DÉBORAH NEIDE DE MAGALHÃES

TOXICIDADE NO COTRATAMENTO DE ESGOTO SANITÁRIO E LIXIVIADO DE ATERRO SANITÁRIO

Dissertação apresentada à Universidade Federal de Viçosa, como parte das exigências do Programa de Pós-Graduação em Engenharia Civil, para a obtenção do título de *Magister Scientiae.*

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APROVADA: 15 de fevereiro de 2012.

Mônica de Abreu Azevedo
(Coorientadora)

Jurandyr Povinelli

Ann Honor Mounteer
(Orientadora)

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BIOGRAFIA

DÉBORAH NEIDE DE MAGALHÃES, filha de Gessy Maria Neide de Magalhães e Albino Teixeira Magalhães (*in memoriam*), nasceu e cresceu em Dores de Campos, Minas Gerais, aos 12 de maio de 1983.

Estudou toda a educação básica na sua cidade, frequentando a educação infantil no Jardim de Infância Branca de Neve, e, do pré-escolar até à 4ª série, a Escola Municipal Randolfo Teixeira. O ensino fundamental foi integralmente realizado na Escola Estadual Duque de Caxias e o ensino médio na Escola Municipal João Cupertino da Silva.

De 2002 a 2006 cursou Licenciatura em Ciências Biológicas na Universidade Federal de São João del-Rei. No ano de sua formatura, matriculou-se no curso de pós-graduação *lato sensu* em Análise Ambiental, na Universidade Federal de Juiz de Fora, o qual concluiu em 2008 com a defesa do TCC que dissertava sobre a gestão dos resíduos sólidos urbanos no município de Dores de Campos. Concomitante a estes estudos (2005 a 2009), retornou à Escola Estadual Duque de Caxias, porém como professora de biologia nos ensinos fundamental, médio e educação de jovens e adultos.

Dando continuidade aos estudos voltados à área ambiental, ingressou então, no ano de 2010, no programa de pós-graduação em Engenharia Civil da Universidade Federal de Viçosa, na área de concentração Sanitária e Ambiental, concluindo em fevereiro de 2012, com a defesa desta dissertação.

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RESUMO

MAGALHÃES, Déborah Neide de, M.Sc., Universidade Federal de Viçosa, fevereiro de 2012. **Toxicidade no cotratamento de esgoto sanitário e lixiviado de aterro sanitário.** Orientadora: Ann Honor Mounteer. Coorientadoras: Mônica de Abreu Azevedo e Ana Augusta Passos Rezende.

O lixiviado de aterro sanitário apresenta elevadas cargas orgânicas e de nutrientes potencialmente tóxicas, que podem causar impactos negativos durante o tratamento biológico combinado com esgoto. Nesta pesquisa foi avaliado o efeito da adição de lixiviado ao esgoto sanitário no cotratamento por lodos ativados em uma ETE no município de Juiz de Fora, sudeste do Brasil. Esgoto bruto, lixiviado, mistura esgoto-lixiviado afluente na ETE e efluente secundário foram coletados mensalmente, durante 6 meses, caracterizados físico-quimicamente (matéria orgânica, sólidos, nutrientes e metais) e avaliadas quanto à toxicidade aguda à *Daphnia similis*. Lodo da linha de reciclo foi coletado para a quantificação da taxa específica de consumo de oxigênio (*SOUR*) e de crescimento de biomassa do lodo no esgoto bruto e em misturas esgoto-lixiviado preparadas em bancada. Os resultados evidenciaram a grande variabilidade na composição do lixiviado, o qual apresentou valores mais elevados em todas as variáveis analisadas, exceto o fósforo. Apesar da elevada carga orgânica adicionada de forma intermitente no sistema, o tratamento por lodos ativados parece remover de forma eficiente a matéria orgânica e nutrientes e produz um efluente secundário com baixas concentrações de DQO e N orgânico. O efluente secundário apresentou concentrações de Hg e Pb acima dos limites estabelecidos para lançamento em corpos d'água no estado de Minas Gerais. Ao contrário do esgoto bruto, todas as amostras de lixiviado foram tóxicas à *D. similis*, o que conferiu toxicidade à mistura afluente na ETE em algumas amostragens. Somente uma amostra de efluente secundário apresentou ligeira toxicidade. A toxicidade do lixiviado correlacionou-se positivamente com o pH, nitrogênio e condutividade. Testes de toxicidade em misturas preparadas em bancada evidenciaram o efeito de potencialização da toxicidade do lixiviado. Há uma tendência do aumento da *SOUR* com o aumento da proporção de lixiviado na mistura e os valores de *SOUR* correlacionaram-se negativamente com As, Cd, Hg e Pb. Os resultados sugerem que os lodos ativados podem se adaptar a misturas com maiores concentrações de lixiviado ao longo do tempo, mas o efeito de altas cargas na entrada do sistema deve ser investigado futuramente.

ABSTRACT

MAGALHÃES, Déborah Neide de, M.Sc., Universidade Federal de Viçosa, February, 2012. **Toxicity in co-treatment of sanitary sewage and landfill leachate.** Adviser: Ann Honor Mounteer. Co-Advisers: Mônica de Abreu Azevedo and Ana Augusta Passos Rezende.

Sanitary landfill leachate has high, potentially toxic organic and nutrient loads that may cause negative impacts during combined biological treatment with sewage. In this study, the effect of intermittent leachate addition at the inlet of the activated sludge municipal sewage treatment plant in Juiz de Fora, Minas Gerais, located in southeastern Brazil, was evaluated. Raw sewage, leachate, mixed leachate-sewage and treated wastewater samples were collected monthly over six months for physicochemical characterization (organic matter, solids, nutrients, metals) and determination of acute toxicity to *Daphnia similis*. Return activated sludge was also collected and used to quantify specific oxygen uptake rate (SOUR) and biomass growth in sewage and leachate-sewage mixtures in the laboratory. Leachate presented higher values than sewage for all parameters measured, except phosphorous, though its composition varied widely. Activated sludge treatment efficiently removed organic matter and nutrients, despite the high intermittent loads associated with leachate added at the treatment plant inlet. The only parameters that regularly surpassed Minas Gerais state discharge limits were Hg and Pb. Contrary to the raw sewage that did not exhibit toxicity, all samples of leachate were acutely toxic to *D. similis* and this led to toxicity in four of the mixed leachate-sewage samples, while only one sample of treated wastewater exhibited slight toxicity. Leachate toxicity correlated positively with pH, nitrogen and conductivity. Potentiation of the toxic effect was observed in leachate-sewage mixtures prepared in the laboratory. SOUR values and sludge production increased in mixtures of 20% leachate-80% sewage over their values in sewage alone. SOUR values correlated negatively with As, Cd, Hg and Pb. The results suggest that activated sludge can adapt to the increased organic and nutrient load associated with landfill leachate over time, but the effect of the impulse loads that enter with leachate should be investigated further.

