Limnologica 54 (2015) 23–32

Contents lists available at ScienceDirect

Limnologica

journal homepage: www.elsevier.com/locate/limno

The role of fungi and invertebrates in litter decomposition in mitigated and reference wetlands

R. Tristan Gingerich[a], Daniel G. Panaccione[b], James T. Anderson[a,*]

[a] School of Natural Resources, Wildlife and Fisheries Resources Program, West Virginia University, PO Box 6125, Percival Hall, Morgantown, WV 26506, USA
[b] Division of Plant and Soil Sciences, West Virginia University, PO Box 6108, Morgantown, WV 26506, USA

ARTICLE INFO

Article history:
Received 5 December 2014
Received in revised form 30 June 2015
Accepted 6 July 2015
Available online 31 August 2015

Keywords:
Litter decomposition
Mitigated wetlands
Reference wetlands
Fungi
Invertebrates

ABSTRACT

Wetland plant litter decomposition influences many wetland processes and is itself driven by a complex web of interacting parameters. Invertebrates and fungi make up one portion of that web by processing organic material; however, their role is poorly understood. To explore invertebrate and fungal influence on plant litter decomposition rate, we measured the decomposition of litter in three mitigated (created wetlands) and three reference wetlands in the Mid-Atlantic Highlands of West Virginia, USA. Litter decomposition rates and most invertebrate metrics were not statistically different between mitigated and reference wetlands; only oligochaetes (worms) and the functional feeding group (FFG) collector/gatherers had numbers that were statistically higher in mitigated wetlands. Invertebrate metrics were able to explain 25% (FFG) to 31% (taxonomic groups) of variance during the first phase of decomposition (<224 days) and 15% (FFG) to 21% (taxonomic groups) during the second phase ($\geq$224 days). Shredders, collector/gatherers, and omnivores were more strongly associated with early phases of decomposition, while oligochaetes and omnivores were most strongly associated with trends in decomposition during the later phase. Fungal biomass, as measured by ergosterol concentration, was similar between mitigated and reference wetlands and was significantly higher in the first phase of litter decomposition than the second phase, but was not statistically correlated with litter decomposition rate. Decomposition influences many aspects of wetland function, making the variables that determine decomposition rates important for assessing and mitigating for lost wetland function.

1. Introduction

Wetlands provide many ecosystem services. When natural wetlands in the United States are filled in or destroyed legally, new wetlands are created or previously existing wetlands are restored or enhanced with the intention of replacing lost net ecological function. In order to accomplish that goal, we need to understand the web of interacting forces that support wetland function. Plant litter decomposition is an important part of the web and influences the physical and chemical properties of wetland soils (Mitsch and Gosselink, 2007), nutrient availability and cycling (Prentki et al., 1978; Facelli and Pickett, 1991), primary productivity (Brinson et al., 1981; Xiong and Nilsson, 1997), and organic matter accumulation (Gambrell and Patrick, 1978; Xiong and Nilsson, 1997). These processes link decomposition to overall wetland services such as invertebrate and wildlife habitat through primary production and detritus availability (Burdett and Watts, 2009; Taylor and Batzer, 2010), to carbon storage through organic matter accumulation (Bridgham et al., 2006), to sediment and mineral retention through primary productivity and organic matter accumulation (Braskerud, 2000; Rooth et al., 2003), and to stream nutrient availability through nutrient cycling (Richardson, 1994; Mitsch and Gosselink, 2007).

Invertebrates contribute to wetland services by playing an important role in litter decomposition (Fazi and Rossi, 2000; Wu et al., 2009). Several studies have implicated invertebrates, particularly invertebrates belonging to the collector/gather and shredder functional feeding groups (FFG) in contributing to plant litter decomposition (Merritt and Lawson, 1979; Brinson et al., 1981; Inkley et al., 2008; Tiegs et al., 2013). Clams (Scatolini and Zedler, 1996), snails (Balcombe et al., 2005a; Meyer and Whiles, 2008), amphipods (Meyer and Whiles, 2008), isopods (Balcombe et al., 2005a), leeches (Meyer and Whiles, 2008), and some hemipterans (Brown et al., 1997) have all been found to have lower abundances in created wetlands, with differences attributed to lower dispersal

* Corresponding author. Tel.: +1 304 293 3825.
E-mail addresses: jim.anderson@mail.wvu.edu, wetland@wvu.edu (J.T. Anderson).

http://dx.doi.org/10.1016/j.limno.2015.07.004
0075-9511/

rates. If differences in invertebrate communities exist in mitigated wetlands, they could affect wetland function through slower litter decomposition.

Microbial colonization also contributes to litter decomposition through bacterial (Kuehn et al., 2000; Jackson and Vallaire, 2007) and fungal processes (Gessner and Chauvet, 1994; Findlay et al., 2002). This study focused on fungal biomass, which is easy to quantify by measuring ergosterol (a sterol present in fungal cell membranes and absent from animal and plant cells) in leaf litter (Newell et al., 1988; Kuehn et al., 2000). Fungal colonization and decomposition begins after senescence, but while plant litter is still standing (Facelli and Pickett, 1991; Kuehn et al., 2000; Chimney and Pietro, 2006) and continues after submergence (Bauer et al., 2003; Kuehn et al., 2011; Zhang et al., 2014). Kuehn et al. (2011) found that 22% of leaf Carbon from *Typha angustifolia* was assimilated into fungal biomass. It is largely unknown how microbial communities in mitigated wetlands compare to those in natural communities.

