Universidade Federal do Rio Grande do Sul
Instituto de Biociências
Programa de Pós-Graduação em Ecologia

Dissertação de Mestrado

DECOMPOSIÇÃO FOLIAR DE ESPÉCIES PIONEIRAS E MACROFAUNA DE SOLO EM ECOSSISTEMAS DEGRADADOS PELA DEPOSIÇÃO DE CINZAS E EXTRAÇÃO DO CARVÃO

Luciana Regina Podgaiski

Porto Alegre, abril de 2009.

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Dissertação apresentada ao Programa de Pós-Graduação em Ecologia, do Instituto de Biociências, 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 2009.

À Odette de Mello Colombo, minha super avó, por todo amor e carinho dedicados a mim durante ¼ de século.

“Just to hear,
just to hear again:
Don’t matter how you go ....
Keep going!”

“Keep pray and trust the way is opening
Keep going!
Just keep going like me”

OS THE DARMA LOVERS

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RESUMO

Dois estudos envolvendo o processo de decomposição foliar de espécies pioneiras e os organismos da macrofauna de solo foram realizados em ecossistemas degradados pela deposição de cinzas e extração do carvão no Rio Grande do Sul, Brasil. Os estudos resultaram em três artigos científicos. O primeiro estudo (1° artigo) acessou a decomposição foliar e a colonização da macrofauna nos folhiços do capim-bermuda (*Cynodon dactylon*- Poaceae), da mamona (*Ricinus communis*- Euphorbiaceae), e da aroeira-vermelha (*Schinus terebinthifolius*- Anacardiaceae), plantas participantes da sucessão natural inicial, em áreas de deposição de cinzas leves e cinzas grossas/escória na mata ciliar do Rio Jacuí, em São Jerônimo. O segundo artigo – originário deste primeiro estudo-tratou exclusivamente sobre os padões de colonização de espécies de tatuzinhos de solo (Isopoda), grupo de animais detritívoros com alta abundância no local, no folhiço das mesmas três plantas. Ainda, compararam-se as suas abundâncias e algumas características reprodutivas entre os dois depósitos de cinzas. O segundo estudo (terceiro artigo) testou uma técnica de manejo para acelerar o processo da decomposição foliar de *Pinus elliottii* (Pinaceae) e enriquecer a fauna de macroartrópodes de solo em uma floresta monodominate de pinus sobre solo minerado e reconstruído topograficamente no município de Minas do Leão. Todos os estudos foram realizados empregando-se a técnica de bolsas-de-folhiço, que consistiu em bolsa de nylon de 30 X 20 cm e malha de 0.2 X 1.0 cm. A decomposição foi medida calculando-se a perda de massa foliar seca nas datas amostrais (6, 35, 70 e 140 dias após a exposição das folhas– 1° e 2° artigo ; 3 e 6 meses– 3° artigo). Foram realizadas análises de macronutrientes do material foliar remanescente. A macrofauna (indivíduos > 2 mm) foi retirada manualmente das bolsas-de-folhiço em laboratório, ou com extrator de Berlese-Tüllgren modificado (3° artigo), separada em ordens e morfoespeciada. Foi constatado que a mamona apresentou decomposição foliar mais rápida (*k*= 20.7) e maior densidade de indivíduos da macrofauna no seu folhiço, mas, no entanto, apresentou menor riqueza de espécies do que as outras plantas pioneiras. Os folhiços do capim-bermuda e da aroeira-vermelha foram similares com relação à decomposição e a diversidade da macrofauna. As espécies de tatuzinhos colonizaram igualmente as três espécies de plantas ao longo do tempo. Apesar da decomposição foliar não ter sido influenciada, a ocorrência de *Atlantoscia floridana* (Isopoda), a abundância e fecundidade de fêmeas de *Benthana taeniata* (Isopoda), e a composição total de espécies da macrofauna foram afetadas pelos diferentes depósitos de cinzas de carvão. Como esperado, a adição de folhas de plantas nativas sobre o solo homogêneo da floresta monodominante de pinus em solo minerado modificou a composição química do folhiço de pinus e aumentou a diversidade dos macroartrópodes de solo, demonstrado ser uma estratégia potencial ao manejo e à restauração ecológica. Estes estudos têm como metas contribur ao entendimento sobre a ecologia (processos ecológicos e interação organismos – folhiço) de áreas degradadas pela deposição de cinzas e áreas de extração e processamento do carvão no sul do Brasil.

ABSTRACT

Two studies concerning the leaf decomposition of pioneer plant species and the soil macrofauna organisms were performed in ecosystems degraded by the deposition of ashes and coal extraction in Rio Grande do Sul, Brazil. These studies resulted in three scientific articles. The first study (primary article) accessed the decomposition and the macrofauna colonization among leaves of bermuda-grass (*Cynodon dactylon* - Poaceae), the castor oil plant (*Ricinus communis* - Euphorbiaceae), and the Brazilian peppertree (*Schinus terebinthifolius* - Anacardiaceae); all are plants that participate in the initial natural succession in areas of deposition of fly ashes and boiler slag in Rio Jacuí's ciliar forest, located in São Jerônimo. The second article (derived from the primary study) exclusively analyzes the colonization patterns of woodlice species (Isopoda), a detritivorous group with high abundance at the site, among the leaf-litter of the same three plants. Their abundances and some reproductive characteristics were compared between the two deposits of ashes. The second study (tertiary article) tested a technique of management to motivate the leaf decomposition of *Pinus elliottii* (Pinaceae), and to enrich the soil macroarthropod fauna in a pinus monospecific forest with spontaneous establishment on a mined and rebuilt soil located in Minas do Leão. All of these studies were accomplished using the litter bags technique, which consists of a 30 X 20 cm nylon bag and a 0.2 X 1.0 cm mesh. The decomposition was measured by calculating the leaf-litter mass loss through multiple sampling dates (6, 35, 70 and 140 days after leaf expositions - primary and secondary articles; or 3 and 6 months after leaf expositions - tertiary article). The nutrient contents in the remaining leaf-litter were analyzed. The macrofauna (individuals > 2 mm) were removed manually from the litter bags in the laboratory, or with a modified Berlese-Tüllgren extractor (tertiary study), and the microfauna were separated by orders and morphospecies. We verified that the castor oil plant presented a faster decomposition rate (*k* = 20.7) and a larger density of macrofauna individuals, but presented a smaller species richness compared to the other pioneer plants. The leaf-litter of the bermuda-grass and the Brazilian peppertree were similar in terms of decomposition and macrofauna diversity. The woodlice species colonized the three species of plants equally along the time scale. Despite no effects on decomposition, the occurrence of *Atlantoscia floridana* (Isopoda), the abundance and fecundity of *Benthana taeniata* females (Isopoda), and the total macrofauna species composition was strongly affected by the different deposits of coal ashes. As expected, the input of leaves from native plants on the homogeneous soil of the pinus forest modified the chemical composition of the pinus leaf-litter and increased the soil arthropod diversity, showing that it could be a potential strategy for management and ecological restoration of damaged areas. These studies aimed to contribute to the understanding about ecological processes and the interaction between organisms and litter in areas with extraction and processing of coal in southern Brazil.

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ANEXOS

INTRODUÇÃO GERAL

A maioria dos ecossistemas terrestres no mundo apresenta áreas degradadas, resultantes de atividades antrópicas tais como a agricultura e a mineração. Os impactos ambientais oriundos da mineração do carvão são bastante complexos nos ecossistemas, causando significativas alterações na paisagem, destruindo a biota e gerando uma grande quantidade de resíduos. O consumo de energia gerada pelo carvão mineral atualmente no Brasil alcança cerca de 6,2 % da nossa oferta de energia interna (Brasil e EPE 2008). As maiores reservas de carvão no Brasil estão situadas na região sul, apresentando o Rio Grande do Sul cerca de 90 % das reservas. Todavia, o estado que contempla a maior produção é Santa Catarina, uma vez que o minério do Rio Grande do Sul apresenta elevado teor de impurezas e é pobre do ponto de vista energético (ANEEL 2008).

A região carbonífera do Baixo Jacuí, depressão central do Rio Grande do Sul, é constituída pelos municípios: São Jerônimo, Barão do Triunfo, Charqueadas, Arroio dos Ratos, Butiá, Minas do Leão, General Câmara, Triunfo e Eldorado do Sul (Fig. 1; Souza e Bittencourt 2000). A mineração do carvão nesta região iniciou-se pela via subterrânea, mas ao passar dos anos, a lavra a céu aberto passou a constituir a principal técnica de extração no estado (Guerra 2000).

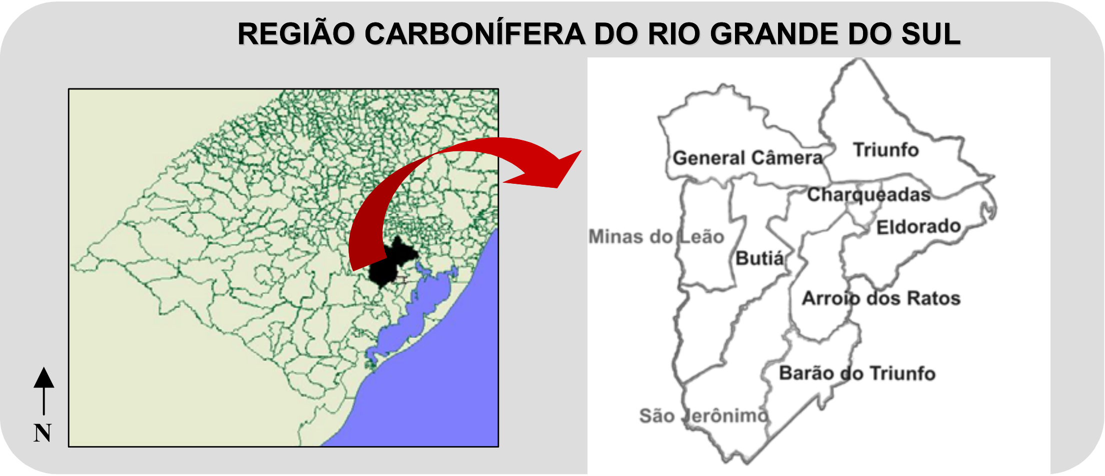

Figura 1: Municípios constituíntes da região carbonífera do Baixo Jacuí, no Rio Grande do Sul. Limites nos paralelos 29°37' e 30°38' de latitude sul e 51°15' e 51°14' de longitude oeste.

De acordo com o § 2.° do art. 225 da Constituição Federal de 1988 todas as áreas prejudicadas pela mineração deveriam obrigatoriamente ser recuperadas no Brasil, mas, poucos estudos têm sido realizados enfocando a recuperação ambiental destas áreas (Prochnow e Porto 2000). A reconstrução de solos após a mineração necessariamente deveria seguir cuidados e regras especiais, entre elas, a reposição das camadas de solo na seqüência original de sua retirada, preservando seus horizontes. No entanto, os "solos reconstruídos" são fundamentalmente antropogênicos, apresentando uma série de desequilíbrios em seus atributos químicos, físicos e biológicos (Kämpf et al. 2000), apesar da topografia recuperada. Kämpf et al. (2000) identificou que solos reconstruídos podem apresentar acidificação, alta densidade, baixa porosidade, baixa condutividade hidráulica e baixa retenção hídrica. Estes parâmetros sugerem a existência de inúmeros fatores limitantes ao estabelecimento da vegetação, assim como à restauração da paisagem e da biodiversidade nestes locais.

O processo de combustão do carvão fóssil em usinas termelétricas produz uma imensa quantidade de resíduos, oriundos do alto teor de matéria mineral associada ao carvão (Guerra 2000; Rohde et al. 2006). Três centrais termelétricas a carvão fóssil estão em atividade no Rio Grande do Sul, sendo duas delas localizadas na região carbonífera tradicional: São Jerônimo e Charqueadas. A termelétrica de São Jerônimo (Companhia de Geração Térmica de Energia Elétrica, CGTEE) está situada às margens do Rio Jacuí, e constitui a menor unidade geradora de eletricidade em operação no Brasil, com potência de 20 MW (ANEEL 2008). Apesar da menor potência, esta usina tem capacidade para gerar cerca de 55.000 toneladas de cinzas por ano (5.000 t – cinzas leves; 50.000 t - cinzas grossas/escória; Rohde et al. 2006).

As cinzas são materiais silico-aluminosos, sendo SiO_2 e Al_2O_3 seus principais componentes (Rohde et al. 2006). Cinzas leves, também denominadas volantes (*fly ash*), são formadas por partículas finas (menores do que 0,15 mm) que são arrastadas pelos gases de combustão nas fornalhas. As cinzas grossas, ou escória (*boiler slag*), são retiradas do fundo das fornalhas, apresentando altos teores de carbono não queimado, em granulometria grosseira e blocos sintetizados (Rohde et al. 2006). A composição química das cinzas varia de acordo com a qualidade do carvão parental e da tecnologia de combustão, mas poluentes como metais pesados e compostos orgânicos são geralmente associados a elas, em menor ou maior quantidade, dependendo do tipo de cinza. Apesar do grande potencial do uso dos resíduos carboníferos na indústria de construção civil (Rohde et al. 2006), grandes quantidades de cinzas são depositadas nas cavas de extração do minério, ao fechamento das mesmas, ou indiscriminadamente nos arredores das unidades geradoras (Teixeira et al. 1999). Os depósitos de cinzas sobre o solo na região carbonífera do Baixo Jacuí datam desde a década de 30 até os dias atuais (Guerra 2000), e consistem em grave impacto ambiental contaminando o ambiente, modificando aspectos estruturais, físicos e químicos do solo, prejudicando a biota e influenciando, de uma maneira indireta, em todos os processos do ecossistema (Carlson e Adriano, 1993).

As áreas afetadas por atividades relacionadas ao uso e processamento do carvão constituem áreas degradadas por definição, uma vez que sua capacidade de “retornar” ao estado original, através de seus meio naturais, é drasticamente diminuída (Reis et al. 1999). Contudo, a colonização vegetal destas áreas por espécies pioneiras agressivas e resistentes é um fato (Prochnow e Porto 2000, Centro de Ecologia 2002, Shu et al. 2005, Azzolini 2008). As plantas pioneiras que se sujeitam a esta colonização primária, partilham características ecofisiológicas peculiares, como o alto potencial de dispersão, taxas de crescimento populacionais rápidas (Gotelli 2007), baixos requerimentos fisiológicos e de substrato e, muitas vezes, resistência a metais pesados e outros poluentes (Tordoff et al. 2000, Whiting et al. 2004). Como conseqüência da colonização primária, as condições abióticas locais podem ser modificadas ao longo do tempo, o solo pode ser estabilizado, sombreado, aerado e fertilizado pela produção de matéria orgânica. Este processo pode se refletir em uma facilitação, ou *nucleação* (Yarraton e Morrison 1974), favorecendo a chegada de outras plantas com maiores requerimentos na comunidade, bem como de animais em busca dos recursos disponíveis, e seus predadores.

O aporte de matéria orgânica no solo é indispensável para a manutenção da ciclagem de nutrientes dentro do ecossistema. Esta matéria orgânica, liberada pelas plantas sob forma de folhas e galhos no solo (folhiço), é passível de desintegração pela interação de agentes abióticos e biológicos (Lavelle et al. 1993). Os agentes abióticos envolvidos neste processo são os fatores climáticos - temperatura, luz, umidade; os bióticos são os organismos detritívoros (invertebrados) e os decompositores (fungos e bactérias). Os invertebrados detritívoros são responsáveis pela fragmentação do folhiço e deposição de grandes quantidades de pelotas fecais no solo, o que estimula enormemente a ação dos decompositores (Weeb 1977). A qualidade química do folhiço, medida em termos de relação carbono-nitrogênio (C:N), concentrações de macronutrientes (especialmente N), lignina e fenóis, também é essencial para predizer a facilidade da ação dos organismos e da decomposição (Zhang et al. 2008). O incremento da produtividade primária é uma conseqüência positiva desta ciclagem.

O folhiço sobre o solo favorece a chegada e o estabelecimento da fauna de invertebrados do solo. Os invertebrados do solo são muito diversos nos ecossistemas, representando 23 % da diversidade total dos organismos vivos descritos até o momento (Lavelle et al. 2006), contudo, eles são reconhecidamente afetados por atividades antrópicas e pela contaminação ambiental (Rusek et al. 2000). Os invertebrados participam de complexas cadeias alimentares, pertencendo aos mais variados níveis tróficos (Coleman et al. 2004). O folhiço oferece importante hábitat e alimentação a estes organismos, e suas características podem ser determinantes à diversidade de espécies que suporta. Como a alta heterogeneidade da superfície do solo é intrínseca a muitos ecossistemas, seja pela diversidade do folhiço ou por outras estruturas como rochas, é fundamental que estratégias de restauração de hábitats levem estes fatores em consideração (Podgaiski et al. 2007). O incremento na heterogeneidade de hábitats é altamente correlacionado ao aumento na diversidade de microhábitats e microclimas, alimentos, refúgios e sítios para atividades oferecidos aos invertebrados de solo, e desta forma vem ao encontro da conservação destes organismos e ao enriquecimento de hábitats.

A inserção de fundamentos ecológicos dentro das práticas de restauração de áreas degradadas e conservação da biodiversidade é um grande desafio. Antes de tomadas de decisões, se faz necessário um entendimento mínimo sobre o funcionamento e a dinâmica dos sistemas a serem contemplados (Engel e Parrota 2003), como, por exemplo, sobre seus históricos de perturbação, composição de espécies vegetais, processos e interações ecológicas (Eviner e Hawkes 2008). Informações de referência e bases teóricas são essenciais ao sucesso das ações. Muitas vezes, a melhor estratégia pode ser não manejar, e sim deixar com que a colonização natural e espontânea dirija à sucessão e à restauração. Outras vezes, é necessário subsidiar o sistema, oferecendo condições para que a biodiversidade e os processos sejam encarecidos. De uma forma geral, estratégias nucleadoras (Reis et al. 2003), que favoreçam a reconstrução da diversidade espacial dos hábitats (Isaacs et al. 2009, Samways 2007) e facilitem a sucessão e as interações interespecíficas (Reis et al. 1999, Silva 2003, Reis e Kageyama 2003) são importantes para manutenção da biodiversidade e a restauração de ambientes degradados.

Artigos científicos

Nesta dissertação, foram realizados estudos contemplando assuntos como o processo ecológico de decomposição de folhiço de plantas pioneiras e interação da diversidade de organismos do solo em áreas altamente degradadas pela ação do homem, servindo como subsídios à ecologia da restauração. Dois estudos foram realizados em áreas de diferentes etapas do processamento do carvão fóssil. Três artigos foram confeccionados: dois referentes ao primeiro estudo, e um referente ao segundo. Os primeiros artigos são entitulados respectivamente como:

- *"Decomposição foliar de plantas pioneiras e a comunidade da macrofauna de solo em um sistema com depósito de cinzas de carvão sofrendo sucessão natural"* e*"Tatuzinhos neotropicais (Crustacea; Isopoda) colonizando o folhiço de plantas pioneiras em um ambiente com depósitos de cinzas de carvão"* e apresentam como problemática a contaminação ambiental provinda em longo prazo por depósitos irregurales de cinzas de carvão em uma area adjacente à usina termelétrica (floresta ripária) de São Jerônimo, RS (Fig. 2).

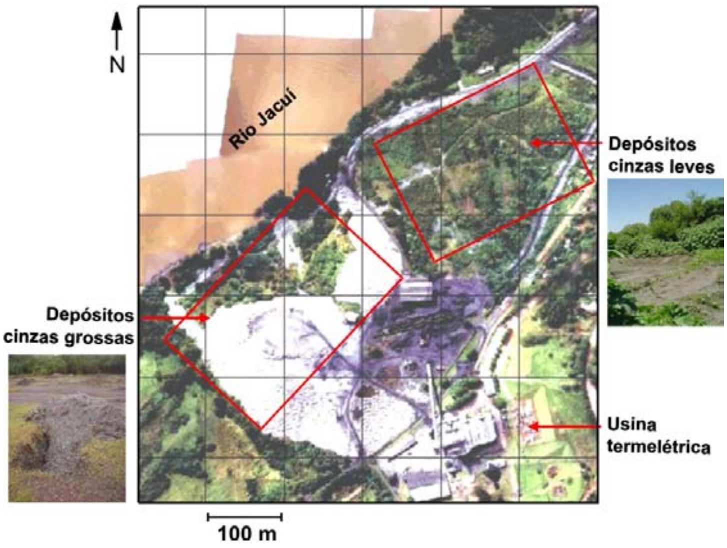

Figura 2: Depósitos irregulares de cinzas de carvão em mata ripária do Rio Jacuí no município de São Jerônimo, RS. Área de estudo dos artigos 1 e 2. Mapa modificado de Centro de Ecologia (2002).

O terceiro artigo da dissertação, entitulado "*A adição de folhas mistas no solo melhora a diversidade de macroartrópodes e a decomposição de serapilheira em uma floresta de pinus pós-minerada?*" é retratado em uma floresta monodominate de pinus, estabelecida espontaneamente sobre um solo minerado e reconstruído topograficamente, em Minas do Leão, RS (Fig. 3).

Figura 3: Mineração, reconstrução topografica do solo e estabelecimento espontâneo de pinus, no município de Minas do Leão, RS. (Área de estudo do artigo 3).

A abordagem do primeiro artigo refere-se especificamente ao processo de decomposição foliar de três plantas pioneiras (Fig. 4) e a colonização deste folhiço pela macrofauna de solo (organismos > 2 mm) no processo de decomposição. As plantas pioneiras escolhidas para o trabalho foram: uma gramínea exótica (capim-bermuda - *Cynodon dactylon*), um arbusto exótico (mamona - *Ricinus communis)* e uma árvore nativa (aroeira-vermelha - *Schinus terebinthifolius*). Todas são consideradas pioneiras e apresentam representatividade na área de depósitos de cinzas, desempenhando um importante papel no processo natural da sucessão ecológical destas áreas (Azzolini, 2008). Ainda neste artigo, são comparados possíveis efeitos de dois tipos de depósitos de cinzas (cinzas leves e cinzas grossas/ escória; Fig.2) sobre a decomposição foliar e a macrofauna de solo.

Capim-bermuda *Cynodon dactylon* — **Mamona** *Ricinus communis* — **Aroeira-vermelha** *Schinus terebinthifolius*

Figura 4: Plantas pioneiras na área de depósito de cinzas, mata ripária do Rio Jacuí, São Jerônimo, RS. Área de estudo dos artigo 1 e 2.