INTRODUÇÃO GERAL

A disposição final de resíduos sólidos urbanos (RSU) em aterros sanitários no Brasil tem crescido nos últimos anos e tende a aumentar mais rapidamente devido ao aumento de consumo em consequência da economia emergente e das pressões governamentais quanto à exigência da adequação do gerenciamento de RSU. Umas das vantagens deste tipo de destinação é a possibilidade de coleta e tratamento do lixiviado, evitando-se assim a contaminação do solo e de águas superficiais e subterrâneas.

O lixiviado de aterro sanitário é resultante das águas que percolam e solubilizam as diversas substâncias químicas da massa de resíduos num aterro sanitário, provindas da infiltração pluvial, da umidade intrínseca dos resíduos, e produzida nos processos bioquímicos de degradação da matéria orgânica (GOMES, 2009). Dessa forma, o lixiviado caracteriza-se geralmente por apresentar matéria orgânica dissolvida ou solubilizada e em suspensão, produtos intermediários da digestão anaeróbia dos resíduos, como ácidos orgânicos voláteis, e substâncias químicas, como metais pesados e organoclorados originados do descarte de agrotóxicos e inseticidas (CASTILHOS JUNIOR, 2006).

A composição do lixiviado apresenta grande variabilidade, pois depende dos constituintes dos resíduos e da idade do aterro, que pode refletir o grau de estabilização em que se encontram os resíduos. Segundo Mc Bean, et al. (2005), o lixiviado novo contém maior teor de matéria orgânica biodegradável e tende a ser ácido (pH entre 5 e 7) devido à presença de ácidos graxos voláteis. Neste estágio predominam as formas nitrogenadas em estado de oxidação mais reduzido, como nitrogênio orgânico e amoniacal. Em lixiviados mais antigos, a carga orgânica diminui, permanecendo a porção recalcitrante, o pH varia em torno da neutralidade, de 7 a 8, e há maiores concentrações de formas oxidadas de nitrogênio.

Muitas pesquisas relacionadas à tratabilidade de lixiviados têm sido desenvolvidas, visando à melhoria na eficiência do seu tratamento. Os processos de tratamento empregados devem combinar, em geral, tecnologias físicas, químicas e biológicas e serem projetados de forma a se adequarem às mudanças na composição e na vazão do lixiviado. Os processos biológicos são mais efetivos para o tratamento do lixiviado jovem, que contém maiores concentrações de ácidos voláteis. Os lixiviados estabilizados, por possuírem em sua constituição concentrações consideráveis de compostos refratários, em especial as substâncias húmicas, nitrogênio amoniacal e baixa concentração de fósforo, dificultam o tratamento biológico e requerem a combinação de processos físicos e químicos de tratamento (TATSI et al., 2003).

Uma prática que vem sendo largamente adotada na Europa, Estados Unidos, Japão e em outros países, incluindo o Brasil, é o tratamento combinado de lixiviado de aterro e esgoto sanitário em estações de tratamento de esgotos (ETE), conferindo uma alternativa vantajosa devido à viabilidade econômica e facilidade técnico-operacional. Nesta técnica, a potencialidade de tratamento do lixiviado é aumentada em decorrência da sua diluição no esgoto. Porém, um dos questionamentos a cerca dessa prática é a respeito das possíveis interferências do lixiviado nos processos de tratamento, especialmente os biológicos (GOMES, 2009).

Neste sentido, várias pesquisas vêm sendo desenvolvidas buscando a definição de critérios que viabilizem o cotratamento desses dois efluentes, principalmente relacionadas à proporção de diluição e tratabilidade pelas diversas tecnologias disponíveis. Del Borghi et al. (2003), assim como outros autores (DIAMADOPAULOS et al., 1997), demonstraram que uma mistura de 10% de lixiviado no esgoto, apresentou eficiência de remoção de DQO e $N\text{-}NH_3$ acima de 70%, quando tratada aerobiamente e usando-se lodo aclimatado por 4h. Mannarino et al. (2010), em concordância com McBean et al. (1995), concluíram que o influxo de até 2% de lixiviado misturado ao esgoto afluente na ETE Icaraí, Rio de Janeiro, não interferiu no processo de tratamento quimicamente assistido. Neste trabalho também foi avaliada a toxicidade dos afluentes, constatando-se que a toxicidade do lixiviado não conferia toxicidade à mistura afluente na ETE e ao efluente final.

No Brasil, o emprego do cotratamento de esgoto e lixiviado em ETEs é uma tendência, uma vez que a Resolução CONAMA 430/11, que dispõe sobre os padrões de lançamento de efluentes, não mais exige o atendimento ao limite de nitrogênio amoniacal (20 mg N/L) no caso de efluentes do tratamento de esgoto sanitário. O cumprimento deste parâmetro é tratado de igual forma no estado de Minas Gerais, regulamentado pela Deliberação Normativa conjunta COPAM/CERH-MG nº. 01/2008.

Embora existam muitos trabalhos que investigam o processo de cotratamento de esgoto e lixiviado, estes têm focado no estudo de frações de diluição, tratabilidade por processos físicos e químicos e sua integração com processos biológicos. Porém, o estudo do possível efeito tóxico da adição de lixiviado ao esgoto durante o tratamento biológico é um aspecto que precisa ser investigado, uma vez que poucos trabalhos a respeito são encontrados na literatura.

O trabalho apresentado a seguir consta de um artigo científico a ser submetido a periódico internacional, cuja proposta foi avaliar a toxicidade de lixiviado de aterro sanitário e seu efeito no tratamento consorciado com esgoto sanitário em um sistema de lodos ativados, por meio de bioensaios de toxicidade aguda e da avaliação da atividade microbiana, além da avaliação da toxicidade do efluente tratado e da possibilidade de impacto deste sobre o corpo receptor.

TOXICITY IN CO-TRETATMENT OF SANITARY SEWAGE AND LANDFILL LEACHATE

1 Introduction

Final disposition of municipal solid wastes (MSW) in sanitary landfills has grown steadily in Brazil, with an increase from 17 to 28% of cities adopting this form of disposal between 2000 and 2008 (IBGE, 2000, 2008). According to the Brazilian Association of Public Sanitation Companies (ABRELPE), 57% of the 173,000 tons of MSW collected daily in 2010 were disposed of in sanitary landfills. Startup of new sanitary landfills is expected to continue increasing, at least in the short term, as a consequence of the emerging Brazilian economy that has lead to increased consumption and generation of solid wastes as well as recent federal policies that requires all cities to adopt adequate MSW management measures. Between 2009 and 2010, MSW generation increased 6.8% while the Brazilian population increased by only 1% in the same period (ABRELPE, 2010).

An important aspect of landfill operation is the adequate collection and treatment of leachate in order to eliminate the potential risk of soil, surface water and groundwater contamination. Leachate treatment represents a technical challenge since leachate is a complex mixture whose composition depends on a variety of factors, including the nature of the solid wastes, the age and operational management of the landfill and season of the year (ABBAS et al., 2009).