Mesh litter bags have long been used to assess both decomposition rates and the role of macroinvertebrates on decomposition (Witkamp and Olson, 1963; Merritt and Lawson, 1979; Stewart and Davies, 1989; Vasilas et al., 2013). In this study, we used two sizes of mesh for the litter bags to create a continuum of invertebrates by size and study the role of invertebrate biomass on decomposition. We hypothesize that decomposition rates are similar between mitigated and reference wetlands and that both invertebrates and fungi influence decomposition rates. Our primary objective was to compare plant litter decomposition among wetland types (mitigated vs reference wetlands) in the Mid-Atlantic Highlands, USA. Our second objective was to determine if invertebrate biomass and fungal biomass was correlated with decomposition rate or wetland type.

2. Materials and methods

2.1. Study area

Leaf breakdown rates were measured at three mitigated and three reference wetlands located in the Mid-Atlantic Highlands region of West Virginia, USA. The three mitigated wetlands (Leading Creek, Sugar Creek, Hazelton) were constructed by the West Virginia Division of Highways (WVDOH) to compensate for wetland losses associated with the Corridor H and Mon-Fayette Expressway system projects (Table 1). The three reference wetlands (Meadowville, Upper Deckers Creek, and Bruceton Mills) were chosen based on the following factors: their proximity to mitigated sites (to minimize differences in climatic events); their similarity in elevation and wetland classification; and their relative degree of disturbance (minimal disturbance on their edge and no disturbance in the interior). Both mitigated and reference wetlands had some level of disturbance on their edge in the form of roads, grazing, or cultivated land. All wetlands were associated with streams and received water from overbank flooding, with hillslope runoff and groundwater as additional sources. All wetlands also had a mixture of flooded and exposed conditions for the majority of the year, with brief periods of deeper flooding, but mitigated wetlands tended to have a higher percentage of open water and ponded areas than reference sites. Reference sites tended to have more scrub–shrub areas than the mitigated sites, and Leading Creek, Meadowville, and Upper Deckers Creek had portions of scrub–shrub and young forest. Although water depth, temperatures, and pH varied throughout the year, there were no statistical differences between wetland types ($p \geq 0.2$; Gingerich, 2010).

2.2. Decomposition (litterbag) procedures

We collected (September–October 2007) three litter species (common rush [*Juncus effusus* L.], brookside alder [*Alnus serrulata* (Ait.) Willd.], and reed canary grass [*Phalaris arundinacea* L.]) based on common dominant species at mitigated and reference sites in West Virginia (Balcombe et al., 2005b; Veselka IV, 2008) and used the litter bag method to compute litter decomposition rates (Benfield, 1996). Not all wetlands studied had the same dominant species or ratio of dominant species; however, litter mixes can have non-additive decomposition rates compared to single species (Gartner and Cardon, 2004). In an attempt to more closely mimic the natural systems in our study wetlands and the most common species across wetlands (Balcombe et al., 2005b; Veselka IV, 2008), 20 g of litter was created from a mix of 3:2:1 reed canary grass (10 g), common rush (6.6 g), and brookside alder (3.3 g).

To minimize variability, reed canary grass and common rush leaves and stems were clipped and collected as they senesced but while still standing (Marsh et al., 2000; Bedford, 2005). We collected brookside alder leaves with a STIHL model SH 85 D Shredder Vacuum/Blower (STIHL Incorporated, Virginia Beach, VI) reversed to suck leaves into the tube. Brookside alder leaves that were not

Table 1
List of three mitigated and three reference wetland study sites in West Virginia, including site name, year created, county and closest town, size (ha), wetland classifications, and differences in mean air temp, water temp, water depth, hydroperiod, and pH. Environmental measurements were taken every two weeks in each wetland from December 2007 to December 2009. Standard error (S.E.) is presented in parentheses under each mean. Analysis of correlations between environmental factors and decomposition rates can be found in Gingerich et al. (2014).

Site name (year created)	County and closest town	Size (ha)	Wetland classifications* at Site	Air Temp. (°C)	Water Temp. (°C)	Water depth (cm)	Hydroperiod†	pH
Mitigated								
Leading Creek (1995)	Montrose, Randolph Co.	17	AB, EP, **SS**‡	9.56 (0.62)	7.16 (0.40)	8.05 (1.30)	0.49 (0.05)	6.30 (0.07)
Sugar Creek (1995)	Meadowville, Barbour Co.	11	**EP**, SS	10.92 (0.58)	7.88 (0.41)	6.34 (1.08)	0.43 (0.05)	6.09 (0.06)
Hazelton (2006)	Hazelton, Preston Co.	2.7	UB, AB, **EP**	6.39 (0.85)	7.70 (0.90)	2.88 (0.69)	0.20 (0.04)	6.93 (0.08)
Reference								
Meadowville	Meadowville, Barbour Co.	11.7	EP, **SS**	10.51 (0.63)	7.83 (0.50)	2.23 (0.51)	0.28 (0.05)	6.37 (0.09)
Upper Deckers Creek	Masontown, Preston Co.	2.1	AB, **SS**, F	10.10 (0.68)	5.37 (0.46)	8.16 (1.32)	0.35 (0.05)	6.21 (0.02)
Bruceton Mills	Bruceton Mills, Preston Co.	1.4	**EP**, SS	8.07 (0.66)	7.06 (0.64)	3.25 (0.41)	0.50 (0.05)	6.55 (0.05)

* Palustrine: unconsolidated bottom = UB, aquatic bed = AB, emergent persistent = EP, scrub–shrub = SS, forested = F (Cowardin et al., 1979).
† Measured as proportion of days inundated.
‡ Bold text indicates dominant classifications.