O segundo artigo se refere estritamente à colonização do folhiço destas três plantas pelas espécies do grupo de artrópodes de solo mais abundante na área degradada: os isópodas, ou mais comumente conhecidos, os tatuzinhos. Estes organismos têm hábitos detritívoros, e uma imensa contribuição ao processamento do folhiço (Quadros e Araujo 2008) e à ciclagem de nutrientes nos ecossistemas. Mudanças em suas características de história de vida, como crescimento e reprodução (Donker et al. 1993), são fortemente relatadas em situações de contaminação ambiental por metais pesados. Desta forma, a influência do tipo de depósito de cinza também é testada sobre a densidade e características reprodutivas dos isopodas neste trabalho.

O terceiro e último artigo é um trabalho de teste de hipóteses ecológicas, com especulações visando estratégias à restauração ecológica. Visto forte impacto ambiental na área de estudo após a mineração, a adição de folhas de espécies nativas mistas (Fig. 5) é testada como uma estratégia de enriquecimento da fauna de solo e incentivo à decomposição de folhiço do *Pinus elliottii* - a espécie pioneira e monodominante no sistema.

Figura 5: Adição de folhas de espécies nativas ao substrato da floresta monodominante de pinus como uma estratégia para incrementar a diversidade da fauna de solo e a ciclagem de nutrientes. (Artigo 3).

Métodos

Os experimentos foram delineados levando-se em consideração a replicabilidade das bolsas-de-folhiço, princípios de aleatoriedade e a utilização de blocos, para redução da heterogeneidade ambiental das áreas de estudo (Gotelli e Ellison, 2004). Para os estudos de decomposição foliar e colonização da fauna de solo foi utilizada uma técnica largamente conhecida (Wieder e Lang, 1982): os "litter bags", ou bolsas-de-folhiço (veja também Rodrigues, 2006). Esta técnica consiste basicamente na adição de material foliar seco, com massa conhecida, em uma bolsa que é disposta ao ambiente para a decomposição. Após um período determinado, as bolsas-de-folhiço são retiradas do ambiente e levadas ao laboratório onde o material foliar remanescente é seco em estufa até temperatura constante, e pesado em balança de precisão. Calcula-se a massa decomposta considerando-se a massa inicial e a final (remanescente) em um determinado tempo. Uma abordagem utilizada em dados de decomposição é o ajuste de um modelo matemático que estima a constante que descreve a perda de massa ao longo do tempo (-*k*). O modelo mais freqüentemente utilizado é o de decaimento exponencial simples, considerado por Olson (1963), o qual se aproxima da biologia da decomposição correspondente a uma perda de massa mais acentuada nos primeiros dias (componentes solúveis e compostos fáceis de serem degradados) e mais demorada em longo prazo (materiais recalcitantes) (Wieder e Lang, 1982). A equação é assim descrita: $X/ X_0= e^{(-kt)}$, em que X_0 é a massa inicial, X é a massa remanscente no tempo t (anos), "*e*" é a base de logaritmo natural e –*k* é o coeficiente de processamento da decomposição.

Dependendo dos objetivos do trabalho, a malha da bolsa-de-folhiço pode apresentar diferentes tamanhos. Malhas pequenas, de aproximadamente 1 mm^2, são as mais utilizadas em ambientes terrestres (Gartner e Cardon, 2004), mas no entanto excluem organismos de solo de maior tamanho a participarem da decomposição. Malhas largas, apesar de permitirem a passagem da macrofauna (> 2 mm), por outro lado também facilitam maior perda de fragmentos foliares. Tendo isto em vista, nesta dissertação, em todos os experimentos, foram utilizadas bolsas-de-folhiço com tamanho 30 X

20 cm e malha de 10 x 2 mm, (Fig. 6). Esta malha permitiu a colonização da macrofauna do solo.

Figura 6: Bolsa-de-folhiço utilizada nos experimentos do projeto.

Em campo, as bolsas-de-folhiço foram coletadas do solo e condicionadas em sacos plásticos. Em laboratório, no caso do experimento de São Jerônimo, as bolsas foram dispostas em bandeijas brancas, nas quais os organismos representantes da macrofauna foram triados manualmente. No caso do experimento de Minas do Leão, as bolsas foram inseridas em extrator de Berlese-Tüllgren modificado, durante uma semana, e após, passaram por uma rápida triagem manual. Este extrator consiste em uma estrutura com forma de funil, que no topo apresenta uma fonte de calor e em baixo um recipiente coletor. O gradiente de temperatura e umidade faz com que os invertebrados migrem para baixo, e caiam nos coletores. Os invertebrados foram acondicionados em potes contendo álcool 80 % e identificados em morfoespécies e grupos tróficos funcionais. O tratamento dos dados e as análises estatísticas foram realizadas de maneira independente e peculiar para cada artigo da dissertação.

ARTIGO 1.

LEAF-LITTER DECOMPOSITION OF PIONEER PLANTS AND THE SOIL MACROFAUNA COMMUNITY ON A COAL ASH DISPOSAL SYSTEM UNDERGOING NATURAL SUCCESSION*

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Summary

We studied the leaf-litter decomposition of spontaneous pioneer plants, and the diversity of the associated soil macrofauna community in a riparian forest affected by coal ash disposals (fly ash and boiler slag) in Brazil. We conducted a litter bag experiment in the damaged area during a period of 140 days. We found that the decomposition rate of *Ricinus communis* leaf-litter was more than 80% faster (*k*-value 20.7) than the other appraised species. This result agrees with its low C:N ratio and high N (%), and increased detritivores. Whereas this leaf-litter had supported the highest densities, it presented the lowest morphospecies richness. The *Cynodon dactylon* and *Schinus terebinthifolius* leaf-litters were similar in decomposition rates and macrofauna diversity. Leaf-litter decomposition and macrofauna densities were not affected by the type of ash disposal system; on the other hand, the morphospecies composition was distinct in the different sites. The physical structure, an unfavorable pH-value and a heavy metal concentration in the ash disposal types may be selecting the more adapted species to the different environmental conditions.

Key words: boiler slag, early successional plants; fly ash; soil invertebrates

Introduction

Anthropogenic habitat modifications are likely to have a major impact on the composition and biodiversity of the Earth (Tilman, 1994). Alterations of biota have modified ecosystem goods and services, which are very difficult to revert (Hooper at al., 2005). In face of the global change phenomena, conservation and restoration of biodiversity requires an immense increase in our knowledge, such as on environmental constraints, drivers of biota diversity and ecological processes. Such knowledge is essential in designing an ecological theory to understand and manage ecosystems, communities, and species in a suitable way (Callaham et al., 2008).

Terrestrial ecosystems impacted by pollutant industrial activities generally present altered biodiversities and ecosystem proprieties. Natural areas that were polluted by waste deposits from coal combustion, for example, show leaching of potentially toxic trace elements, such as heavy metals, a lack of essential nutrients and an inappropriate physical structure of the substratum, which lead to: (1) reductions in plant establishment and growth, (2) changes in plant elemental composition, and (3) an increased cycling of toxic elements through the food chain (Carlson and Adriano, 1993). Also, natural succession processes in mine tailings are generally very slow, requiring many years for changes to become apparent (Shu et al., 2005). Environments with heavy metal contamination show decreases in litter decomposition rates (Coughtrey et al., 1979; Giller at al., 1998; McEnroe and Helmisaari, 2001), mainly due to inhibited abundance, the diversity and feeding performance of soil detritivores and the microbial activity (Strojan, 1978; Van Wensem, 1997; Loureiro et al., 2006; Filzek et al., 2004; Kools et al., 2008).

Current approaches to the restoration of contaminated environments include the use of technical reclamation, generally relating to the usage of covering systems, and sowing or planting target species (Tordoff et al., 2000; Dutta and Agrawal, 2003; Casselman et al., 2005). However, lands are sometimes abandoned after use, allowing for spontaneous biota establishment through primary succession (Hodacová and Prach, 2003). Early successional plants need to have suitable adaptive strategies (Shu et al., 2005), and over time, they might facilitate the arrival of other species by ameliorating the harsh soil conditions (Schulze, 2005). Although slower, this natural process may lead to a more natural and richer vegetation cover (Hodacová and Prach, 2003).

As belowground and aboveground compartments of terrestrial ecosystems are closely linked, the effects of one may be felt by the other (Hooper et al., 2000). Through plant litter input, resources are provided to soil biota; which in turn is responsible for most of the decomposition of the system. The decomposition process releases nutrients for plants, which increases their productivity (Wardle, 1999; Hooper et al., 2000). Impacts that affect plant community structure and abundance can induce changes in the soil food webs and decomposition rates by altering the decaying litter material entering the soil. As plant species with different traits differ in both the quality and quantity of resources offered, they might support soil animal communities that demand different requirements of food and shelter (Wardle et al., 2004; Wardle, 2006). The litter decomposition rates are greatly influenced by their chemical properties, especially increasing in litters with a low C:N ratio and high N, which varies among species and across habitats (Gartner and Cardon, 2004; Zhang et al., 2008). However, leaf-litter from herbaceous plant species generally decomposes more easily than that from woody plant species (Zhang et al., 2008), which could represent a trade-off between food and shelter for animals from the belowground communities (Hooper et al., 2000). Both soil invertebrates and soil processes may be driven by the forest dynamic and management (Salmon et al., 2008; Lindsay and Cunningham, 2009).

Animals in soils are numerous and constitute a diverse group of species that are organized into complex food webs (Coleman et al., 2004). They have a fundamental role in the delivery of ecosystem services and goods by the soil, contributing to water storage and detoxification, nutrient cycling, soil formation, primary production, flood and erosion control, and climate regulation (see Lavelle et al., 2006). Within the soil fauna, the macroinvertebrates represent animals that are relatively large (body size > 2 mm; Swift et al., 1979) and that live in the surface litter, in nests or in burrows (Lavelle et al., 2006). This group supports organisms that interact in different ways with their environment, belonging to several trophic levels in the food web (Coleman et al., 2004), representing animals from detritivores (e.g., Isopoda and Diplopoda) to true herbivores (e.g., some Hemiptera and Gastropoda), predators (e.g., Araneae and Chilopoda) and omnivores (e.g., Opiliones and some Hymenoptera).

In this study, we assessed leaf-litter decomposition of early successional plants and the associated soil macrofauna community structure in a riparian forest that was affected by activities of a coal power plant. This ecosystem has been receiving coal combustion waste disposals (fly ash and boiler slag) for a long time, and after having disturbances interrupted, it is undergoing a spontaneous succession process. This early succession has been driven mainly by exotic plants, which are improving the substrate and microsite conditions for other native species (Azzolini, 2008). For our experiments, we chose three abundant plants, which represent different life forms: the bermuda-grass *Cynodon dactylon* (L.) Persoon (Poaceae; exotic), the shrub castor oil bean *Ricinus* *communis* L. (Euphorbiaceae; exotic) and the Brazilian peppertree *Schinus terebinthifolius* Raddi (Anacardiaceae; native). Specifically, we addressed three questions about this system. (1) Which pioneer leaf-litter decays more quickly, releasing nutrients to the ash substrate? (2) Does the leaf-litter identity influence the diversity of the colonizing soil macrofauna and its functional trophic groups? (3) Does the ash disposal type affect leaf-litter decomposition and soil macrofauna?

Methods

Study area

The study area consisted of about 9 ha and belonged to the riparian forest of the Jacuí river, in São Jerônimo (29°57’55.6”S; 51°44’14.9”W), in the Central Depression region of Rio Grande do Sul state, Brazil. In this region, the climate is temperate, with a hot summer, but without a dry season (Cfa type, according to the Köppen-Geigen climate classification; Peel et al., 2007). Coal combustion wastes from São Jerônimos’s thermal power plant were landfilled in the study area for more than 30 years. Nowadays, the area stopped receiving additional residues, and natural succession is proceeding (Fig. 1A).

Two different kinds of coal wastes were disposed in the area: fly ash and boiler slag (Fig. 1B, 1C). Fly ash is a fine-grained powder with spherical particles, and boiler slag is molten-grained with angular particles. The trace element affinities are similar for both wastes, but fly ash is generally enriched by elements with calcium oxide-sulfate affinities, and boiler slag, by elements with iron oxide affinities (Querol et al., 1995). The elemental concentrations of wastes vary according to the parent coal composition and combustion technology. In São Jerônimos's thermal power plant, which burns coal from the Rio Grande do Sul, there is a production of fly ash and boiler slag. The most abundant heavy elements from wastes are Sn, Ni and Mo, and those lower in proportion are Cr, As, Hg, Al, Pb, Mn, V, Cd, Ba and Zn (Rohde et al., 2006).

Throughout the decades, the waste disposals in the study area were heterogeneously accomplished at several times. Old and new deposits can be found side by side. According to the landscape management history, we can classify two distinct sites: one that received more fly ash and another that had more boiler slag disposals. The sites are 200 m apart. Hence forth, they will be called the fly ash site and the boiler slag site, respectively.

Vegetation from the early successional stages dominates both sites. However, as the plant establishment was driven by the chronology of the ash deposits, the vegetation physiognomy is also heterogeneous. Exotic and spontaneous plants such as the Bermuda-grass *C. dactylon* and the castor oil plant *R. communis* assume a prominent role in the recovery of new disposals in both sites. *Mimosa bimucronata* (De Candolle) O. Kuntze (Leguminosae) is found constituting patches, especially in the boiler slag site. Common wood species in both sites are the Brazilian peppertree *S. terebinthifolius*, camboatá-vermelho *Cupania vernalis* Cambessedes (Sapindaceae) and açoita-cavalo *Luehea divaricata* Martius (Tiliaceae). Azzolini (2008) provides a list of plant species from the area.

Plant species

For our study, we chose three abundant pioneer plants from the study area, representing species from three different life forms and successional stages: a grass (*C. dactylon*), a shrub (*R. communis*), and a tree (*S. terebinthifolius*) (Fig. 1D, 1E). The three species are broadly known by their spontaneous behavior in disturbed and natural habitats around the world (GISP, 2005). *Cynodon dactylon* is a perennial grass, which has both rhizomes and stolons; its leaf blades measure about 8.0 X 0.4 cm. *Ricinus communis* is a perennial shrub, reaching 2–3 m in height, has palmate leaves measuring around 15 X 45 cm and has 7-11 lobes. *Schinus terebinthifolius* is a native tree from the study region, which grows 5-10 m in height, has composite leaves with 3 to 10 pairs of leaflets that measure around 5 X 2.5 cm.

Physical and chemical characteristics of the ash disposals sites

We used twenty-four substrate core samples (approx. 10 cm deep and 5 cm diameter) from the disposal sites (12 *per* site) to analyze the pH and the water retention capacity. Sampling was done in eight plots; three samples were nested within each plot. In the laboratory, the same dry weights of substrate (100 g) were conditioned in pots with several homogeneous perforations in the bottom. We added water to the pots until saturation was reached. Water retention was calculated by dividing the weight of the water retained by the dry weight of the substrate. As the soil's ability to retain water is related to particle size, we expected that the fly ash site would retain more water than the boiler slag site. However, the sites did not differ in their capacity of water retention (Nested ANOVA, $F_{1,6}$= 1.87, P= 0.22). The pH-values of both sites were acidic, but we found that the boiler slag disposal site was significantly more acidic than the fly ash site (Nested ANOVA; $F_{1,6}$ = 32.38, P = 0.001; Fig. 2).

Further, two samples of substrate from each of the sites were analyzed for nitrogen (N), phosphorus (P), potassium (K), calcium (Ca), magnesium (Mg), sulfate

(SO_4^{-2}) and boron (B) content at the UFRGS Laboratory of Soil Analysis. After digestion in H_2O_2-H_2SO_4, N was accessed by distillation. P and K were analyzed by using the Mehlich I method (Mehlich, 1953). Ca and Mg exchangeable were extracted in KCl mol L^{-1}; sulfate was extracted in $CaHPO_4$ 500 mg L^{-1} of P; and B was extracted with hot water. The elemental concentrations found in the sites are shown in the Table 1. The boiler slag site presented more N (%) and B (mg/dm^3) than the fly ash site, which in turn, had higher K concentrations (mg/dm^3).

Litter bag experiment

For the experiments of leaf-litter decomposition and colonization by macrofauna, we constructed nylon litter bags measuring 30 X 20 cm, with a coarse mesh size (10 x 2 mm). This mesh allows for the entry of macrofauna (including small individuals). We filled the litter bags with 20.3 ± 0.2 g of air-dried fresh leaves of the three pioneer plant species, which were collected from the study region. In June 2007, we placed a total of 96 litter bags in the field: 48 in the fly ash, and 48 in the boiler slag site. Four blocks were settled *per* site. Each plant species was replicated in four litter bags *per* block. At each sampling occasion (6, 35, 70 and 140 days of leaf-litter exposure), one litter bag *per* plant species was randomly removed from each block.

Laboratory procedures

In the laboratory, we dried (60°C; 72 h) and weighed the remaining leaf-litter from the litter bags. In the initial (t = 0), leaf contents of C, N, P and K (% of dry mass), and of litter from 140 days of decomposition, were analyzed for each plant in each site. C from the leaves was assessed by moisture combustion/Walkley-Black (Walkley and Black, 1934). N was determined using the Kjeldahl method (Kjeldahl, 1883). P and K were analyzed using nitric/perchloric acid digestion followed by determination on an ICP-OES (Inducted Coupled Plasma- Optical Spectrometer). All of the methods were performed at the UFRGS Laboratory of Soil Analysis.

Soil macrofauna

We collected the soil macrofauna by hand from the litter bags before drying the litter. We counted and separated the individuals into class, order, and family (whenever possible). Within each group, we classified the individuals in species or morphospecies. Immature insects (larvae), spiders, and very small cockroaches and snails were not classified. The individuals were assigned to four functional trophic groups: carnivores, detritivores, herbivores, or omnivores. Feeding assignments were based on the literature (e.g., Marinoni et al., 2003) and through personal communication with systematists. Carnivores include spiders, cursorial hunters and hematophagous insects; detritivores comprise saprophagous invertebrates, fungi feeders and litter grazers; herbivores consist of sap feeders and leaf chewers; and omnivores include species with mix feeding habits.

Data analyses

We estimated the single constant (k), which describes the loss of mass over time, for the three species while considering all litter bag data used in the study. The constant was estimated by fitting the single exponential decay model (Olson, 1963; Wieder and Lang, 1982):

$X/ X_0= e^{(-kt)}$,

where X_0 is the initial mass, X is the mass remaining at time t (year), *e* is the base of natural logarithm, and *–k* is the processing rate coefficient. This approach was performed solely in terms of ecological knowledge (facilitating comparison with other data sets; as in Zhang et al., 2008), and to obtain insights into the biology of the decomposition process of these species (Wieder and Lang, 1982). To compare the efficiencies of decomposition processes among the leaf-litter species and sites, we expressed the decomposition as the percentage of initial leaf-litter mass remaining after 6, 35, 70 and 140 days.

We used a nested analysis of variance (ANOVA) with repeated measures for testing the effects of site, leaf-litter species and time, in the leaf-litter mass remaining in the litter bags and the soil macrofauna density (individuals g^{-1} d.w. litter). The blocks were nested in the sites, and the time of leaf-litter exposure was used as a repeated measure. Prior to analysis, the data of fauna density were square root transformed to satisfy the parametric assumptions of normality of the residuals and the homogeneity of variances. Comparisons of the leaf-litter masses remaining from each of the species between the sites were performed with ANOVA, also considering time as a repeated measure.

As species richness is largely influenced by an individual's abundance in samples (Gotelli & Colwell, 2001), we used rarefaction curves with confidence intervals for the comparison of morphospecies richness among the leaf-litter species and the two sites. To compare the morphospecies composition between these same treatments, we performed ordinations using non-metric multidimensional scaling (NMDS; Kruskal, 1964) with the Bray-Curtis distance measure as an index of dissimilarity. We used a matrix with macrofauna morphospecie densities *per* plant specie within each block (temporal litter bags data summed). To reduce the noise of rare species, we removed the species that occurred only once in the matrix, thereby improving the interpretability of the ordination plot (McCune and Grace, 2002). Analysis of similarity (ANOSIM; Clarke 1993) on a Bray-Curtis distance matrix was performed to determine whether potential clustering patterns identified visually were statistically different. Both, rarefaction, NMDS and ANOSIM were performed with PAST 1.9 (Hammer et al. 2001).

Results

Leaf-litter decomposition

Leaf nutrient concentration is shown in Table 2. The N and P contents in *R. communis* leaves are more than 70% and 50% higher than the other leaf species, but their K content was low. The C:N ratio was lower for *R. communis* (8.04) than for *C. dactylon* (26.7) and *S. terebinthifolius* (34.17). K content was highest in *C. dactylon*, and the N and P contents were lowest in *S. terebinthifolius* leaves.

The decomposition rate of leaf-litter over time (single constant *k*) was more than 80% faster for *R. communis* (k = 20.7 g g^{-1} yr^{-1}, R^2 = 0.93), followed by *C. dactylon* (k = 3.5 g g^{-1} yr^{-1}, R^2 = 0.38) and *S. terebinthifolius* (k = 3.2 g g^{-1} yr^{-1}, R^2 = 0.73). More than 90% of the *R. communis* leaves decayed in 35 days. Less than 70% of *C. dactylon* and *S. terebinthifolius* leaves showed decay through the end of the experiment (140 days). The leaf-litter mass remaining in the litter bags was influenced by the interaction of leaf-litter species and the time of leaf-litter exposure (Table 3; Fig. 3A). *Ricinus communis* presented a higher mass loss at 35, 70 and 140 days than the other leaf-litter species. *Cynodon dactylon* presented more mass loss in 70 days than *S. terebinthifolius*, but in the other times their leaf-litter masses remained to not differ.

The leaf-litter mass decaying was not different among sites (Table 3; Fig 3B), neither when considering the individual leaf-litter species in an interaction with time (Table 4). Fly ash and boiler slag showed similar effects on the decomposition process. However, after 140 days of leaf-litter exposure, some chemical elemental contents were shown to be decreased in the boiler slag site than in the fly ash site, such as C in *C. dactylon*; K in *R. communis*; and C, N and K in *S. terebinthifolius* (Table 5).

Soil macrofauna diversity

We found a total of 2,573 individuals and 126 morphospecies of soil macroinvertebrates belonging to the classes Arachnida, Chilopoda, Crustacea, Diplopoda Gastropoda, Hexapoda and Oligochaeta colonizing the litter bags (Appendix 1). Rare species (≤ 2 individuals) were about 53% of the total richness. Isopoda was the most abundant group in terms of individuals (n = 987; 38%), followed by Oligochaeta (n = 501; 19%) and Hymenoptera (n = 407; 16%). Coleoptera and Hymenoptera presented the highest morphospecies richness; 54 and 24 morphospecies, respectively. The invertebrates with detritivorous habits represented 70.2% of the total individuals and 28 morphospecies. Omnivores represented 17.8% of individuals and 26 morphospecies. Carnivores had 10% of the individuals and 51 morphospecies, and herbivores had 2% of the individuals and 21 morphospecies.

In one litter bag, from the 70 day period, we found a great abundance of one morphospecie of ant (Formicidae; 258 individuals). As such insects have a social behavior, we considered this data as an outlier.

