UNIVERSIDADE FEDERAL DO RIO GRANDE DO SUL
INSTITUTO DE BIOCIÊNCIAS
PROGRAMA DE PÓS-GRADUAÇÃO EM ECOLOGIA

DECOMPOSIÇÃO FOLIAR E MACROINVERTEBRADOS AQUÁTICOS EM UM SISTEMA LÓTICO NEOTROPICAL

André Frainer Barbosa

PORTO ALEGRE, ABRIL DE 2008

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Dissertação apresentada ao Programa de Pós-Graduação em Ecologia, do Instituto de Biocências da Universidade Federal do Rio Grande do Sul, como parte dos requisitos para obtenção do título de Mestre em Ecologia

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PORTO ALEGRE, ABRIL DE 2008

À vó Carmen Laizola Frainer
À vó Onella Fantinel Barbosa

“*A PRÁTICA É O CAMINHO DA VERDADE*”

KARL MARX

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RESUMO

O aporte de material foliar em sistemas lóticos de pequeno e médio porte é responsável pela maior parte da energia e matéria que entra nesses ambientes. Esse material passa por um processo de decomposição natural, o qual é afetado por diversos fatores, como pela presença de microorganismos decompositores, macroinvertebrados fragmentadores ou devido às condições físicas e químicas da água. Porém, enquanto que em regiões temperadas os organismos retalhadores são reconhecidamente importantes para o processamento do material foliar, em regões tropicais e sub-tropicais esses organismos parecem não participar tão ativamente da decomposição foliar. Nesse trabalho, estudamos áreas influenciadas por diferentes graus de antropização, objetivando verificar a influência de atividades agrícolas na decomposição foliar, bem como a relação dos invertebrados retalhadores neste processo de decomposição. Procuramos, também, avaliar a validade do uso de índices biológicos já existentes para países da América Latina na detecção de impacto ambiental na bacia hidrográfica em estudo. Os experimentos de decomposição foram realizados na Bacia Hidrográfica Lajeado Grande, noroeste do Rio Grande do Sul. Nessa bacia hidrográfica, oito sítios amostrais foram selecionados e tiveram suas características físicas e químicas mensuradas. Em cada sítio, bolsas-de-folhiço contendo 4g de *Ocotea puberula* (Lauraceae) foram expostas e retiradas mensalmente até o fim da decomposição foliar. Verificou-se que constituintes físicos dos trechos amostrados, como frações da fácie areia, influenciam a decomposição foliar negativamente ($p = 0,0022$), enquanto que o gradiente de uso da terra analisado não demonstrou afetar a decomposição foliar ($p = 0,3328$). Por outro lado, a vazão, característica desse sistema de corredeiras, mostrou-se responsável pela ação de disruptura do material foliar com maior intensidade do que a ação de macroinvertebrados bentônicos. Os valores de densidade, abundância e riqueza dos macroinvertebrados fragmentadores foram afetados negativamente pela vazão da água. Esses organismos apresentaram, assim, maior abundância e densidade nos tratamentos com menor vazão d'água e com a decomposição foliar mais lenta ($p = 0,0019$). Estes resultados reforçam outros estudos realizados em sistemas neotropicais que atestam que a decomposição foliar nessa região sofre maior influência das variáveis físicas do que da atividade dos macroinvertebrados aquáticos, ao menos no que se refere aos invertebrados retalhadores. O uso dos organismos relacionados ao processo de decomposição foliar, na aplicabilidade de índices biológicos, também necessita de estudos mais específicos, uma vez que os índices existentes para a América Latina contemplam poucas regiões do continente.

ABSTRACT

Litter fall in small and medium sized lotic systems is responsible for most of the energy and matter input into these ecosystems. After the input of this material, it will be decomposed by several factors, such as some physical water properties or the feeding behavior of shredders and the presence of decomposer microorganisms. In temperate regions, invertebrate shredders are well recognized to influence leaf decomposition. In tropical and sub-tropical regions, however, these organisms seem not to present such important function. In this work we studied some areas affected by human disturbance aiming to verify the importance of land-use gradients to leaf decomposition as well as the role of shredders on this ecological process. We also analyzed the applicability of rapid biological assessments on the studied catchment using indexes already proposed for other Latin America regions. The study was conducted on Lajeado Grande Basin, southern Brazil. There, eight sites were selected and had some physical and chemical properties measured. In each site, four riffles were chosen which were considered as replicates. In these riffles, litter-bags made of 4g *Ocotea puberula* (Lauraceae) leaves were exposed and retrieved monthly until the end of the decomposition. We verified that some physical constituents of the sites, as percentage of sand, influenced leaf decomposition negatively ($p = 0.0022$), while the analyzed land-use gradient did not affect leaf decomposition ($p = 0.3328$). On the other hand, water discharge - another physical characteristic from these systems - was responsible for the leaf-litter breakdown with more intensity than shredders were. Shredders' density and abundance were affected negatively by water discharge and presented greater numbers at the slow breakdown riffles ($p = 0.0019$). These results are in agreement with studies from other neotropical systems, which suggest that leaf decomposition in this region is more affected by physical variables than by biological activity, at least for macroinvertebrate shredders. The use of the organisms that colonize leaf material to the applicability of biological assessments also needs more studies, once the existing indexes for their use in Latin America are appropriate for just a few regions.

APRESENTAÇÃO

O estudo apresentado nesta dissertação teve início com a realização do projeto de análise ambiental desenvolvido pela Fundação Estadual de Proteção ao Meio Ambiente do Estado do Rio Grande do Sul (FEPAM-RS), juntamente com a Secretaria Estadual do Meio-Ambiente (SEMA), intitulado Programa Nacional de Monitoramento Ambiental, fase 2 (PNMA-II) do Ministério do Meio Ambiente. Os resultados preliminares do projeto foram compilados no Manual Técnico [1] da SEMA. Nesse estudo, aplicou-se o método de decomposição de matéria orgânica, através do cálculo da sua taxa de decomposição, junto com a análise da fauna associada ao material foliar exposto, como alternativa para a detecção de fragilidades ambientais.

Ecossistemas aquáticos: o compartimento detrital

Estudos sobre a decomposição foliar em águas continentais remontam à década de 70, quando diversos pesquisadores perceberam a importância da grande quantidade de material orgânico proveniente da vegetação ripária em sistemas lóticos (e.g., Kaushik & Hynes, 1971, Fisher & Likens, 1973; Petersen & Cummins, 1974), e também lênticos (e.g., Smock & Stoneburner, 1980; Oerti, 1993), para a fauna aquática. Esse material orgânico formado em sua maioria por folhas, mas também por outras partes vegetais, como galhos, troncos, e diásporos (frutos e sementes), compõe grandemente a base energética dos trechos de rios de pequena ordem em regiões temperadas (Webster *et al.*, 1999).

O processo de decomposição foliar em sistemas lóticos inicia-se com o aporte de folhas em ambientes aquáticos oriundos da vegetação ripária. Com a entrada do material foliar na água, ocorre uma disruptura mecânica da folha juntamente com a dissolução de diversos nutrientes presentes na mesma. Nessa primeira parte do processo pode ocorrer a perda de cerca de 20% da biomassa original (Petersen & Cummins, 1974; Suberkropp *et al.*, 1976) em um período aproximado de 24 horas. Em seguida, há a colonização de fungos hyphomicetes sobre esse material (Gessner *et al.*, 2007), responsáveis pelo enriquecimento

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[1] Rodrigues, G. G. ; Barbosa, A. F. 2006. Concepção Ecossistêmica para Avaliação da Qualidade da Água na Bacia do Lajeado Grande. **In:** Niro Afonso Pieper. (Org.). *Controle da Contaminação Ambiental Decorrente da Suinocultura no Rio Grande do Sul: MANUAL TÉCNICO*. Secretaria Estadual do Meio Ambiente. 1ed. Porto Alegre, v. 2, p. 85-96.

das folhas com nutrientes retirados da água (Suberkrop & Klug, 1974; Bärlocher & Kendrick, 1974; Bärlocher, 1985; Suberkropp & Chauvet, 1995), ao mesmo tempo em que iniciam a ação biológica de decomposição (Gessner & Chauvet, 1994; Gessner *et al.*, 1999). Neste mesmo período ocorre também a formação do biofilme bacteriano, o qual pode se constituir em um importante componente da fase inicial de decomposição foliar (Hieber & Gessner, 2002; Buesing & Gessner, 2005). Após essa etapa de colonização por fungos e formação de biofilme bacteriano, denominada também de fase de condicionamento, há a colonização por macroinvertebrados bentônicos. Neste último grupo, a presença de organismos retalhadores irá intensificar a decomposição foliar, sendo responsável pela degradação do restante da biomassa foliar (Webster & Benfield, 1986).

A entrada de material foliar, geralmente, está associada a trechos superiores de rios (até terceira ou quarta ordem), onde o sombreamento realizado pela vegetação ripária impede, ou diminui consideravelmente, a produção autotrófica realizada pelo fitoplâncton e por algas perifíticas. Por esse motivo, a produção heterotrófica torna-se fonte de matéria e energia predominante para os sistemas lóticos de ordens inicias. Esse material alóctone se acumula, principalmente, em zonas de remanso dos cursos d'água de ordens iniciais ou nos trechos intermediários, ou de transição, de rios. Em regiões temperadas, no entanto, a maior velocidade de decomposição está relacionada aos trechos de corredeira, nos quais há maior atividade microbiológica (Ferreira & Graça, 2006) e, dentre os macroinvertebrados, há maior presença do tipo funcional denominado retalhador (Cummins *et al.*, 1980; Kobayashi & Kagaya, 2005).

Dessa forma, a disrupção do material foliar, ou material orgânico particulado grosso (CPOM, do inglês *coarse particulate organic matter*), em partículas menores, ditas material orgânico particulado fino (FPOM, do inglês *fine particulate organic matter*), une as cabeceiras de rios com os trechos inferiores a partir da cadeia de detritos iniciada com a deposição de material, oriundo, principalmente, da vegetação ripária (Vannote *et al.*, 1980; Webster *et al.*, 1999). Vinculados à cadeia de detritos, a fauna de invertebrados aquáticos - formada por organismos filtradores, coletores de partículas finas (denominados unicamente *coletores*), raspadores de substrato (denominados unicamente *raspadores*) e predadores - presentes em trechos intermediários e inferiores dos cursos d'água se beneficiarão e, em última instância, dependerão do processo de decomposição foliar (Graça, 2001). Do mesmo modo, a megafauna, ou macroconsumidores, - que compreende a íctiofauna, anurofauna, e crustáceos - que se sucede nessa trama ecológica estará diretamente relacionada à cadeia de detritos, iniciada nas cabeceiras dos cursos d'água (Vilella *et al*., 2005). Ressalta-se ainda que, apesar da disruptura do material foliar ser melhor compreendida atualmente, a decomposição de outras partes vegetais menos estudadas, como caules, ramos, e diásporos (frutos e sementes), também apresenta importância para a cadeia de detritos em sistemas lóticos (Díez *et al*., 2002; Harmon *et al*., 1986; Webster & Benfield, 1986).

Devido aos diversos compartimentos relacionados à decomposição foliar, essa dissertação apresenta, primeiramente, uma introdução aos principais componentes atuantes no processo de decomposição foliar. Após a introdução, seguem-se dois capítulos estruturados na forma de artigos científicos, os quais discutem, cada um, parte dos compartimentos apresentados na introdução e que, em linhas gerais, são os principais atuantes na decomposição foliar.

O primeiro capítulo trata das características de mesoescala que afetam esse processo ecológico. Inserem-se, nessa escala, os modos de uso da terra e as conseqüências desses usos em escalas de menor abrangência (mesoescala), como a deposição de sedimentos devido à supressão da mata ciliar. O segundo capítulo discute a influência dos macroinvertebrados do tipo funcional retalhador no processo de decomposição foliar e os motivos pelo quais esses organismos apresentam baixa importância em sistemas lóticos na região neotropical em relação às regiões temperadas.

O terceiro capítulo, também estruturado como artigo científico, utiliza os valores de abundância e riqueza de macroinvertebrados aquáticos presentes no material foliar para determinar valores de qualidade da água, com base em índices biológicos pré-elaborados para duas regiões distintas da América do Sul. Esse capítulo foi uma primeira tentativa de estabelecer valores de qualidade de água para a região noroeste do Rio Grande do Sul e de relacionar os macroinvertebrados presentes nessa região com valores químicos e físicos da água, a partir de uma proposta dos órgãos ambientais do Estado do Rio Grande do Sul.

Os três artigos serão submetidos a diferentes revistas e, por esse motivo, estão aqui formatados de maneira a manter a coerência estrutural da dissertação. Após o terceiro capítulo, apresentam-se as considerações finais que pretendem vincular (i) os distúrbios físicos oriundos do uso da terra para a decomposição foliar com as (ii) conseqüências dessas alterações na estrutura e função da fauna aquática o que ocasiona, assim, uma (iii) diminuição nos indicadores biológicos de qualidade da água. Estabelecer essa relação significa entender o quanto os processos de decomposição foliar são afetado por distúrbios antrópicos e, também, compreender a validade do uso desse processo ecológico na detecção de alterações ambientais.

Introdução

Escalas de abrangência: variáveis abióticas da água (mesoescala) e uso da terra (macroescala)

Diversos fatores atuam sobre o processo ecológicos de decomposição e influenciam na velocidade com que o material foliar se degrada em sistemas lóticos (Leroy & Marks, 2006). Entre os fatores físicos, inclui-se a velocidade da água como o mais importante, seguido de efeitos de deposição de sedimentos (i.e., fácies de areia, silte e argila). Os picos de velocidade da água podem acelerar o processo de decomposição consideravelmente, ao mesmo tempo em que interferem na estrutura - composição, distribuição e abundância - da comunidade bentônica (Arrington & Winemiller, 2006). A deposição de sedimentos finos, por sua vez, atenua a intensidade desse processo ecológico em diferentes níveis de velocidade (Niyogi *et al.* 2003; Sponseller & Benfield, 2001). Certos organismos com tipos funcionais específicos, que atuam na decomposição (i.e., retalhadores), apresentam maior perda energética em microhábitats onde haja alta velocidade d'água e formação predominante de areia, do que em áreas com presença de cascalhos de maior diâmetro e de refúgios contra a correnteza (Franken *et al.*, 2006).

A maior incidência dos fatores físicos de correnteza e sedimentação pode estar associada ao uso da terra no entorno dos cursos d'água, seja ele em função de atividades agrícolas, seja devido à urbanização que intensifica consideravelmente essas variáveis. Através das práticas agrícolas, o incremento de nutrientes como Nitrogênio e Fósforo pode acelerar a decomposição foliar através do enriquecimento do material vegetal e, com isso, proporcionar uma maior, ou mais rápida, colonização por fungos e/ou macroinvertebrados aquáticos (Suberkropp & Chauvet, 1995; Robinson & Gessner, 2000; Gulis *et al.*, 2006; Bergfur *et al.*, 2007). Por outro lado, o incremento de nutrientes pode, até mesmo, diminuir a velocidade e/ou intensidade da decomposição por meio do impacto negativo sobre a fauna bentônica (Baldy *et al.*, 2007). Alguns estudos apontam, porém, para a inexistência de efeitos da adição de nutrientes sobre o processo de decomposição. Raviraja *et al.* (1998) verificaram que, apesar da alta poluição orgânica em um rio localizado na Índia setentrional, e do efeito negativo dos poluentes sobre os fungos hyphomycetes, a taxa de decomposição não diferiu de outro sistema aquático lótico sem a presença de poluentes, na mesma região.