The few leachate biological treatment systems implanted in Brazil have presented mediocre performance, because of the variability in composition, seasonal flowrate variations and the fact that many were projected based on criteria defined for sanitary sewage that presents physicochemical characteristics quite different from those of leachates (CASTILHOS JUNIOR, 2006).

The high levels of organic matter, conductivity and nutrients typical of leachates can pose challenges to biological treatment. The high load of macro and micro nutrients in leachates can cause imbalances leading to perturbation of the biochemical processes occurring during biological treatment. Some organic compounds in the leachates, such as humic and fulvic acids, are relatively recalcitrant to biological treatment and can confer toxicity to wastewaters (KALKA et al., 2010, DEL

BORGHI et al., 2003). Combined treatment of landfill leachate with sanitary sewage in municipal sewage treatment plants (STP) has been adopted in many countries, including Brazil, since it allows for leachate dilution and avoids the technical and operational difficulties posed by the large seasonal variations in leachate flowrate and composition (DIAMADOUPOULUS et al., 1997; GOMES et al., 2009). Although improved degradation of the high organic load in leachates can be achieved by dilution with sanitary sewage and by adaptation of biological sludge (DEL BORGHI et al., 2003), the impacts on STP operation arising from mixing these two types of effluents needs to better evaluated (GOMES et al., 2009).

Results of combining complex mixtures such as leachate and wastewater cannot be anticipated, since interactive effects between toxic substances are often observed that can increase or decrease toxicity. Many toxic substances in these types of wastewater may be present at levels at or below analytical detection limits making their identification technically or economically unfeasible (MATEJCZYK et al., 2011). Therefore, toxicity testing of leachate, sewage and their mixture, allied with appropriate chemical analyses can provide a more direct measure of possible toxic effects and their associated causes.

The objective of the present study was to evaluate the toxicity of landfill leachate and its effect on treatment of domestic sewage in an activated sludge treatment plant, and to identify possible correlations between toxicity and leachate chemical components, such as organic matter, nitrogen compounds and trace metals. Possible toxic effects were evaluated by means of acute toxicity assays and activated sludge respirometry and biomass growth tests.

2 Material and methods

2.1 Sampling site

Samples were collected at the Barbosa Lage STP located in the city of Juiz de Fora (population 512,000), Minas Gerais, southeastern Brazil. The STP receives on average 7000 m^3/d of raw sewage and treatment consists of screening, grit removal, equalization (2h hydraulic retention time, HRT) and biological treatment in an extended aeration activated sludge system (15 h HRT). Treated wastewater is discharged to the Paraibuna River and excess biological sludge is centrifuged and disposed of in the municipal sanitary landfill. The leachate generated at the Salvaterra landfill is collected and transported for treatment at the Barbosa Lage STP, The landfill occupies an area that was operated as the Juiz de Fora city dump from 1999 to 2005, when it was converted to a landfill and operated until reaching its holding capacity in April of 2010. An average of 45 m^3/day of leachate is transported from Salvaterra to the STP in 15 m^3 capacity trucks, generally two in the morning (8 and 11 am) and one in the afternoon (4 pm). The leachate is discharged directly at the inlet before the screens.

2.2 Samples

Samples were collected monthly from January through June of 2011, a period that covered parts of both the dry (April through October) and rainy (November through March) seasons in the region. The samples included raw sewage and landfill leachate, collected before their mixture, the leachate-sewage mixture, before equalization and treated wastewater collected at the STP outlet to the Paraibuna River. The samples were chilled and transported to the Water Quality Control Laboratory at the Federal University of Viçosa, within 3 hours of collection. One part of each sample was acidified to $pH \leq 2$ with nitric acid and stored refrigerated for metals analyses and another part was filtered qualitatively and stored frozen for toxicity bioassays. Return activated sludge (RAS) was also collected from the recycle line in some months. The sludge was kept under constant aeration and at constant temperature in the laboratory and held for no more than one week before use.

2.3 Physicochemical characterization

Raw sewage, leachate, leachate-sewage mixture and treated wastewater samples were characterized using standard methods (APHA, 2005) by quantifying the parameters shown in Table 1. Total (TSS) and volatile (VSS) suspended solids contents were determined in the sludge samples by the standard gravimetric method on the days they was used in respirometric studies. All chemical analyses were performed in duplicate at the Federal University of Viçosa Water Quality Control and Instrumentation and Chemometrics Laboratories.

Table 1. Parameters and analytical methods used in sewage and leachate characterization

Parameter	Method (equipment)
pH	4500-H^+ B (Digimed DM-20 pH meter)
Electrical conductivity (EC)	2510 B (Conductivity cell Digimed DM-3)
5-day biochemical oxygen demand (BOD_5)	5210 B
Chemical oxygen demand (COD)	5220 D
Total Kjeldahl (TKN) and ammonia-nitrogen (NH_3-N)	4500-N_{org} B and 4500-NH_3 B
Oxidized nitrogen (nitrate and nitrite, N_{ox}^-)	4500-NO_3^- E (NitraVer 5® commercial kit)
Total phosphorous (total-P)	4500-P D
Dissolved (TDS) and suspended (TSS) solids	2540 B and 2540 D
Trace metals (Cd, Cr, Cu, Hg, Pb) and AS	3111 B (Varian SpectrAA-200) and 3112 B (coupled hydride generator Varian VGA-17A)

2.4 *Acute toxicity*

Acute toxicity bioassays were performed using a static, 48 hour test with the water flea *Daphnia similis*, according to the Brazilian standard (NBR 12.713, ABNT, 2009). In each assay, five sample dilutions were prepared in four replicates. Each replicate contained five 6 to 24 hour-old test organisms. Mixtures containing different proportions of leachate and sewage were also prepared in order to determine the proportion of leachate that resulted in acute toxicity. Two types of negative controls were included, one containing only dilution water and the other only raw sewage. The test solutions were kept for 48 hours at 22°C under a 12 h light/dark cycle, after which the number of immobile organisms were counted. The endpoint calculated was the EC_{50}, the percent of wastewater, leachate or their mixture that resulted in immobility of 50% of the test organisms.

2.5 *Oxygen uptake rate*

Specific oxygen uptake rates (SOUR) of activated sludge in sewage (control) and leachate-sewage mixtures were determined using the standard method (2710 B, APHA et al., 2005). Mixtures were prepared with 5, 10 and 20% (v/v) leachate in sewage. Test solutions were maintained at 20°C and aerated for fifteen minutes using porous stones attached to an air pump. Aeration was turned off, 10 mL RAS were added to the mixtures and dissolved oxygen concentrations measured in 15 second intervals for 15 minutes using an oxygen meter (Digmed, model DM-4). The oxygen uptake rate (OUR mg O_2/L.h) was calculated as the inclination of the curve of oxygen concentration versus time. The SOUR (mgO_2/h.gVSS) was obtained by dividing the OUR by the concentration of VSS in the test mixture.