Soil macrofauna in the treatments

The total density of individuals presented significant effects from the interaction between leaf-litter specie and time (Table 3, Fig. 3C). *Ricinus communis* sheltered more individuals than the other species in 6, 35 and 140 days of decomposition. On the other hand, rarefaction curves showed that this leaf-litter supported the smaller morphospecies richness (Fig. 4A). *Schinus terebinthifolius* seemed to have a higher density of individuals in 70 days than the other species, but it could have been the reflex of an outlier produced by the increased ant abundance. In relation to density in other times and number of morphospecies, *C. dactylon* and *S. terebinthifolius* did not differ.

The disposal sites did not influence the total individual colonization (Table 3), although the tendency was increased in the fly ash site (Fig. 3D). The morphospecies richness did not differ between sites (Fig. 4B). As revealed by the ordination in two dimensions (NMDS, stress = 0.16), the overall community macrofauna composition was responsive to the site treatments, instead of the leaf-litter species (Fig. 5). Analysis of similarity showed weak but significant effects of the sites in the macrofauna (ANOSIM, R = 0.14, P = 0.014). Mainly due the increased number of rare species, we did not use ordination analysis for the functional trophic groups separately.

Functional trophic groups in the treatments

Detritivores

We found a significant interaction between leaf-litter species and time of leaf-litter decomposition for the detritivores (Table 6). *Ricinus communis* supported higher detritivore density during the first days of litter decomposition (Fig. 6A), but it had a lower total morphospecies richness (Fig. 7A). *Cynodon dactylon* and *S. terebinthifolius* did not appear to differ in terms of density and richness. The sites were similar concerning density (Table 6, Fig. 6B) and richness of detritivores (Fig. 7B).

Carnivores

We found significant interaction between leaf-litter species and time, and the effects of site in carnivore density (Table 6). *Schinus terebinthifolius* had lower densities at 6 days, and *R. communis* had higher densities at 140 days than the other species (Fig. 6C). The plants did not differ in number of carnivore morphospecies (Fig. 7C). The fly ash site had the highest density of individuals; on the other hand, the boiler slag site had the most morphospecies richness (Fig. 6D, Fig 7D).

Omnivores

We did not find any effects of leaf-litter species on the omnivore densities and richness (Table 6; Fig. 6E, Fig. 7E). The fly ash site had more morphospecies richness colonizing the litter bags than the boiler slag site (Fig. 7F), but no effects on density were detected (Table 6, Fig. 6F).

Herbivores

The densities and richness of herbivores were similar concerning the treatments (Table 6; Fig. 6G, H, Fig. 7G, H).

Discussion

Leaf-litter

Of the three species of pioneer leaf-litter appraised in this study, *R. communis* had the fastest leaf-litter decay. The *k* value found for this plant (20.7 g g^{-1} yr^{-1}) is very high when compared to other studies. For example, Zhang et al. (2008) reviewed 70 studies around the world and found *k* values ranging from 0.006 to 4.993 g g^{-1} yr^{-1}. Besides, *R. communis* leaves present a high N (%) and a low C:N ratio; they also supported the highest densities of detritivores during the first days of decomposition, which clearly explain its mass loss rates (Gartner and Cardon, 2004). Some spontaneous exotic species, such as *R. communis*, often maintain higher leaf N concentrations, decomposing more rapidly and releasing more nitrogen to the soil than native species (Levine et al., 2003; Ashton et al., 2005). The increase in decomposition rates and nutrient cycling may be negative within the equilibrium of natural ecosystems (Ehrenfeld, 2003; Wolfe and Klironomos, 2005); however, in our reclaiming system, the effects are certainly positive through the amelioration of nutrient availability into the coal ash substrate. Another aspect of *R. communis* is that its leaves contain substances such as ricin, ricinin and flavonoids, which present insecticidal and antimicrobial actions (Okongkwo and Okoye, 1992; Bigi et al., 2004). However, the amount of these substances may be reduced in the leaf-litter, which did not appear to directly affect the macrofauna once the organisms had an increased density in this leaf-litter, and the morphospecies composition had a negative response to the leaf-litter identity.

Leaf-litter is food for detritivores, who prefer to feed from palatable plants with a low C:N ratio and a high N (e.g., Zimmer 2002). Leaf litter is also a microhabitat for organisms of all functional trophic groups of soil fauna. The decomposition process may result in a trade-off between these two resources (food and microhabitat). A greater consumption of litter decreases the structural integrity that suits it for use as a microhabitat (Hooper et al. 2000). The high-quality litter of *R. communis,* as expected, was decomposed (consumed) very quickly; while at the same time, it supported a soil community with low species richness. This case may represent an example of such a trade-off, because the rapid loss of leaf-structure and its associated microhabitat diversity can be related to a decrease in the soil invertebrate diversity (Hooper et al. 2000). Despite marked differences in the initial leaf nutrients content and leaf structure, *C. dactylon* and *S. terebinthifolius* were similar in the decomposition rates and macrofauna diversity that they supported. *Cynodon dactylon* is more palatable for detritivores and easier to decompose than *S. terebinthifolius*. On the other hand, the broad leaves of *S. terebinthifolius* offer greater resource heterogeneity (as a role of habitat space to fauna) than the narrow leaves of *C. dactylon*. Taking into account these characteristics, we supposed that these species would differ in relation to the appraised ecological aspects. However, they did not appear to be distinguished in our study system.

In many ecosystems, and especially in the system studied here, plant establishment, productivity and changes related to succession basically depend on the recycling of nutrients and the amelioration of microsite conditions. Soil biota development also requires adequate organic resources to be sustainable, which is provided by the plant community by the production of surface litter layers. Our results concerning litter decomposition and the associated soil fauna community complement the research of Azzolini (2008) in some presuppositions about the natural plant succession in this coal ash-contaminated area. Firstly, C. *dactylon* has an important role in rapidly establishing a cover in recent ash disposals, which stabilizes the substrate. Plantules of *R. communis* occur together with *C. dactylon*, but in a little number. With the growth of *R. communis* plantules, the cover of *C. dactylon* is diminished due to the shading, which favors the establishment of other *R. communis* plants. In this time, the leaves of *C. dactylon* (with high K) undergo a decomposition process that supplies resources for soil biota and nutrients to substrate. By falling, *R. communis* plants also add their leaves (with high N and P) to the decomposer subsystem, increasing macrofauna densities (detritivores in the beginning, and carnivores in the end) and quickly transferring nutrients to the soil. Like other shrubs, *R. communis* offers its branches for perching, which facilitates the arrival of birds that disperse native seeds. Due to the amelioration of the soil conditions, *S. terebinthifolius* and other native plants get to establish and grow in the substrate (around four to five years after ash disposals). Finally, with the succession proceeding, the litter of native trees would be put into the soil, representing heterogeneous shelters for soil fauna.

Coal ash disposals effects

Major constraints to leaf-litter decomposition and soil macrofauna communities on the two coal ash disposal sites studied could particularly be the high concentration of heavy metals or other chemical elements, acidity and poor physic structure. In relation to heavy metals, for this same thermal plant, Rohde et al. (2006) found that boiler slag has increased concentrations of Sn, Mo, Cr, Al, and Pb than fly ash, which, in turn, had increased concentrations of Ni, Hg, Cd and Zn. Concerning other chemical elements, we found that the site with a major influence of boiler slag had higher N and B concentrations than the fly ash. The fly ash site, in its time, had an increased K ion concentration, which could improve salinity and electrical conductivity. Related to the pH-value, both disposal sites are acidic, which probably reduces microflora activities and the availability of major nutrients (such as N, P, K, Mg, B), as well as the increase of the availability of other toxic elements (such as Mn, Zn, Cu). These effects are certainly higher in the boiler slag site because it had the lowest pH-value. Substrate conditions are also different; while the fly ash has fine-grained particles, boiler slag has larger sized particles with an amount of unburned Carbon.

In view of all that, we found that the ash disposal type did not affect the integrity of the leaf-litter decomposition of the pioneer plants in spite of some chemical elements having been showed to be mineralized faster in the boiler slag site than in the fly ash site. On the other hand, the total morphospecies composition was clearly responsive. There are several works that have been registering changes in survivorship, physiological and morphological traits of some soil organisms when in contact with polluted food, substrate, and a non-favorable pH-value (Jones and Hopkin, 1998; Rusek, 2000; Grumiaux et al., 2007). There are recognized evidences of heavy metal adaptations in organisms such as Isopods (Donker et al., 1993) and Gastropods (Beeby and Richmond, 1989), which always accompany altered life histories as part of a complex adaptation syndrome (Posthuma and Van Straalen, 2002). Soil nutrient stoichiometry can also be explicitly linked to the invertebrate litter fauna densities, explaining a possible bottom-up regulation of the higher trophic levels (McGlynn et al., 2007). Thus, the difference in species composition between the two disposal sites may reflect the selection of the more adapted or resistant species to the specific environmental conditions. As the level of functional redundancy by a diverse range of taxa is significant in soil communities (Wardle, 2006), the integrity of the leaf-litter decomposition process was not altered.

Conclusions

We examined the fundamental aspects of the ecology of a polluted environment, which include: the decomposition of early successional plants, and colonization by soil fauna in two distinct sites of coal ash disposal. Similar works have not yet been conducted in Brazil that relates the ecosystem process and soil fauna in the assessment of natural areas that are affected by the coal residues. We conclude that (1) the leaf-litter of *R. communis* decayed more quickly than the other pioneer plants; (2) the leaf-litter identity influenced the diversity of the colonizing soil macrofauna; the leaf-litter of *R. communis* supported a higher density of individuals but a lower species richness than the other plants; and (3) the coal ash disposal type did not affect the leaf-litter decomposition, but instead, selected for distinct communities of soil macroinvertebrates, showing the existence of singular environmental conditions and suggesting the effects of different constraints in the long term polluted environment.

According to Eviner and Hawkes (2008), efforts to understand the feedback between plants and soils have the potential to discover a major tool for restoration (e.g., colonizing plants that ameliorate poor soil conditions) or a major obstacle to restoration (e.g., invasive species that alter soil conditions to benefit themselves). We have evidence that exotic plants have been ameliorating poor substrate conditions through litter decomposition and they benefit the proceeding of natural succession for a longer time; these results are in agreement with previous findings by Azzolini (2008). Also, the soil fauna development in the damaged area, especially the detritivorous animals, has brought clear benefits to the ecological restoration process because these organisms greatly affect the soil structure and chemistry, and facilitate the ecosystem processes (Snyder and Hendrix, 2008).

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Tables

Table 1. Mean ± SE (in brackets) of elemental concentration of substrate samples from fly ash and boiler slag disposal sites.

Chemical elements		Fly ash disposal		Boiler slag disposal	
N	(%)	0.6	(0.11)	0.9	(0.08)
P	(mg/dm^3)	32.0	(10.61)	21.5	(4.60)
K	(mg/dm^3)	> 313.0		185.0	(5.66)
Ca	(cmol/dm^3)	9.6	(1.70)	9.0	(0.64)
Mg	(cmol/dm^3)	1.4	(0.18)	1.2	(0.07)
$S\text{-}SO_4^{-2}$	(mg/dm^3)	15.5	(1.77)	15.5	(0.35)
B	(mg/dm^3)	0.3	(0.03)	0.5	(0.03)

Table 2. Leaf nutrient content of the three pioneer plants.

Plant species	C (%)	N (%)	P (%)	K (%)
C. dactylon	40	1.5	0.18	1.9
R. communis	41	5.1	0.38	0.25
S. terebinthifolius	41	1.2	0.15	1

Table 3. Summary of the results of the nested ANOVA for repeated measures (F values with *P* values in brackets) testing for effects of treatments on leaf-litter mass remaining (%) and the total soil macrofauna density (individuals g^{-1} d.w. litter- square root transformed). All statistically significant numbers ($p < 0.05$) are in boldface. The degrees of freedom are: site (1, 6), blocks within site (6, 14), leaf-litter species (2, 14), time (3, 42), leaf-litter species X time (6, 42).

Response variables	Site	Blocks within site	Leaf-litter species (A)	Time (B)	A x B
Leaf-litter mass	0.16 (0.70)	2.25 (0.10)	**285.97 (< 0.01)**	**249.93 (< 0.01)**	**19.68 (< 0.01)**
Total macrofauna	3.63 (0.105)	1.02 (0.45)	**3.83 (0.05)**	**2.87 (0.05)**	**5.00 (< 0.01)**

Table 4. Summary of the results of the ANOVA for repeated measures (F values with *P* values in brackets) testing for the effects of the treatments on the leaf-litter mass remaining (%) from the leaf-litter species. All statistically significant numbers ($p < 0.05$) are in boldface. The degrees of freedom are: site (1, 6), blocks within site (6, 14), time (3, 42), and leaf-litter species x time (6, 42).

Leaf-litter species	Site (A)	Time (B)	A x B
C. dactylon	3.46 (0.112)	**138.39 (<0.001)**	0.36 (0.786)
R. communis	0.04 (0.842)	**196.25 (<0.001)**	0.42 (0.743)
S. terebinthifolius	0.04 (0.857)	**138.39 (<0.001)**	0.36 (0.786)

Table 5. Percentage of nutrient loss of leaf-litter species after exposure (140 days) in the fly ash (F) and boiler slag (B) disposal sites. * Not detected.

Disposal sites/	C (%)		N (%)		P (%)		K (%)	
Leaf-litter species	F	B	F	B	F	B	F	B
C. dactylon	66.9	62	61.2	62.9	78.4	79.4	97.3	97.1
R. communis	*	97.1	99	98.4	99.2	98.8	98.3	96.2
S. terebinthifolius	71.4	67.4	51.2	45.7	78.5	72.6	94.7	94.5

Table 6. Summary of the results of the nested ANOVA for repeated measures (F values with p values in brackets) testing for the effects of the treatments on the functional trophic groups' density of soil macrofauna (ind. g^{-1} d.w. litter – square root transformed). All statistically significant numbers ($p < 0.05$) are in boldface. The degrees of freedom are: site (1, 6), blocks within site (6, 14), leaf-litter species (2, 14), time (3, 42), leaf-litter species x time (6, 42).

Response variables	Site	Blocks within site	Leaf-litter species (A)	Time (B)	A x B
Detritivores	0.6.4 (0.45)	2.41 (0.08)	1.13 (0.35)	**19.34 (<0.01)**	**5.37 (<0.01)**
Carnivores	**10.13 (0.02)**	0.79 (0.59)	**7.58 (0.01)**	**7.45 (< 0.01)**	**4.43 (< 0.01)**
Omnivores	1.48 (0.27)	0.66 (0.68)	1.10 (0.36)	0.80 (0.50)	0.78 (0.59)
Herbivores	1.26 (0.30)	0.68 (0.67)	0.45 (0.64)	1.82 (0.16)	0.47 (0.83)

FIGURES

Figure 1. Study areas with different ash disposal types and pioneer plants. A) Recent ash deposit showing the plant community structure around; B) Boiler slag disposal site; C) Fly ash disposal site; D) *Cynodon dactylon* (grass) and *Ricinus communis* (shrub); E) *Schinus terebinthifolius* (tree).

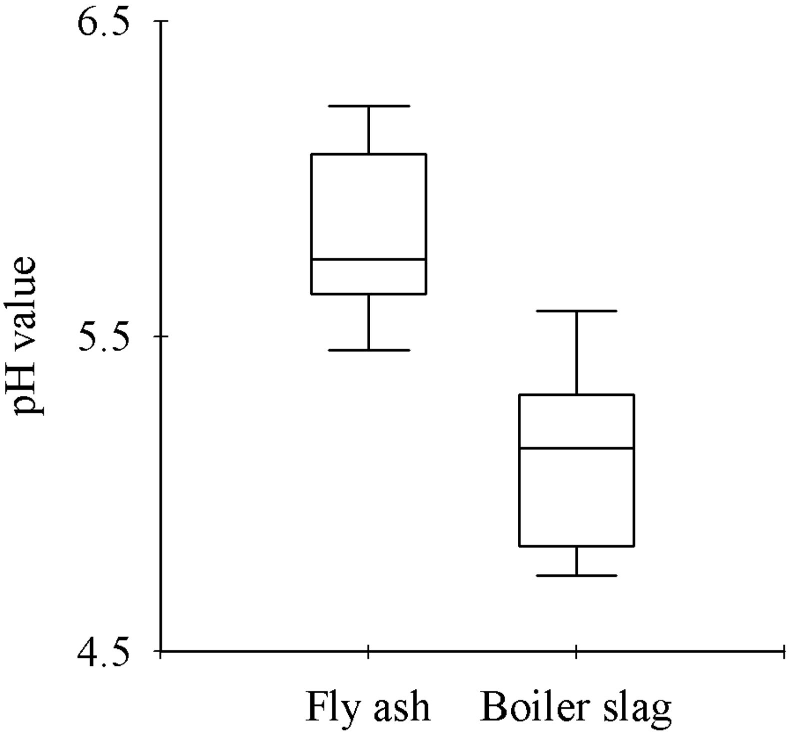

Figure 2. Box plot for pH values of the fly ash and boiler slag disposal sites.

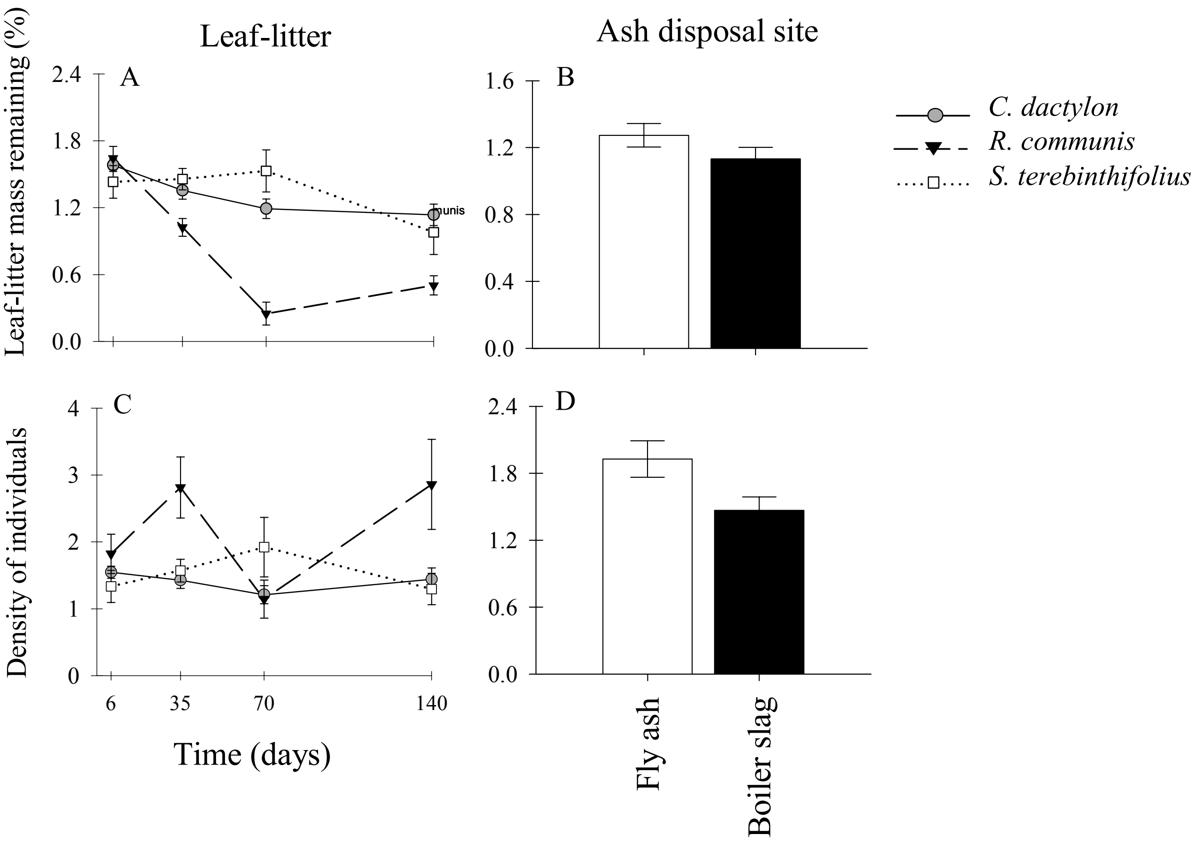

Figure 3. Means ± SE of the leaf-litter mass remaining (A, B) and individuals. g^{-1} d.w. litter (square root transformed) from macrofauna (C, D) in response to the effects of the treatments: leaf-litter species X sampling time (A, C), and ash disposal site(B, D).

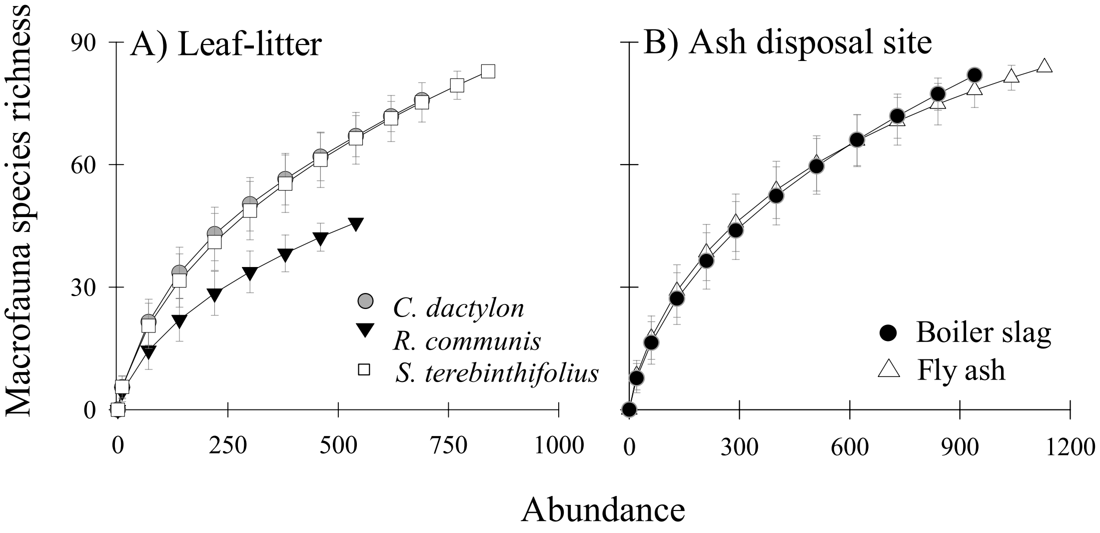

Figure 4: Rarefaction curves of the total soil macrofauna in response to the leaf-litter species and ash disposal sites. Error bars represent ± 1 confidence interval (C.I. 95%).