Outro interferente que apresenta impacto negativo na fauna aquática e, por esse motivo, no processo ecológico de decomposição subseqüente, é a aplicação de inseticidas em áreas adjacentes aos cursos d'água (Cuffney *et al*., 1990; Wallace *et al*., 1995). Em áreas urbanas e industriais, o acúmulo de metais pesados apresenta-se como fator de grande impacto no retardamento do processo ecológico (Niyogi *et al*., 2001; Carlisle & Clements, 2005; Duarte *et al*., 2008). A presença de focos pontuais de poluição (despejo de esgotos) tende a enriquecer organicamente os sistemas aquáticos, de forma a ocasionar os mesmos efeitos que o incremento de nutrientes em sistemas agrícolas exerce.

Ainda, o uso inadequado da terra, como o desmatamento da vegetação ciliar – entre os mais comuns - irá diminuir o aporte de matéria orgânica para o sistema aquático (Niyogi *et al*., 2004; Roberts *et al*., 2007), além de alterar a fonte de energia e matéria de um sistema heterotrófico para autotrófico, devido à ausência de sombreamento exercido pela vegetação marginal (Vannote *et al*., 1980). Áreas de desmatamento, mesmo quando realizadas dentro do melhor critério possível, ou seja, sem corte de vegetação entre 30 e 100 metros de distância da calha do rio, apresentam fortes impactos sobre a fauna aquática e, assim, sobre os processos ecológicos de produção, consumo e decomposição foliar (Kreutzweiser *et al*., 2008). Ao mesmo tempo em que a retirada da mata ciliar tende a modificar a fonte de energia e matéria dentro do sistema aquático - de heterotrófico para autotrófico -, o aumento no aporte de sedimentos devido à erosão e/ou lixiviação decorrentes do desmatamento também provocará mudanças estruturais no curso d'água (Santmire & Leff, 2007). Essas mudanças tendem a diminuir os valores obtidos em estudos de decomposição foliar através do soterramento do material vegetal (Sponseler & Benfield, 2001). Devido à decomposição foliar ser afetada por diversas alterações ambientais relacionadas ao uso da terra, esse processo foi proposto para ser utilizado na indicação ambiental em cursos d'água (Gessner & Chauvet, 2002; Royer & Minshall, 2003).

Macroinvertebrados aquáticos

Estudos em regiões temperadas demonstram que a presença de macroinvertebrados alteram a velocidade de decomposição do material foliar (e.g., Webster *et al*., 1999) e, com isso, agem na ciclagem de nutrientes dos ecossistemas aquáticos. Na região tropical e sub-tropical os retalhadores *strictu sensu* (e.g., Plecoptera:Gripopterygidae) e também macroconsumidores (e.g., Crustacea:Trichodactilydae), se destacam (Rosemond *et al*., 1998;

Vilella *et al*., 2005) em oposição a outros *taxa* descritos para as regiões temperadas (e.g., Crustacea:Gammaridae). Tem-se, contudo, pouca evidência do papel desempenhado por esses organismos na decomposição foliar (Mathuriau & Chauvet, 2002; Moretti *et al*., 2007), em especial na região neotropical.

Apesar do aporte de material foliar ser em grande quantidade nos sistemas aquáticos tropicais e sub-tropicais, durante o ano todo (Nin *et al*., no prelo), dentre o grupo de macroinvertebrados aquáticos, encontram-se poucos insetos verdadeiramente retalhadores nessas regiões (Wantzen & Wagner, 2006). Um dos motivos dessa diferença em relação às regiões temperadas aparece devido à grande quantidade de predadores presentes em sistemas netropicais, os quais influenciam na dieta dos insetos, e fazem, assim, com que os organismos detritívoros apresentem comportamento mais generalista para obtenção de alimentos (Covich, 1988; Wantzen & Wagner, 2006). Além disso, insetos aquáticos tropicais e sub-tropicais apresentam ciclos de vida pronunciadamente mais curtos do que insetos de regiões temperadas. Essa característica, juntamente com a maior predação, tende a levar os insetos a possuirem uma dieta onívora, a fim de acelerarem seu desenvolvimento em insetos adultos.

Alguns autores (e.g., Graça, 2001) atestam que a natureza morfológica e química do material foliar de espécies de regiões neotropicais, por apresentarem maior diversidade devido às pressões ecológicas, possui, também, maior quantidade de compostos secundários utilizados na defesa contra a herbivoria (Feeny, 1970; Grime *et al*., 1996; Poorte *et al*., 2004; Wantzen *et al*., 2002; Santiago, 2007). Esses compostos, junto com características morfológicas (e.g., tricomas) poderiam continuar atuando depois da entrada do material nos sistemas aquáticos, de forma a impedir ou, ao menos, prejudicar a colonização das folhas por macroinvertebrados aquáticos.

Outro ponto ainda, seriam as fortes chuvas tropicais que podem carrear o substrato dos sistemas lóticos e suprimir a fonte de recursos alóctone que serviria para a alimentação dos retalhadores (Winterbourn *et al*., 1981). Por este aspecto, organismos detritívoros não podem depender de um único recurso natural, que, apesar de ter aporte constante em regiões tropicais e sub-tropicais, é instável dentro do sistema aquático. Assim, acredita-se que a menor abundância e riqueza de retalhadores verificados em regiões neotropicais (Wantzen & Wagner, 2001) façam com que a decomposição foliar seja influenciada mais diretamente pela ação de microorganismos como fungos hyphomicetes, ou mesmo pela ação física da velocidade da água.

Uso de macroinvertebrados no biomonitoramento

Os macroinvertebrados bentônicos são uma categoria ecológica amplamente utilizada para a detecção de alterações ambientais (Rosenberg & Resh, 1993) devido ao fato de terem suas características estruturais (e.g., ocorrência e abundância) e funcionais (e.g., presença de grupos alimentares específicos) influenciadas pelas condições físicas, químicas e geoquímicas do ambiente (Robinson & Minshall, 1986; Lake, 2000; Gafner & Robinson, 2007). Além disso, esse grupo apresenta larga distribuição espacial e são encontrados em alta abundância em ambientes lóticos e lênticos, o que faz deles organismos propícios para a coleta e estudo em praticamente qualquer sistema aquático continental (Rosenberg & Resh, 1993).

A ampla distribuição e alta abundância de macroinvertebrados aquáticos, aliado ao fato de eles serem identificados com razoável facilidade e de serem coletados e armazenados com equipamentos simples, fez com que diversos índices e métricas para a detecção de alterações ambientais pudessem ser elaborados a partir da ocorrência desse grupo de organismos (Resh & Jackson, 1993). Em países como Inglaterra, Austrália e Estados Unidos, os macroinvetebrados bentônicos são utilizados juntamente com análises químicas e físicas da água para o monitoramento, análise e indicação da qualidade da água em sistemas naturais ou antropicamente alterados (e.g., Armitage *et al*., 1983). Na América Latina e no Brasil, em especial, esses índices estão sendo recém estudados e aplicados (Barbosa *et al*., 2001; Cota *et al*., 2001). Por esse motivo, estudos sobre macroinvertebrados aquáticos que foquem o biomonitoramento ambiental devem ser realizados no Brasil para, com isso, aumentar tanto o conhecimento taxonômico pertinente, quanto o entendimento sobre a distribuição espacial dos organismos e suas relações com os fatores ambientais.

OBJETIVOS

Objetivo geral

- Compreender a importância dos macroinvertebrados aquáticos na decomposição foliar em um sistema lótico neotropical e compreender se esse processo ecológico pode ser utilizado para verificar efeitos antropogênicos em um gradiente de alterações ambientais

Objetivos específicos

- Verificar se diferentes tipos de usos da terra afetam a decomposição foliar.

- Determinar a importância de variáveis bióticas e abióticas para o processo de decomposição foliar em um sistema neotropical.

- Estabelecer índices biológicos de qualidade da água para os trechos estudados a partir da fauna de macroinvertebrados associada ao folhiço.

CAP 1

SEDIMENT ACCUMULATION AND NOT AGRICULTURAL GRADIENTS INFLUENCES LEAF BREAKDOWN

(Com Gilberto G. Rodrigues)

(Artigo submetido à *Freshwater Biology*, e em preparação para ser re-enviado.)

"*If I had a world of my own, everything would be nonsense. Nothing would be what it is, because everything would be what it isn't*."

Alice in wonderland,
Lewis Carroll

Abstract

Leaf-litter is a major resource that enters streams and is responsible for most of the headwaters energy budget. Its decomposition is influenced by macroinvertebrate shredders and fungi hyphomicetes and by chemical and physical water properties. Several studies have tried to link leaf processing to anthropogenic disturbs. We aimed to verify whether land use types would influence leaf decomposition. Eight stream reaches were selected in southern Brazil and classified into three groups: heavy agriculture, moderate agriculture and light agriculture. A few chemical parameters differed from heavy agriculture to moderate and light agriculture. Our results showed that leaf breakdown rates were lower both in the heavy and in the moderate agriculture sites in comparison to the light agriculture sites ($p = 0.03$). The main cause for slowing down leaf breakdown rate was sediment accumulation in the heavy agriculture sites. Leaf decomposition was positively related to percentage of sand texture ($p = 0.002$) and this texture was negatively related to silt and clay. The sites with lower decomposition rates had more clay and silt on its sediment composition. These findings suggest that leaf decomposition is influenced by agriculture status due to land use alterations, as riparian vegetation removal, which can increase local sedimentation and imply negative consequences to litter breakdown.

Key-words: ecological assessments, leaf breakdown, land use, sediments.

Introduction

The ecological importance of allocthonous material input and its decomposition in canopy covered stream systems has been widely reported (Kaushik & Hynes, 1971; Vannote *et al*., 1980; Gessner *et al*., 1999; Webster *et al*., 1999). According to Gessner and Chauvet (2002), ecosystem processes are very useful when one wants to understand the health (*sensu* Karr, 1999) of an ecosystem and determining litter breakdown is one potentially useful way to achieve this goal (Bunn & Davies, 2000; Gessner & Chauvet, 2002, Royer & Minshal, 2003). However, recent field tests of the idea have led to some apparently conflicting results. While some authors have found a close relation between litter breakdown and measures of human disturbance (Pascoal *et al*., 2003; Niyogi *et al*., 2003, Leroy & Marks, 2006), other authors argued that it is difficult to relate decomposition rates to types of land use (Hagen *et al*., 2006, Sponseller & Benfield, 2001).

Leaf breakdown involves several organisms, primarly aquatic hyphomycete fungi (Suberkropp *et al*., 1983; Gessner & Chauvet, 1994; Baldy *et al*., 1995; Pascoal *et al*., 2005) and benthic macroinvertebrates (Bird & Kaushik, 1992; Linklater, 1995; Rodrigues, 2001; Gonçalves *et al*., 2006). Higher nutrient (N or P) input from agriculture may result in higher microbial activity and higher breakdown rate (Suberkropp & Chauvet, 1995; Robinson & Gessner, 2000; Ferreira *et al*., 2006). Elevated phosphorous concentrations have been related to increased macroinvertebrate density, especially of shredders (Rosemond *et al*., 2001, Paul *et al*., 2006), and may therefore positively affect decomposition rates. Agricultural activities are also usually accompanied by removel of natural riparian vegetation and substitution by crop plantations. This changing on riparian structure results in increased sedimentation on streams (Townsend & Riley, 1999) which, in turn can decrease both microorganisms (Santmire & Leff, 2007) and invertebrates density and richness (Zweig & Rabeni, 2001; Chaves *et al*., 2005) and thus may slow down litter decomposition (Sponseler & Benfield, 2001, Niyogi *et al*., 2003).

In this study we used exponential leaf breakdown rates obtained as *k* values (Petersen & Cummins, 1974) from eight stream sites along a small catchment (˜ 500 ha) in southern Brazil to assess leaf breakdown response to environmental disturbances. The hypothesis was that leaf decomposition rate will be negatively affected by the intensification of agriculture in a region affected by livestock grazing and other intensive farming activities.

Material and Methods

Study site

Lajeado Grande Basin is located in southern Brazil, Rio Grande do Sul State and is in the Broadleaf Subtropical Forest area. The area presented human activities linked specially to pig farming, a well known non-point source water contaminant. Several small catchments (usually around 90 km length each) present this economic activity in Rio Grande do Sul State, and the Lajeado Grande Basin is among those with the highest density of pig growing farms.

The cacthment area of Lajeado Grande Basin is 525.38 km^2. Stream depth is around 0.5 m in the headwaters and 4 m near the river mouth. Together with Rio Grande do Sul State Environmental Agency (FEPAM) eight reaches were chosen to determine leaf breakdown rates. Four of these reaches were located on the main stream channel, the Lajeado Grande, and the other four reaches were located along its main tributary stream, the Lajeado Erval Novo. Stream substrate was composed predominantly of gravel and sand while catchment's soil is predominantly made of fine particles (FEPAM, *unpublished data*).

Stream reaches located in the mean river are called LG after Lajeado Grande and stream reaches located in the mean tributary are called LEN, after Lajeado Erval Novo. All sites are numbered from one to four due to its location in the system: smaller numbers refer to sites close to river mouth and higher numbers refer to sites located close to the headwaters (Figure 1).

Leaf-bags

Ocotea puberula (Rich.) Nees (Lauraceae family) is a very common tree in northern Rio Grande do Sul and occurs all over Brazil and mostly South America region, from the French Guyana to northern Argentina. Leaves were collected from one single tree in order to avoid possible leaves components differences between individuals and between eco-regions. Leaves were then air-dried until necessary and weighed to the nearest 0.01g.

Leaves were weighed and packed into 10 mm mesh size ordinary nylon bags. Each pack contained 4.0 g of *O. puberula* leaves (exact weight of each bag was recorded) and labelled one-by-one with PVC marked tags. A total of 320 leaf-bags was made and divided into eight groups corresponding to the eight study stream sites. For each reach 40 leaf-bags were placed among four sample points fixed on the river banks (10 leaf-bags for each sampling point). To maximize representativeness of within-site heterogeneity, main environmental conditions in each reach (pools and riffles) were sampled.

Four replicates were collected at each site after the following periods of exposure: 15, 30, 60, 90, 120, 150 and 180 days. Litter-bags were gently removed from water using a 200µm mesh handy-net and immediately stored on previously labelled plastic bags. For transport to the laboratory litter-bags were kept in a cooler. Leaf-bags were opened in laboratory, washed over another 200µm mesh size screen with tap water for cleaning leaves from associated material and for retaining the macroinvertebrate associated fauna for further analyses. Leaves were oven-dried at 60 °C for 48h and weighed to the nearest 0.01g.