2.6 Sludge growth

The effect of leachate on activated sludge growth was evaluated using samples collected in May, 2011. Bench-scale sequencing batch reactors with one liter working volumes were filled with filtered raw sewage or filtered leachate-sewage mixture (20/80 v/v) in duplicate. The reactors were inoculated with RAS at an initial concentration of 100 mg VSS/L and kept at constant temperature (20°C ± 2) under aeration by porous stones attached to an air pump. The reactors were operated on a 24 hour cycle for 13 days. Each cycle consisted of 23 hours aeration and one hour sludge settling, followed by withdrawal of 700 mL clarified wastewater, and replenishment with 700 mL aliquots of filtered sewage or the leachate-sewage mixture for the start of a new cycle. COD was measured in the clarified wastewater and TSS and VSS determined in the settled sludge at the end of each cycle.

2.7 Statistical analyses

Results of physicochemical characterizations were analyzed using Statistica 8.0 (Statsoft, Tulsa, OK). Normality was checked using the Kolmogorov-Smirnov & Lilliefors test. Student´s t-test was used for comparison of means. Pearson´s correlation coefficient (r) was calculated to evaluate relationships between biological tests and physicochemical components of the wastewater samples. All statistical analyses were run at a 5% level of significance.

Toxicity assay results were analyzed and EC_{50} values estimated using Probit 1.5 or Trimmed Spearman-Karber (TSK) 1.5 programs, available for download from the US Environmental Protection Agency.

3 Results and discussion

3.1 Sample characterization

Characterization of the santitary sewage and landfill leachate entering the STP and treated wastewater are presented in Table 2. Leachate samples presented much higher average values and broader ranges in values than raw sewage, except for phosphorous. The largest differences between sewage and leachate were found for organic matter (BOD and COD), solids, nitrogen and conductivity. Leachate conductivity was on average two orders of magnitude (10^4 µS/cm) higher than sewage conductivity (10^2 µS/cm). This is a common characteristic of landfill leachates inherent to the chemical hydrolysis and partial biological degradation and hence solubilization of solid wastes that leads to its formation.

Table 2. Characteristics and variability of raw domestic sewage, landfill leachate and treated wastewater, together with state discharge limits

Parameter	**Raw sewage**		**Landfill leachate**		**Treated wastewater**		**Discharge limit**
	Average	**Range**	**Average**	**Range**	**Average**	**Range**	
pH	7.1	7.0-7.2	7.7	7.4-8.4	7.3	7.0-7.8	6.0-9.0
EC, µS/cm	576	401-629	14,913	7,290-24,940	480	300-741	none
COD, mg O_2/L	399	210-568	16,028	2,385-23,600	92	34-206	180
BOD, mg O_2/L	141	57-212	7709	1,218-12,669	31	7-60	60
Organic N, mg/L	13	12-15	251	29-624	5	ND-14	none
NH_3-N, mg N/L	37	31-44	1060	434-1,896	17	3-39	none
N_{ox}^{-}-N, mg/L	32	5-52	178	93-271	7	1-17	none
Total-P, mg/L	7.0	4.9-8.9	4.5	0.7-8.6	3.1	0.9-8.4	none
TSS, mg/L	247	192-408	11,602	490-65,125	79	20-140	100
TDS, mg/L	249	148-352	11,804	4,120-18,051	402	202-992	none
As, mg/L	0.008	0.001-0.021	0.011	ND-0.014	0.0001	ND-0.0005	0.2
Cd, mg/L	0.041	ND-0.091	0.125	ND-0.396	0.050	ND-0.118	0.1
Cr, mg/L	0.014	ND-0.048	0.542	ND-1.307	0.023	ND-0.083	1.0
Cu, mg/L	0.001	ND-0.004	0.423	ND-2.540	ND	ND-ND	1.0
Hg, mg/L	0.003	ND-0.007	0.013	ND-0.030	0.037	0.036-0.039	0.01
Pb, mg/L	0.391	ND	0.641	ND	0.453	ND-1.072	0.1

ND = not detected.

High conductivity can be a problem in biological treatment by the activated sludge process, since high cation concentrations can interfere in biological floc aggregation, and therefore lead to poor sludge settling in the clarifier (KARA *et al.*, 2008). This can have disastrous results, both in terms of final effluent quality and stability of the system, with regard to sludge recycling and wasting. Potentially toxic metals were also present in higher concentrations in leachate, except for As and Pb, which had similar concentrations in sewage.

Biodegradability, measured as the BOD/COD ratio, averaged 0.48 in leachate, but only 0.34 in sewage, suggesting that the high organic load in leachate was not as recalcitrant to biological treatment as has been reported in other studies (YU et al., 2010; MANNARINO et al., 2010). The considerable leachate biodegradability indicates it is a relatively "young leachate". Słomczyńska and Słomczyński (2004) reported that leachates less than five years old exhibit BOD/COD ratios from 0.22 to 0.7, whereas older leachates exhibit much lower biodegradabilities (YU et al., 2010).

Nitrogen, in all its forms (organic, ammonium and oxidized) was several fold higher in leachate than in sewage. Toxicity of leachates has been attributed to high ammonium levels (DIAMADOPOULOS et al., 1997) and removal of nitrogen is considered an important objective of leachate treatment (MANNARINO et al., 2010).

Combined treatment of leachate and sewage was proposed as a means to dilute the high organic and nitrogen loads and salinity in leachates (DEL BORGHI et al., 2003). Furthermore, Gomes et al. (2009) and Abbas et al. (2009) have reported that combined leachate-sewage treatment can be advantageous since biological treatment of landfill leachate alone is often limited by phosphorous. However, if not managed correctly, impulse loads of leachate may lead to instability in biological treatment (YU et al., 2010). The impact of leachate addition on the STP loading rate is illustrated in Figure 1, in which organic, solids and Kjeldahl nitrogen loads (concentration x flowrate, kg/h) at the entrance of the STP before and after leachate addition are compared. It is clear that without adequate equalization the activated sludge system would be susceptible to an organic shock that could exhaust oxygen at the entrance to the aeration tank. The high dissolved solids load could also lead to a toxic shock, with a negative effect on sludge activity and settleability.