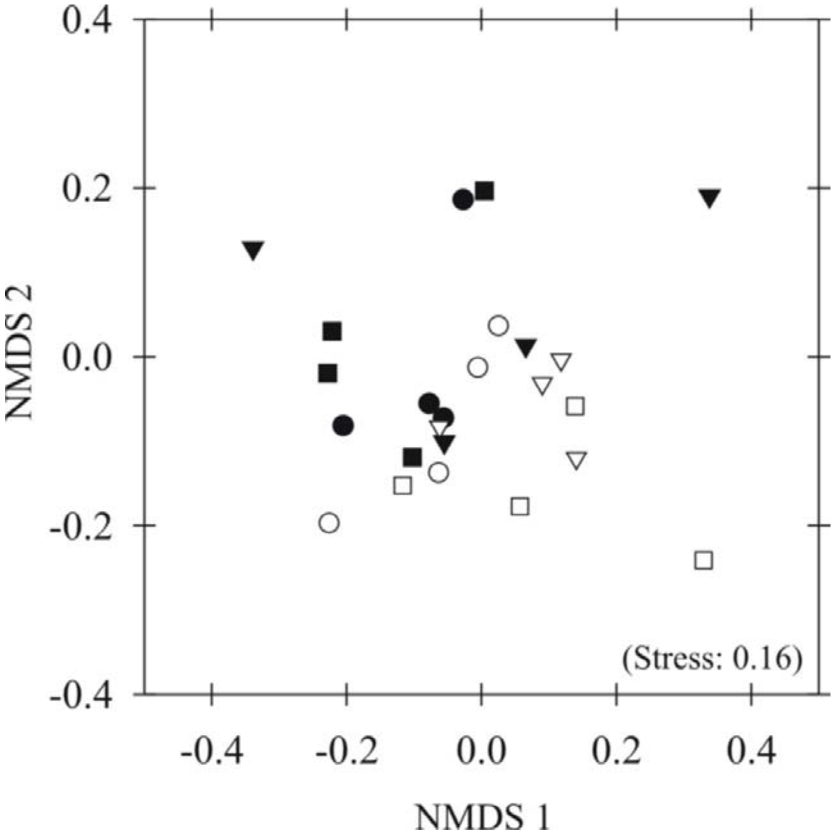

Figure 5. NMDS ordination in two dimensions of sample units showing the total soil macrofauna community structure in response to the leaf-litter species and ash disposal sites. Circle = *C. dactylon*; triangle = *R. communis*; square = *S. terebinthifolius*; white symbols = fly ash site; black symbols = boiler slag site.

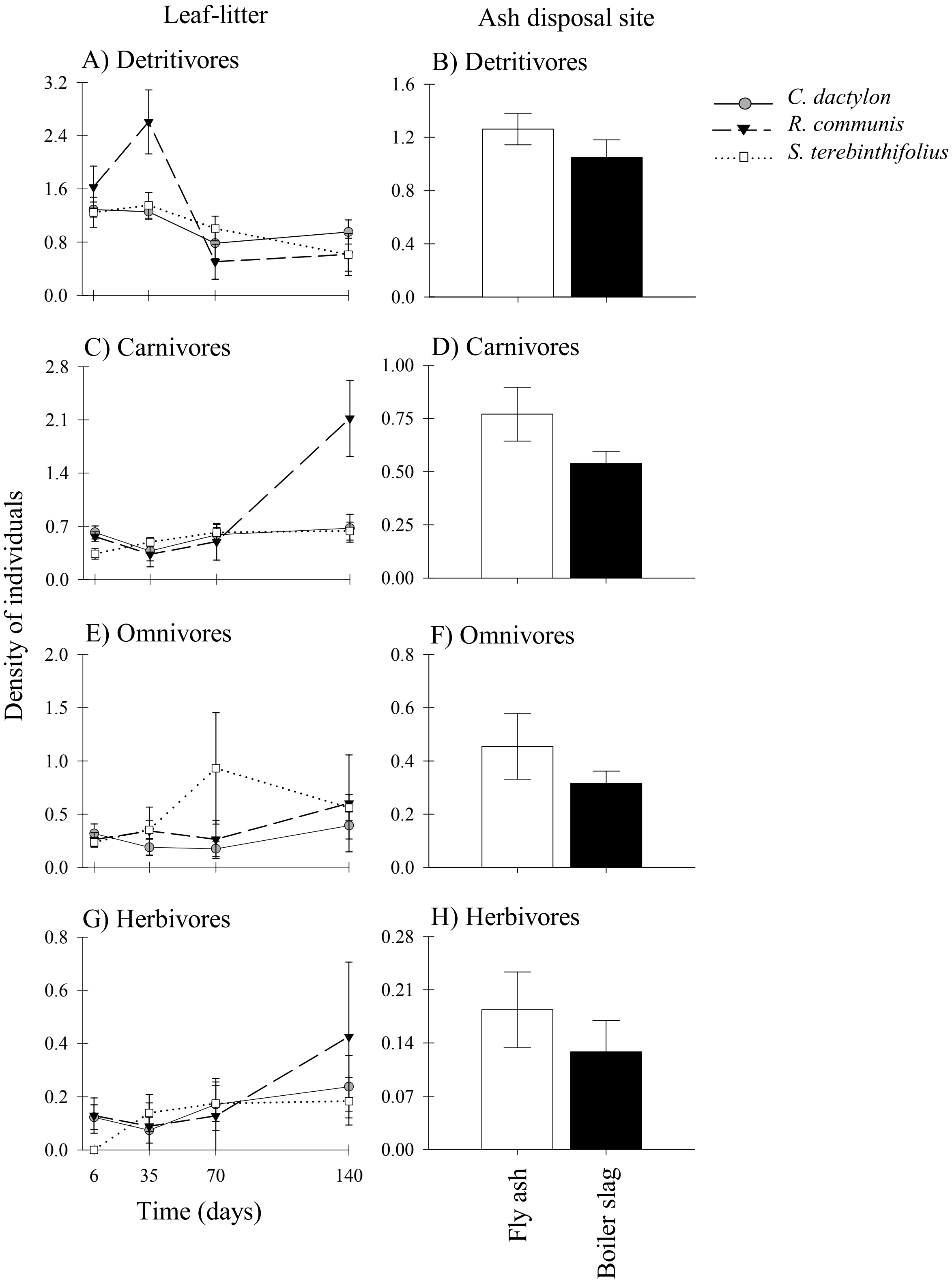

Figure 6. Means ±SE of individuals.g^{-1} d.w.litter (square root transformed) *per* functional trophic group of soil macrofauna under the effects of treatments (leaf-litter species X sampling time, and ash disposal site).

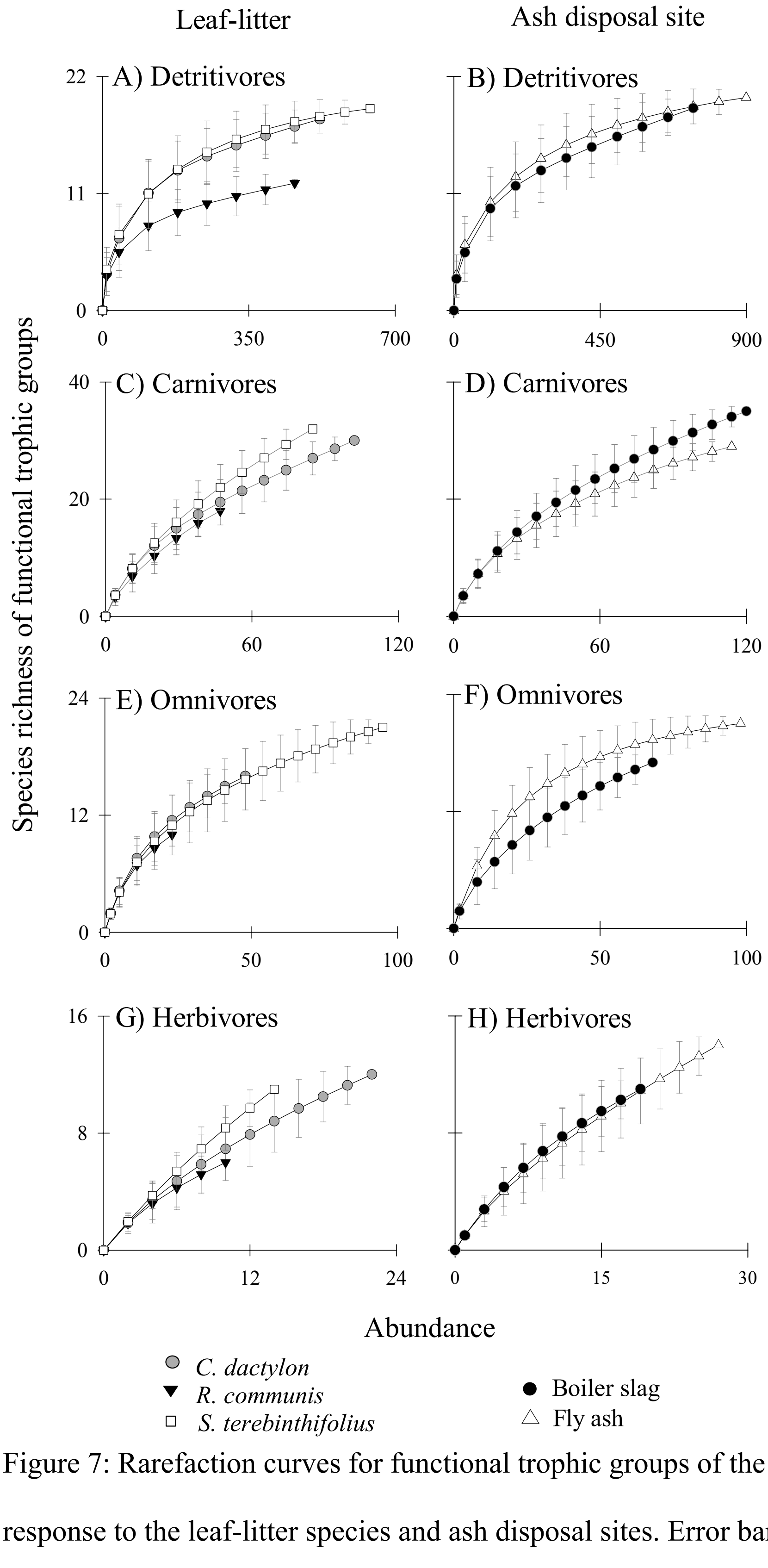

Figure 7: Rarefaction curves for functional trophic groups of the soil macrofauna in response to the leaf-litter species and ash disposal sites. Error bars represent ± 1 confidence interval (C.I. 95%).

Appendix 1. Morphospecies composition, and the abundance of the soil macrofauna colonizing the pioneer leaf litter (C-*Cynodon dactylon*, R-*Ricinus communis*, S-*Schinus terebinthifolius*) at 6, 35, 70 and 140 days of leaf-litter exposure, in two different coal ash disposal sites (fly ash and boiler slag). The functional trophic groups (FTG) are: detritivores (D), carnivores (C), omnivores (O) and herbivores (H). N.I. = not identified in morphospecie or trophic group.

Class / Order	Family / Species or Morphospecies	FTG	**Fly ash site**												**Boiler slag site**											
			C				**R**				**S**				**C**				**R**				**S**			
			6	**35**	**70**	**140**	**6**	**35**	**70**	**140**	**6**	**35**	**70**	**140**	**6**	**35**	**70**	**140**	**6**	**35**	**70**	**140**	**6**	**35**	**70**	**140**
Arachnida																										
Araneae	Anyphaenidae																									
	Immatures	C	0	1	3	0	0	0	0	0	0	0	0	0	0	0	1	0	1	0	0	0	0	1	1	0
	Araneidae																									
	Immatures	C	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Corinnidae																									
	Immatures	C	0	0	0	0	1	0	0	1	0	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0
	Gnaphosidae																									
	Jovens	C	0	2	1	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
	Hahniidae																									
	Hahnidae sp.	C	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0
	Linyphiidae																									
	Erigone sp.	C	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	2	0	0	0	0	0	0	0	1
	Laminacauda sp.	C	0	0	0	0	0	0	1	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0
	Lepthyphantes sp.	C	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0
	Scolecura cambara Rodrigues, 2005	C	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Immatures	C	0	0	0	7	0	0	0	0	0	1	1	1	0	0	0	2	0	0	0	1	1	0	0	0
	Lycosidae																									
	Lycosidae sp.	C	4	0	0	0	0	0	0	0	0	0	0	0	1	0	1	0	2	0	0	0	0	0	0	0
	Immatures	C	5	1	0	2	2	0	0	5	6	4	0	1	3	1	1	0	2	1	0	0	0	0	0	1
	Oecobidae																									
	Immatures	C	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
	Onoopidae																									
	Oonopinae sp.	C	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1
	Immatures	C	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

	Salticidae																									
	Aphirape uncifera (Tullgreen, 1905)	C	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0
	Unidentatae sp.1	C	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Unidentatae sp.2	C	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
	Immatures	C	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	1	1	0
	Scytodidae																									
	Scytodes sp.	C	0	0	0	1	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Immatures	C	0	0	1	10	0	0	0	1	0	1	3	0	0	0	0	0	0	0	0	0	0	0	0	1
	Segestriidae																									
	Immatures	C	0	0	0	1	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Tetragnathidae																									
	Immatures	C	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0
	Theridiidae																									
	Coleosoma sp.	C	0	1	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0
	Thymoites sp.	C	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
	Immatures	C	0	0	2	1	0	0	0	1	0	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0
	Thomisidae																									
	Immatures	C	0	0	4	0	1	0	0	0	0	0	0	0	0	0	1	1	0	0	2	1	0	1	1	0
Opiliones	Opiliones sp.	O	0	0	0	0	0	0	0	1	0	2	0	0	0	0	0	0	0	0	0	0	0	1	0	0
Chilopoda	Chilopoda sp.1	C	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
	Chilopoda sp.2	C	0	1	1	0	2	0	0	0	0	0	0	1	0	1	0	0	0	0	0	0	0	1	4	0
	Chilopoda sp.3	C	0	0	0	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0
Crustacea																										
Amphipoda	Talitridae																									
	Talitroides sylvaticus (Haswell)	O	0	0	0	0	2	0	0	0	2	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0
Isopoda	Balloniscidae																									
	Balloniscus sellowii (Brandt, 1833)	D	0	0	2	0	3	0	1	0	11	1	5	1	1	2	1	2	2	2	0	0	0	6	8	0
	Philosciidae																									
	Atlantoscia floridana (van Name, 1940)	D	7	1	4	0	20	9	0	0	27	32	14	2	0	0	0	0	0	0	0	0	0	0	0	0
	Benthana taeniata Araujo & Buckup, 1994	D	81	28	13	8	88	11	0	1	159	72	39	16	85	24	12	1	68	22	2	0	54	16	17	0
	Plathyarthridae																									
	Trichorhina sp.	D	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
Diplopoda	Diplopoda sp.1	D	0	0	0	0	0	0	0	0	3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Diplopoda sp.2	D	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	1	4	2	1
Gastropoda																										
Soleolifera	Veronicellidae																									
	Belocaulus angustipes (Heynemann, 1885)	H	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Stylommatophora	Agriolimacidae																									
	Deroceras laeve (Müller, 1774)	D	0	1	0	1	2	0	0	0	0	0	0	3	0	0	1	2	3	2	0	0	0	0	4	1
	Bradybaenidae																									

	Bradybaena similaris (Férussac, 1821)	H	0	0	6	1	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0
	Bulimulidae																									
	Drymaeus sp.	H	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	1	2	0	0	1	0	0	0	0
	Euconulidae																									
	Habroconus sp.	D	0	0	0	1	1	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	1	6	0
	Systrophiidae																									
	Happia sp.	C	0	0	1	0	2	0	0	0	5	0	1	2	0	0	0	0	0	0	0	0	0	0	0	2
	aff. Drepanostomella sp.	C	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0
	Gastropoda sp.		0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	N.I.		0	0	0	1	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0
Hexapoda																										
Blattodea	Blattodea sp.1	D	0	0	0	0	2	0	0	0	1	0	0	0	1	0	0	0	7	0	0	0	1	0	0	0
	Blattodea sp.2	D	0	0	0	0	0	0	0	0	3	0	1	0	0	0	0	0	0	0	0	0	1	0	0	0
	Blattodea sp.3	D	1	2	9	3	0	0	0	0	10	3	0	0	0	0	0	0	0	0	0	0	0	1	0	0
	N.I.	D	0	0	0	5	0	0	0	0	0	0	0	5	0	0	0	0	0	0	0	1	0	0	0	0
Coleoptera	Anthicidae																									
	Anthicidae sp.	C	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
	Chrysomelidae																									
	Chrysomelidae sp.1	H	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Chrysomelidae sp.2	H	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0
	Chrysomelidae sp.3	H	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
	Chrysomelidae sp.4	H	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Diabrotica speciosa (Germar, 1824)	H	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Colydiidae																									
	Colydiidae sp.	D	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Curculionidae																									
	Curculionidae sp.1	H	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Curculionidae sp.2	H	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1	0	0	0	0	0	1	0	0
	Curculionidae sp.3	H	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
	Curculionidae sp.4	H	1	0	0	0	0	0	0	0	0	0	0	0	2	0	0	0	1	0	0	0	0	0	0	0
	Elateridae																									
	Elateridae sp.	D	3	1	3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Leiodidae																									
	Leiodidae sp.	D	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0
	Nitidulidae																									
	Nitidulidae sp.1	D	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0
	Nitidulidae sp.2	D	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Nitidulidae sp.3	D	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Phalacridae																									
	Phalacridae sp.1	D	0	7	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Phalacridae sp.2	D	4	0	0	0	6	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0
Ptilidae																									
Ptilidae sp.	D	0	1	0	2	0	0	0	0	0	0	0	1	0	0	0	5	0	0	0	0	0	0	2	0
Scarabaeidae																									
Scarabaeidae sp.1	D	0	0	0	0	0	0	0	0	0	0	0	0	3	1	1	1	1	0	0	0	0	1	2	1
Scarabaeidae sp.2	D	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0
Scydmaenidae																									
Scydmaenidae sp.	C	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
Staphilinidae																									
Staphilinidae sp.1	C	0	1	0	1	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
Staphilinidae sp.10	C	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Staphilinidae sp.11	C	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0
Staphilinidae sp.12	C	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Staphilinidae sp.13	C	0	0	0	1	0	1	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0	0	0
Staphilinidae sp.14	C	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Staphilinidae sp.15	C	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0
Staphilinidae sp.16	C	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
Staphilinidae sp.17	C	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
Staphilinidae sp.18	C	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Staphilinidae sp.19	C	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Staphilinidae sp.2	C	0	0	0	0	0	0	0	0	0	0	0	0	6	0	0	0	0	0	0	0	0	0	0	1
Staphilinidae sp.20	C	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Staphilinidae sp.21	C	0	0	0	0	0	1	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	1
Staphilinidae sp.3	C	0	0	0	0	1	0	0	0	0	0	0	0	16	1	0	0	14	0	0	0	3	0	0	0
Staphilinidae sp.4	C	0	0	0	0	0	0	0	0	0	0	0	0	8	0	0	0	1	0	0	0	1	0	0	0
Staphilinidae sp.5	C	0	2	2	0	0	1	0	0	1	2	2	0	0	1	3	0	1	0	0	0	0	0	0	1
Staphilinidae sp.6	C	0	0	0	0	0	0	0	0	0	4	14	0	0	0	0	0	0	0	0	0	0	0	0	0
Staphilinidae sp.7	C	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0
Staphilinidae sp.8	C	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0
Staphilinidae sp.9	C	6	0	0	0	7	0	0	0	0	0	0	0	1	0	0	1	3	0	0	0	0	0	0	0
Pselaphinae																									
Pselaphinae sp.1	C	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0
Pselaphinae sp.2	C	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Pselaphinae sp.3	C	0	0	0	1	0	0	0	0	0	1	0	1	0	0	0	0	0	0	0	1	0	0	0	0
Pselaphinae sp.4	C	3	6	1	0	2	0	0	0	0	3	3	1	0	0	3	0	0	0	0	0	0	0	1	0
Pselaphinae sp.5	C	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0
Pselaphinae sp.6	C	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Pselaphinae sp.7	C	0	0	0	0	0	0	0	0	0	0	0	0	0	0	6	1	0	0	0	0	0	0	1	0
Pselaphinae sp.8	C	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	1	0	0
Scaphidiinae																									

	Scaphidiinae sp.	D	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0
	Tenebrionidae																									
	Tenebrionidae sp.1	D	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
	Tenebronidae sp.2	D	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Dermaptera	Dermaptera sp.1	O	0	0	0	0	0	0	0	0	4	1	3	0	0	0	0	0	0	0	0	0	0	0	0	0
	Dermaptera sp.2	O	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Hemiptera	Fulgoroidea																									
	Fulgoroidea sp.	H	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0
	Cicadellidae																									
	Cicadellidae sp.1	H	0	0	1	1	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	1	0
	Cicadellidae sp.2	H	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	1	0	0
	Membracidae																									
	Membracidae sp.	H	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
	Cydnidae																									
	Cydnidae sp.	H	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0
	Miridae																									
	Miridae sp.	H	0	0	1	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Pentatomidae																									
	Oebalus poecillus (Dallas, 1851)	H	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Reduviidae																									
	Reduviidae sp.	C	0	0	0	0	0	1	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Ryparochromidae																									
	Ryparochromidae sp.	H	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
	NI	H	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0
Hymenoptera	Formicidae																									
	Dolichoderinae																									
	Dorymyrmex sp.	O	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Formicinae																									
	Brachymyrmex sp.1	O	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	1	0	0
	Brachymyrmex sp.2	O	0	0	0	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	3	0
	Camponotus sp.2	O	1	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	1	0
	Camponotus sp.3	O	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0
	Paratrechina sp.1	O	3	0	0	0	1	0	0	0	1	0	258	0	3	0	0	0	0	0	0	0	1	0	2	0
	Paratrechina sp.2	O	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	1	0	0
	Myrmicinae																									
	Acromyrmex sp.	D	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	2	0
	Camponotus sp.1	O	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0
	Pheidole sp.1	O	1	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0
	Pheidole sp.2	O	6	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
	Pheidole sp.3	O	0	0	0	7	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

	Pheidole sp.4	O	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0
	Pyramica sp.	O	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0
	Solenopsis sp.1	O	0	3	0	1	0	4	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Solenopsis sp.2	O	0	0	0	0	0	0	0	0	0	1	1	2	0	1	3	4	2	0	1	1	0	2	10	6
	Solenopsis sp.3	O	0	0	0	0	0	0	0	1	0	0	1	1	2	0	0	1	0	0	0	0	0	3	1	3
	Solenopsis sp.4	O	0	0	0	1	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	1	0
	Solenopsis sp.5	O	0	0	0	1	0	0	0	0	0	0	0	6	0	0	0	0	0	0	1	0	0	0	0	0
	Strumigeny sp.	C	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	3	0
	Wamannia sp.1	O	0	0	0	0	5	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	Wasmannia auropunctata (Roger)	O	1	0	0	0	0	0	0	0	0	0	2	7	0	1	1	0	0	0	0	0	1	0	0	0
	Ponerinae																									
	Hypoponera sp.1	O	0	0	0	0	0	1	0	0	0	0	0	0	1	0	1	0	0	0	0	0	0	0	0	0
	Hypoponera sp.2	O	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
	N.I.		0	0	0	0	0	0	0	1	0	0	2	0	0	0	2	0	0	0	0	0	0	0	0	0
Orthoptera	Orthoptera sp.	H	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0
Immatures (larvae)	N.I.	-	6	6	0	0	5	0	1	0	7	6	5	5	2	4	1	5	4	3	0	0	0	2	42	9
Oligochaeta	Megascolecidae																									
	Metaphire californica (Kinberg, 1867)	D	17	49	10	20	19	13	0	2	2	17	3	8	22	36	27	8	156	16	0	0	6	49	21	0
TOTAL			**155**	**116**	**67**	**82**	**182**	**42**	**5**	**17**	**250**	**166**	**367**	**76**	**162**	**80**	**71**	**46**	**272**	**49**	**8**	**7**	**72**	**97**	**150**	**34**

ARTIGO 2.

