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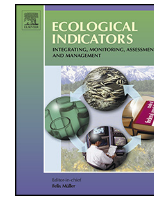

Aligning indicators of community composition and biogeochemical function in stream monitoring and ecological assessments

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ABSTRACT

Reliable and inexpensive indicators of ecosystem function are essential for accurately monitoring and describing ecosystem integrity. Currently, most state and federal assessments of aquatic ecological integrity rely on structural indicators and assume tight coupling of structure and function. We used fluorescent composition of dissolved organic matter as a metric for certain ecosystem functions and compared the resulting index of autochthonous microbial dissolved organic matter (DOM) to macroinvertebrate indicators and classifications of water quality attainment reported by the Maine Department of Environmental Protection (Maine DEP) at 142 stream sites. We observed that metrics of sensitive insect orders such as relative Plecoptera generic richness, relative Ephemeroptera abundance, and generic richness of EPT (Ephemeroptera, Plecoptera, and Trichoptera) were negatively correlated with higher values of metrics based on autochthonous microbial DOM sources. At the same time we observed an increase in the Hilsenhoff Biotic Index with increasing microbial DOM. We compared the abundance of this microbial DOM component to Maine DEP measured attainment classes and found that microbial DOM generally separated sites with high biological integrity from sites where the biotic community was highly degraded. This highlights that measures of biogeochemical ecosystem function complement measures of structure in biological assessments.

1. Introduction

In many countries, legislation has been enacted to protect water bodies for a variety of human and natural uses. The ultimate goal of such legislation is to sustain ecosystem services that provide benefits such as clean water and consumable biomass production by protecting both ecosystem structure and function (e.g. United States Clean Water Act, CWA; European Union Water Framework Directive, WFD). Many regulatory tools relying on chemical and biological indicators have been developed to help determine the ecological status of these water bodies (Dolédec and Statzner, 2010; Birk et al., 2012; Carter and Resh, 2013). However, monitoring agencies generally rely heavily on structural indicators such as fish, macroinvertebrate, or algal community composition metrics to determine if a water body is meeting its water quality goals (Birk et al., 2012; Carter and Resh, 2013).

Macroinvertebrates, fish, and algae have long been used as structural indicators of water quality because they integrate environmental conditions and stressors over long periods (Hilsenhoff, 1987). Taxonomic composition of the community reflects both acute and chronic exposure to anthropogenic stressors such as ‘flashy’ discharge, elevated pollutants, and resource availability (Walsh et al., 2005; Wenger et al., 2009). The widespread use of ecosystem structural metrics to assess ecosystem integrity and to protect ecosystem services presupposes that ecosystem function matches ecosystem structure (Davies and Jackson, 2006; Sandin and Solimini, 2009; Woodward et al., 2012). This assumption may be particularly problematic in aquatic ecosystems where measures of biodiversity structure may poorly reflect ecosystems services like water purification (Cardinale et al., 2012). This may occur because community composition does not always reflect function in disturbed ecosystems (Sandin and Solimini, 2009; Young and Collier, 2009; Woodward et al., 2012), especially if the community assayed does not directly drive the function of interest. For

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example, many biogeochemical cycles are controlled by microbial communities, while most water quality monitoring protocols focus on macroinvertebrate and fish assemblages. This suggests that structural metrics should be complemented by functional metrics to accurately assess the condition of an ecosystem (Sandin and Solimini, 2009).

Direct measures of stream ecosystem function as ecological indicators have become more common, and frameworks have been proposed for their use and interpretation (Young et al., 2008; Palmer and Febria, 2012). Most of the emphasis to date has been on decomposition of leaves and leaf analogs (Paul et al., 2006; Simon et al., 2009; Imberger et al., 2010), microbial extracellular enzyme activities (Harbott and Grace, 2005; Lehto and Hill, 2013), and ecosystem metabolism (Bunn et al., 1999; Mulholland et al., 2005; Young et al., 2008). However, the response of the latter to urban watershed disturbances can be inconsistent across studies (Meyer et al., 2005; Bernot et al., 2010). Studies using such functional measures have rarely related them systematically to structural measures, but the few that have done this found that they do not always align neatly (Young and Collier, 2009; Del Arco et al., 2012). While the importance of directly measuring ecosystem function is recognized by researchers and practitioners, the effort required to obtain functional measurements remains beyond the budgetary capacity of many agencies (Gessner and Chauvet, 2002; Davies and Jackson, 2006).

