2000), in this case scouring events, which kept periphyton in an early stage of succession more often than a late successional stage. Our colonization data suggested that the early assemblage was edible. Therefore, these successional resets by floods might have promoted assemblages dominated by small, single-cell pennate diatoms, and hindered the dominance of a lower

quality food source for invertebrate grazers such as the late-stage, chain-forming centric diatoms and filamentous red algae (Cattaneo, 1983; Sommer, 1997).

5. Conclusion

The goal of this study was to demonstrate that altering a single system component might not always improve overall stream health, even when that alteration is the restoration of a greatly impaired stream component back to a more natural state. Our data indicated a strong interrelationship amongst the physical, biological, and chemical stream components in this urbanized stream. By restoring only stream flow to a more natural state without altering nutrient loading, extremely high biomass, late-succession periphyton assemblages would most likely dominate. And conceivably, without regular flood reset events, periphyton would only be limited by substrate space on which to colonize. Because these late-stage assemblages may provide a lower quality food source for grazers, the amount and efficiency of nutrients passed to higher trophic levels may possibly be reduced. In the case of Carter Creek, a successful restoration strategy must consider coupled processes relating to hydrology, chemistry, and biota.

Acknowledgements

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