Neotropical woodlice (Crustacea, Isopoda) colonizing early successional plant leaf-litter in a coal ash disposal environment*

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Abstract

An experimental study was conducted to investigate the colonization patterns of woodlice in leaf-litter from three spontaneous pioneer plants in an environment exposed to long-term coal ash pollution (fly ash and boiler slag). Three species were found: *Atlantoscia floridana* (Philosciidae; n=116), *Benthana taeniata* (Philosciidae; n=817) and *Balloniscus sellowii* (Balloniscidae; n=48). Woodlice colonized equally the plant species along the time of leaf exposure (6, 35, 70 and 140 days). Occurrence of *A. floridana* and abundance and fecundity of *B. taeniata* females was strongly influenced by the ash disposal type. These results are discussed under ecological aspects.

Key words: Brazil, soil invertebrates; detritivores, litter decomposition, fly ash; boiler slag.

Introduction

Coal has been the fastest-growing major fuel in the world (BP 2008). However, its extraction and combustion implies in serious environmental impacts. The combustion of coal in thermal power plants produces vast quantities of wastes, and instead of utilizing those residues for the building industry (Scheetz & Earle 1998; Rohde et al 2006), thermal power plants usually make landfills or minefills. Coal and associated coal combustion products contain polycyclic aromatic hydrocarbons and trace elements of “heavy metals” (Querol et al 1995; Teixeira et al 1999). These substances may contain mobile toxic constituents with potentially genotoxic effects in the surrounding environment (Silva et al 2000).

One of the major potential adverse impacts of the disposal of coal combustion residue in terrestrial ecosystems is decline of plant establishment and growth (Carlson & Adriano 1993). This decline results from changes in edaphic traits (Tordoff et al 2000; Gupta et al 2002), that leads to unfavorable physical (compacted layers, reduced bulk density) and chemical conditions (nutrient depletion, high toxicity and low pH-value) and reduction in microbial and soil biota activity. Despite these challenges, early successional plant species that show heavy metal tolerance or resistance are able to colonize such environments (Whiting et al 2004). A positive feedback loop may then occur between the pioneer plants and the substrate (Wilson & Agnew 1992) by the improvement of the microclimate and resource availability through the stabilization of coal wastes, shading, aerating, depositing litter and offering habitat and shelter for animals (Carlson & Adriano 1993). In this way, early successional plants may pave the way for subsequent species in a facilitation model (Connel & Slatyer 1977).

Soil invertebrates play a crucial role in the early succession of the restoration process in polluted/damaged ecosystems. Woodlice (Crustacea, Isopoda) are commonly found in metal-polluted environments (Jones & Hopkin 1996; Jones & Hopkin 1998; Grelle et al 2000; Tajovský 2001). As abundant litter detritivores in terrestrial ecosystems they participate in the leaf-litter processing (Quadros & Araujo 2008) by breaking up the organic matter and returning nutrients to soil through their feces, which strongly enhance microorganism activity (Hassall et al 1987). To avoid the toxic effects from contaminated food, they are capable of immobilizing high levels of heavy metals in their hepatopancreas (Raessler et al 2005). This may affect the life history traits by changing resource allocation of woodlice (Donker 1992). These changes are related to increased mortality, slower growth rates, reduced body size and reproduction effects (Donker 1992; Donker et al 1993; Jones & Hopkin 1996; Jones & Hopkin 1998). However, there is strong evidence for the occurrence of heavy-metal adaptation in natural populations of isopods (Posthuma & Van Straalen 2002).

The southern region of Brazil is very rich in coal reserves, which have been exploited, extracted and destined to generation of energy in thermal power plants for decades (Pires & Querol 2004). Consequently, this region has a number of environments exposed to long-term coal ash pollution (Teixeira et al 1999). A recent research in one of those environments investigated the leaf decomposition of three abundant early successional plant species and the diversity of associated soil invertebrate macrofauna (Podgaiski & Rodrigues unpubl. data). *Ricinus communis* L. (Euphorbiaceae) and *Cynodon dactylon* (L.) Persoon (Poaceae) are exotic plants in Brazil, and they occurred in patches of recent ash deposits. *Schinus terebinthifolius* Raddi (Anacardiaceae), a native tree, is very abundant in South Brazil and it was found in older ash deposits. The research revealed woodlice as the most abundant group of soil macrofauna in leaf-litter. In view of that, the present study investigates (1) the colonization patterns of different woodlice species along leaf decomposition of these plant species and (2) the woodlice abundance and some reproductive traits in different types of coal ash disposals (fly ash and boiler slag).

Materials and methods

Study area

The study was conducted in the city of São Jerônimo in the state of Rio Grande do Sul, Brazil (29°57’55.6”S; 51°44’14.9”W), in a site along a riparian forest of the Jacuí river (Figure 1). The climate of this region is temperate (Cfa type of Köppen-Geiger climate classification; Peel et al 2007) with hot summers but without a dry season.

The study area has been a coal combustion residue disposal site for more than 30 years, but has recently stopped receiving additional waste. Two different sites are found in the area: one that was more influenced by fly ash disposal and another that was more influenced by boiler slag disposal (Figure 1). Fly ash and boiler slag are aluminosilicate minerals, with SiO_2 and Al_2O_3 being the predominant components (Rohde et al 2006), and they differ in both physical and chemical properties. Fly ash is a fine-grained powder, composed of spherical glassy and hollow particles captured by air pollution control equipment in thermal power plants. Boiler slag is a specific type of bottom ash, which is vitreous molten grained and composed of angular particles with high C content from unburnt coal derived from wet ash removal of wet-bottom furnaces (ECOBA 2008). The elemental concentrations vary according to parent coal composition and combustion technology. Rohde et al (2006) studied the results of leaching fly ash and boiler slag from the São Jerônimo thermal power plant and found the most abundant elements to be Sn, Ni and Mo, and in lower proportion Cr, As, Hg, Al, Pb, Mn, V, Cd, Ba and Zn. Boiler slag had higher concentrations of Sn, Mo, Cr, Al, Pb and Mn than fly ash, which, in turn, had higher concentrations of Ni, Hg, Cd and Zn. The boiler slag disposal site has a lower pH (mean 5.1) than the fly ash disposal (mean 5.8) (Podgaiski & Rodrigues unpubl. data).

Experimental design and laboratory procedures

Ninety-six nylon litter bags of 30 x 20 cm, made of coarse mesh (1.0 x 0.2 cm) were filled with 20.3 ± 0.2 g of air-dried freshly fallen leaves of *C. dactylon, R. communis* and *S. terebinthifolius* (32 litter bags per plant species). In June 2007, the litter bags were placed on bare ground in eight blocks: four within the site influenced by fly ash disposal and four within the site influenced by boiler slag disposal (Figure 1). Each block was composed of 12 litter bags (four litter bags per plant species), distant at least 2 m from each other (Figure 1). There were four successive sampling occasions (6, 35, 70 and 140 days after exposure), when one litter bag of each plant species was randomly removed from each block (Figure 1).

The isopods from the litter bags were manually collected, identified at species level and counted. The reproductive traits measured were the size of ovigerous females (cephalothorax width; Araujo & Bond-Buckup 2004) and their fecundity (marsupial content). After the inspection of the litter bags to sort out the animals, the remaining leaf-litter in each bag was dried and weighed.

Keeping in mind that isopods prefer leaf-litter with a low Carbon-Nitrogen (C:N) ratio (Zimmer 2002), the C and N content of green leaves of each plant leaf-litter species was determined using the methods of moisture combustion/Walkley-Black (Walkley & Black, 1934) and Kjeldahl (Kjeldahl 1883) with 0.01 % of detection limit, respectively. The C:N ratio was lowest for *R. communis* (8.0), followed by *C. dactylon* (26.7) and *S. terebinthifolius* (34.2) (Podgaiski & Rodrigues unpubl. data).

Statistical analysis

Isopod abundance in litter bags was standardized and expressed as individuals.g^{-1} d.w. (dry weight) litter. The isopod abundance (total and for each isopod species) was compared in plant leaf-litter species and sampling dates with repeated measures analysis of variance (ANOVA) in blocks for each of two sites influenced by coal ash disposals. The relationship between isopod ovigerous female size (mm) and fecundity was assessed between sites using analysis of covariance (ANCOVA). The size of the ovigerous females in both sites was compared with ANOVA. Residual analyses were carried out to check all models used in this study.

Results

Three native neotropical isopod species were found, totaling 981 individuals (Table 1): *Benthana taeniata* Araujo & Buckup, 1994 (Philosciidae), *Atlantoscia floridana* (van Name, 1940) (Philosciidae) and *Balloniscus sellowii* (Brandt, 1833) (Balloniscidae). The species behave differently regarding the two sites of disposal. *Atlantoscia floridana* occurred exclusively in the fly ash site (116 ind.), *B. taeniata* was found in a much higher abundance than the others (817 ind.), with 63% of individuals occurring in the fly ash. *Balloniscus sellowii*, on the other hand, was rarer than the others (48 ind.) and showed no differences between the two sites (Table 1).

According to results of repeated measures ANOVA, the total abundance of isopods as well as the abundance of each isopod species was not significantly different among leaf-litter species (Table 2; Figure 2) in both sites. There was no interaction between treatments (leaf-litter species) and time (sampling dates). However, a tendency towards a higher abundance of all isopod species in *R. communis* leaf-litter could be observed at 35 days of leaf exposure (Figure 2). The time of leaf exposure only had a significant effect in the abundance of *B. taeniata* in the fly ash disposal site. In this case, *B. taeniata* was more abundant in 6 days than in 70 days of leaf-litter decomposition (Tukey test; P=0.04).

Benthana taeniata was the only species that presented a sufficient number of ovigerous females for the analysis of reproductive traits (53 individuals: 34 in the fly ash site and 19 in the boiler slag site). ANCOVA indicates that fecundity in relation to female size ($F_{1,\,50}$= 107.36; P< 0.001) was lower in females from the boiler slag disposal site than in females from the fly ash disposal site ($F_{1,\,50}$= 10.79; P= 0.002; Figure 3). Nevertheless, ovigerous female size was not significantly different between sites ($F_{1,\,51}$= 0.02; P=0.88).

Discussion

The woodlice species found in the coal waste disposal sites studied are native to the Neotropical region. Among them, *A. floridana* is the most common and abundant species in southern Brazil. Its populations can reach up to 1040 ind. m^{-2} (Araujo & Bond-Buckup 2005), with an average biomass of 1 kg ha^{-1} (Quadros & Araujo 2008).

Balloniscus sellowii is also commonly found in southern Brazil (Araujo et al. 1996; Lopes et al. 2005). *Benthana taeniata* occurs exclusively in this region (Araujo et al. 1996) and no data on its biology and ecology is available. The fragment of riparian vegetation situated between the disposal sites and the adjacent river probably serves as a source for these colonizing populations. Interestingly, synantropic woodlice as *Armadillidum* spp., *Porcellio* spp. and *Porcellionides* spp., which are very abundant in urban areas in Brazil (Araujo et al. 1996), were not present in the area of the thermal power plant and disposal sites.

Woodlice in coal ash disposal sites

The woodlice species seems to have different levels of tolerance to the environments created by the coal ash disposal. The boiler slag is clearly less suitable for isopods, especially *A. floridana.* The increased amount of contaminants such as Se, Mo and Mn in boiler slags (Rohde et al 2006) probably contributed to the low abundances verified in this site, if they imply in higher mortality and/or slower growth rates, then decreasing or inhibiting colonization by woodlice. Compared to *A. floridana* and *B. taeniata*, *B. sellowii* seems to be more tolerant to this habitat, since it is commonly found in human managed habitats, such as monocultures of exotics plants (*Eucaliptus* spp. and *Pinus* spp.) and urban parks. Moreover, *B. sellowii* shows an increased fecundity, which is a key reproductive trait that enhances its colonization ability (Quadros et al. 2008).

Fecundity of *B. taeniata* females was differently affected in the two sites. Previous studies have shown evidence of changes in life history traits of woodlice inhabiting contaminated environments (Donker et al 1993; Jones & Hopkin 1996; Van Brummelen et al 1996) due to changes in resource allocation (Donker 1992). The lower fecundity of *B. taeniata* females in the most polluted site was may be an example of such trade-off, but since there are no other studies concerning the ecology and reproduction of *B. taeniata*, especially in non-polluted sites, further research needs to be conducted.

Woodlice in pioneer leaf-litter along decomposition

Terrestrial isopods prefer to feed from leaf litter of decayed, dicotyledonous (Rushton & Hassall 1983; Hassall et al 1987) plants with low C:N ratio (Zimmer 2002). Considering that they may be attracted to a litter patch that offers more palatable food (Tuck & Hassall 2005), a higher abundance was expected in *R. communis* bags (based on its lower C:N ratio) and in leaf-litter from the latest days of decomposition (more decayed leaves). However, our predictions were not supported as woodlice used the three plant species in all times of leaf decomposition. Another possibility is that woodlice were attracted to the litter bags for sheltering and protection from direct light, high temperatures (Hassall & Tuck 2007) and potential predators. For organisms that utilize the litter both as food and habitat, there is a trade-off between these two resources, because the decomposition process that increases the palatability of the litter (Hassall et al 1987) at the same time decreases the structural integrity that makes it best suited for use as persistent shelter (Hooper et al 2000). The nitrogen-rich leaf-litter of *R. communis* is an example of such a trade-off. It may constitute a valuable food source, however, it decomposes much faster than the others (Podgaiski & Rodrigues unpubl. data). If the litter was not suitable for feeding in the beginning of the experiment, woodlice may have been attracted to the litter bags mainly for sheltering and, along the decaying process and after considerable microbial degradation (Wolters 2000), they may have fed from the litter.

The woodlice populations studied here inhabit a contaminated and highly modified riparian ecosystem of southern Brazil. Considering that they are abundant detritivores, they influence the soil restoration processes in this area by the acceleration of humus-forming processes that contribute to an efficient nutrient availability for the establishment of the plant community. On the other hand, as bioaccumulator organisms that are predated by a wide range of animals, both invertebrates and vertebrates (Sunderland & Sutton 1980), they are likely to take part in the process of biomagnification of heavy metals through the food chain (Paoletti & Hassall 1999). Despite some studies have been conducted with woodlice and other invertebrates in polluted-environments in Europe and other regions of the world (e.g., Majer et al 2007; Tajovský 2001; Grelle et al 2000), these kinds of studies are not yet found in Brazil with neotropical invertebrates. Studies relating plants and invertebrate detritivores, ecosystem processes (such as litter decomposition) and effects of contaminants on invertebrate biology would be very important to the understanding of the ecology of polluted environments. This knowledge would be essential to meet restoration and diversity conservation objectives.

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Tables

Table 1. Total abundance of isopod species colonizing leaf-litter from three pioneer plant species (C- *Cynodon dactylon*, R- *Ricinus communis* and S- *Schinus terebinthifolius*) in two coal ash disposal sites. N= 96 litter bags.

Isopod species	Fly ash disposal site				Boiler slag disposal site				Total
	C	R	S	Total	C	R	S	Total	
Philosciidae									
Atlantoscia floridana	12	29	75	116	0	0	0	0	116
Benthana taeniata	130	100	286	516	122	92	87	301	817
Balloniscidae									
Balloniscus sellowii	2	4	18	24	6	4	14	24	48
Total isopods	144	133	379	656	128	96	101	325	981

Table 2. Results of repeated measures ANOVA (F and P-values) for factors affecting the total abundance of isopods and *B. taeniata*, *B. sellowii* and *A. floridana* abundances for each of two coal ash disposal sites. Sources of variation are: treatment (leaf-litter species: *C. dactylon*, *R. communis* and *S. terebinthifolius)*, time (6, 35, 70 and 140 days after leaf exposure) and blocks (n= 4 *per* site).

Response variable	Fly ash disposal site				Boiler slag disposal site			
	Treatment (A)	Time (B)	A x B	Block	Treatment (A)	Time (B)	A x B	Block
Total abundance	1.4 (0.306)	2.3 (0.110)	1.0 (0.442)	1.4 (0.319)	0.9 (0.455)	1.5 (0.256)	1.2 (0.346)	0.8 (0.556)
B. taeniata	0.6 (0.567)	3.7 (0.031)	0.6 (0.730)	0.3 (0.817)	0.9 (0.448)	1.4 (0.267)	1.2 (0.365)	0.8 (0.543)
A. floridana	1.0 (0.417)	1.4 (0.280)	1.0 (0.461)	2.9 (0.120)	*	*	*	*
B. sellowi	0.4 (0.695)	2.2 (0.128)	0.6 (0.705)	0.7 (0.565)	0.4 (0.677)	2.5 (0.093)	1.7 (0.169)	0.2(0.908)

P-values are in brackets. Degrees of freedom are 2, 3, 6 and 3 for Treatment, Time, Treatment x Time and Block, respectively. * *A. floridana* was not found at the boiler slag disposal site.

Figures

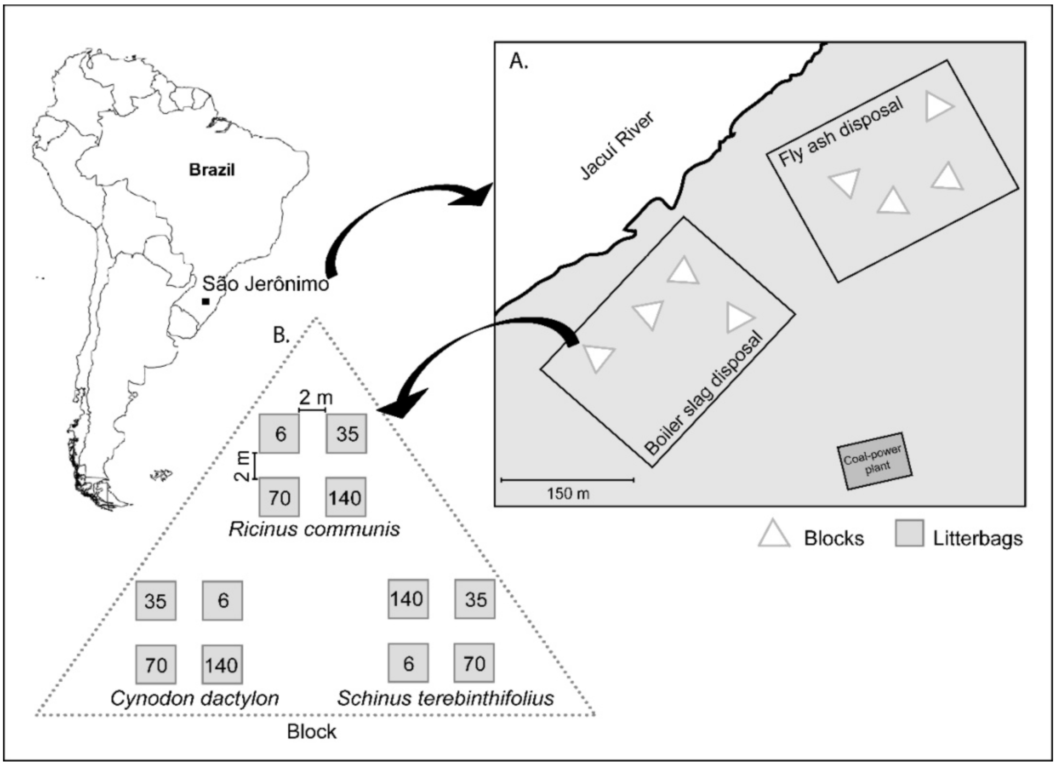

Figure 1. Study area and sites in São Jerônimo, Rio Grande do Sul, Brazil, showing the distance of the two sites from the coal power plant and to the Jacuí river (A). Each study site had four blocks with four litter bags of each plant species, disposed as illustrated inside the triangle (B). The numbers in the litter bags indicate sampling dates (days after exposure).