Four reaches were chosen to have the one-day exposure period. The mean weight obtained between these reaches was used as initial leaf-bags weight. Breakdown rate was measured by the *k* coefficient formula $M_t = M_o e^{-k.t}$ (Petersen & Cummins, 1974), where: M_t = mass at time t; M_o = leaf initial mass; t = time in days; and k = exponential decay coefficient.

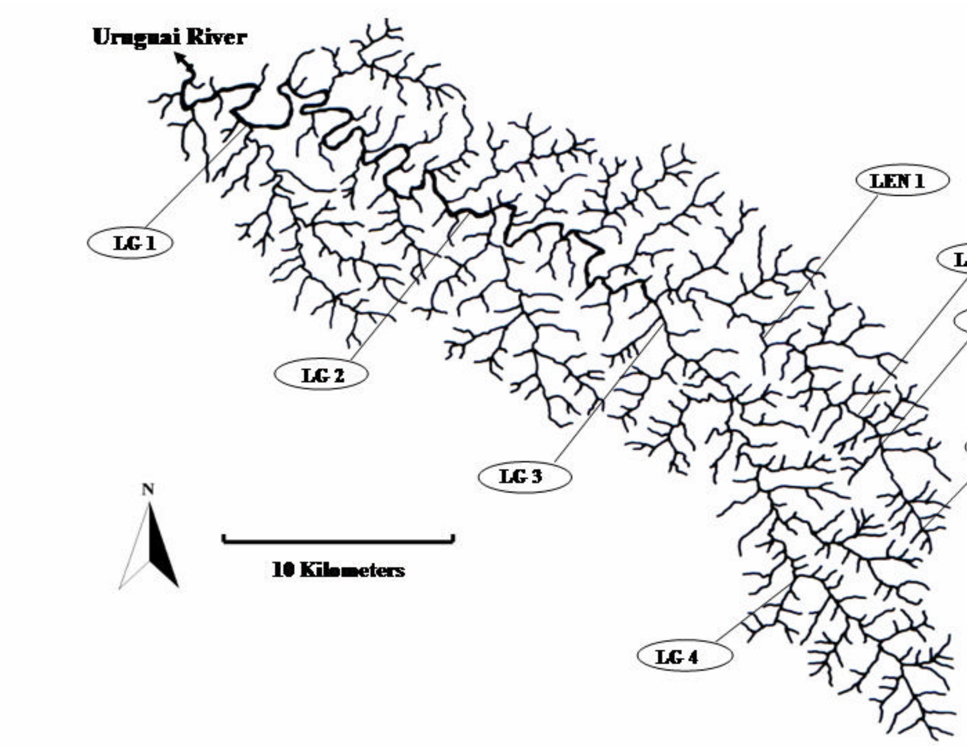

Figure 1. Lajeado Grande catchment in southern Brazil. Sampling sites are shown.

Chemical and physical variables

Total nitrogen, nitrate, phosphorous, dissolved oxygen, specific conductance, pH-value, and water discharge were taken monthly from October 2004 to January 2005.

Environmental variables

During field procedures, surrounding environment features (agricultural intensity, particle composition, riparian vegetation width and riparian canopy cover on the stream) and characteristics of the channel (predominant substrate, maximum depth and channel width) were taken. Agricultural intensity was measured as a modification of Hagen *et al*. (2006) site description: 0 = no agricultural activity next to the stream; 1 = active agriculture next to the stream but no livestock grazing; 2 = agriculture and livestock grazing reaching stream banks. Surface refers to the mean ground composition. 1 = mainly gravel; 2 = gravel and silt; 3 = mainly silt. Riparian vegetation width was measured as: 0 = zero to one meter; 1 = one to five meters; and, 2 = five to ten meters vegetation width. Riparian canopy cover was measured as percentage of stream shelter: 0 = 0%; 1 = 25%; and 2 = 50% (Table 1).

Table 1. Environmental variables used to determine the land use types: percentage of canopy cover, riparian vegetation width and agriculture intensity.

	Low agriculture	Moderate agriculture	Heavy agriculture
Discharge [m^3/s]	0.53 – 10.32	0.32 – 7.34	0.0009 – 0.004
Channel width [meters]	6 – 10	6 – 10	1
Channel max. depth [meters]	0.4 – 2	0.35 – 3	0.2 – 0.5
Surface	1 – 3	2	3
Canopy cover	1 – 2	0 – 2	0 – 2
Riparian vegetation width	2	1	0
Intensive agricultural area	0 – 1	1	2

Sediment process

During the experiment we observed that one of the light agriculture areas, namely LG2, lacked on its riparian vegetation in one of the stream banks. This fact increased local sedimentation over the leaf packs. Also, the presence of a small dam (one meter high) downstream could help sediment accumulation backwards the dam localization. We thought that sedimentation could be another factor influencing breakdown rate rather than only land-use type. We used sediment analysis from the river banks measured by FEPAM (*data unpublished*) from October 2004.

Statistical analyses

Each water chemistry variable was tested by one-way ANOVA. The environmental variables of the land-use classification were analyzed after vector transformation within variables by standardizing by the range. A matrix of Bray-Curtis dissimilarity resemblance measure between the sampling units was obtained. A Principal Component Analysis (PCA) was then executed. These procedures were executed on the MULTIV software (version 2.3.20. Copyright © Pillar, V. D. 2006).

Leaf breakdown was compared among land-use categories after ln(x+1) transformation of remaining mass in an Analysis of Co-Variance test (ANCOVA), followed by *a posteriori* Tukey's honest significant difference (HSD). A linear regression model between *k* values as response variables and the first PCA axis as independent variable was executed. Also an one-way ANOVA between sites with high sedimentation levels and sites with low or no sedimentation was done. We, then performed a backward stepwise multiple regression analysis, in order to find which of the sediment components (sand, gravel, clay and silt) would be influencing the breakdown rates. Sediment levels original data was as percentual substrate cover. We transformed this data into arcsine square root (Zar, 1999). Correlation between sediment components was verified. These tests were performed in the R-Program (version 2.3.0. R Development Core Team, 2006).

Results

Leaf breakdown

Three levels of breakdown rate could be classified among the stream reaches fitting Petersen & Cummins (1974) breakdown velocity classification: fast breakdown rates (LG1, LG3 and LG4 sites, k ˜ 0.010 day^{-1}), slow-moderate breakdown rates (LG 2, LEN 1 and LEN 2 sites, k ˜ 0.004 day^{-1}) and slow breakdown rates (LEN 3 and LEN 4, k ˜ 0.003 day^{-1}). Leaf breakdown among agriculture land uses types (Table 2), however, did not differ statistically (p = 0.33, Table 3). At LG1, leaves were totally decomposed after 90 days. At LG3 and LG4 there were no more leaves in litter-bags in 120 days, while LG2, LEN1, LEN2, LEN3 and LEN4 decomposed only after 180 days.

Table 2. Breakdown rates expressed in k value (± standard error) from light, moderate and heavy agriculture areas.

Agriculture intensity	Stream reach	Breakdown rate (± SE)	r^2
Low agriculture	LG1	0.0113 (± 0.1377)	0.8756
	LG 2	0.0040 (± 0.08681)	0.8659
Moderate agriculture	LG 3	0.0133 (± 0.06388)	0.9837
	LG 4	0.0096 (± 0.2348)	0.8057
	LEN 1	0.0056 (± 0.08385)	0.944
	LEN 2	0.0047 (± 0.1526)	0.8316
Heavy agriculture	LEN 3	0.0025 (± 0.2071)	0.8876
	LEN 4	0.0033 (± 0.1736)	0.9261

Land use gradient

After conducting the PCA, the first stream classification made had to be changed, as LG3 moved to the moderate agriculture group (Figure 2). The first axis of the PCA corresponds to 74.16% of total variation and the second axis corresponds to 22.13% of total variation. Original descriptors with the highest correlation coefficients in the first axis are agriculture land use (0.96), channel width (-0.91), water discharge (-0.83) and stream depth (-0.82). The second axis main descriptor is channel surface (0.86).

The linear regression between *k* values as response variable and the first PCoA axis as independent variable was not significant (p = 0.13). However, LG2 showed a higher sedimentation level and when this site was removed, we could see a negative relation between these parameters (r^2 = 0.5754, p = 0.029).

Table 3. Analysis of variance for the breakdown rate among heavy, moderate and light agriculture areas.

	Df	S.S.	M.S.	F value	P
Treatments	2	4.0864×10^{-5}	2.0432×10^{-5}	1.3823	0.3328
Residuals	5	7.3905×10^{-5}	1.4781×10^{-5}		

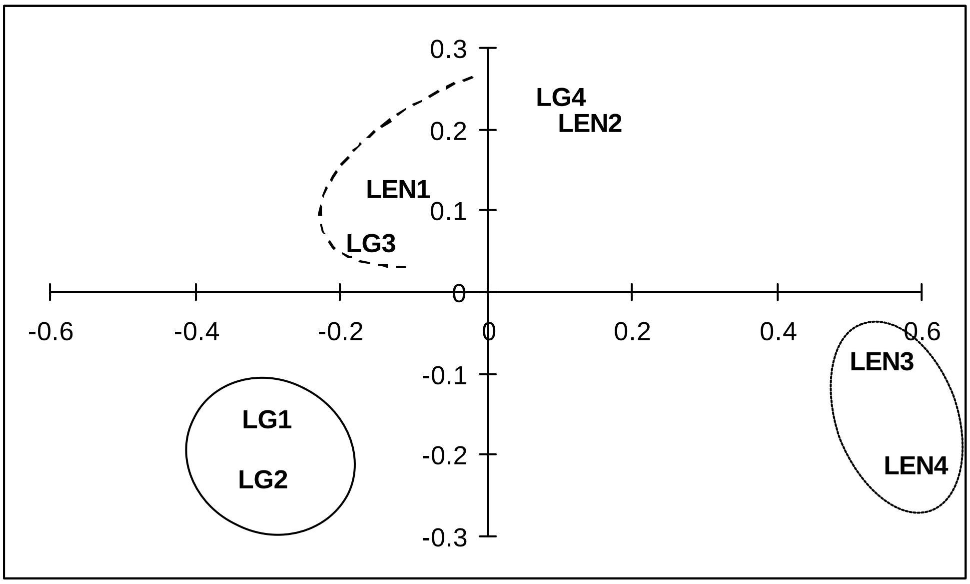

Figure 2. PCA for the environmental variables shown on table 1. Three groups are distinguished: light agriculture (LG1 and LG2); moderate agriculture (LG3, LG4, LEN1 and LEN2); and, heavy agriculture (LEN3 and LEN4).

Chemical and physical water composition

Chemical analyses detected differences in five of the measured variables (Table 4). Dissolved oxygen (DO) and pH-value were slower in high agriculture than in the other two areas ($p < 0.001$). DO mean level was of 5.55 mg/L and pH-value was 6.85 at high agriculture sites. In moderate and low agriculture mean pH level ranged from 7.25 to 7.40 and DO mean concentration level ranged from 7.2 to 7.27 mg/L. Nitrate concentrations were higher in high agriculture (1.45 mg/L) than in moderate (0.1 mg/L) and light agriculture (0.17 mg/L) areas ($p = 0.001$). Phosphorous, however, presented a higher concentration on both light (0.065 mg/L) and moderate (0.1 mg/L) agriculture, while high agriculture presented 0.03 mg/L ($p = 0.02$). Temperature, although not statistically different, was slightly higher in the heavy agriculture areas. Turbidity, on the other hand, showed a slight increase in the low agricultural areas. Water discharge was more pronounced at both light and moderate agriculture areas ($p = 0.01$).

Table 4. Chemical and physical variables (mean values ± standard deviance) for light, moderate and heavy agriculture areas. Small letters beside the values indicate significant difference ($p < 0.05$) between treatments.

Variables	Light agriculture	Moderate agriculture	Heavy agriculture	p
Temperature (°C)	20.8 (± 2.6)	20.9 (± 1.8)	21.6 (± 2.8)	0.75
pH	7.2 (± 0.3)**a**	7.4 (± 0.1)**a**	6.8 (± 0.3)**b**	<0.001
Dissolved O_2 (mg/l)	7.3 (± 0.7)**a**	7.2 (± 0.3)**a**	5.5 (± 0.7)**b**	<0.001
Specific conductance (μS/cm)	63.5 (± 11.3)	62.5 (± 12.6)	63.2 (± 14.8)	0.97
Total Phosphorus	0.0 (± 0.0)**a**	0.1 (± 0.0)**a**	0.3 (± 0.0)**b**	0.02
Nitrate (NO_3)	0.2 (± 0.2)**a**	0.1 (± 0.1)**a**	1.4 (± 1.4)**b**	0.001
Total Nitrogen	1.20 (± 0.52)	1.2 (± 0.4)	1.6 (± 1.3)	0.36
Turbidity	13.8 (± 5.05)	12.8 (± 9.9)	10.9 (±8.5)	0.58
Discharge (m^3/s)	3.3 (± 2.8)**a**	2.1 (± 2.1)**a**	0.0 (± 0.0)**b**	0.01

Sediments

LG2, LEN3 and LEN4 presented a high amount of sedimentation, with silt and clay texture presenting between 65 and 70% of the total sample (FEPAM, data unpublished). The other sites had no more than 45% of silt and clay and a minimum of 20% of these two components. High levels of sedimentation led to a negative effect on leaf breakdown ($p = 0.043$) (Table 5). Backwards stepwise multiple regression data showed that sand texture is the best predictor of the model ($R^2 = 0.7812$, $p = 0.002225$; Table 6; Figure 3). Sand is negatively related to silt and clay (Table 7).

Table 5. Analysis of variance between sediment accumulated sites.

	Df	Sum Sq	Mean Sq	F value	P
Sediment	1	5.9502×10^{-5}	5.9502×10^{-5}	6.4598	0.04398
Residuals	6	5.5267×10^{-5}	9.2110×10^{-6}		

Table 6. Sediment backward stepwise multiple regression analysis. Leaf breakdown rates (as *k* values) are the responsible variables and sediment texture (sand, gravel, clay and silt) are the independent variables. Sediment original data (FEPAM, *unpublished data*) was used after arcsine square root transformation. Only the last significant term is shown (95% CI, n = 8).