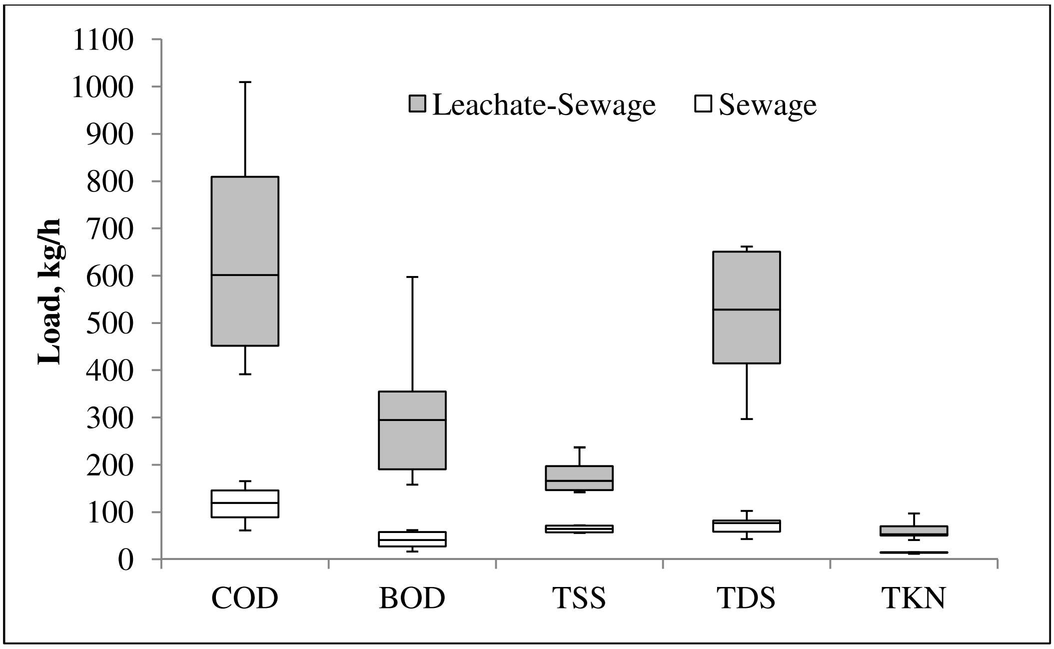

Figure 1. Boxplot representation of impact of leachate addition on organic (COD, BOD), solids (TSS, TDS) and Kjeldahl nitrogen (TKN) loads at the entrance of the STP (lines = median, bars = 25%-75% range, error bars = minimum and maximum values).

The activated sludge system was generally able to absorb and treat the leachate-sewage mixture to the level required by Minas Gerais state regulations (COPAM/CERH, 2008) (Table 2). Organic matter removal was sufficient, with only one treated wastewater sample exceeding the COD discharge limit of 180 mg/L. However, in the case of municipal sewage treatment plants, state legislation permits discharge above this level, if average COD removal is over 55%, which is clearly the case at this plant. TSS exceeded the discharge limit of 100 mg/L twice, with values of 126 mg/L in February and 140 mg /L in June, which was likely due to solids loss in the secondary clarifier caused by poor floc formation in the aeration tank, possibly as a result of high influent conductivity.

High leachate conductivity can interfere in COD analyses, especially when associated with high chloride concentrations (above 2000 mg $C\ell^{-}$/L). Chloride analyses of leachate samples showed that only one sample presented $C\ell^{-}$ above the recommended limit, reaching 2880 mg $C\ell^{-}$/L. However, because of the high concentrations of organic matter, samples were diluted several fold for COD determination, resulting in chloride levels that were eliminated by the standard procedure (addition of $HgSO_4$).

Hg was the only parameter that was consistently detected at levels above the discharge limit at the STP outlet. Pb and Cd were also detected above the legal limit in some treated wastewater samples. Both sewage and leachate contributed to the high Hg and Pb concentrations, whereas leachate was responsible for input of most of the Cd.

3.2 Sample toxicity

One of the challenges to combined leachate-sewage treatment is leachate toxicity that may hinder biological treatment and lead to discharge of toxic effluents. Raw sewage did not present acute toxicity to *D. similis*, whereas all leachate samples were toxic, with EC_{50} ranging from 4 to 18%. Significant ($p < 0.05$) positive correlations were found between leachate acute toxicity and pH, conductivity, TKN, ammonium and oxidized nitrogen (Table 3) which is similar to results reported by various authors (ISIDORE et al., 2003, SILVA et al., 2004, KALKA et al., 2010). Cotman and Gotvajn (2010) observed that ammonia stripping of leachates reduced their toxicity to activated sludge microorganisms. The significant correlation with pH may also be related to the presence of ammonia whose toxic unionized form is present at higher pH (pKa = 9.3).

Table 3. Values of significant correlations (r, $p < 0.05$) between leachate toxicity and physicochemical variables

Variable	**Correlation coefficient**
pH	0.84
Conductivity	0.90
TKN	0.90
NH_3-N	0.92
N_{OX}-N	0.89

Isidore et al. (2003) reported that leachate toxicity was reduced after chelation with EDTA and therefore attributed the toxicity to divalent metal cations. Although no significant correlations were found between leachate toxicity and metals contents, Cd, Cr and Hg were present in leachate at concentrations shown to be acutely toxic to *D. similis* (RODGHER et al., 2010). Furthermore, mixtures of metals can exhibit a synergistic effect, with toxicity detectable at lower metals concentrations in the mixture than in the presence of the individual metals.

Leachate-sewage mixtures collected at the STP presented a relatively broad range of EC_{50} values (38% to >100%, in the samples of January and May), reflecting the large variability in leachate composition. Treated wastewater was only slightly toxic in March (EC_{50} = 92%). This toxicity correlated directly with the presence of As (r = 1.0). However, the presence of arsenic did not correlate with toxicity of the other sample types.

Leachate-sewage mixtures prepared in the laboratory consistently presented greater acute toxicity (lower EC_{50} values) than the leachate itself (Figure 2), indicating a potentiation effect between leachate and sewage, since raw sewage never showed toxicity and leachate toxicity increased (lower EC_{50} values) when added to sewage. Kalka et al. (2010) observed the same trend in acute toxicity bioassays with *Daphnia magna, Vibrio fischeri* and *Thamnocephalus platyurus.* However, Mannarino et al. (2010) reported no increase in toxicity of domestic sewage or leachate to *D. similis* or the fish *Danio rerio* after mixing.

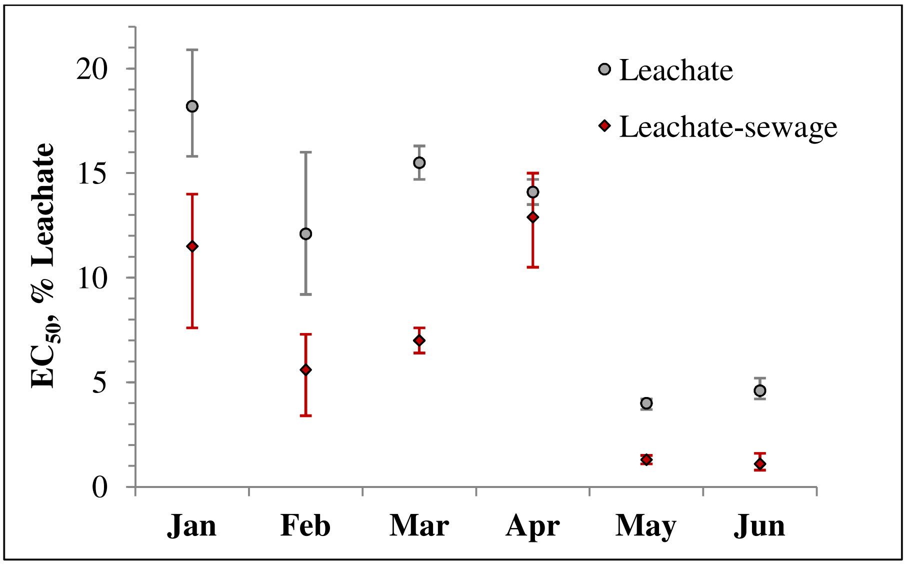

Figure 2. Comparison between leachate and leachate-sewage mixture acute toxicities to *D. similis* (error bars represent 95% confidence intervals around EC_{50} values).