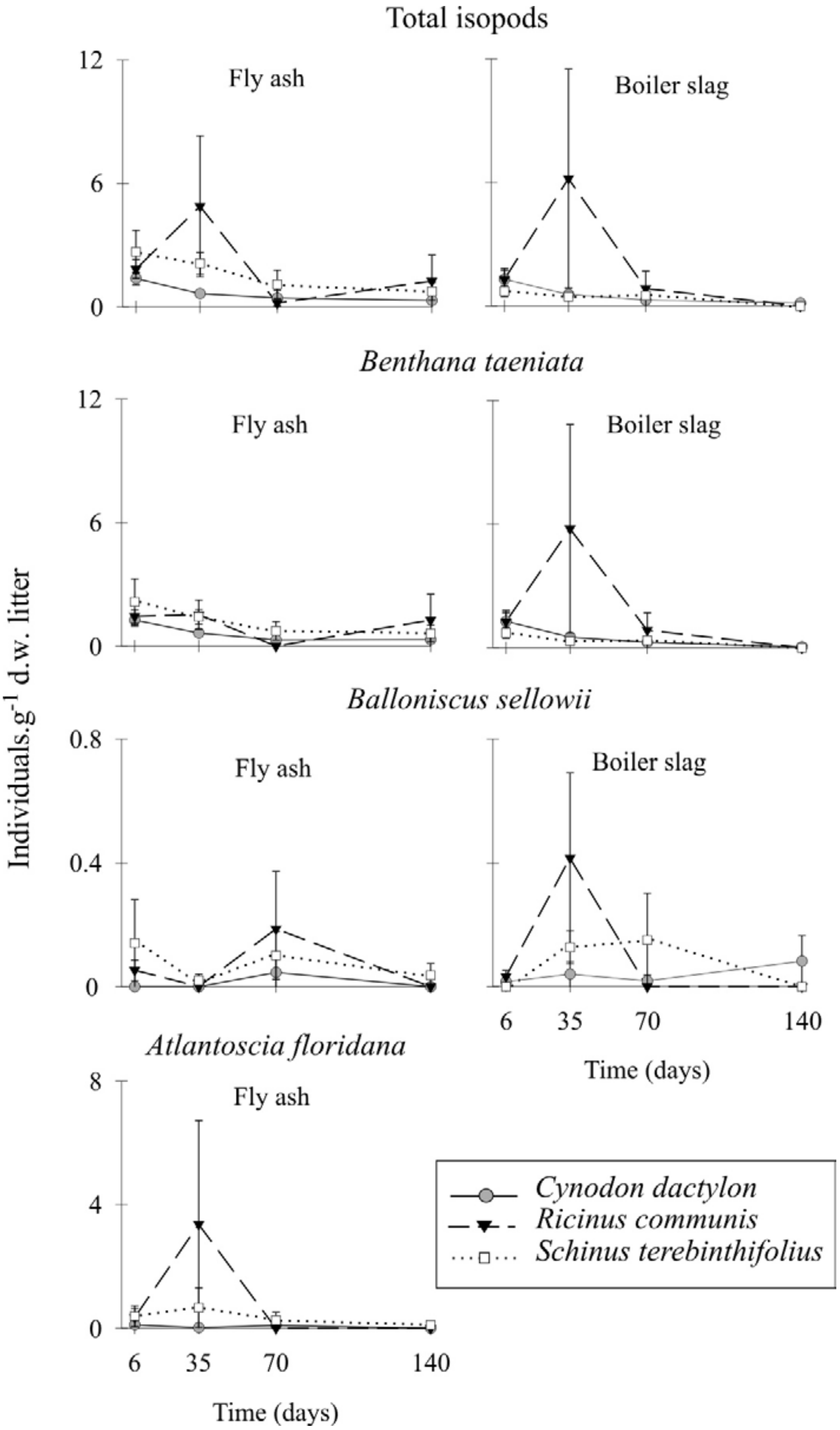

Figure 2. Mean ± standard error of abundances (individuals.g^{-1} d.w. litter) of terrestrial isopods in different plant species in litter bags and in four sampling occasions (6, 35, 70 and 140 days) in two coal ash disposal sites.

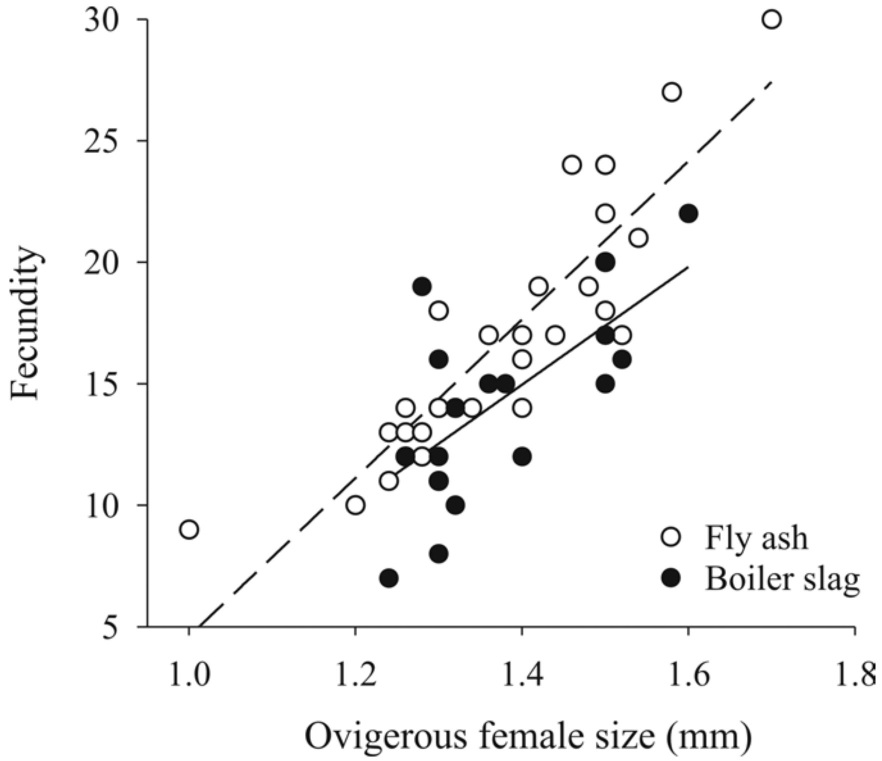

Figure 3. Relationships between fecundity (marsupial content) of *Benthana taeniata* females and female size (cephalothorax width) in two coal ash disposal environments (fly ash and boiler slag). Each point represents one ovigerous female (n= 53).

ARTIGO 3.

Does enhancement of mix leaves on the soil improve macroarthropod diversity and litter decomposition in a post-mining pinus forest?*

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Summary

In ecological systems, the diversity of one component may promote the diversity in other components (e.g., belowground resources vs. aboveground biota). In the case of plant litter vs. soil invertebrates, the diversity of the latter would be promoted mainly due to the microhabitat spatial heterogeneity and a diverse set of food resources. In this study, we hypothesized that the enhancement of mix native leaves on the floor of a mined *Pinus elliottii* monospecific forest in southern Brazil would increase the soil macroarthropod community diversity. Furthermore, because high quality and structured litters may promote the decomposition of the associated litters by interactions between them, we also hypothesized that our experiment could alter the decomposition dynamics of the *P. elliottii* litter. We showed that mix leaf-litter plots attracted a higher individual density, which was 25% more species and a more distinct macroarthropods species composition when compared to the *P. elliottii* leaf-litter plots. Moreover, considering spiders separately, the increase in species richness reached upwards to 58%. The mass and nutrient loss of the *P. elliottii* leaf-litter were not altered when in contact with mixed leaf-litter. On the other hand, the nutrient content composition of litter was responsive to the treatment, where we found a higher P (%) and K (%), but less of C (%). Thus, it appears that the input of diverse resources on a damaged monospecific forest can improve the soil biodiversity and alter the dynamic of an ecosystem process. These results may have implications for forest management and the restoration of damaged systems.

Key words: Brazil, leaf-litter decomposition, litter mixing, soil fauna

Introduction

The human effects on biodiversity are known worldwide. Changes in land use through a variety of activities increases the rates of deforestation, soil exhaustion, alien invasion, and species extinction. Habitat degradation and the changes in biotic functional diversity are usually accompanied by changes in the ecosystem properties (Hopper et al. 2005). The impact of the conversion of natural to agricultural systems, for example, reflects drastically in the decrease on plant diversity, which in turn reduces the range of invertebrates and the functions that they drive, such as those related to decomposition (Swift and Anderson 1994, Hooper et al. 2000). Because biodiversity influences, and is influenced by, the system components at many scales, an enhancing of our understanding about the drivers of diversity are required in front of environmental changes (Tilman 1994, Wardle et al. 2004), so as to seek the best decisions about ecosystem management and conservation.

In terrestrial ecosystems, diversity of one component may promote the diversity in other components (Siemann et al. 1998, Tews et al. 2004, Armbrecht et al. 2004). This linkage is clearly recognized mainly when we look at the influences of the aboveground communities towards the belowground communities. A high diversity of plants can input high litter diversity into the soil, which favors a diversity of decomposers, detritivores and organisms from the higher trophic levels in the soil food web (Hooper et al. 2000). Several litter qualities and litter types offer both diverse food and microhabitats to the soil biota, favoring an increase in niches and resource-use differences and a reduction of interespecific competition, and fostering possibilities of more species coexistence with a greater use of resources (MacArthur 1970, McKane et al. 2002, Finke and Snyder 2008). Several recent studies have emphasized the responses of soil microbes, nematodes and mites to the plant species diversity (reviewed in Wardle 2006). Despite the large representativeness of macroarthropod groups in the soil diversity, a great importance of their ecological roles (Seastedt and Crossley 1984, Giller 1996, Lavelle et al. 2006) and their potential to be linked with plant communities (Siemann et al. 1998, Shaffers et al. 2008), to our knowledge, they have been rarely addressed in studies concerning relationships between litter resource diversity and the belowground diversity (e.g., Armbrecht et al. 2004).

Litter decomposition is a key process to determining soil fertility and quality in ecosystems; it is governed by interactions among physical parameters (such as climate and soil proprieties), resource quality and organisms (Swift et al. 1979, Lavelle et al. 1993). Decomposer invertebrates directly affect litter decomposition through litter fragmentation and modifications of the structure, and the activity of the microbial community, which in turn is responsible for most of the carbon and energy flow (Hättenschwiler et al. 2005). The chemical properties of leaves, especially nitrogen, lignin and polyphenols are important predictors for biota action and litter decomposition (Palm and Sanchez 1991, Zhang et al. 2008). Therefore, the litter decomposition of different plant species has the potential to influence each other by affecting the dynamics of litter mass loss, litter nutrient content and decomposer abundance and activity (Gartner and Cardon 2004). The pattern is that high quality litters may promote the decomposition of any of the associated litters, while poor quality litters have negative effects on the decomposition of other litters (Seasted 1984, Quested et al. 2002, Wardle et al. 2003, Gartner and Cardon 2004).

Despite the fact that many natural ecosystems have a high diversity of plants in the aboveground community, damaged anthropized systems with monospecific plant dominance are easily found in silvicultures, constituting single homogeneous litter layers. If located in infertile conditions, these systems may undergo slow decomposition rates, in which nutrients are conserved and soil carbon sequestration is promoted (Wardle et al. 2004). Litter of monocultures, and soil under infertile conditions, may present a reduced soil biodiversity, altered food webs, and negative feedbacks in linkages between the aboveground and belowground compartments (Swift and Anderson 1994, Hansen 2000, Wardle et al. 2004, Haase et al. 2008).

A common assumption about suitable management practices is that the goals to restore and rebuild the landscape diversity/heterogeneity within anthropized systems will help to maintain the biota diversity and preserve a range of ecosystem functions (Isaacs et al. 2009, Samways 2007). In the present work, we added an amount of mixed leaves from native trees to the floor of a post-mining monospecific forest of *Pinus elliottii* Engelm (Pinaceae) in Brazil, with a goal of enhancing the soil fauna diversity and the *P. elliottii* leaf litter decomposition. Specifically, we experimentally tested two hypotheses: (1) that different species of native leaves would create soil substrates with different structural, chemical, physical and microbial community properties, and that these differences would in turn increase the diversity of the soil macroarthropod community; and (2) that the decomposition dynamics of the *P. elliottii* litter could be altered both directly by the interactions with mix native litters and indirectly through the interaction with an enhanced soil biota community.

Methods

Study area

The experiment was performed from December 2007 to July 2008 in a monodominate forest of *P. elliottii* (approx. 20 ha; 30°09’44. 2”S; 52°00’17. 2”W), in the city of Minas do Leão, Rio Grande do Sul state, Brazil. The landscape in this region is characterized by a mosaic of land uses, including coal mining activities and agricultural systems (especially exotic silvicultures). Native forests mainly remain along the streams as riparian vegetation. The climate represents a Cfa type of Köppen-Geigen climate classification: temperate with a hot summer, and precipitation in all of the months (Peel et al. 2007).

Coal was exploited by opencast mining from the study area around the year 1980 and between 2001-2002. The cave was filled with soil, coal wastes and coal combustion residues. Whereas revegetated with grasses, a spontaneous *P. elliottii* colonized itself on the land (Fig. 1A), quickly establishing a monodominate forest (personal communication, Pedro Paulo da Silva Batista- mining work chief of CRM, Companhia Rio-Grandense de Mineração). Plants from this tree genus are exotic and invasive in South America, threatening natural ecosystems and advancing in deforested areas (GISP 2005).

At the same time of our experiments, we obtained data of litterfall production of this forest. Starting from 12 litterfall collectors (1 m^2; mesh 1 X 3 mm), we found an average production of 2,005.3 $kg.ha^{-1}$ of *P. elliottii* leaf-litter in the summer period (December 2007 to March 2008), and 1,531.8 $kg.ha^{-1}$ in the autumn period (March to July 2008).

Experimental design

We assigned sixteen 90 X 60 cm plots on the forest floor in pairs to either a mix leaf-litter or a control treatment (8 replicates/treatment; Fig. 2). The pairs were separated by more than 100 meters from each other, and the plots from the same pair were separated two meters from each other. Each plot was composed of nine leaned litter bags (30 X 20 cm length; mesh 10 X 2 mm; Fig. 1B). Control plots presented all litter bags containing *P. elliottii* leaves. The mix leaf-litter plots had one litter bag containing *P. elliottii* leaves in the center of the plot, and eight surrounding litter bags with native leaves from four different tree species. Of these, four litter bags had single species, and four had a mix of native species. We based the experiment on leaves that were cut from living trees, collected from the study region. The leaves were placed in litter bags after oven-drying. Each litter bag was filled with 40 ± 0.016 g of leaves, and each plot consisted of about 360 g. The mix litter bags were standardized by volume basis of the four native species.

The native species were: *Cupania vernalis* Cambessedes (Sapindaceae)*, Inga marginata* Willdenow (Fabaceae), *Luehea divaricata* Martius (Tiliaceae) and *Schinus terebinthifolius* Raddi (Anacardiaceae). These species represent common trees in the forests of the study region (Reitz et al. 1983). Leaves of *C. vernalis* (Fig. 1C) and *S. terebinthifolius* (Fig. 1D) are composts, and their leaflets measure about 10 x 3.5 cm and 5 X 2.5 cm, respectively. Leaves of *L. divaricata* (Fig. 1E) and *I. marginata* (Fig. 1F) are generally found completely curly on the forest floor; they have a mean measure of 10.5 X 5 and 7.5 X 2.5 cm, respectively. Leaf nutrient contents (%) of the plants are presented in Table 1.

Sample analyses

We sampled the litter bags after three (March - Summer) and six months (July - Autumn) of the exposure of the experiments. At each sampling time, we removed four pairs of plots randomly. Soil macroarthropods (> 2 mm; Swift 1979) were immediately extracted from the litter bags by hand and by using modified Berlese-Tüllgren funnels (6 days).

The macroarthropods were classified into morphospecies. Immature spiders and insect larvae were not classified into morphospecies. The mass remaining in each litter bag was determined after oven-drying the leaf-litter (60°C; 72 h). The contents of carbon and macronutrients (nitrogen, phosphorous, potassium, calcium and magnesium) were obtained for the *P. elliottii* litter from the central litter bag of the plots. The carbon content was determined by moisture combustion/Walkley-Black method (Walkley and Black 1934); nitrogen was analyzed by the Kjeldahl method (Kjeldahl 1883). Phosphorous and potassium were assessed by nitric/perchloric acid digestion followed by determination on an ICP-OES. Calcium and Magnesium were extracted in KCl mol L^{-1}. All analyses were conducted at the Laboratory of Soil Analysis at UFRGS.

Data analysis

We assessed the effects of the mix leaf-litter addition on the macroarthropod diversity by comparing the plots of both treatments, considering data of all litter bags from the plots. For the assessment of the effects on the decomposition and nutrient dynamics of *P. elliottii* leaf-litter, we took into account the data of the central litter bag inside each plot.

Arthropod abundance was standardized to a density measure in each plot:

individuals.g^{-1} dry weight (d.w.) litter. We used analysis of covariance (ANCOVA) in blocks to test the effects of the treatments in the total macroarthropod density. The sampling dates (time) were used as covariate factors in the analysis. Also, we evaluated the effects of the treatments in the density of the six most abundant macroarthropod groups (> 3% of total abundance) using the multivariate analysis of covariance (MANCOVA) in blocks. In cases where MANCOVA yielded a significant result, we performed univariate ANCOVAs to test for a difference among each variable separately. Data from these analyses were square root transformed to satisfy the parametric assumptions of normality of residuals and homogeneity of variances.

We compared the morphospecies richness of the macroarthropods (considering all groups together, the two richest groups separately, and the other remaining groups together) between the leaf-litter treatments using rarefaction curves with confidence intervals (95%). We performed this analysis in order to remove the effects of sample abundance on morphospecies richness (Gotelli and Colwell 2001, Vasconcelos and Melo 2008). We used nonmetric multidimensional scaling analyses (NMDS, Kruskal 1964) to ordinate sample units (plots) and to assess whether the morphospecies composition (considering all groups together, the two richest groups separately, and the other remaining groups together) was responsive to the treatments. Rare species that occurred only in one plot were deleted from the analyses to improve the interpretability of the ordination plot (McCune and Grace, 2002). We used the Bray-Curtis distance metric in the data of species density. To determine whether the treatments were statistically different, we used a two-way analysis of similarity (ANOSIM; Clarke 1993) on a Bray-Curtis measure matrix. Both rarefaction, NMDS and ANOSIM were performed with PAST 1.9 (Hammer et al. 2001).

To test *P. elliottii* mass loss in the treatments, we used the analysis of covariance (ANCOVA) in blocks. The mass loss of entire plots (considering all litter bags from the treatments) was also assessed. We evaluated the nutrient content proportions (%), and the nutrient content loss in *P. elliottii* litter using the multivariate analysis of covariance (MANCOVA) in blocks.

Results

A total of 2,291 individuals from the orders Araneae, Blattodea, Chilopoda, Coleoptera, Diplopoda, Hemiptera, Hymenoptera, Isopoda, Opiliones, Orthoptera and Pseudoscorpiones were collected (Table 2, Appendix 1). Predators (spiders, centipedes, and pseudoscorpions) were dominant in the number of individuals (49%) in our experiment; 98% of this was represented by spiders. Ants, beetles and harvestman, which have species with a range of feeding strategies inside the major group, represented 26% of the total individuals. Herbivores, such as hemipterans and crickets, were around 14%; and groups with detritivorous habits, such as millipedes, cockroaches and woodlice had 11% of the individuals. A total of 109 morphospecies were identified, with 75% of these being represented by beetles (45%) and spiders (30%; Table 2, Appendix 1). About 58% of the total species were rare, being 50% *singletons* (represented by only one individual) and 8% *doubletons* (represented by only two individuals). For the beetles, 79% were rare species (67% *singletons* and 12% *doubletons*), and for the spiders, 53% were rare (47% *singletons* and 6% *doubletons*).

Macroarthropods response to mix leaf-litter input

As we hypothesized, the addition of mix native leaves onto the forest floor would show positive effects on the macroarthropod diversity. Mixed leaf-litter plots supported more individuals (ANCOVA, $F_{1,7} = 24.48$, $P < 0.001$) and attracted 25% more morphospecies (Fig. 3A) than the control plots. The morphospecie composition (Fig. 4A) was strongly responsive to the leaf-litter treatment (ANOSIM, $R = 0.72$; $P < 0.001$) and to the time of the samplings (ANOSIM, $R = 0.75$; $P < 0.001$).

From the eleven classified groups of macroarthropods, ten and seven groups had their respective total abundances and morphospecies richness increased in at least one sampling time in the mix leaf-litter plots (Table 2). We found a significant response concerning the densities of the six most abundant groups (Table 2) in the leaf-litter treatments (MANCOVA, Pillai Trace = 0.99, $F_{1,7} = 50.72$, $P = 0.019$). Separate analyses yielded a significant positive effect of the mix litter in the spider, cockroach, beetle, millipede, and ant densities, but did not for the hemipteran densities (Table 3).

Spiders had 58% more of a species richness in the mix leaf-litter than in the control plots (Fig. 3C) and their morphospecie composition (Fig. 4B) was clearly different in the treatments (ANOSIM, leaf-litter $R = 0.72$, $P = 0.001$; sampling time $R = 0.54$, $P = 0.002$). The rarefied species richness of beetles in the mix leaf-litter plots did not differ from the control, indicating that the increase in the total species richness (show in the Table 2) was a sampling artifact that resulted from an increased sample abundance (Gotelli and Colwell 2001). Mainly due to the increased number of species that occurred in only one plot (84%) and having been excluded from analysis, we decided to analyze the composition data of the beetles (8 species remaining) together with the other arthropod groups.

The rarefied species richness of the other arthropod groups analyzed together (excepting the richest orders- i.e., spiders and beetles) did not differ between the leaf-litter treatments. On the other hand, their species composition (in this case, only excepting spiders) was significantly responsive to the treatments (Fig. 4B; ANOSIM, leaf-litter R=0.45, P< 0.001; sampling time R=0.69, P< 0.001).

Pinus elliottii leaf-litter response to mix leaf-litter

Pinus elliottii needles have a lower litter quality compared to the native broad leaves (Table 1). Thus, mass loss of entire plots (all litter bags *per* plot analyzed together) was very high in the mix leaf-litter plot ($F_{1,7}$ = 135.05, P < 0.001; Fig. 5A) compared to control plots.

Not as expected, the *P. elliottii* mass loss ($F_{1,7}$ = 0.51, P = 0.50; Fig. 5B) and nutrients loss (Pillai Trace = 0.84, $F_{1,7}$ = 1.83, P = 0.39) were not influenced by the contact with the mix leaf-litter. However, the proportions of the nutrients content (%) exhibited responses to the treatments (Pillai Trace = 0.98, $F_{1,7}$ = 17.56, P = 0.05). Univariate analysis revealed significant differences in K (%) and P (%), and marginal differences in C (%) (Table 4; Fig. 6). Potassium and P concentrations were higher in *P. elliottii* leaf-litter when in contact with mix leaf-litter. On the other hand, C concentrations seemed to be increased in the control plots.

Discussion

The main conclusion of this paper is that the input of diverse resources upon damaged soil in a monospecific forest favored an increase in the soil arthropod diversity and differential species composition. Our experimental technique strongly influenced diversity of many groups such as spiders, cockroaches, beetles, harvestman, millipedes, ants and woodlice. In particular, characteristics such as microhabitat spatial heterogeneity and a range of food resources could attract diverse organisms to the mix native litter patches. Our experiment also shows that, unlike the expected, the input of native mix leaf-litter did not change the *P. elliottii* litter decomposition rates. On the other hand, our strategy altered the *P. elliottii* litter nutrient contents, probably trough nutrient transferation or differential microbial activity.