	Estimate	Std. Error	t value	P
Intercept	-0.006442	0.002683	-2.401	0.05320
Sand	0.019721	0.003868	5.098	0.00222

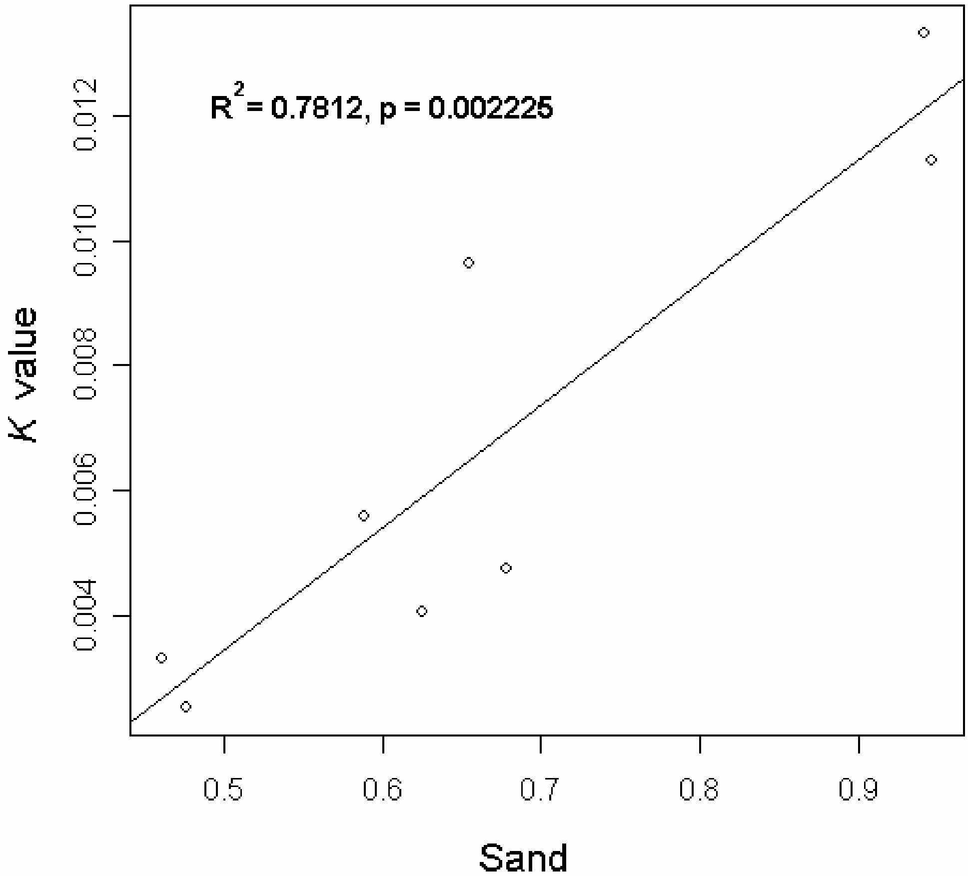

Figure 3. Sediment backward stepwise multiple regression significant term. Independent variables original data were used after arcsine square root transformation.

Table 7. Correlation between sediment components. In bold are shown the components which presented statistically significant correlations ($p < 0.05$).

	Gravel	Sand	Silt	Clay
Gravel	1			
Sand	0.2091	1		
Silt	-0.60719	**-0.86102**	1	
Clay	-0.40288	**-0.81568**	0.68233	1

Discussion

In this work, we could not find a clear relationship between leaf breakdown and land uses. However, we could make some speculations. First of all, the categories made after landscape evaluation were not efficient as indicators of the ecological process. Royer and Minshall (2003) attested that the use of leaf breakdown rates for biological assessments should not be used for describing different scales than the reach scale where the leaves are being exposed. Hawkins and Vinson (2000) also described the problem of weakness of land-use classifications in bio-assessments to the invertebrates' independent and continuous variation over environmental gradients. As invertebrates are important feeders on leaves (Kaushik & Hynes, 1971; Webster & Benfield, 1986), it is supposed that this non-detectable variation when streams were *a priori* classified could have affected the non correspondence between land use and breakdown rate. According to Whittier *et al*. (2007), even in choosing reference sites large mistakes can be done, as reference sites not always follow researches "expectance".

The studied catchment presents an almost homogeneous chemical composition of elements. Data collected by Rio Grande do Sul State Environmental Agency about metal concentration on the substrate of the same sites studied here, detected only a small increase of these compounds towards the river mouth but no real difference between sites (FEPAM, *unpublished data*). Chemical composition differed for a few parameters from the heavy agriculture to the moderate and light agriculture (Table 3). Although no significant difference was observed for the breakdown rates, leaves in the heavy agricultural land-use type presented a slower decomposition rate than the other agriculture areas (Table 2). Greater amounts of phosphorous were detected in light and moderate agriculture in comparison to heavy agriculture, which can have positively affected the leaf breakdown of these sites (Suberkropp & Chauvet, 1995). Nitrate, on the other hand, was higher on heavy agriculture than on moderate and light agriculture areas which attests the heavy agriculture sites to be more influenced by agriculture itself and by live-stock grazing. However, several studies have found a positive relation between nutrient enrichment (e.g., nitrogen and phosphorus) and leaf breakdown rates (Robinson & Gessner, 2000; Ferreira *et al*., 2006; Bergfur *et al.*, 2007).

Both low levels of dissolved oxygen and low pH-value presented on the heavy agriculture areas can be explained by the slow discharge observed on these sites. These physical-chemical properties can limit the macroinvertebrates activities, thus reducing their abundance and richness, which could have influenced also shredders occurrence.

Another issue that influenced the non-observable relationship between litter decomposition and environmental gradient is that one of the light agricultural areas (LG2) presents a dam downstream the sampling site and a lack of riparian vegetation. These factors contributed to a higher level of sedimentation which affected negatively the breakdown rates (Table 5). Other works have also reported the negative affects of sedimentation to breakdown rates (Niyogi *et al*., 2003, Sponseller & Benfield, 2001). For Sponseller and Benfield (2001), sediments were the main cause of the non-detectable relation between breakdown rates and land-uses. This negative relation found between sediments and breakdown rates in our work was further examined as to verify which amount and which type of sediment interferes in the decomposition rates. From the four sediment categories, sand was the only significant texture related to breakdown rates and these two variables were positively related (Table 6; Figure 3).

Zweig and Rabeni (2001) found sand related to high sediment deposition and thus responsible for decreasing invertebrates' densities and richness. However, in their work sand was related to silt and other small particles while our work detected higher presence of sand related to lower amounts of silt and clay (Table 7) and, thus, lower sedimentation level. Santmire and Leff (2007) also found larger particles related to higher microbiological colonization and Sponseler and Benfield (2001) reported smaller particles as silt and clay to be responsible for burying the leaves and slowing down its decomposition. From our data set sand was an indirect (through negative relation to silt and clay) but significant describer (by its positive relation with breakdown rates) of the effects of sedimentation on leaves decomposition.

Our linear regression between PCA main axis and k values, after removing LG2, showed a relation between some environmental variables (agriculture status, channel width, water discharge and channel depth) and the breakdown rate. Although the relation detected between PCA first axis and the k values without LG2 would support the idea that agriculture status affects breakdown rates, most properties influencing this linear regression are related to channel morphology and hydrology (channel width and depth and water discharge). Water discharge, which can be affected by channel morphology, is known to affect breakdown rates by physical abrasion (Niyogi *et al*. 2003) and this was the case of the low and moderate agriculture areas. In the studied reaches, polluted areas are mainly located in upper sites of the catchments, which present lower channel width and channel depth and thus, lower water discharge. In these areas there are better soils available for growing vegetables and so there are no upstream sites that present no human disturbance in this bio-region. This catchment's characteristic caused a problem with our data since pollution in this area is negatively related to water discharge (FEPAM, *unpublished data).*

In the last few decades many works have focused on leaf breakdown in running water systems due to its ecological importance especially to streams. As a consequence, assumptions of its ecological process to the water bio-assessments have been constantly reported (Gessner & Chauvet, 2002, Royer & Minshall, 2003). Some works have found a relation between water pollution (e.g., metal contamination or organic input) and leaf breakdown rates (Niyogi *et al*., 2003, Pascoal *et al*., 2003). However, some few authors trying to find a relation between breakdown rates and land use gradient have verified that this relation is to be at least discussible (Hagen *et al*. 2006, Sponseller & Benfield, 2001). Paul *et al*. (2006) found fast breakdown rates in agricultural areas related to nutrient enrichment and fast breakdown rates in urban areas related to storm runoff, while Hagen *et al.* (2006) found low breakdown values in both heavy agriculture and forest sites. We found a relation between sedimentation and leaf breakdown rates but not directly with agricultural areas, as destruction of the riparian zone is not necessarily related to the agriculture practices.

Intensive agricultural fields may change an ecosystem structure through increasing of nutrients (Lowrance *et al.*, 1984; Smith, 1992) and contamination by insecticides and other chemical products that may affect the invertebrates life-cycle (Niyogi *et al*., 2003, Gafner & Robinson, 2007). On the other hand, the absence of a vegetated riparian zone leads to two contradictory effects. First, it allows the increasing of algae biomass around leaves because of more light availability, thus accelerating breakdown (e.g., Franken *et al.*, 2005). In the same way, the lack of a riparian zone will contribute for increasing the organic input from the surrounding land uses (Smith, 1992; Watzin & McIntosh, 1999), when that is the case. As an opposite effect, there is a higher sedimentation level at these areas which can bury leaves and then slow down their breakdown rate (Sponseller & Benfield, 2001; Niyogi *et al*. 2003), or at least, influence negatively invertebrates' occurrence (Niyogi *et al*., 2007) as it was observed in our work. For Niyogi *et al.* (2007) the restoration of the riparian zone could be an effective improvement for stream habitats and invertebrates health.

Royer and Minshal (2003) argued that when leaf breakdown is to be used in biological assessments, care should be taken as this ecological process is more directly related to the reach scale than to catchment scale. Thus, we attest that caution should be taken for using this ecological process in order to find relations between them and human disturbances. However, as the studies have gone further, and as the problems on these studies so far have arisen, one could start integrating them as to carefully understand such magnitude of impacts on this important ecological process.

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Cap 2

Influence of hydrological factors on leaf decomposition in a neotropical stream: consequences for the shredders

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"Não é o mais forte da espécie que sobrevive,
nem o mais inteligente. É aquele indivíduo que
é mais adaptável à mudança".

Charles Darwin

Abstract

Shredders are important components of litter decomposition on temperate regions. However, there is a paucity of this feeding-group in warmer areas, especially in the neotropics. Some reasons have been hypothesized for this scarcity of shredders, as (i) lower leaf quality, (ii) hydrological factors influencing litter decomposition and (iii) high occurrence of predators controlling shredders abundance in tropical and sub-tropical regions. This work aimed to verify the presence of shredders on litter in a neotropical stream in southern Brazil and to check to which extent decomposition rate is related to shredders occurrence. We placed litter-bags in eight stream reaches and retrieved four samples each month, until six months of exposure. Litter-bags were made of 4g of *Ocotea puberula* (Lauraceae) leaves collected nearby the studied catchment. Reaches were chemically similar and were divided to its discharge values, as our experiment presented a high water discharge and a low water discharge treatment. Litter breakdown rate was significantly different between treatments ($p = 0.03$). Macroinvertebrates assemblage values, as total number, density and *taxa* richness did not differ between treatments ($p > 0.05$) but abundance and density of shredders differed significantly ($p = 0.01$; $p = 0.04$, respectively). The highest discharge reaches presented the fastest decomposition but the lowest number of shredders. On the other hand, slow discharge reaches presented low decomposition rates and greater amount of shredders. The rate between predators and other invertebrates was similar between treatments ($p = 0.37$) suggesting that predators were not important in controlling shredders abundance. These findings suggest that shredders are not so important for leaf decomposition in the neotropical region and, in opposition to what happens in temperate areas, shredders need resource availability (which is promoted by lower physical breakdown rates).

Key-words: leaf decomposition, neotropics, shredders, resource, water discharge.

Introduction

Studies conducted in the last thirty years in temperate streams have verified the importance of shredders to the detritus chain on aquatic systems (e.g. Kaushik & Hynes, 1973; Fisher & Likens, 1973; Petersen & Cummins, 1974; Vannote *et al*., 1980). This ecological food-chain starts on litter input from surrounding areas on streams (Webster & Benfield, 1986) and further colonization of the leaves by fungi hyphomycetes (Gessner *et al*., 2007). Following this first step, shredders colonize leaves and feed on these fungi and/or in the leaves' mesophyllo (Graça, 2001; Wantzen & Wagner, 2006).

Although fungi are a very important step on leaf decomposition (Gessner *et al*., 2007), shredders are responsible for most of the leaves' breakdown (e.g., Niyogi *et al*., 2001). Nevertheless, on this detritivorous food-chain, shredders main function is to available the great amount of resource that enters temperate streams for a large variety of other organisms, once the coarse particulate matter will be reduced to fine particulate matter (Graça *et al*., 2001). Thus, collectors and filters located on lower areas of the streams will be directly linked to this step (Allemano *et al*., 2007).

It is claimed that shredders are either scarce in tropical regions (Wantzen & Wagner, 2006), especially on neotropic areas, or they play a minor role on leaf decomposition (Mathuriau & Chauvet, 2002; Moretti *et al*., 2007). Contradictorily, there is a great input of leaves into these systems during all year, which one could expect to be a reason for the presence of shredders. Some authors suggested that leaves in the tropics have more unpalatable substances, as secondary compounds (Lavelle *et al*., 1993; Aerts, 1997), which would inhibit the feeding of aquatic invertebrates on them. Others, such as Wantzen & Wagner (2006) argue that tropical regions present a greater amount of predators and so, invertebrates in general should have a more diversified feeding diet.

Hydrological factors would also be reasons for the paucity of shredders (Winterbourn *et al*., 1981; Youle, 1996) as tropical rain storms can easily "wash" stream substratum, taking the food source away from the invertebrates, and again, inhibiting its feeding behavior. On one hand, some authors have suggested that reduced discharge should reduce invertebrates density and richness (e.g., Dewson *et al*., 2007b) or invertebrates abundance (e.g., Wood *et al*. 2000) by altering availability and suitability of instream habitats (Dewson *et al*., 2007a). On the other hand, an increasing in invertebrates abundance due to slower current velocity was also detected by Suren *et al*. (2003). In this way, hydrological effects on macroinvertebrates assemblage is still discussible (see Dewson 2007a for a review). We hypothesize that, besides leaf quality, shredders abundance will be related to resource availability mediated by hydrological factors if predators are not controlling their abundance.

Methods

Sites characterization

The stream used in this experiment is Lajeado Grande stream. It is located in southern Brazil and presents a sub-tropical climate (27ºS, 54ºW). Rain fall average is of 1800 mm a year and mean temperature in summer and winter are 25 °C and 8 °C, respectively. This catchment is on deciduous broadleaf forest area which originally covered great part of Southern and Central Brazil. Leaves fall during all year and Lauraceae and Myrtaceae comprise the main tree families presented on this region.

In order to verify water discharge influence on shredders occurrence, we selected six sites differing in discharge values (Table 1). They were selected in the same basin to avoid differences in water chemistry (Table 2) but located as far from each other as possible, to have them as independent sites. After the beginning of the experiment, one of the slow discharge sites had to be removed from analyses as higher sedimentation was observed which changed its characteristics making it difficult to compare with the other five sites. All chemical and physical data was taken by Rio Grande do Sul State Environmental Agency (FEPAM).

Table 1. Water discharge (m^3/s) measured monthly between October 2004 and January 2005. Data was kindly supported by FEPAM.

Treatments	Water discharge in m^3/s (± sd)
1	0.97 (± 0.57)
2	3.28 (± 2.87)

Table 2. Mean chemical variables (± sd) taken monthly from October 2004 to January 2005 on each sampling site (Treatment 1: n = 3; treatment 2: n = 2). Data was kindly supported by FEPAM.