Leachate can represent up to almost 5% of the STP inlet flow (292 m^3/h sewage and 15 m^3/h leachate) at the time of addition, and even after the two hour equalization tank, still corresponds to 2.5% of the inlet flow to the activated sludge aeration tank, which is above the EC_{50} values of the leachate-sewage mixtures prepared in the laboratory in May and June. Some authors have suggested that the final leachate concentration in co-treatment should not exceed from 2% (McBEAN, et al., 1995; MANNARINO et al., 2010). On the other hand, Çeçen and Çakıroğlu (2001) stated that the proportion of leachate should not exceed 20% of the flowrate, or 50% of the initial COD in an activated sludge system. The potentiation effect observed between leachate and sewage could compromise microbial metabolism and thus biological treatment itself. The intermittent addition of leachate to the STP also might not permit adequate adaptation of microorganisms in the biological reactor.

3.3 SOUR

In order to evaluate possible toxic effects of leachate on activated sludge microorganisms, sludge oxygen uptake was measured in the presence of sewage and leachate-sewage mixtures (Table 4). Although no significant differences were found in respiration rates, SOUR tended to increase with the increase in the proportion of leachate in the mixture. The higher organic content in the sewage-leachate mixtures apparently lead to greater microbial activity, which as previously mentioned, could exhaust the oxygen supply in aerobic treatment.

Table 4. Specific oxygen uptake rate (SOUR) in raw sewage (S) and leachate-sewage mixtures ($L_{\%}S$)

	SOUR (mgO_2/g VSS.h)			
Sample	**S**	**L_5S**	**$L_{10}S$**	**$L_{20}S$**
Jan	1.93	2.52	2.58	2.91
Feb	1.97	2.39	3.42	4.05
Mar	1.35	0.93	1.18	4.24
Abr	1.42	2.03	1.72	1.55
May	1.92	2.12	2.16	1.92
Jun	2.26	2.56	2.30	2.41

Sludge SOUR values in the mixture containing 20% leachate presented significant negative correlations (Table 5) with As, Cd, Hg and Pb, indicating a possible toxic effect of these metals on microbial biomass leading to reduced respiration rate in sludge.

Table 5. Significant correlations (r, $p < 0.05$) between 20% leachate-sewage mixture ($L_{20}S$) SOUR values and sewage and leachate physicochemical variables.

	SOUR $L_{20}S$	
Variable	*Sewage*	*Leachate*
As	-0.91	ns
Cd	-0.82	-0.85
Hg	ns	0.90
Pb	-0.84	-0.85

*ns = no significant correlation.

3.4 Biomass Growth

To evaluate the possible impact of leachate on sludge biomass growth, sequencing batch reactors fed with sewage and leachate-sewage mixtures were inoculated with activated sludge. VSS production and soluble COD were monitored after each of 13 cycles (Figure 3). Very little solids production was observed in the reactors treating only sewage, whereas in the reactors treating the 20% leachate-sewage mixture, solids reached approximately 500 mg VSS/L after 13 cycles.

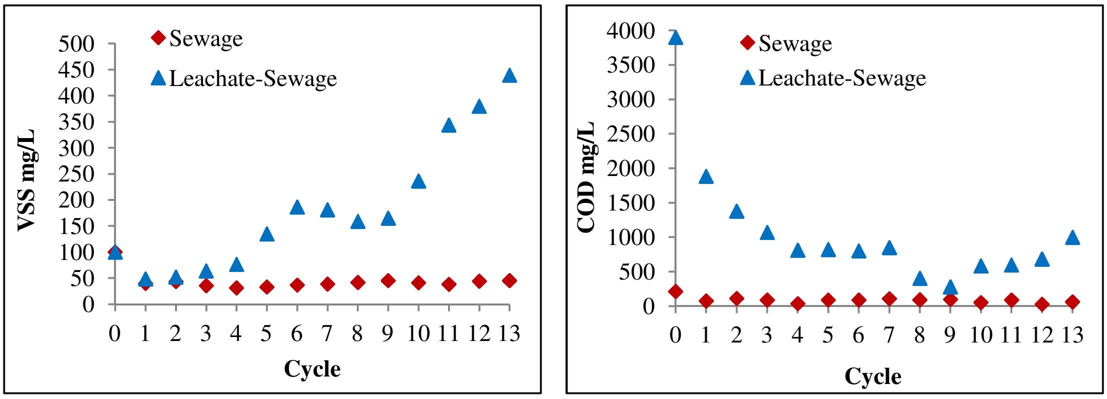

Figure 3. Variation in VSS production and COD removal in sequencing batch reactors treating sewage or a 20% leachate-sewage mixture.

Soluble COD after each treatment cycle remained fairly constant in the reactors fed sewage, while it decreased rapidly within the first 4 cycles in the reactors fed the leachate-sewage mixture. COD removal efficiency averaged 65% for sewage and over 96% for the leachate-sewage mixture. The pause in increased COD removal observed from cycles 5 to 7 in the mixture was reflected in the slightly delayed pause in increased VSS production, observed from cycles 6 to 9, that may reflect adaptation of the activated sludge biomass to the mixture. Initial COD in sewage was only 210 mg/L while in the 20% leachate-sewage mixture it was 3900 mg/L. Clearly, the higher organic and nitrogen contents of the leachate-sewage mixture afforded greater biomass growth than the sewage. These results suggest that activated sludge is able to adapt to the higher more concentrated wastewater over time, however, the effect of impulse loads such as those occurring in the STP still need to be evaluated.

4 Conclusions

This study evaluated the toxicity of sanitary landfill leachate and its effect on combined treatment with domestic wastewater in the activated sludge process. Leachate contained significantly higher levels of all physicochemical variables evaluated than sewage, except phosphorous. Addition of leachate to sewage increased organic (BOD and COD), nitrogen (Kjeldahl and oxidized forms) and solids loads, but these were consistently reduced during activated sludge biological treatment. Treated wastewater presented Pb and Hg concentrations above state discharge limits.

All leachate samples were acutely toxic to *D. similis*, and conferred toxicity to the wastewater after mixture in the STP, however treated wastewater was only slightly toxic on one sampling date. Leachate toxicity correlated positively with nitrogen compounds, principally, ammonia, as well as with electrical conductivity and pH. Potentiation of the toxic effect of leachate occurred after mixture with sewage in the laboratory.

The sludge oxygen uptake rate tended to increase with the increase in the proportion of leachate in leachate-sewage mixtures. Negative correlation was found between oxygen uptake and Pb, Cd, As and Hg. Addition of leachate to sewage resulted in increased biomass growth rate in laboratory scale reactors.