Macroarthropods response to mix leaf-litter input

As other experimental studies with resource manipulation (Chen and Wise 1999, Leroy et al. 2007), we have an overlap in the factors (microhabitat and food enhancement) enriching the soil fauna diversity in the mixed litter plots. Concerning microhabitat enhancement, it is known that a diverse litter layer can support higher spatial structure heterogeneity through different leaf-sizes, leaf shapes and leaf-surface structures than monospecific litter layers (Hättenschwiler et al. 2005). This prediction is yet more convincing when we compare the structure of the needle litter of *P. elliottii* with the native broadleaf litters studied. Uetz (1974) reported that more spatially structured litter layers produce more habitat space and then can offer more suitable shelters against harsh climatic conditions and predators, microsites for foraging, mating, oviposition, etc. In this way, all trophic levels of the soil fauna could be positively influenced by the increase in microhabitat complexity (Stevenson and Dindal 1982, Kaneko and Salamanca 1999, Hansen 2000, Tews et al. 2004, Lassau et al. 2005). On the other hand, habitat preferences also reflect foraging habits. Bottom-up controls propagated through trophic levels are very common in detritus food webs and are related in experiments that exclude the enhancement in microhabitats (Scheu and Schaefer 1998). The input of mix leaves, which play the role of food with diverse qualities and chemical compositions, may lead to a diversity of detritivores through food selectivity and resource partitioning (Hooper et al. 2000, Wardle 2006). Perhaps, the microenvironment, which is created by the mix of different species of trees and influenced by differentiate water retention and decomposition, favors the growth of different types of fungi and bacteria (Armbrecht et al. 2004, Hättenschwiler et al. 2005); these conceivably constitute part of the diet of detritivores. The diversity of predators may be also responded to the diversity of preys (Seetle et al. 1996, Chen and Wise 1999, Scheu and Schaefer 1998, Miyashita et al. 2003).

The interaction between species through top-down controls in the soil food web can limit the influence of the aboveground diversity toward the belowground species (Wardle and Yeates 1993, Hooper et al. 2000). Spiders were approximately 48% of the total macroarthropod individuals found and they had a significant enhanced density in the mixed leaf-litter plots. This result reflects a consistent pattern from literature, in which spiders tend to increase their abundances in a more intensive way than did other groups in front of an increased habitat complexity (reviewed in Langellotto and Denno 2004). Structurally rich habitats can diminish the antagonistic interactions among predators, such as intraguild predation and have effective implications to prey suppression (Finke and Denno 2002, Langellotto and Denno 2006, Sanders et al. 2008). Thus, the increased densities of spiders may lead to negative effects in diversity of other invertebrate groups (their preys; Wise et al, 1999) as in the example of the case of hemipteran density and beetle richness, therefore hiding some positive correlations between the preys and the enhancement of the mix litter.

Pinus elliottii leaf-litter response to mix leaf-litter

Despite many works that have demonstrated the additive effects of mixes in the decomposition rates of individual litters (Chadwick 1998, Gartner and Cardon 2004), we did not find any direct effects between the litter mass and the loss nutrients of *P. elliottii*. Indirectly, we show significant changes in its chemical composition, which is related to the differential concentrations of phosphorous, potassium and carbon. We identify three plausible explanations. First, as nutrient concentrations usually increase in the lower quality litter, while decreasing in the higher quality litter though nutrient transfer (Wardle et al. 1997, Gartner and Cardon 2004), the greater concentrations of phosphorous and potassium may have leached from the surrounding high quality native litters. Second, in view of the increased diversity of soil macroarthropods in the more diverse litter layer, we suppose that a greater production of diverse fecal materials may have occurred in this treatment. As assimilation efficiency is low in the food digestion by soil organisms (Wolters 2000), an increased number of nutrients may be coming from the animal's defecation. Third, as the soil carbon and energy flow is mainly driven by microbial activities (Coleman et al. 2004), which are substantially influenced by the two previous topics, the changed chemical composition of *P. elliottii* in the mix litter treatment may be reflecting the action of the differential microbial communities. Infertile soils tend to support a fungal-based energy channel (Wardle et al. 2004). Increased potassium and decreased carbon concentrations may be related to greater fungal activities and respiration (Coleman et al. 2004, Sayer 2006).

Then, why did we not find a major mass loss in the *P. elliottii* litter when surrounded by heterogeneous native litter? One plausible possibility is that our experimental time frame was too short to find an effect in the decomposition rates;

because in the end of the experiment we still had more than 50% of initial remaining leaf-litter in the plots. Another speculative reason is the existence of top-down trophic cascades that are induced by predators (Wise et al. 1999). Spiders are evolved in decomposition food webs and have the potential to generate positive or negative impacts on the rates of litter mineralization. Wise et al. (1999) hypothesized that in a four-level food chain, an increase in spiders should lead to a decrease in detritivores/fungivores (e.g., Collembola). By relieving the predation pressure in fungi, the nutrients may become immobile in senescent fungal hyphae, not entering in the soil.

Ecological implications

We found that the input of diverse resources on a damaged monospecific system can improve the soil macroarthropod biodiversity and alter the dynamic of an ecosystem process. These results certainly have great implications in conservation biology and restoration ecology. Isaacs et al. (2009), attempting to promote the conservation of the beneficial aboveground arthropods in agricultural landscapes, showed that the increase in native plants provides local adaptation, habitat permanency, and support of native biodiversity, therefore maximizing arthropod-mediated ecosystem services, such as crop pollination and pest control. Here, we showed that the input of native leaves on the floor of an exotic monospecific forest benefits belowground arthropods, probably invigorating biotic and abiotic relationships and also maximizing the soil arthropod-mediated ecosystem services (Lavelle et al. 2006). Moreover, the technique of incorporating diverse organic matter in poor anthropized systems have clear benefits to the soil nutrient enrichment, and consequently, to primary productivity (Neher 1999).

Finally, based on our data and taking into account the increased rates of habitat degradation and the changes in biotic diversity and ecosystem properties around the world, we suggest two aspects related to ecological restoration practices. First, it should maintain the quality of the landscape heterogeneity, simulating conditions of natural ecosystems (Samways 2007, Podgaiski 2007) to benefit fauna diversity permanency. Second, it should seek soil fertility maintenance, which will in turn have positive consequences for above–belowground linkages and their effects on natural communities (Haase et al. 2008).

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Tables

Table 1. Initial leaf nutrient content (%) of the plants.

Plants	C	N	P	K	Mg	Ca
Pinus elliottii	44.75	1.02	0.09	0.32	0.18	0.52
Cupania vernalis	47	2.8	0.29	1.7	0.24	0.42
Inga marginata	44	3.7	0.24	0.87	0.15	0.42
Luehea divaricata	45	2.4	0.20	0.93	0.28	0.66
Schinus terebinthifolius	43	1.2	0.12	0.65	0.36	1.8

Table 2. Total abundance and morphospecies richness (in brackets) of soil macroarthropods in the mix leaf-litter plots and control plots in two sampling times.

Class	Order	Mix leaf-litter plot		Control plot		Total
		3 months	6 months	3 months	6 months	
Arachnida	Araneae	296(16)	538(24)	63(3)	213(11)	1110(33)
	Opiliones	0	23(1)	0	1(1)	24(1)
	Pseudoscorpiones	1(1)	0	0	0	1(1)
Chilopoda		0	12(2)	0	6(2)	18(2)
Crustacea	Isopoda	8(1)	13(1)	0	0	21(1)
Diplopoda		8(1)	128(2)	0	16(2)	152(2)
Hexapoda	Blattodea	29(3)	46(2)	3(1)	3(1)	81(3)
	Coleoptera	13(10)	72(31)	6(5)	29(13)	120(49)
	Hemiptera	6(3)	187(4)	3(1)	118(2)	314(5)
	Hymenoptera	58(5)	271(3)	32(6)	88(3)	449(11)
	Orthoptera	0	0	0	1(1)	1(1)
Total		419(40)	1290(70)	107(16)	475(36)	2291(109)

Note: Each column represents the diversity values from four plots.

Table 3. F and P-values (in brackets) from ANCOVA in block results for square root transformed densities (individuals.g-1 d.w. litter) of five macroarthropod groups.

Response variable	Treatment (df=1,7)	Blocking (df=6,7)	Time (df=1,7)
Araneae	**54.93 (<0.001)**	3.51 (0.06)	**21.79 (0.002)**
Blattodea	**20.49 (0.003)**	0.65 (0.689)	0.42 (0.537)
Coleoptera	**8.12 (0.024)**	0.85 (0.567)	**16.26 (0.005)**
Diplopoda	**6.24 (0.041)**	1.25 (0.385)	**10.19 (0.015)**
Hemiptera	1.25 (0.301)	1.49 (0.304)	**24.02 (0.002)**
Hymenoptera	**5.73 (0.048)**	2.80 (0.102)	**8.89 (0.020)**

Note: Sources of variation are: treatment (mix leaf-litter and control plots), blocks (*n* = 8), and time of samplings (3 and 6 months). All statistically significant numbers ($P < 0.05$) are in boldface.

Table 4. F and P-values (in brackets) from ANCOVA in block results for nutrients content (%) of the *P. elliottii* leaf-litter.

Response variable	Treatment (df=1,7)	Blocking (df=6,7)	Time (df=1,7)
N (%)	0.179 (0.684)	1.645 (0.264)	**11.487 (0.012)**
P (%)	**7.986 (0.025)**	**4.831 (0.029)**	**16.662 (0.005)**
K (%)	**10.287 (0.015)**	1.400 (0.332)	**32.200 (<0.001)**
Ca (%)	0.011 (0.918)	3.502 (0.063)	**36.505 (<0.001)**
Mg (%)	0.080 (0.785)	3.203 (0.077)	0.438 (0.529)
C (%)	4.200 (0.079)	1.296 (0.367)	1.296 (0.292)

Note: Sources of variation are: treatment (mixed leaf-litter and control plots), blocks (*n* = 8), and time of samplings (3 and 6 months). All statistically significant numbers ($P < 0.05$) are in boldface.

Figures

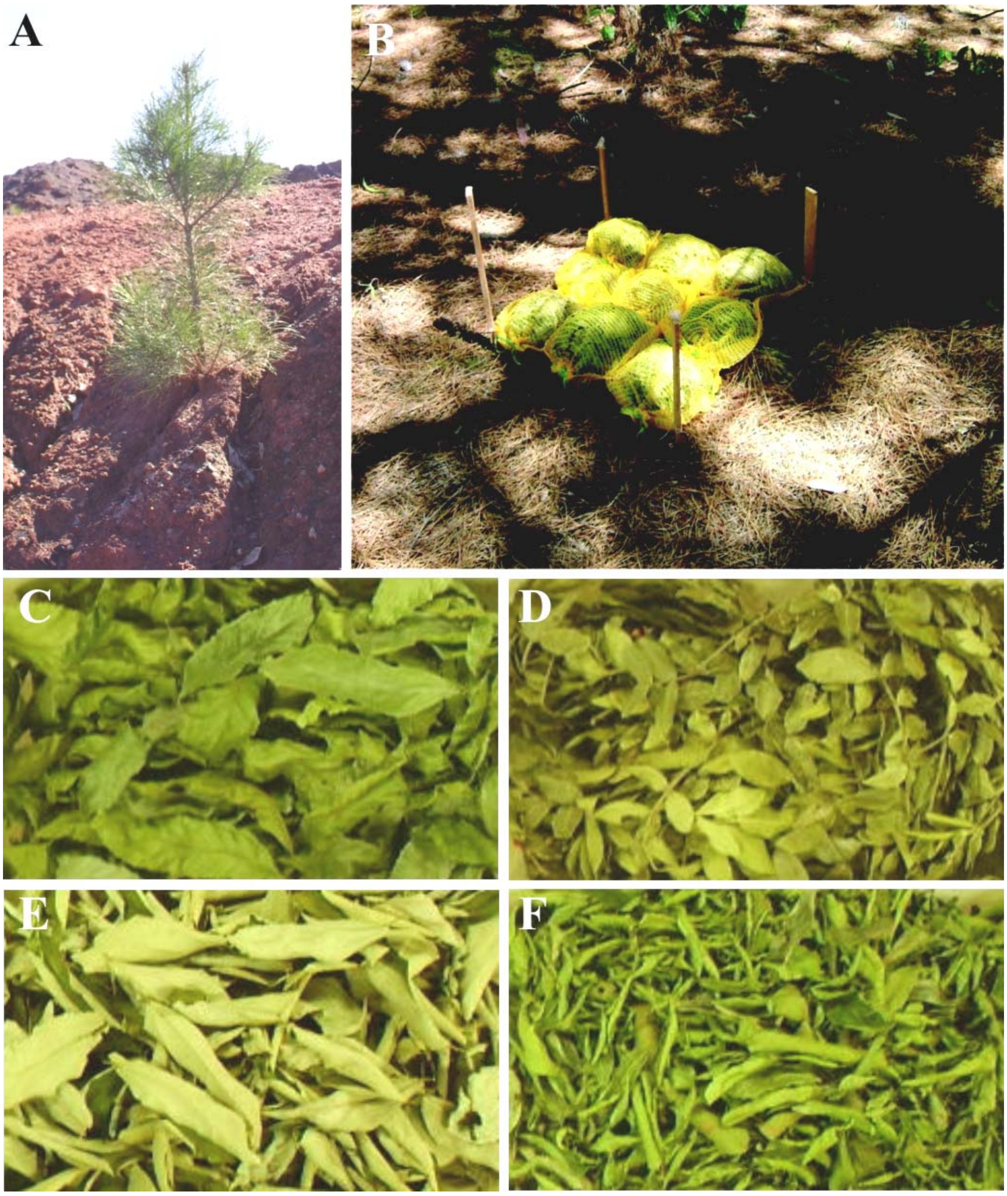

Figure 1. Plantule of *Pinus elliottii* (A) growing up in front of a land with mining activities in the study region, and experimental plot (B) in the post-mining *P. elliotti* forest. Leaves of trees: *Cupania vernalis* (C), *Schinus terebinthifolius* (D), *Luehea divaricata* (E) and *Inga marginata* (F) showed in the same scale.

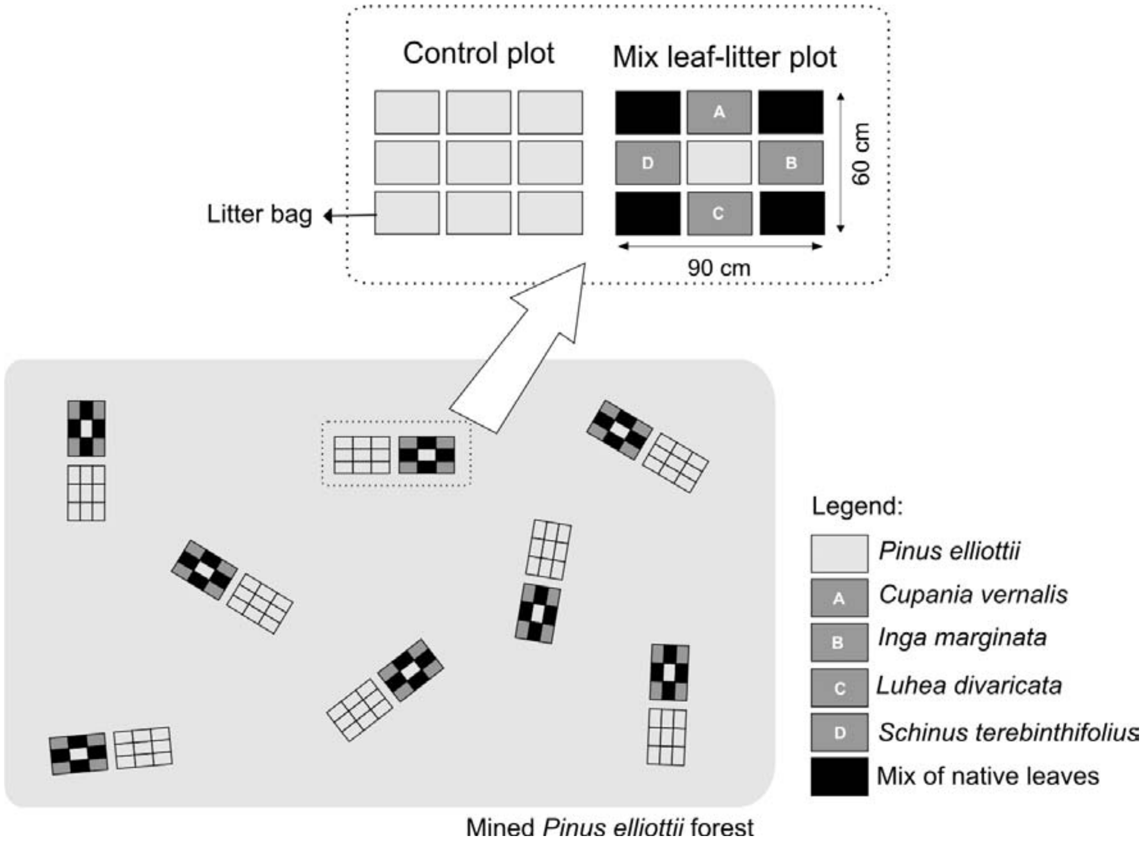

Figure 2. Experimental design testing the effects of the enhancement of leaves from native trees on the floor of a post-mining *P. elliottii* forest on the soil macroarthropod diversity and on the *P. elliottii* litter decomposition.

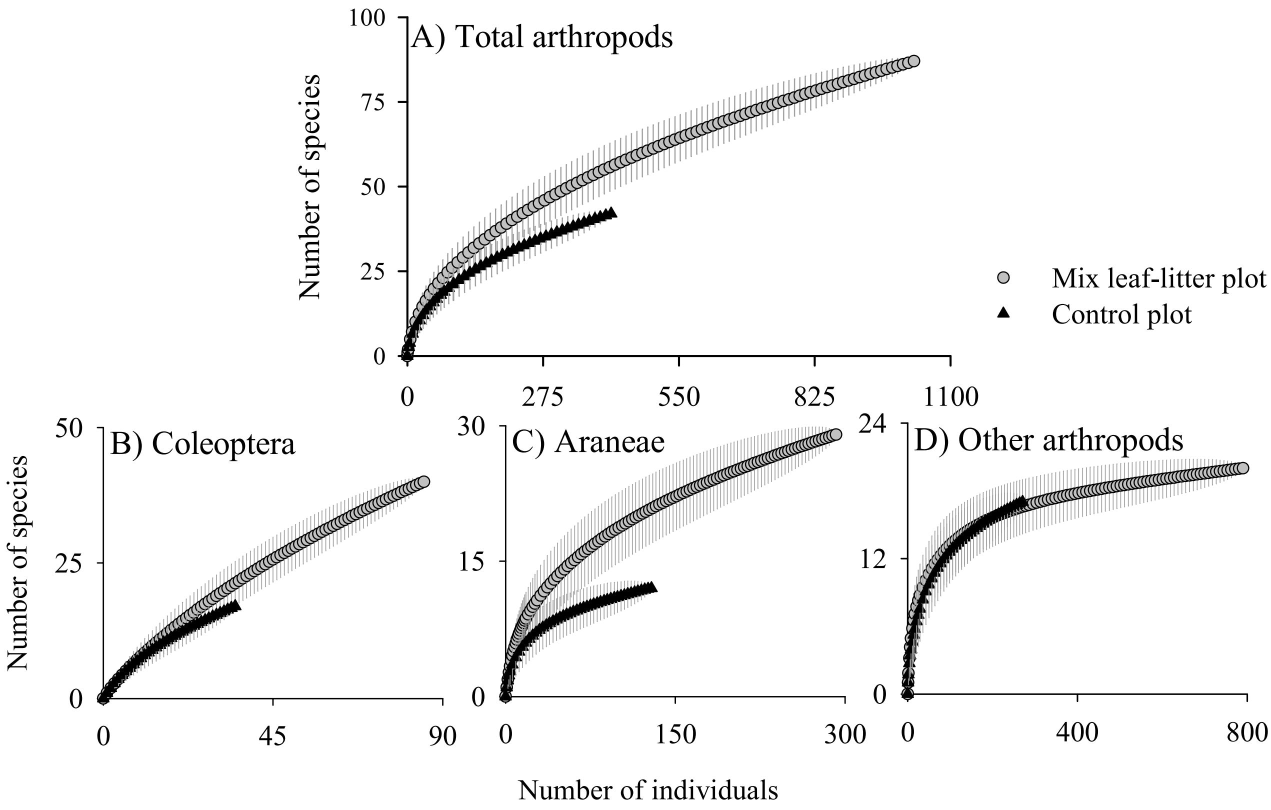

Figure 3. Rarefaction curves for the species richness of: the total soil arthropod abundances (A), Coleoptera (B), Araneae (C) and other arthropods (D) in relation to the effects of the mix leaf-litter and the control plot treatments. Error bars represent a ± 1 confidence interval (CI).

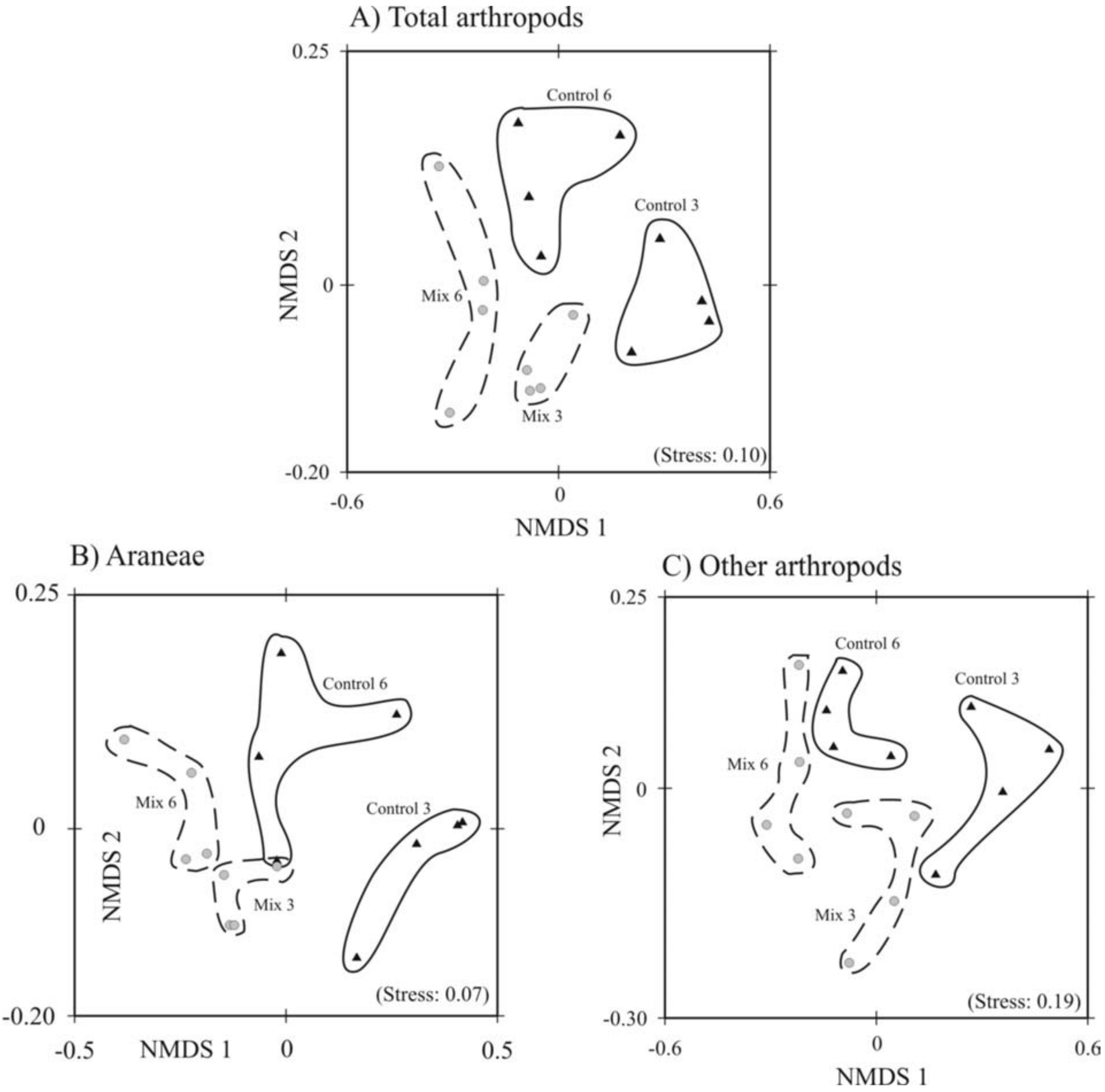

Figure 4: NMDS ordinations of the mix leaf-litter (Mix) and control plots from two sample dates (3 and 6 months) in the first two principal axes, considering total soil arthropods (A), Araneae (B) and other arthropods (C).