	Treatment 1 average (± s.d.)	**Treatment 2 average (± s.d.)**	**p**
Temperature	20.9 (± 2.6)	20.5 (± 1.73)	0.7418
pH	7.3 (± 0.3)	7.3 (± 0.11)	0.9034
Dissolved Oxygen	7.3 (± 0.8)	7.3 (± 0.35)	0.9007
Conductivity	63.6 (± 11.3)	56.5 (± 9.34)	0.1621
Total phosphorus	0.1 (± 0.0)	0.1 (± 0.03)	0.3557
Total nitrogen	1.2 (± 0.5)	1.2 (± 0.36)	0.954
DBO	2.2 (± 0.3)	2.1 (± 0.14)	0.1717
Turbidity	13.9 (± 5.1)	13.6 (± 4.8)	0.9177
Total solids	106.8 (± 42.8)	109.0 (± 32.6)	0.9012

Leaf-bags

Leaf-bags were made of 4g (± 0.01g) of *Ocotea puberula* (Lauraceae) leaves collected just after abscission and air-dried until necessary. Bags were 30 x 20cm large and presented a mesh size of 10mm to allow the presence of invertebrates. As some studies have shown that shredders in the neotropics are related to wetland pools or to river banks (Wantzen & Wagner, 2006; Ríncon & Martínez, 2006; Graça *et al.*, 2001) our leaf-bags were intentionally placed next to the stream banks, divided in four riffles which were considered as replicates. At each sampling date (i.e., after 15, 30, 60 and 90 days from incubation) one leaf-bag from each riffle was retrieved and stored in a cooler for transportation to the laboratory. Then, leaves were gently washed over a 200µm sieve to remove invertebrates for further analyses. Invertebrates were kept in 70% alcohol and identified using stereomicroscope. Leaves were oven-dried for 48 hours and weighed to the nearest 0.01g. Breakdown rates were then calculated on a negative exponential formula (Petersen & Cummins, 1974).

Functional types: Shredders and predators

Macroinvertebrates were identified to genera level according to Fernandéz and Dominguéz (2001) for most insects and according to Costa *et al*. (2004) and Pes *et al*., (2005) for most of the Odonata and Trichoptera, respectively. Plecoptera was identified following Olifiers *et al*. (2004). Total genera richness, total abundance and density were measured and a ratio between predators and other macroinvertebrates number was also calculated.

Statistical analysis

Breakdown rates obtained from *O. puberula* leaves in slow and fast current velocity sites were compared in a one-way analysis of variance (1-way ANOVA). Analyses of Co-Variance (ANCOVA) was used for comparing genera richness, macroinvertebrates abundance and density between treatments where time was used as covariate. Predator and prey ratio between treatments and during exposure time was analyzed on a two-way ANOVA. Shredders abundance and density were compared between treatments and time (used as a covariate) on an ANCOVA analysis. All statistical analyses were conducted on R program (2006).

Results

Decomposition

Leaf breakdown differed between treatments ($p = 0.004$) and during time ($p < 0.001$). At the fast discharge, *O. puberula* breakdown was two times faster than at the slow discharge sites. At the fast current sites average k was of 0.0114 and in the slow current sites average k was of 0.0053 (Table 3).

Table 3. Breakdown rates (log-transformed) obtained from *Ocotea puberula* leaves in slow and fast current velocity sites. ($p = 0.004$ between treatments and $p < 0.001$ during time)

Current velocity	Site	Breakdown rate (± SE)	r^2
Fast discharge	F1	0.0113 (± 0.1377)	0.8756
	F2	0.0133 (± 0.06388)	0.9837
	F3	0.0096 (± 0.2348)	0.8057
Slow discharge	S1	0.0056 (± 0.08385)	0.944
	S2	0.0047 (± 0.1526)	0.8316

Macroinvertebrates assemblage

There was a general trend for invertebrates presenting lower values at the end of the experiment. However, genera richness, density and abundance did not differ between treatments ($p > 0.05$) nor during time ($p > 0.05$) (Table 4).

The ratio between predators and other invertebrates did not differ between treatments (Table 5) and was around 1:8 (predators:invertebrates). Most predators belong to Odonata group, from both Zygoptera and Anisoptera sub-orders which main families were Calopterygidae, Coenagrionidae and Libellulidae. Other predators found were Hydroptilidae *Neotrichia* sp. and *Hydroptila* sp. and Perlidae *Anacroneuria* sp. Invertebrates and predators ratio did not differ during exposure time.

Table 4. Analyses of Co-Variance (ANCOVA) for genera richness, macroinvertebrates abundance and density. Numbers in genera richness are related to total number of genera found. Abundance is related to total macroinvertebrates number found in litter-bags. Predictor variables are treatment and time (covariate).

Response variable	Independent variable	D.f.	Sum Sq	Mean Sq	F value	P
Abundance	Treatment	1	1	1	0.0002	0.9890
	Time	3	10464	3488	0.9256	0.4524
	Treatment:Time	3	2745	915	0.2428	0.8651
	Residuals	15	56526	3768		
Density	Treatment	1	36663	36663	3.0674	0.1003
	Time	3	86113	28704	2.4015	0.1083
	Treatment:Time	3	46961	15654	1.3097	0.3079
	Residuals	15	179287	11952		
Genera richness	Treatment	1	15.653	15.653	3.5962	0.07734
	Time	3	7.816	2.605	0.5986	0.62575
	Treatment:Time	3	13.806	4.602	1.0573	0.39641
	Residuals	15	65.292	4.353		

Table 5. Analysis of variance (ANOVA) of predator and prey ratio between treatments and during exposure time.

Predator:prey ratio	Df	Sum Sq	Mean Sq	F value	p
Treatment	1	0.000895	0.000895	0.803	0.3746
Time	3	0.005641	0.001880	1.6876	0.1821
Residuals	48	0.053480	0.001114		

Shredders found on the leaf-bags were mainly insects but some other groups were considered shredders as well. Crustaceans found in this work (*Aegla* sp. and *Trichodacthylus* sp.) belong to Decapoda group and are usually classified as macroconsumers that can feed on invertebrates and act as shredders, especially when searching for miners into the leaves. Besides these, true shredders as *Phylloicus* sp. and *Limnoperla* sp. from Calamoceratidae and Grypopterygdae families, respectively, and dipterans Tipulidae were found on the leaf packs.

Shredders abundance and density, however, differed significantly between treatments ($p < 0.05$) but not during time ($p > 0.05$) (Table 6; Figure 1 and 2). Most shredders were found on the slow current sites and belong to Grypopterygidae (Plecoptera), Calamoceratidae (Trichoptera), Tipulidae (Diptera), Trichodactylidae and Aeglidae (Crustacea). The main difference between treatments is that *Phylloicus* sp. (Calamoceratidae) was present at a higher grade in the slow stream flow sites and Trichodactylidae (Crustacea) was only found in these slow current sites (Table 7).

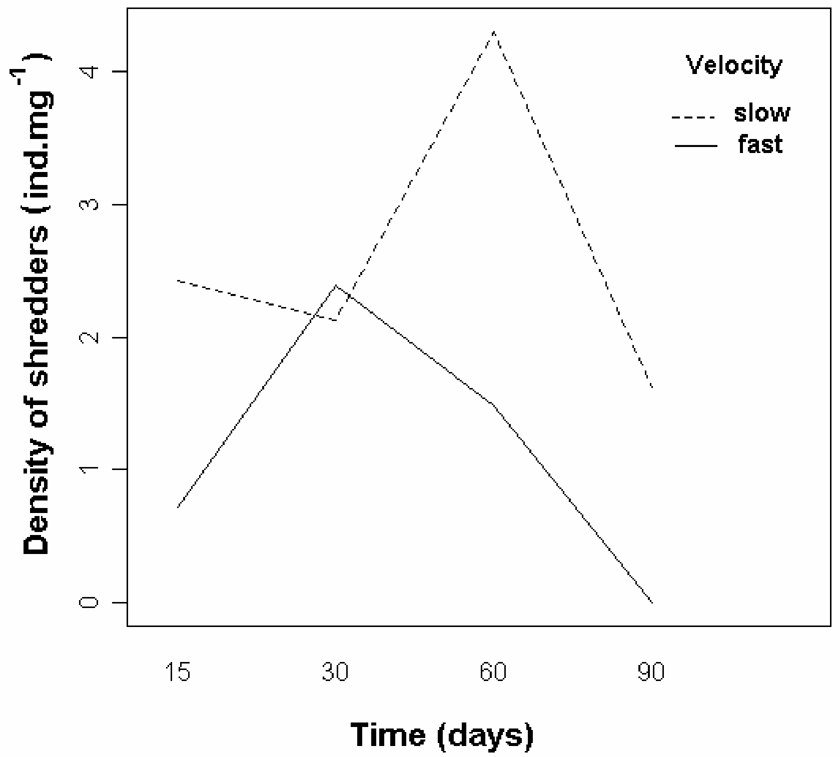

Figure 1. Analysis of covariance (ANCOVA) showing higher shredder density in the low discharge sites than in the high discharge sites. Density does not change during time. (Treatments: $p = 0.0462$; Time: $p = 0.3268$).

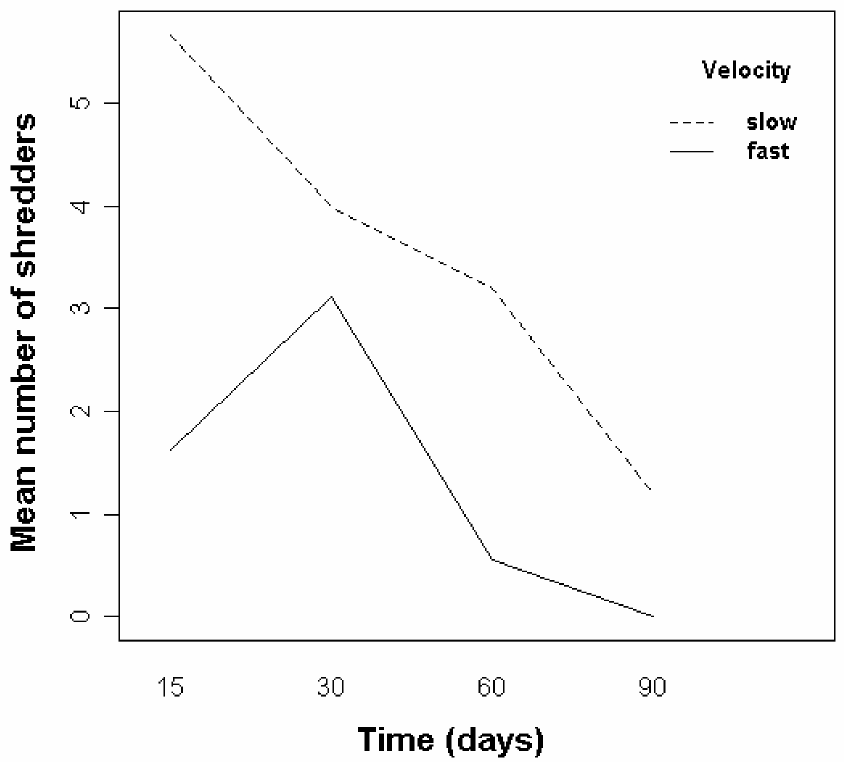

Figure 2. Analysis of covariance (ANCOVA) showing higher shredder abundance in the low discharge sites than in the high discharge sites. Abundance does not change during time. (Treatments: $p = 0.01562$; Time: $p = 0.078$).

Table 6. Analysis of covariance among Shredders mean numb er, treatments and time (as covariate). In bold, the statistically significant terms ($p < 0.05$).

Response variable	Independent variables	D.f.	Sum Sq	Mean Sq	F value	p
Shredder density	Time	3	21.568	7.189	1.1811	0.3268
	Velocity	1	25.492	25.492	4.1882	**0.0462**
	Residuals	48	292.162	6.087		
Shredders abundance	Time	3	73.61	24.54	2.4091	0.07852
	Velocity	1	64.00	64.00	6.2839	**0.01562**
	Residuals	48	488.85	10.18		

Table 7. Shredders richness (X indicates presence and O indicates absence of a certain *taxa*. Fast discharge treatment: n = 3; slow discharge treatment 2: n = 2)

Time	**Treatment**	**Trichoptera** Calamoceratidae *Phylloicus* sp.	**Plecoptera** Gripopterygidae *Limnoperla* sp.	**Diptera** Tipulidae Sp1	**Crustacea** Aeglidae *Aegla* sp.	**Crustacea** Trichodactylidae *Trichodactylus* sp.
15	Fast	X	X	X	0	0
	Slow	0	X	X	0	X
30	Fast	X	X	X	X	0
	Slow	X	X	X	X	X
60	Fast	0	X	X	0	0
	Slow	X	0	X	0	X
90	Fast	0	0	0	0	0
	Slow	X	0	0	0	X

Discussion

Functional role: Shredders, predators and resource quality

There is a non-agreement of the current literature on the feeding-functional classifications for some *taxa* found on this work. *Smicridea* sp. can be considered both as collector-gather (Poff *et al*., 2006) or as generalist (Wantzen & Wagner, 2006); *Trichorythods* sp. could be classified as collector-gatherer (Poff *et al*., 2006) or as collector that could behave as shredder or scraper (Wantzen & Wagner, 2006); *Nectopsyche* sp. is considered as generalist herbivore by Poff *et al*. (2006) although some Leptoceridae are considered shredders (e.g., Graça *et al*., 2006; Wantzen & Wagner, 2006); and Pyralidae presents the same problem as *Nectopsyche* sp. besides the fact that Pyralidae has its classification based on family level. So, these *taxa* were not considered as shredders even though the number of *Nectopsyche* sp., in especial, found in this work would make the differences between treatments even larger, helping our results to become more easily confirmed (*data not shown*).

Several reasons have been hypothesized in relation to the paucity of shredders in the neotropics in comparison to temperate regions: (i) lower quality of food, regarding to leaf toughness and presence of tannins and secondary compounds (Graça & Barlocher, 1998; Wantzen *et al*., 2002; Poorter *et al*., 2004); (ii) higher amount of predators; and, (iii) hydrological factors that could disturb aquatic systems in higher frequency (as in tropical rain storms) (Wantzen & Wagner, 2006). These factors linked with the non-correspondence of larvae developing time with resource input, as it happens in temperate regions, could make shredder-specialist behavior even more difficult to exist.

Even though leaves in tropic regions can have more secondary compounds, many organisms can avoid its chemicals as it has been shown for temperate invertebrates (Canhoto & Graça, 1999) or they can even choose among leaf species (Rincon & Martínez, 2006). Rincon and Martínez (2006) showed that *Phylloicus* sp. could select leaves among native species which differed in nutrient quality, in Venezuela. Graça *et al*. (2001) showed that tropical shredders survived and grew in similar ways on leaves from both temperate (*Alnus glutinosa* (L.) Gaertn.) and tropical regions (*Hura crepitans* L.). In a tropical stream Ardón and Pringle (2008) demonstrated that leaf secondary compounds did not influence leaf breakdown in Costa Rica, but structural components as cellulose did.