Monitoring efforts are largely focused on small streams, as they are important loci of ecosystem services due to their wide spatial extent (>80% of river miles; Leopold et al., 1964) and intensity of biogeochemical cycling (Alexander et al., 2000; Peterson et al., 2001). Microbial community structure and function mediate many ecosystem services in streams via element cycling processes. These microbial functions are driven by the production or consumption of organic matter (Lindeman, 1942). Dissolved organic matter (DOM) is an important basal source of energy in stream food webs (Hall and Meyer, 1998). In small closed-canopy streams, much of this DOM is derived from allochthonous sources (Vannote et al., 1980), which includes leaching from vascular plants and soils. Anthropogenic disturbance of the vegetative land cover and soil forming processes may influence the production and delivery of DOM to small streams (Kominoski and Rosemond, 2012; Stanley et al., 2012; Parr et al., 2015), affecting the structure and function of these ecosystems (Tanentzap et al., 2014).

Whereas the concentration of aquatic DOM may respond unpredictably to anthropogenic watershed disturbance (Stanley et al., 2012), the composition, or mix of organic molecules in DOM may change in more predictable ways (Hosen et al., 2014; Parr et al., 2015). Undisturbed watersheds are characterized by a high proportion of allochthonous DOM (typically humic-acids) derived from wetland and forest covers (humic acid-like DOM). Humic-like DOM can have a variable response to watershed disturbance, but it tends to decrease as wetland or forest dominated landscapes are urbanized (Parr et al., 2015). This response may depend on the type of anthropogenic disturbance; in more continuously disturbed agricultural catchments, humic-like DOM may increase, perhaps due to soil organic matter destabilization and export (Petrone et al., 2011; Graeber et al., 2012). Other forms of DOM characteristic of autochthonous freshwater microbial or algal sources (Williams et al., 2010; Yang et al., 2012; Parr et al., 2015) or wastewater discharges (Baker and Inverarity, 2004; Holbrook et al., 2006) typically increase. These forms of DOM may be more bioavailable (Hosen et al., 2014; Parr et al., 2015) and have the potential to affect the balance of aquatic ecosystem metabolism. As such, DOM integrates multiple watershed processes spanning scales from microbes to landscapes, and can potentially serve as a useful indicator of ecosystem function.

The goal of this study was to determine whether metrics of DOM composition, which reflect ecosystem biogeochemical function and watershed response to disturbance, align with water quality attainment classifications based on structural macroinvertebrate community assemblages. We hypothesized that macroinvertebrate structural and biogeochemical functional indicators would generally align, but with some important differences because drivers of invertebrate community structure and biogeochemical processes only partially overlap. To evaluate this hypothesis, we compared fluorescent DOM composition as our indicator of DOM bioavailability and biogeochemical function to macroinvertebrate-based metrics of community structure and ecosystem biological condition collected by the Maine Department of Environmental Protection (DEP) at sites across an urbanization gradient in the postglacial northeastern U.S.

2. Methods

2.1. Overview

We selected 142 stream sites along a gradient of urbanization in southern and central Maine, U.S.A. that are Maine DEP macroinvertebrate biomonitoring sites. In Maine, urbanization typically occurs in a landscape matrix of second growth forest and wetlands. Most of the watersheds in this study have been reforested for ~50–70 years after logging or agriculture. We excluded sites influenced primarily by agriculture or point source discharges. At each site, we measured a suite of physicochemical parameters and collected water samples to measure optical DOM composition and basic water chemistry (Table A1). Maine DEP's water quality monitoring program assesses the macroinvertebrate community structure of 50–60 streams per year. Maine DEP follows a 5-year cycle and focuses sampling in 5 geographic regions on a rotating basis.

2.2. Study design and site selection

The 142 sites used were selected from the DEP database based on three criteria. First, we selected sites for which macroinvertebrate monitoring data was available. Second, we limited our sampling to 1st and 2nd order streams because these should be most tightly coupled to the terrestrial environment (land cover) and to minimize the effects of natural longitudinal changes on stream ecology and physicochemical parameters (Vannote et al., 1980). Third, we used geographic information systems (GIS) to analyze the degree of urbanization as the percent of total watershed impervious surface area (hard surfaces such as roofs, roads, parking lots, and sidewalks; ISA). 113 of these watersheds were visited 3 times and the remaining 29 were visited once during the summer of 2011 to overlap directly with Maine DEP macroinvertebrate assessment. It is important to note that because the Maine DEP sampling program rotates every ~5 years, we only have temporally matched DOM and macroinvertebrate data for ~63 sites. For the remaining sites, macroinvertebrates were mostly sampled within 1–3 years of DOM sampling. These 142 sites ranged from 0 to 60% ISA (Maine Land Cover Database (MELCD) Impervious Cover layer, 5 m resolution, Maine DEP, 2005) and represented the complete range of urbanization in the state of Maine. Average watershed slopes ranged between 5 and 13% with high and low slopes distributed evenly across the urbanization gradient.

2.3. Macroinvertebrate sampling and water quality attainment determination

To comply with the Clean Water Act, the State of Maine has classified its fresh waters into 4 statutory classes: AA, A, B, and C. The biological standards for each of these classes are summarized as