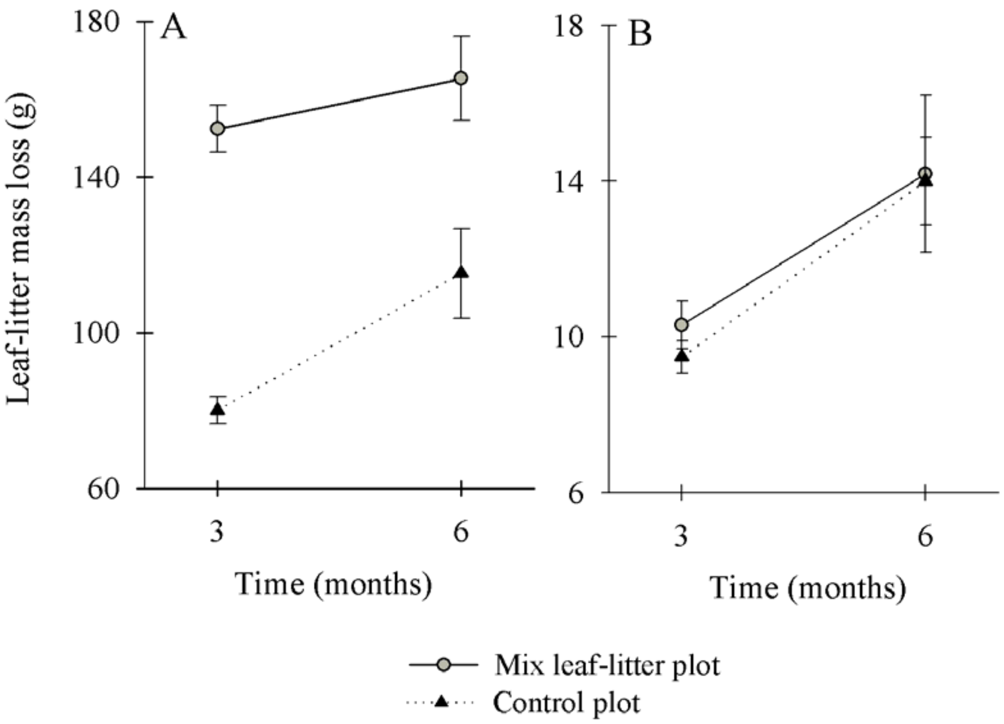

Figure 5: Leaf-litter mass loss of entire plots (A), and single litter bags of *P. elliottii* from the center of the plots.

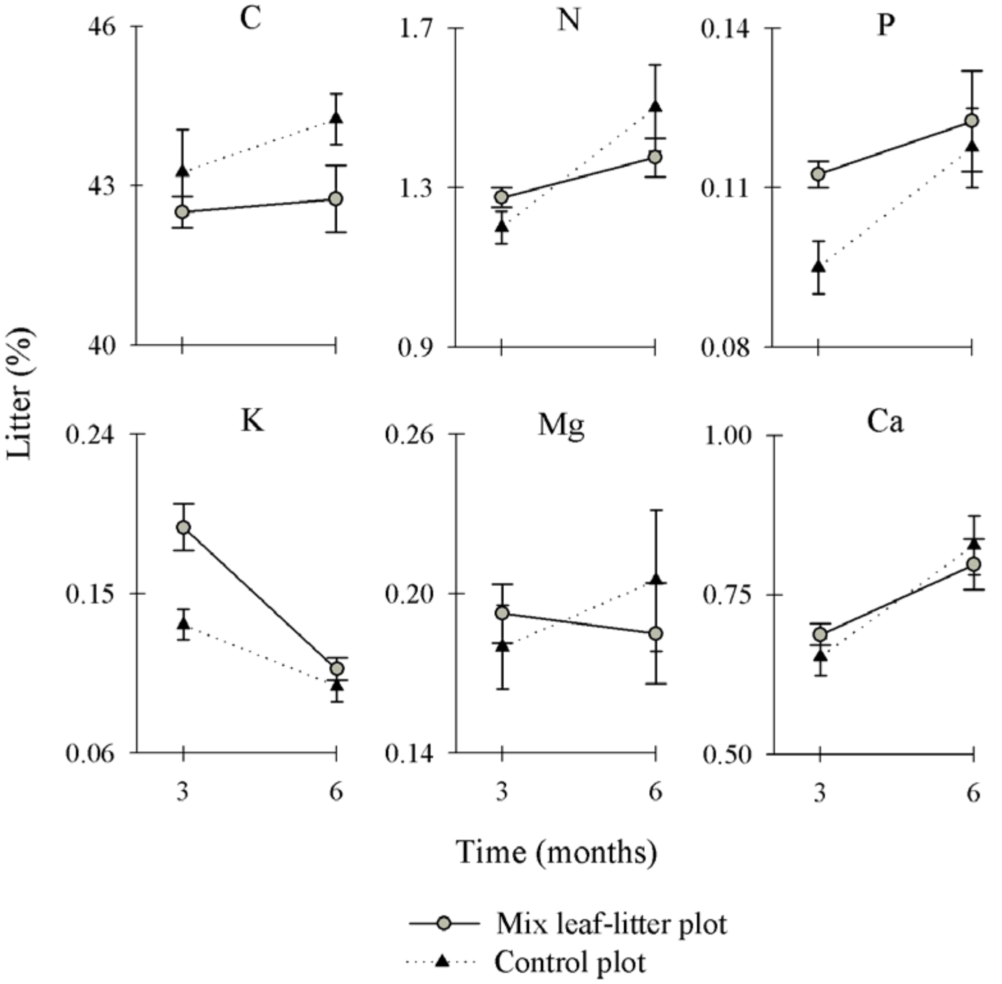

Figure 6: Nutrient concentration (%) dynamics in of *P. elliottii* leaf litter in the mix leaf-litter and control plots.

Appendix 1: Morphospecies composition, and abundance of soil macroarthropods in the mix leaf-litter and control plots at 3 and 6 months of exposure.

Class / Order	Family / Specie or Morphospecie	Mix leaf-litter plot		Control plot	
		3 months	6 months	3 months	6 months
Arachnida					
Araneae	Amphinectidae				
	Immature	0	4	0	0
	Araneidae				
	Immature	0	1	0	2
	Caponiidae				
	Nops meridionalis Keyserling, 1891	6	4	0	0
	Immature	3	5	0	0
	Corinnidae				
	Meriola mauryi Platnick & Ewing, 1995	5	0	0	0
	Orthobula sp.1	21	7	0	3
	Orthobula sp.2	0	1	0	0
	Orthobula sp.3	0	1	0	0
	Orthobula sp.4	0	1	0	0
	Immature	8	0	0	0
	Ctenidae				
	Isoctenus sp.	0	1	0	0
	Immature	9	20	1	4
	Gnaphosidae				
	Apopyllus silvestrii (Simon, 1905)	0	1	0	0
	Gnaphosidae sp.	1	0	0	0
	Immature	23	23	0	3
	Hahniidae				
	Hahniidae sp.1	2	30	0	10
	Hahniidae sp.2	0	1	0	0
	Immature	7	35	4	11
	Linyphiidae				
	Erigone sp.	0	2	0	0
	Gigapassus octarine Miller, 2007	1	5	0	0
	Lygarina sp.	0	55	0	42
	Mermessus sp.	0	1	0	0
	*Neomaso arundicola*Millidge, 1991	1	0	0	0
	Neomaso damocles Miller, 2007	0	15	0	10
	Neomaso sp.1	0	1	0	0
	Neomaso sp.2	0	0	0	1
	Scolecura cambara Rodrigues, 2005	1	1	0	5
	Smermisia vicosana (Bishop & Crosby, 1938)	2	3	1	6
	Sphecozone sp.	1	5	0	1
	Immature	22	86	1	38
	Lycosidae				
	Immature	0	3	1	0
	Migalomorphae				
	Immature	1	0	0	0
	Mysmenidae				
	Itapua sp.	0	2	0	0
	Immature	0	2	0	1
	Oecobidae				
	Oecobius navus Blackwall, 1859	0	0	1	0
	Immature	1	0	1	0

	Oonopidae				
	Oonopinae sp.	4	37	0	2
	Immature	4	15	4	1
	Oxyopidae				
	Immature	0	1	0	0
	Salticidae				
	Aphirape sp.	0	1	0	0
	Breda sp.	1	2	0	0
	Unidentatae sp.	0	0	0	1
	Salticidae sp.1	3	0	0	0
	Salticidae sp.2	1	0	0	0
	Immature	54	98	10	23
	Scytodiidae				
	Immature	7	1	0	0
	Theridiidae				
	Thymoites sp.1	4	1	0	0
	Thymoites sp.2	44	16	23	23
	Immature	59	49	14	25
	Thomisidae				
	Immature	0	1	2	1
Opiliones	Opiliones sp.	0	23	0	1
Pseudoscorpiones	Pseudoscorpiones sp.	1	0	0	0
Chilopoda	Chilopoda sp.1	0	11	0	4
	Chilopoda sp.2	0	1	0	2
Crustacea					
Isopoda	Philoscidae				
	Atlantoscia floridana (van Name, 1940)	8	13	0	0
Diplopoda	Diplopoda sp.1	8	99	0	9
	Diplopoda sp.2	0	29	0	7
Hexapoda					
Blatodea	Blatodea sp.1	20	43	3	3
	Blatodea sp.2	8	3	0	0
	Blatodea sp.3	1	0	0	0
Coleoptera	Carabidae				
	Carabidae sp.1	0	0	0	1
	Carabidae sp.2	0	1	0	0
	Crisomelidae				
	Crisomelidae sp.	0	1	0	0
	Curculionidae				
	Curculionidae sp.1	0	0	0	2
	Curculionidae sp.2	0	1	0	0
	Curculionidae sp.3	1	0	0	0
	Disticidae				
	Disticidae sp.	0	1	0	0
	Nitidulidae				
	Nitidulidae sp.1	0	1	0	0
	Nitidulidae sp.2	1	0	0	0
	Scarabaeidae				
	Scarabaeidae sp.	0	1	0	0
	Staphilinidae				
	Staphilinidae sp.1	0	3	0	0
	Staphilinidae sp.2	0	1	0	0
	Staphilinidae sp.3	0	1	0	0
	Staphilinidae sp.4	0	1	0	0
	Staphilinidae sp.5	0	1	0	0
	Staphilinidae sp.6	0	1	0	0
	Staphilinidae sp.7	0	0	0	1
	Staphilinidae sp.8	0	0	1	0
	Staphilinidae sp.9	0	3	0	0

	Staphilinidae sp.10	0	2	0	0
	Staphilinidae sp.11	0	1	0	0
	Staphilinidae sp.12	0	2	0	0
	Staphilinidae sp.13	0	2	0	0
	Staphilinidae sp.14	0	1	0	0
	Staphilinidae sp.15	0	0	1	0
	Pselaphineae				
	Pselaphineae sp.1	0	1	0	0
	Pselaphineae sp.2	0	1	0	0
	Pselaphineae sp.3	0	1	0	0
	Pselaphineae sp.5	0	4	0	0
	Pselaphineae sp.4	0	0	0	1
	Coleoptera sp.1	0	0	0	1
	Coleoptera sp.2	0	2	0	1
	Coleoptera sp.3	0	0	0	1
	Coleoptera sp.4	1	0	1	0
	Coleoptera sp.5	0	1	0	0
	Coleoptera sp.6	0	1	0	0
	Coleoptera sp.7	0	4	0	3
	Coleoptera sp.8	2	2	0	4
	Coleoptera sp.9	0	23	0	4
	Coleoptera sp.10	0	0	0	1
	Coleoptera sp.11	0	1	0	0
	Coleoptera sp.12	0	5	0	8
	Coleoptera sp.13	1	0	0	0
	Coleoptera sp.14	1	0	0	0
	Coleoptera sp.15	1	0	0	0
	Coleoptera sp.16	2	0	1	0
	Coleoptera sp.17	0	1	2	1
	Coleoptera sp.18	1	0	0	0
	Coleoptera sp.19	2	0	0	0
Hemiptera	Hemiptera sp.1	0	178	0	115
	Hemiptera sp.2	1	6	0	3
	Hemiptera sp.3	2	2	0	0
	Hemiptera sp.4	0	1	0	0
	Hemiptera sp.5	3	0	3	0
Hymenoptera	Formicidae				
	Formicinae				
	Camponotus rufipes (Fabricius, 1775)	4	0	0	0
	Camponotus sp.	0	0	6	0
	Myrmicinae				
	Crematogaster sp.	0	0	1	0
	Pheidole fallax (Mayr, 1870)	2	205	0	0
	Pheidole sp.1	7	0	0	0
	Pheidole sp.2	0	0	6	10
	Pheidole sp.3	0	0	1	0
	Solenopsis (*Diplorhoptrum*) sp.	4	9	0	0
	Solenopsis invicta Buren, 1972	41	57	3	77
	Solenopsis sp.1	0	0	13	0
	Solenopsis sp.2	0	0	1	0
	Solenopsis sp.3	0	0	1	0
	Ponerinae				
	Pachycondyla sp.	0	0	1	0
Orthoptera	Orthoptera sp.	0	0	0	1
TOTAL		**419**	**1290**	**108**	**474**

CONSIDERAÇÕES FINAIS

Estudos ecológicos sobre decomposição foliar de plantas pioneiras e a macrofauna de solo associada em ecossistemas degradados pela deposição de cinzas e extração do carvão a céu aberto foram elaborados e realizados em curto prazo. Apesar de vários estudos terem sido realizados em ambientes extremamente poluídos na Europa e outras regiões do mundo (e.g., Rodrigues 2001; McEnroe et al 2001; Majer et al 2007), abordagens similares a estes estudos, relacionando estas variáveis ecológicas em ecossistemas extremos, ainda são incipientes no Brasil. Há carência de informações tradicionais sobre o funcionamento dos sistemas degradados, o que prejudica a tomada de decisões acerca do manejo, restauração e conservação da biodiversidade destes sistemas.

Em São Jerônimo (RS), o local exato onde cinzas de carvão fóssil (cinzas leves e cinzas grossas) foram despejadas irregularmente durante longos anos constitui área integrante da mata ripária do Rio Jacuí, que por lei (Código Florestal, nº 4771, de 1967) é considerada área de preservação permanente (APP). De acordo com a resolução do CONAMA nº 303 (2002), a função ambiental das APPs é de preservar os recursos hídricos, a paisagem, a estabilidade geológica, a biodiversidade, o fluxo gênico de fauna e flora, proteger o solo e assegurar o bem estar das populações humanas. Com base nestas funções essenciais, dever-se-ia fazer o possível para manter e preservar as características originais destas áreas, não subjugando nenhum uso antrópico degradante a elas. As cinzas de carvão fóssil produzidas na Usina Termelétrica de São Jerônimo (CGTE – Companhia de Geração Térmica de Energia Elétrica) são classificadas como “resíduos sólidos– não inertes” (NBR 10004), e desta forma, seu despejo em áreas naturais deveria ser altamente restrito, uma vez que contêem elementos com propriedades químicas e toxicológicas que podem causar danos à saúde e ao ambiente (Rohde et al. 2006). Sua disposição no solo (NBR 8419), deveria ser necessariamente em um aterro sanitário, local em que apresentaria mínimos impactos ambientais. Segundo a constituição Federal (1998), compete ao gerador de resíduos sólidos a sua inteira responsabilidade sobre eles, compreendendo desde as etapas de acondicionamento, disponibilização para coleta, coleta, tratamento, até sua disposição final ambientalmente adequada. Desta forma, a CGTE, como produtora de resíduos sólidos altamente poluidores (cinzas), e agente de disposição imprópria dos resíduos em APP (mata ripária), apresentou graves irregularidades com a lei.

Esta mata ripária em São Jerônimo, que felizmente não vem mais recebendo os depósitos de cinzas de carvão, está sofrendo o importante processo ecológico da sucessão natural a partir de plantas nucleadoras (Azzolini, 2008). A partir do nosso estudo, foi constatado que a mamona (*R. communis*) –abundante planta pioneira na área - apresentou decomposição foliar muito mais rápida do que outras plantas pioneiras avaliadas (capim-bermuda *C. dactylon* e aroeira-vermelha *S. terebinthifolius*). Isto significa que esta espécie está disponibilizando nutrientes mais rapidamente ao substrato (cinzas), e desta forma, pode estar contribuindo ao melhoramento das condições bióticas locais de uma maneira mais eficiente do que as outras espécies. No entanto, é importante reconhecer o papel de cada espécie dentro da sucessão ecológica na área (Azzolini, 2008). Apesar das três espécies avaliadas apresentarem caráter pioneiro, elas participam em momentos distintos do processo de sucessão, apresentando uma certa ordem na qual ocorrem facilitações e desfavorecimentos, e assim devem ser julgadas com importâncias singulares na transformação do ambiente degradado. A partir dos resultados de Azzolini (2008), podemos supor que as duas espécies exóticas avaliadas (mamona e capim-bermuda) alterem as condições do substrato (cinzas), beneficiando elas mesmas em um primeiro momento, mas também as espécies nativas que se estabelecem no andamento da sucessão, como a aroeira-vermelha. Nesta área, percebe-se o processo de restauração ecológica ocorrendo naturalmente.

O folhiço das três espécies de plantas pioneiras (mamona, capim-bermuda e aroeira-vermelha) avaliadas no estudo de São Jerônimo foram largamente colonizados pela macrofauna do solo nos experimentos, o que demonstra claramente a importância desta camada orgânica para os organismos. Os folhiços da aroeira-vermelha e do capim-bermuda foram similares com relação à diversidade de espécies que suportaram. A mamona, por sua vez, apesar de ter apresentado maiores densidades de indivíduos, apresentou a menor colonização por diferentes espécies, visto a sua rápida desintegração foliar.

Embora a decomposição de folhiço não tenha sido influenciada por diferentes depósitos de cinzas (cinzas leves X cinzas grossas), a diversidade de invertebrados carnívoros, a ocorrência de *A. floridana* (Isopoda), a abundância e fecundidade de fêmeas de *B. taeniata* (Isopoda), e as composições totais de espécies da macrofauna foram fortemente afetadas pelos depósitos. A estrutura física do substrato, o pH não favorável, as diferentes concentrações de metais pesados ou até mesmo características peculiares de cada área, como a composição da vegetação, podem estar influenciando e selecionando as diferentes espécies nos ambientes. Assim como outros experimentos naturais realizados em campo, nosso estudo tem limitações em apontar os mecanismos estritamente responsáveis pelos padrões encontrados. Seriam necessários estudos manipulativos - que padronizassem condições físicas/químicas – mais análises químicas e réplicas amostrais para melhor compreender os efeitos dos tratamentos sobre a decomposição foliar e a macrofauna de solo. Mas, em geral, apesar da não replicação do fator cinza no experimento, parece que as cinzas grossas apresentam mais fatores condicionantes e limitaram mais a macrofauna, o que poderia ser levado em consideração na recuperação destas áreas.

Com relação à área de estudo em Minas do Leão (RS), que teve seu solo minerado e reconstruído topograficamente após a extração do carvão, pode se considerar que apresenta uma série de desequilíbrios nos atributos químicos, físicos e biológicos do seu solo (Kämpf et al. 2000). A colonização espontânea da exótica *P. elliottii* neste sistema pós-minerado não permite sucessão natural, ao contrário do que vem ocorrendo nos depósitos de cinzas. Benefícios sobre o ambiente pretérito, sem dúvida, são intrínsecos ao estabelecimento desta floresta monodominante de pinus, mas intervenções de manejadores e restauradores, neste caso, são bem vindas em ordem a estimular processos carentes e incrementar a biodiversidade.

Tendo isto em vista, a adição de folhiço de plantas nativas sobre as camadas homogêneas de acículas de pinus no solo da floresta de pinus demonstrou ser uma prática de manejo viável, uma vez que modificou a composição química do folhiço de pinus, representando uma diferenciação na ciclagem de nutrientes. Esta prática também enriqueceu a fauna de artrópodes de solo nesta área, provavelmente pelo aumento na heterogeneidade de recursos oferecidos a fauna, aumento de espaço de hábitat e qualidade de alimento.

Com base nos resultados deste estudo, e certamente em um forte embasamento da literatura (ver artigo 3), nós pudemos propor dois aspectos a serem levados em conta ao sucesso de práticas de restauração ecológica:

1) almejar a manutenção da heterogeneidade estrutural da paisagem, simulando as condições de ecossistemas naturais, e
2) auxiliar na manutenção da fertilidade do solo, a qual trará positivas conseqüências a todas os outros elementos do sistema.

Com relação a macrofauna encontrada nestes dois projetos de pesquisa pertencentes à dissertação, observou-se a existência de um grande número de indivíduos e espécies nas áreas degradadas estudadas. Duas extensas listas de espécies são apresentadas, e constituem informação importante ao inventário de biodiversidade do Rio Grande do Sul. Tatuzinhos (Isopoda) e aranhas

(Araneae) foram os dois grupos mais representativos em indivíduos nos depósitos de cinzas e na floresta de pinus pós-minerada, respectivamente. Besouros (Coleoptera) foram os mais ricos em espécies nas duas áreas.

Visto todas estas questões na presente dissertação, esperamos estar contribuindo um pouco mais ao entendimento de processos ecossistêmicos e a interação dos organismos do solo em áreas degradadas pelo carvão no sul do Brasil, principalmente no Rio Grande do Sul. A decomposição foliar e a interação fauna – serapilheira servem como ferramentas e podem auxiliar em estratégias e o desenvolvimento prático-teórico da ecologia da restauração.

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Anexos

NORMAS DAS REVISTAS CIENTÍFICAS PARA A PUBLICAÇÃO

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