We used the same leaf species in all reaches and so leaf quality should not influence differences between treatments. The very low decomposition rate verified on this work (k = 0.0047 and 0.0052), which are included among the smallest rates already verified in the neotropics (Rueda-Delgado *et al*., 2006), is in agreement with the decomposition rate of another *Ocotea* used in an experiment in Brazilian Cerrado (Moretti *et al*., 2007). Moretti *et al*. (2007) found, for this species, the highest density and biomass of invertebrates, although in Cerrado streams a decomposition rate of 0.005 was considered as fast.

It should be expected, though, that either predaceous or lack of resource would account for treatments exhibiting a poor shredder abundance. In this work we could verify that predators' abundance was similar in both treatments ($p > 0.1$) and the predators:prey ratio did not present variation between treatments ($p > 0.05$).

Thus, the last reason for influencing shredders abundance would be the lack of resource. It has been shown that neotropic shredders are abundant in lateral wetland pools (Wantzen & Junk, 2000) or in slow flowing areas (Wantzen *et al*., 2002; Cheshire *et al*., 2005). We sampled leaf-bags next to the stream bank where, following this assumptions, shredders would be more abundant, and we avoided middle parts of the streams. With the same chemical composition, the reaches studied here differed only to its water flow (discharge). As macroinvertebrates abundance and density were similar between treatments ($p > 0.05$ for both parameters) but shredders abundance and density were higher in the slow discharge treatment ($p = 0.03$) we could assume that leaves presented higher breakdown due to physical fragmentation (disruption) and not to invertebrates feeding (Figure 2 and 3). Although most literature agree that flow reductions (artificially or natural) reduce invertebrates richness because of reducing habitat availability (Dewson *et al.*, 2007a), the slow flow detected here was only enough to reduce physical fragmentation of leaves and not to change in-stream habitats (e.g., reducing wetted areas).

Rueda-Delgado *et al.* (2006) verified that leaves of different qualities in a headwater Amazon stream were faster decomposed at the fast-discharge season, where Amazon River is low and stream discharge is more variable than the high-water period, when stream discharge is very low. Although the proportion of functional feeding groups found on their work did not differ statistically between rain seasons, the proportions of shredders as *Phylloicus* was slightly higher in the slow discharge season than in the fast discharge season. However, a work on Amazonian litter-banks during slow-flowing season revealed a high diversity of invertebrates on litter but no litter-feeding specialist was found (Henderson & Walker, 1986) as most organisms were shrimps and the primary consumers fed mainly on algae or fungi on litter, but not on litter itself.

Cheshire *et al.* (2005) verified a higher richness and biomass of shredders in pool habitats rather than riffles in Australia. Higher occurrence of shredders in slow flowing streams, in South America, was verified by Chara *et al.* (2007), who detected a high importance of shredders on leaves in a slow flowing stream in Colombia, where this functional group represented over 50% of invertebrates biomass on leaf bags.

Some studies on other tropical regions have also verified a paucity of shredders in streams, as in Kenya (Dobson et al., 2002) and in Papua New Guinea (Yule, 1996). Besides Australian tropical streams (Cheshire *et al.*, 2005), most reports indicate a climate correspondence effect for this feeding-group around the world. Winterbourn *et al.*, (1981) had already noted a paucity of shredders in New Zealand streams for which they pointed New Zealand non-retentive instable stream environments as an explanation for this effect. Hence, according to provided literature for the tropics, in special to that data from the neotropics, it seems that in this region there is a change on the current overview about shredders influence on leaves (Graça *et al*., 2001), as shredders do not play a major role on decomposition (Mathuriau & Chauvet, 2002) but may be influenced by resource availability. However, we emphasize that more studies are needed for neotropical regions to verify to which extent shredders might influence or not litter decomposition and to understand which shifts might occur in shredders distribution among habitats (Cheshire *et al*., 2005) or from slow decomposing to fast decomposing sites.

One ecological consequence outcome from this different shredder-resource-water flux link is their possible occurrence fragility in the neotropics. This is because in spite of several measures that have been undertaken in the last few years aiming conservation and restauration, the neotropical region are still often subjected to non-sustainable development (e.g., Malhi *et al*., 2008). This fact encourages land exploitation for agriculture and destruction of the riparian vegetation by forest logging which can increase water flux events and thus move out the resource needed for shredders (Naiman & Décamps, 1997). Kreutzweiser *et al*. (2008), for instance, could detect effects of logging on macroinvertebrates assembly and on leaf-decomposition even under best management practices in Canada. In this scenario, the few shredders that occur in these tropical and sub-tropical regions can become even more endangered, which can lead to the break of a very important trophic link, even though small in the neotropics (Graça *et al*., 2001), between headwater shredders and lower stream reach invertebrates.

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CAP 3

USE OF BMWP AND ASPT ON SOUTHERN BRAZIL THROUGH CERRADO AND ARGENTINEAN VALUES

(Com Gilberto G. Rodrigues)

(Artigo em preparação para ser submetido à *Acta Limnologica Brasiliensia*)

"O rio da minha aldeia não faz pensar em nada.
Quem está ao pé dele está só ao pé dele."

Poemas de Alberto Caeiro
Fernando Pessoa

Abstract

The use of macroinvertebrates in biological assessments has been widely suggested by researchers and environmental agencies. While some countries have it well developed, as England, USA and Australia, other ones, especially in southern hemisphere, still do not have these metrics established. In Brazil, studies focusing macroinvertebrates as biological indicators have just started and, due to Brazilian continental size, these studies are valid for a few areas of the country. In this work we assessed eight reaches in southern Brazil using biological indexes (BMWP and ASPT) already proposed from Brazilian Cerrado and Argentinean Patagonia. Our studied catchment presented low index values for both Cerrado and Patagonean scores. However, differences were found between both indexes. One site was classified as more impacted when using Argentinean values but was classified as less impacted when used Cerrado values. This first approach of bio-assessment in southern Brazil shows the urgent need of more ecological and taxonomical studies so that one could classify more accurately running waters in this region of Brazil.

Key-words: Biological assessment, BMWP, ASPT, macroinvertebrates, subtropical climate, southern Brazil.

Introduction

Benthic macroinvertebrates are very useful organisms in environmental assessments. They are easy to be collected and handled, have appropriate identification keys, are abundant and present many different ecological traits (Rosemberg & Resh, 1993). Because of this, the use of macroinvertebrates in biological assessments has been well developed, especially in northern hemisphere countries (Rosenberg & Resh, 1993) as several indices have been proposed (Stribling *et al*., 2008). Southern hemisphere, and South America in especial, in the other hand, is just starting to have these indexes used (Cota *et al*., 2002; Junqueira & Campos, 1998; Junqueira *et al*., 2000, Miserendino & Pizzolón, 1999) although countries as Australia and New Zealand can be considered as outliers.

These bio-assessments follow well-studied protocols which classify the waters into classes after the observation of specific benthic invertebrates *taxa* occurrence. Generally, these protocols are based on family-*taxa* level, due especially to its easier identification when compared to genera and species level (Barbour *et al*., 1996; Chessman *et al*., 2007). To date most researches and protocol developing have been conducted based on the Biological Monitoring Working Party index (BMWP) developed in England (Armitage *et al*., 1983). This index scores macroinvertebrates *taxa* from one to ten, as the lowest values stand for poor water quality indicator organisms and the highest values stand for good water quality indicator organisms.

Another index used is called ASPT (Average Score per Taxon) and is calculated based on the values obtained from BMWP but divided by the number of *taxa* found in each site. Some assessment analyses have also used the EPT values, which stands for three main groups of macroinvertebrates: Ephemeroptera, Plecoptera and Trichoptera (e.g., Zweigg & Rabeni, 2001). However, the use of an index at the order level from only three orders is less sensitive in water analyses once these orders can have many families that respond to different levels of pollution (but see Melo, 2005).

In this work, we assessed water quality of several stream reaches in southern Brazil, through BMWP modified values from Cerrado and Argentinean Patagonia. Our goal is to identify which index is more appropriated for this region and to provide literature with new data from macroinvertebrates biological assessments.

Material and Methods

Study site

The studied catchment is located in southern Brazil, in a sub-tropical area influenced by the Deciduous Forest. The stream, called Lajeado Grande, presents 83.4 km length and mean depth of 0.6 m. Its basin covers 525.38 km^2, presenting its headwaters on southern Brazil Plato and its mouth in Uruguay River. The area presents an intense agricultural activity, most of which uses the riparian zone for cropping. The most frequent tree species on this area belong to Lauraceae and Myrtaceae families.

Eight reaches (Table 1, Figure 1) were chosen to have the aquatic macroinvertebrates sampled. These sites present different land uses around the stream, including pig growing farms and soy/corn monocultures. Water chemistry was measured monthly and substrate composition was verified in October 2004 by Rio Grande do Sul State Environmental Agency (FEPAM). The chemical and physical measurements were done for pH-value, dissolved oxygen, electrical conductivity, turbidity, temperature, water discharge, total phosphorus, total nitrogen, nitrate. Substrate was classified into percentages of gravel, sand, silt and clay (FEPAM , *unpublished data*).

Macroinvertebrates

For sampling the invertebrates we used litter-bags attached to the stream bank. This procedure should allow us to collect both invertebrates that use leaves as a substratum or those invertebrates which actively feed on leaves. The purpose on doing so is to be able to analyse invertebrates specifically related to the very important feeding group of shredders which, for feeding on leaves, can be good indicators of human activities on the riparian zone.

The 10mm mesh-size litter-bags were made of 4g of *Ocotea puberula* leaves. These leaves were collected from one single tree next to Lajeado Grande catchment. Bags were placed on 6 reaches of the stream. In each reach, groups of ten litter-bags were divided into four points on the stream bank where riffles were present. These points were used as replicates in statistical calculations. At 15, 30, 60 and 90 days the 4 litter-bags from each reach were retrieved from water into plastic bags and stored in a cooler for transportation to the lab. Then, leaves were gently washed with tap water over a 250µm sieve and the material was kept in 70% alcohol for further invertebrates' identification. Fernandéz and Dominguéz (2001) was the main source for identifying Insecta. Costa *et al*. (2004) was used for most Odonata, Pes *et al*., (2005) for Trichoptera and Olifiers *et al*. (2004) for Plecoptera. Mollusca were identified by specialists.

Bio-assessment

For assessing the invertebrates assemblage we used literature source from Argentinean Patagonia and from Brazilian Cerrado. Miserendino & Pizzolón (1999) were the reference from Argentina and Cota *et al*. (2002) and Junqueira *et al*. (2000) were the reference from Brazil (Table 1).

Statistical analyses

An one-way Analysis of Variance (1-way ANOVA) was used to compare both BMWP and ASPT values among sampling sites. Whenever differences were significant, *a posteriori* TukeyHSD test was then executed in order to verify which sampling sites differed. Time was used as a block. The eighteen most abundant genera were related to the chemical and physical measurements and to substrate composition in a Canonical Components Analysis (bi-plot cca).

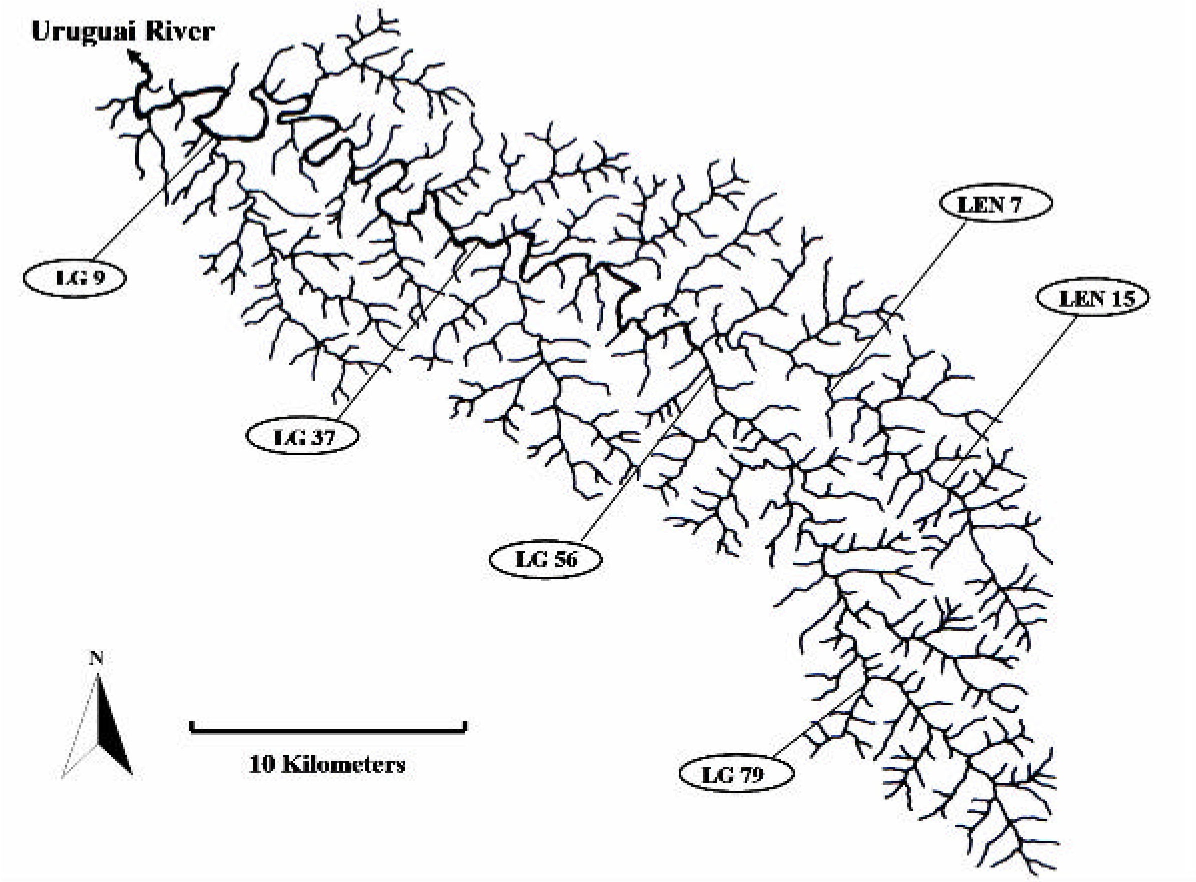

Figure 1. Lajeado Grande catchment, southern Brazil. Sampling sites for aquatic macroinvertebrates analyses are shown.

Results

BMWP and ASPT

From all *taxa* found on the Lajeado Grande sites, Noteridae, Leptohyphidae, Megapodagrionidae, Protoneuridae, Hemiptera, Trichodactylidae, Hirudinea, Ampullariidae, Nematyelminthe could not be included in any classification due to lack of taxonomic information for this *taxa*. However, 30 *taxa* found on Lajeado Grande catchment were included in the analyses. Both Cerrado and Patagonian indexes did not have values for four *taxa* (Table 2).