The effects of intermittent addition of leachate on the microorganisms responsible for biological treatment of leachate and sewage should be investigated in future studies to establish appropriate operational criteria for combined treatment of these wastewaters.

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ANEXOS

DADOS EXPERIMENTAIS DAS ANÁLISES FÍSICO-QUÍMICAS, DOS ENSAIOS ECOTOXICOLÓGICOS, RESPIROMÉTRICOS, DE CRESCIMENTO DE BIOMASSA E ESTATÍSTICOS.

ANEXO A – RESULTADOS DAS ANÁLISES FÍSICO-QUÍMICAS

Tabela A1. Médias obtidas nos ensaios de caracterização físico-química das amostras de esgoto bruto afluente na ETE Barbosa Lage, Juiz de Fora – MG

		Amostragens					
Parâmetro	Unid.	*Janeiro*	*Fevereiro*	*Março*	*Abril*	*Maio*	*Junho*
pH	-	7,2	7,1	7,0	7,1	7,2	7,1
cond.	µS/cm^2	401	627	607	562	628	629
DQO	mg/L	273	568	528	416	210	400
DBO	mg/L	87	167	212	208	57	112
NTK	mg/L	43	47	50	50	59	53
$N\text{-}NH_3$	mg/L	31	35	35	38	44	39
N_{ox}^- N	mg/L	17	44	43	52	5	32
P-total	mg/L	6	8	9	6	5	8
ST	mg/L	476	680	599	423	348	450
SST	mg/L	192	408	247	242	200	194
SF	mg/L	274	356	323	280	245	262
SV	mg/L	202	324	276	143	103	188
Pb	mg/L	nd	nd	nd	0,685	0,796	0,867
Cd	mg/L	nd	nd	nd	0,063	0,091	0,089
Cu	mg/L	nd	nd	nd	0,004	nd	nd
Cr	mg/L	nd	nd	nd	0,018	0,016	0,048
As	mg/L	0,001	0,001	0,001	0,021	0,014	0,013
Hg	mg/L	0,002	nd	0,004	0,003	0,007	0,002

Tabela A2. Médias obtidas nos ensaios de caracterização físico-química das amostras de efluente secundário da ETE Barbosa Lage, Juiz de Fora – MG

		Amostragens					
Parâmetro	Unid.	*Janeiro*	*Fevereiro*	*Março*	*Abril*	*Maio*	*Junho*
pH	-	7,8	7,5	7,0	7,0	7,2	7,4
cond.	$\mu S/cm^2$	391	573	300	339	535	741
DQO	mg/L	53	92	62	34	103	206
DBO	mg/L	16	37	23	7	60	45
NTK	mg/L	18	30	6	5	24	52
$N\text{-}NH_3$	mg/L	18	29	3	3	13	39
N_{ox}^{-} N	mg/L	1	17	2	3	1	16
P-total	mg/L	1	5	2	2	2	8
ST	mg/L	1023	356	271	416	363	455
SST	mg/L	31	126	69	20	88	140
SF	mg/L	139	52	61	64	132	194
SV	mg/L	884	304	210	352	231	261
Pb	mg/L	nd	nd	nd	0,82	0,83	1,07
Cd	mg/L	nd	nd	nd	0,09	0,10	0,12
Cu	mg/L	nd	nd	nd	nd	nd	nd
Cr	mg/L	nd	nd	nd	0,005	0,08	0,05
As	mg/L	nd	nd	0,0005	nd	nd	nd
Hg	mg/L	0,037	0,039	0,038	0,036	0,039	0,036

Tabela A3. Médias obtidas nos ensaios de caracterização físico-química das amostras de mistura esgoto-lixiviado afluente nas ETE Barbosa Lage, Juiz de Fora – MG

		Mistura esgoto-lixiviado					
		Amostragem					
Parâmetro	Unid.	*Janeiro*	*Fevereiro*	*Março*	*Abril*	*Maio*	*Junho*
pH	-	7,4	7,6	7,2	7,4	7,6	8,1
cond.	$\mu S/cm^2$	1343	2470	2110	2950	1551	5530
DQO	mg/L	1457	841	2474	2957	909	1660
DBO	mg/L	917	344	1050	1848	371	704
NTK	mg/L	89	134	127	204	118	282
$N\text{-}NH_3$	mg/L	78	116	106	176	100	233
N_{ox}^{-} N	mg/L	25	50	51	115	15	55
P-total	mg/L	6	4	2	6	5	6
ST	mg/L	1680	1460	2196	2440	978	2272
SST	mg/L	567	245	358	455	242	288
SF	mg/L	1080	626	1152	1252	552	1104
SV	mg/L	600	834	1044	1188	426	1168
Pb	mg/L	nd	nd	nd	1,11	1,07	1,14
Cd	mg/L	nd	nd	nd	0,11	0,13	0,14
Cu	mg/L	nd	nd	nd	nd	nd	nd
Cr	mg/L	nd	nd	nd	0,28	0,23	0,24
As	mg/L	nd	nd	nd	0,013	0,012	0,012
Hg	mg/L	0,012	0,029	0,002	0,001	0,001	0,0003

Tabela A4. Médias obtidas nos ensaios de caracterização físico-química das amostras de lixiviado bruto do Aterro Sanitário Salvaterra, Juiz de Fora – MG

		Amostragens					
Parâmetro	Unid.	*Janeiro*	*Fevereiro*	*Março*	*Abril*	*Maio*	*Junho*
pH	-	7,4	7,8	7,4	7,5	8,0	8,4
cond.	µS/cm^2	7290	7490	11410	14250	24940	24100
DQO	mg/L	23600	2385	16396	18830	19305	15653
DBO	mg/L	12669	1218	10707	10879	6414	4369
NTK	mg/L	1173	463	750	1246	2100	2136
$N\text{-}NH_3$	mg/L	549	434	680	980	1823	1896
$N_{ox}^{-}\ N$	mg/L	174	93	169	115	271	245
P-total	mg/L	1	5	2	5	6	9
ST	mg/L	71750	4610	13260	13780	18716	18320
SST	mg/L	65125	490	965	1560	665	805
SF	mg/L	24800	1790	6500	6455	8474	7684
SV	mg/L	46950	2820	6760	7325	10242	10636
Pb	mg/L	0,03	nd	nd	1,23	1,16	1,42
Cd	mg/L	nd	nd	nd	0,40	0,16	0,19
Cu	mg/L	2,54	nd	nd	nd	nd	nd
Cr	mg/L	1,31	nd	nd	0,59	0,49	0,87
As	mg/L	0,012	0,013	0,014	0,0005	0,012	0,012
Hg	mg/L	0,025	0,024	0,030	nd	nd	nd

ANEXO B – RESULTADOS DOS BIOENSAIOS DE TOXICIDADE AGUDA

Tabela B1: Resultados dos ensaios de toxicidade aguda para o esgoto bruto sobre *D. similis,* 48h

Concentração, %	**Nº de organismos/organismos afetados**					
	Janeiro	*Fevereiro*	*Março*	*Abril*	*Maio*	*Junho*
0	10/0	10/0	20/0	20/0	20/0	20/1
6,25	10/1	10/0	20/0	20/0	20/0	20/0
12,5	10/0	10/0	20/0	20/0	20/0	20/0
25	10/0	10/0	20/1	20/0	20/0	20/0
50	10/0	10/0	20/3	20/0	20/0	20/0
100	10/0	10/0	20/1	20/0	20/0	20/0

Tabela B2: Resultados dos ensaios de toxicidade aguda para o efluente secundário sobre *D. similis,* 48h, após cotratamento de esgoto e lixiviado.