On the BMWP values (Figure 2), differences among sites could be found only through Patagonian index (Table 3). However, Tukey HSD *a posteriori* test did not find any significant difference between sites. Time of sampling (related to litter decomposition) did not differ for any of the indexes used.

Table 2. BMWP values from two distinct regions: Brazilian Cerrado (after Junqueira *et al*., 2001 and Cota *et al*., 2000) and Argentinean Patagonia (after Miserendino & Pizzolón, 1999). Some of the families were not present in one or both the indexes shown below.

Class	Order	Family	Patagonia	Cerrado
Insecta	Trichoptera	Calamoceratidae	8	-
		Hydropsychidae	5	6
		Hydroptilidae	-	7
		Leptoceridae	10	7
		Policentropodidae	7	7
		Gyrinidae	3	5
	Coleoptera	Elmidae	5	5
		Dytiscidae	3	4
	Ephemeroptera	Baetidae	6	5
		Caenidae	4	4
		Leptophlebiidae	10	10
	Plecoptera	Gripopterygidae	10	10
		Perlidae	10	8
	Zygoptera	Calopterygidae	-	8
		Coenagrionidae	6	7
	Anisoptera	Aeshnidae	6	8
		Cordulidae	-	-
		Gomphidae	7	5
		Libellulidae	6	8
	Lepidoptera	Pyralidae	-	8
	Diptera	Ceratopogonidae	4	4
		Chironomidae	2	2
		Simulidae	5	5
		Tipulidae	5	5
Crustacea	Decapoda	Aeglidae	5	-
Annelida	Oligochaeta	Oligochaeta	1	1
Mollusca	Gastropoda	Ancylidae	5	6
		Hydrobiidae	3	-
		Planorbidae	3	3
	Bivalvia	Sphaeriidae	3	3

Table 3. Analysis of Variance (ANOVA) for the BMWP values from Argentinean Patagonia and Brazilian Cerrado. Significant results are shown in bold, ($p < 0.05$).

	BMWP	Df	Sum Sq	Mean Sq	F value	P
	Time	3	183.8	61.3	0.4936	0.688159
Patagonia	Site	5	3508.6	701.7	5.6545	**0.000274**
	residuals	56	6949.6	124.1		
	Time	3	119.0	39.7	0.2096	0.8893
Cerrado	Site	5	2138.8	427.8	2.2609	0.0607
	residuals	56	10594.8	189.2		

For the ASPT values (Figure 3), both Cerrado and Patagonia indexes showed differences among southern Brazilian sites (Table 4). As for the BMWP values, time did not present difference for biological values in any of the indexes used. Through Tukey HSD *a posteriori* test we could verify differences between LG 79 and LG 37 (p=0.0456869) from

Cerrado ASPT values and between LG 79 and LEN 7 ($p = 0.0078286$) from Patagonia ASPT values.

Table 4. Analysis of Variance (ANOVA) for the ASPT values from Argentinean Patagonia and Brazilian Cerrado. In bold, the significant results ($p < 0.05$).

	ASPT	Df	Sum Sq	Mean Sq	F value	P
	Time	3	3.6475	1.2158	2.5812	0.06246
Patagonia	Site	5	7.2503	1.4501	3.0784	**0.01592**
	Residuals	56	26.3781	0.4710		
	Time	3	0.7075	0.2358	0.4209	0.73870
Cerrado	Site	5	6.8641	1.3728	2.4502	**0.04456**
	Residuals	55	31.3766	0.5603		

Relation to physical and chemical variables

The CCA analysis for the eighteenth most important invertebrate families and the chemical and physical variables of the streams showed significance for the permutated model (Table 5). Some families were closed related to presence of silt/clay, as Tipulidae and Gripopterygidae. Simulidae was related to high levels of dissolved oxygen, high turbidity and presence of gravels. Oligochaeta and Leptohyphidae were related total nitrogen concentrations, phosphorus concentrations and to sand texture. Elmidae, Ceratopogonidae and Leptoceridae were related to high levels of phosphorus concentrations. Coenagrionidae, Gyrinidae and Calamoceratidae were related to nitrate concentrations, high pH-value and silt. Baetidae, Chironomidae and Caenidae were only related to temperature. Leptophlebidae was related to water discharge and Sphaeridae was related to electric conductivity (Figure 4).

Table 5. Permutation test for CCA under direct model

	Df	Chisq	F	N. Perm	P
Model	13	0.7302	1.8964	800	0.02875
Residual	4	0.1185			

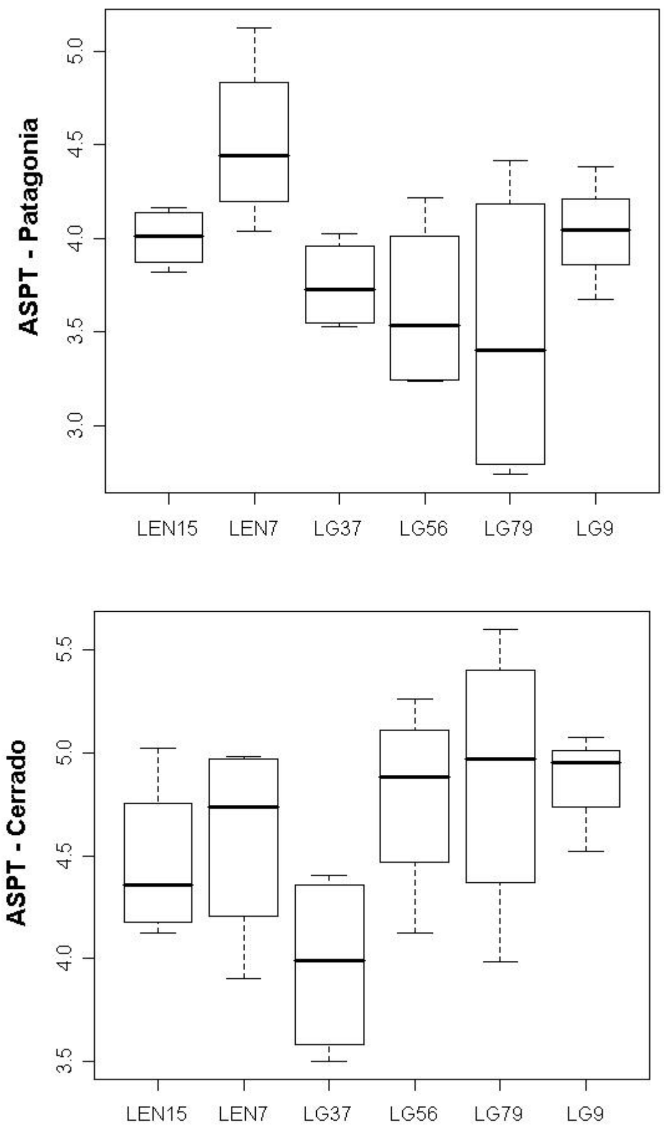

Figure 2. ASPT values for sampling sites from (a) Cerrado and (b) Patagonia. Cerrado and Patagonian ASPT values showed significant difference among southern Brazilian sites ($p < 0.05$).

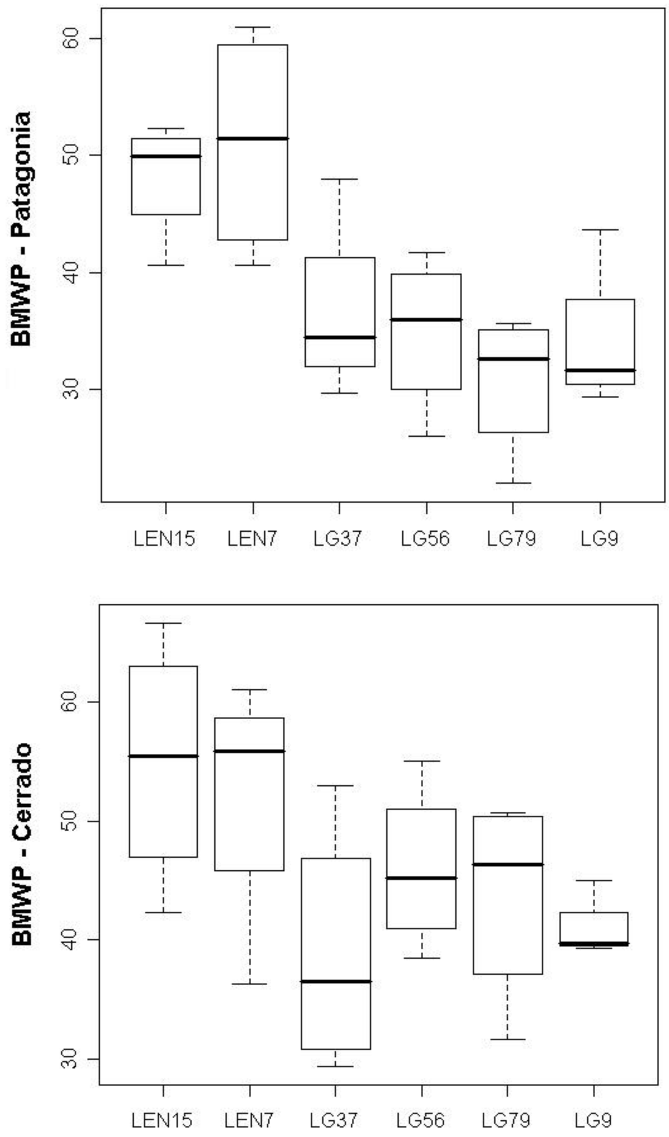

Figure 3. BMWP values for sampling sites from (a) Cerrado and (b) Patagonia. Only Patagonian BMWP values showed significant difference among southern Brazilian sites ($p < 0.05$).

LEN7 presented the highest score from Argentinean ASPT and BMWP values. This site presented a greater number of Leptoceridae and Gripopterygidae organisms which score ten on Argentinean BMWP index. These organisms are manly shredders and can indicate the presence of riparian vegetation around the stream.

LG37, on the other hand, presented the lowest value from Cerrado ASPT and BMWP index. This site presented the highest number of Oligochaeta and Chironomidae which are indicators of poor water quality presenting the lowest values on Cerrado BMWP. LG9 differed from both mentioned sites and presented a large number of organisms, mainly Coleoptera and Zygoptera.

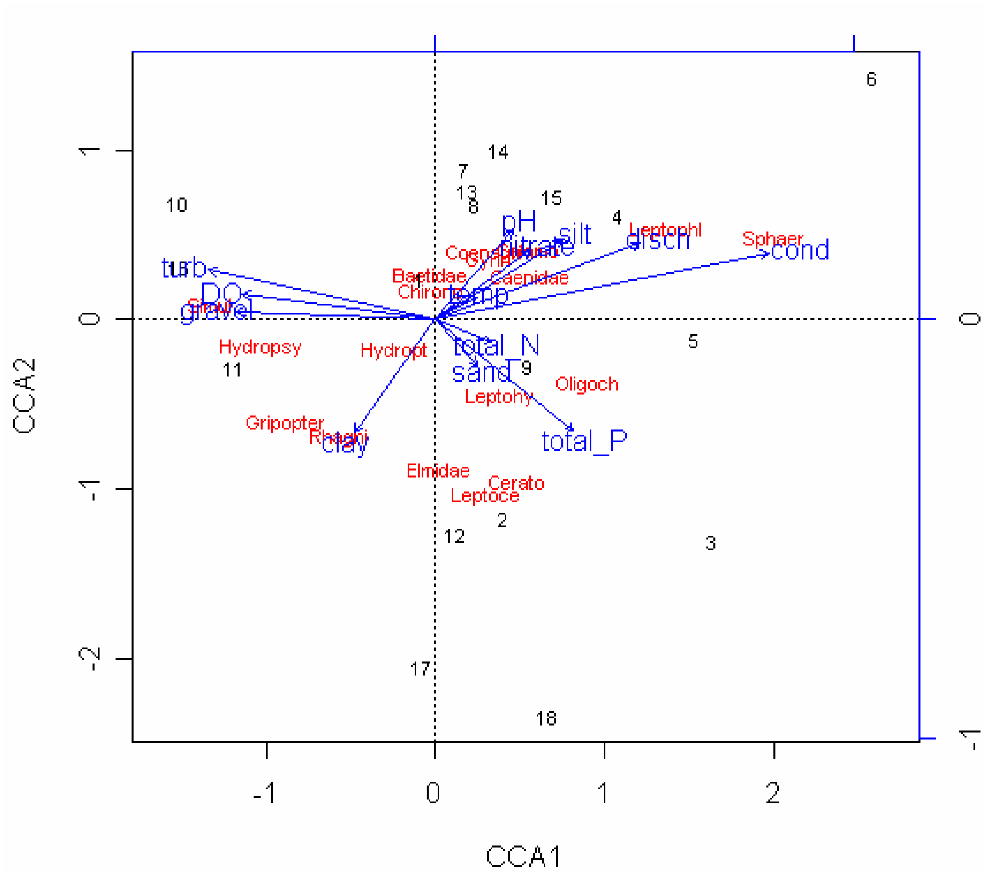

Figure 4 CCA obtained for invertebrates` families and chemical and physical variables (arrows). Invertebrate families: "Coena" = Coenagrionidae; "Calamo" = Calamoceratidae; "Hydropsy" = Hydropsychidae; "Hydropt" = Hydroptilidae; "Leptoce" = Leptoceridae; "Elmidae"; "Gyrin" = Gyrinidae; "Leptophl" = Leptophlebiidae; "Caenidae"; "Leptohy" = Leptohyphidae; "Baetidae"; "Gripopter" = Gripopterygidae; "Coenagri" = Coenagrionidae; "Chirono" = Chironomidae; "Simul" = Simulidae; "Cerato" = Ceratopogonidae; "Rhaghi" = Rhagionidae; "Oligoch" = Oligochaeta; "Sphaer" = Sphaeridae. Chemical and physical variables: "temp" = temperature; "pH"; "DO" = dissolved oxygen; "cond" = conductivity; "total_P" = total Phosphorus; "nitrate"; "total_N" = total Nitrogen; "turb" = turbidity; "disch" = discharge; "gravel"; "sand"; "silt"; "clay".

Discussion

Southern Brazilian sites presented low classification values from both Argentinean and Brazilian indexes. The highest BMWP values were found on those sites where more shredders were present. This feeding-group is related to areas that have well-preserved vegetation cover and great amounts of litter input. According to Alba-Tecedor (1996), however, all sites should be classified as contaminated water, ranking from 36 – 60 points on the BMWP index on both Patagonian and Cerrado values.

Different classifications were obtained from Cerrado and Patagonian values to the fauna found on Lajeado Grande stream. From the Cerrado index, only ASPT showed differences among sites ($p = 0.044$). From the Patagonian values, however, both ASPT and BMWP presented differences ($p < 0.05$) among sites.

All *taxa* were included, even rare *taxa*. Melo (2005) attested that the use of simplified data-methods are mostly reliable when comparing streams at local scale, which was the case in our study. Sickle *et al*. (2007), on the other hand, argues that the exclusion of rare taxa affects results on bio-assessments. We also did not have reference sites as this catchment presents a high disturbance over all region, and, in this case, reference sites could act as weak-references (Whittier *et al*., 2007).

Through the CCA analysis we were unable to make a precise correspondence between the physical and chemical variables and the invertebrate families, according to the BMWP indexes obtained from South America. Elmidae and Ceratopogonidae families, for instance, which are considered as indicators of low-intermediate water quality (ranging 4 and 5, respectively, in the BMWP values of both Cerrado and Patagonia) were related to high total phosphorus concentrations but Leptoceridae, which is considered as indicator of intermediate-high water quality (scoring 10 and 7 in Cerrado and Patagonia indexes, respectively) was related to high total phosphorus as well. In the same way, Gyrinidae, Coenagrionidae and Calamoceratidae were related to high concentration of nitrate, high pH-value and presence of silt texture, even though some can be considered as low water quality indicators (Gyrinidae) and other as intermediate to high water quality indicators (Coenagrionidae and Calamoceratidae). Baetidae, Chironomidae and Caenidae scored between 2 to 6 in Cerrado and Patagonian values and were related to higher temperature which can occur in areas without sufficient canopy cover, which allows sunlight to increase water temperature.

Although literature on biological assessment subject is very rich for Northern hemisphere countries (see Rosemberg & Resh, 1993) we attest that more studies should be conducted in South America in general, but in southern Brazil in especial. Once we have a data-base of invertebrates distribution and their relation to water parameters we will be able to standardize this method for biological assessments and make use of these organisms as water-quality indicators. For now, the use of aquatic macroinvertebrates on biological assessments may produce weak classifications for streams in regions where these organisms occurrence and environmental tolerance is not yet well known.

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CONSIDERAÇÕES FINAIS

Os resultados obtidos nesse estudo indicam que a decomposição foliar pode ser um indicador de qualidade ambiental em sistemas lóticos de ordens iniciais. Cuidados, porém, são necessários quanto ao uso desse método. Neste trabalho, a detecção de alteração ambiental em escalas de maior abrangência, como no caso da atividade agrícola ao redor dos cursos d'água, não foi possível, muito provavelmente, devido ao pequeno número amostral e à alta correlação entre os pontos de atividade agrícola intensa com os ambientes próximos às nascentes na bacia hidrográfica em estudo. Esse problema referente ao delineamento amostral pode ter camuflado informações pertinentes ao vínculo da decomposição foliar com o gradiente de atividades agrícolas.

Visto a atividade agrícola, ou até mesmo, a atividade urbana e industrial, apresentar diferentes formas de alteração na estrutura e função da paisagem (e.g., aplicação de inseticidas, enriquecimento de nutrientes na terra, alteração do pH do solo, modificações na formação vegetacional, supressão da vegetação ripária, alteração da vazão dos cursos d'água, soterramento da calha dos cursos d'água por material rochoso de menor granulometria, entre outros), deve-se compreender mais profundamente o quanto cada uma dessas alterações ambientais (mas também as alterações ambientais conjuntamente) afetam o processo ecológico de decomposição. Algumas modificações da paisagem podem agir de modo direto sobre a decomposição foliar, como é o caso do soterramento da calha dos rios ou da alteração da vazão dos cursos d'água por meio de atividades de irrigação (com retirada de água diretamente dos sistemas lóticos e não de uma bacia de contenção, por exemplo). Essas modificações, quer sejam originadas naturalmente (como no caso da maior vazão provocada por chuvas), quer sejam provocadas pelo uso da terra (como no caso da sedimentação), foram detectadas como agentes no processo de decomposição foliar neste estudo.

Há, ainda, efeitos indiretos da alteração estrutural e funcional da paisagem, por meio de atividades agrícolas ou urbano-industriais (como o enriquecimento do solo por nutrientes ou a sua contaminação por metais pesados) que podem ocasionar alterações nesse processo ecológico. Essas alterações podem, através da lixiviação dos nutrientes ou do carreamento de metais pesados para os cursos d'água, atingir a fauna aquática e, com isso, alterar a decomposição foliar. Neste estudo, não foi possível detectar esses efeitos, uma vez que a bacia hidrográfica em questão apresentava propriedades químicas semelhantes entre os pontos estudados. Sabe-se, porém, que o enriquecimento de nutrientes pode alterar a composição da assembléia de macroinvertebrados bentônicos ou provocar modificações na assembléia de fungos decompositores. A contaminação por metais pesados pode, do mesmo modo, interferir nesses grupos aquáticos, por meio da diminuição da abundância ou riqueza dos diversos grupos presentes. Esses efeitos sobre a fauna podem, assim, modificar a taxa com que o material foliar é decomposto.

A mudança na estrutura da vegetação ripária verificada neste trabalho alterou, em primeiro lugar, os processos de deposição de sedimento, oriundos da margem do rio, e ocasionou um retardamento no processo de decomposição foliar. Partículas de menor granulometria, como silte e argila, apareceram relacionadas com o retardamento da decomposição foliar, ao passo que partículas de maior granulometria - no caso, a areia - estavam relacionadas aos trechos com maiores velocidades de decomposição. Ainda que a areia seja uma textura de pequena granulometria, a sua presença estava negativamente relacionada com a presença de silte e argila, que apresentam texturas ainda menores. Trabalhos que focam esse aspecto da sedimentação têm verificado a influência das partículas de menor granulometria como agentes do retardamento da decomposição foliar.

Outras mudanças na estrutura da vegetação ripária, como o corte e supressão dessa vegetação, podem acarretar, ao mesmo tempo, um aumento do aporte aluvial devido à não-retenção da água das chuvas pelo que antes seria a mata ciliar. Esse fato pode provocar (i) uma maior perda de biomassa no material orgânico alóctone, proveniente de trechos superiores do rio, retido em remansos ou corredeiras ou ocasionar, até mesmo, (ii) um completo carreamento desse material em direção às partes baixas da bacia hidrográfica. Nos locais onde o material orgânico alóctone é carreado devido ao aumento da vazão, há perda de matéria e energia responsáveis por grande parte da produção primária. Já nos trechos onde a biomassa vegetal apenas se degrada mais rapidamente devido ao maior volume de água circulante, há mudanças estruturais e funcionais em suas assembléias aquáticas, o que afetará, conseqüentemente, a cadeia alimentar envolvida.

Mudanças na composição da vegetação ripária, em termos de abundância e riqueza da flora, podem, também, ser importantes na modificação do processo de decomposição foliar. Neste trabalho, utilizou-se a mesma espécie vegetal em todos os pontos amostrados, o que excluiu a possibilidade de haver diferenças químicas e físicas quando se trata do material foliar exposto nos cursos d’água. Contudo, a exposição de folhas com alta taxa de decomposição, ricas em nutrientes, misturadas com folhas de espécies pobres quanto aos constituintes químicos, e, por isso, com baixa taxa de decomposição, podem alterar o processo de decomposição foliar das espécies envolvidas. Isso ocorre porque os nutrientes da espécie que se degrada mais rapidamente podem modificar a palatabilidade do material mais pobre nutricionalmente, e fazer com que essa última espécie apresente uma decomposição mais acelerada. O efeito dessa diversidade foliar na decomposição está, contudo, sendo recém estudado, e as conseqüências dessa mistura de espécies são, ainda, desconhecidas.

Ao contrário do que acontece em sistemas temperados, contudo, a decomposição foliar em sistemas tropicais ou sub-tropicais parece mais suscetível a ações físicas, como a deposição de areia ou a alta velocidade da água, mencionadas anteriormente, e menos suscetível a ações biológicas, ao menos no que se refere aos macroinvertebrados bentônicos. Enquanto que a maior deposição de silte e de argila afeta negativamente a decomposição foliar por soterramento do material foliar, o aumento da vazão influencia positivamente esse processo ecológico. Esse aumento na velocidade de decomposição, por meio do aumento da vazão, gera maiores efeitos sobre a fauna bentônica que mais diretamente "depende" do recurso foliar, a saber, os organismos retalhadores. Quando a velocidade da água diminui, e assim, a decomposição foliar desacelera, a fauna bentônica responde positivamente à presença prolongada do recurso alimentar. Os organismos do tipo funcional retalhador podem, então, ocorrer sobre esse material e retirar os nutrientes necessários para seu crescimento. Sem a presença tão constante do recurso foliar, tais organismos retalhadores não poderão adquirir a matéria e energia necessárias para seu crescimento a partir do material orgânico alóctone. Além disso, pressões ecológicas de cima para baixo na cadeia alimentar (relação top-down) - nomeadamente a predação, a qual é considerada como muito importante no controle da fauna em sistemas tropicais e subtropicais - fazem com que a fauna detritívora apresente dieta mais generalista para obtenção de alimentos do que o demonstrado para sistemas temperados, onde essa fauna pode ser especificamente retalhadora de folhiço, sem estar sujeita à pressão exercida pelos predadores.

O fato de sistemas neotropicais não apresentarem quantidade idêntica de retalhadores em relação aos sistemas temperados parece estar relacionado, assim, aos motivos hidro-metereológicos, típicos de sistemas tropicais e sub-tropicais (i.e., chuvas intensas e freqüentes), que não permitem a manutenção tão prolongada do recurso foliar nos cursos d’água, comum nas regiões mais frias e menos chuvosas do hemisfério norte. A pressão ecológica por parte dos predadores, contudo, não foi verificada neste trabalho, uma vez que a razão entre predadores e presas permaneceu a mesma nos dois tratamentos (com alta e baixa vazão). Sugere-se que a pressão por predação seja, de qualquer modo, importante na regulação da ocorrência de grupos detritívoros com hábitos específicos em regiões tropicais e subtropicais.

Desse modo, a hipótese de que o processo de decomposição foliar seja afetado pela poluição, tanto urbana quanto agrícola, através de mudanças estruturais e funcionais na assembléia de macroinvertebrados aquáticos que, por sua vez, ocasionaria modificações no modo como o material foliar é consumido, não é sustentada a partir dos resultados obtidos no presente estudo. A poluição por atividades agrícolas parece ocasionar efeitos mais intensos na decomposição através de alterações físicas do ambiente do que através de alterações biológicas. Contudo, mesmo que o aumento no fluxo da água seja responsável por aumento na decomposição, não se sabe, ainda, o quanto os organismos retalhadores realmente atuam na decomposição (ou seja, o quanto eles decompõem o material foliar), nem o quanto outros organismos microbiológicos, como os fungos aquáticos, são importantes para o processo de decomposição. Visto, porém, que os retalhadores não aparecem como principais causadores da decomposição foliar, ao contrário das regiões temperadas, talvez sejam os fungos aquáticos os principais agentes biológicos envolvidos no consumo do material foliar.

Este trabalho também indicou que todos os pontos amostrados se enquadram em categorias de risco ambiental com baixa qualidade das águas, quando a fauna aquática é analisada. A partir do uso de índices BMWP com valores da Patagonia argentina e do Cerrado brasileiro, obtiveram-se maiores valores biológicos, isto é, melhor qualidade de água, justamente nos mesmos trechos que apresentaram maior quantidade de retalhadores. O fato de os retalhadores se apresentarem como grupo funcional indicador de qualidade ambiental explica-se devido ao vínculo desse grupo de invertebrados com o material alóctone e, assim, com a presença de mata ciliar em melhor estado de conservação. Porém, estudos mais amplos sobre a fauna bentônica e suas respostas frente a alterações ambientais são necessários antes que esses organismos possam ser aplicados integralmente no continente latino-americano, em especial, no sul do Brasil, onde ainda há escassez de informações referentes à ocorrência dos macroinvertebrados bentônicos.

O processo ecológico de decomposição foliar, influenciado mais fortemente por fatores físicos ou biológicos, é, ainda, de grande importância para os sistemas aquáticos continentais. Esse processo transforma a matéria orgânica particulada grossa, consumível por poucos organismos, em matéria orgânica particulada fina, a qual é a base da cadeia alimentar detritívora em sistemas aquáticos. Entender como esse processo ocorre em sistemas tropicais e subtropicais e compreender as conseqüências que as atividades humanas trazem para a decomposição desse material orgânico, é peça fundamental para melhorar a compreensão ecológica dos sistemas hídricos (em especial, daqueles de água corrente) e, a partir daí, estabelecer critérios ambientais quanto ao uso das terras em bacias hidrográficas. Assim, para que as políticas públicas de uso dos cursos d’água em países como o Brasil sejam desenvolvidas visando à conservação da funcionalidade desses ambientes, deve-se, primeiramente, saber como esses sistemas aquáticos realmente funcionam em termos ecológicos e hidrológicos.

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ANEXOS

Tabela 1. Tabela filogenética do total de amostras realizadas na Bacia do Lajeado Grande, de novembro de 2004 a julho de 2005.

Classe	Ordem	Família	Gênero	Nº de indivíduos
Insecta	Trichoptera	Calamoceratidae	*Phylloicus*	19
		Hydropsychidae	*Smicridea*	128
		Hydroptilidae	*Neotrichia*	34
			Hydroptila	64
		Leptoceridae	*Nectopsyche*	350
			Oecetis	2
		Policentropodidae	*Cernotina*	6
	Coleoptera	Elmidae	*Elm_sp1*	668
			Elm_sp2	5
			Elm_sp3	4
			Elm_sp4	10
			Elm_sp5	1
		Dytiscidae	*Dyt_sp1*	7
		Noteridae	*Not_sp1*	3
		Gyrinidae	*Gyr_sp1*	24
	Ephemeroptera	Leptophlebiidae	*Ulmeritoides*	173
			Miroculis	40
			Simothraulopsis	2
		Caenidae	*Caenis*	399
		Leptohyphidae	*Traverhyphes*	452
			Lepto_sp1	7
			Trichorythodes	4
		Baetidae	*Bae_sp1*	31
	Plecoptera	Perlidae	*Anacroneuria*	10
		Gripopterygidae	*Limnoperla*	56
	Zygoptera	Calopterygidae	*Hetaerina*	23
		Coenagrionidae	*Acanthagrion*	6
			Coen_sp1	6
			Coen_sp2	43
		Megapodagrionidae	*Allopodagrion*	23
		Protoneuridae	*Peristicta*	7
	Anisoptera	Libellulidae	*Lib_sp1*	2
			Lib_sp2	4
		Cordulidae	*Navicordulia*	3
			Cord_sp2	3
		Gomphidae	*Cacoides*	4
			Phyllocycla	7
		Aeshnidae	*Aesh_sp1*	1
	Lepidoptera	Pyralidae	*Nymphulinae*	1
		Lep sp1	*Lep_sp1*	1
	Diptera	Chironomidae		5120

		Simulidae		541
		Ceratopogonidae		73
		Tipulidae		38
	Hemiptera			10
Crustacea	Decapoda	Trichodactylidae	*Trichodactylus*.	11
		Aeglidae	*Aegla*	2
Annelida	Oligochaeta			326
	Hirudinea			86
Mollusca	Gastropoda	Ampullariidae	*Pomacea*	3
		Ancylidae	*Gundlachia*	14
			Drepatonema	5
		Hydrobiidae	*Potamolithus*	15
			Heleobia	1
		Planorbidae	*Drepanotrema*	4
	Bivalvia	Sphaeriidae	*Pisidium*	95
Nematyelminthes				2