Concentração, %	**Nº de organismos/organismos afetados**					
	Janeiro	*Fevereiro*	*Março*	*Abril*	*Maio*	*Junho*
0	10/1	10/0	20/0	20/0	20/0	20/1
6,25	10/1	10/0	20/0	20/0	20/1	20/4
12,5	10/0	10/1	20/1	20/0	20/0	20/2
25	10/0	10/1	20/1	20/0	20/0	20/0
50	10/0	10/0	20/0	20/0	20/0	20/1
80	-	-	20/2	-	-	-
85	-	-	20/3	-	-	-
90	-	-	20/8	-	-	-
95	-	-	20/12	-	-	-
100	10/0	10/0	20/19	20/0	20/0	20/0

Tabela B3: Resultados dos ensaios de toxicidade aguda para a mistura esgoto-lixiviado afluente na ETE, sobre *D. similis,* 48h

Concentração, %	**Nº de organismos/organismos afetados**					
	Janeiro	*Fevereiro*	*Março*	*Abril*	*Maio*	*Junho*
0	10/0	10/0	20/0	20/0	20/0	20/2
6,25	10/1	10/0	20/0	20/0	20/0	20/0
12,5	10/0	10/0	20/0	20/0	20/0	20/0
25	-	10/0	20/0	20/0	20/0	20/1
35	10/0	10/2	-	-	-	-
37,5	-	10/5	-	-	-	-
40	-	10/9	-	20/0	-	20/1
42,5	-	10/6	-	-	-	-
45	-	10/10	-	-	-	-
50	-	10/10	20/1	20/9	-	20/7
60	-	-	-	20/19	-	20/20
100	10/0	10/0	20/11	20/20	20/7	20/20

Tabela B4: Resultados dos ensaios de toxicidade aguda para o lixiviado bruto sobre *D. similis,* 48h

Concentração, %	**Nº de organismos/organismos afetados**					
	Janeiro	*Fevereiro*	*Março*	*Abril*	*Maio*	*Junho*
0	10/0	10/0	20/0	20/0	20/0	20/0
2	-	-	-	-	-	20/1
3	-	-	-	-	20/0	-
4	-	-	-	-	20/12	20/2
5	-	-	-	-	20/17	-
6	-	-	-	-	20/20	20/18
6,25	10/1	10/2	20/0	20/0	20/20	-
8	-	-	-	-	-	20/20
10	-	-	20/0	20/1	-	-
12,5	10/0	10/4	20/1	20/1	20/20	20/20
15	-	10/7	20/9	20/16	-	-
17,5	-	10/10	20/15	20/18	-	-
20	-	10/9	20/20	-	-	-
25	10/9	10/10	20/20	20/20	20/20	20/20
50	10/10	10/10	20/20	20/20	20/20	20/20
100	10/10	10/10	20/20	20/20	20/20	20/20

Tabela B5: Resultados dos ensaios de toxicidade aguda para a mistura esgoto-lixiviado preparada em bancada, sobre *D. similis,* 48h

Proporção de lixiviado, %	**Nº de organismos/organismos afetados**					
	Janeiro	*Fevereiro*	*Março*	*Abril*	*Maio*	*Junho*
0	10/0	10/0	20/2	20/0	20/0	20/0
0,25	-	-	-	-	-	20/1
0,5	-	-	-	-	-	20/2
0,75	-	-	-	-	-	20/9
1	-	-	-	-	20/6	20/9
1,25	-	-	-	-	-	20/11
2	-	-	-	-	20/18	-
2,5	-	-	20/1	-	-	-
3,5	-	-	-	-	20/20	-
5	-	10/4	20/2	20/0	-	-
5,5	-	-	-	-	20/20	-
7,5	-	-	20/13	20/1	-	-
10	10/4	10/9	20/20	20/0	20/20	20/20
12,5	-	-	20/19	20/10	-	-
15	10/7	10/10	-	20/13	-	-
20	10/9	10/10	-	-	-	-
25	10/10	10/10	20/20	20/20	20/20	20/20

Tabela B6: Toxicidade (CE_{50} e intervalo de confiança, 95%) do esgoto bruto (EB), lixiviado (L), misturas esgoto-lixiviado preparada em bancada (EB-L preparada) e afluente (EB-L afluente) a *D. similis*

Amostra	**CE_{50} (%)**					
	Janeiro	*Fevereiro*	*Março*	*Abril*	*Maio*	*Junho*
EB	NT	NT	NT	NT	NT	NT
L	18,2 (15,8 – 20,9)	12,1 (9,2 – 16,0)	15,5 (14,7 – 16,3)	14,1 (13,5 – 14,7)	4,0 (3,7 – 4,2)	4,6 (4,2 – 5,2)
EB-L preparada	11,5 (7,6 – 14,0)	5,6 (3,4 – 7,3)	7,0 (6,4 – 7,6)	12,9 (10,5 – 15,7)	1,3 (1,1 – 1,5)	1,1 (0,8 – 1,6)
EB-L afluente	NT	NT	91,9 (89,7 – 94,2)	NT	NT	NT

*NT = amostras não tóxicas.

Tabela B7: Correlações entre a CE_{50} e os parâmetros físicos e químicos das amostras de efluente secundário, lixiviado bruto e mistura esgoto-lixiviado afluente.

	Efluente secundário																		
	pH	*cond.*	*DQO*	*DBO*	*NTK*	$N\text{-}NH_3$	N_{OX}^-	*P*	*ST*	*SST*	*SF*	*SV*	*Pb*	*Cd*	*Cu*	*Cr*	*As*	*Hg*	CE_{50}
CE_{50}	-0,55	-0,53	-0,24	-0,21	-0,46	-0,50	-0,33	-0,24	-0,38	-0,10	-0,40	-0,31	-0,44	-0,44	-	-0,32	**1,00**	0,17	1,00
	Lixiviado bruto																		
CE_{50}	**-0,86**	-0,80	-0,16	0,42	**-0,87**	**-0,84**	**-0,83**	-0,62	-0,08	0,06	-0,10	-0,07	-0,61	-0,13	0,05	-0,39	-0,28	0,62	1,00
	Mistura esgoto-lixiviado afluente																		
CE_{50}	-0,16	0,39	0,65	0,38	0,37	0,36	0,52	-0,41	0,72	-0,14	0,52	0,81	-0,13	-0,16	-	-0,10	-0,14	-0,19	1,00

Valores em negrito: correlações significativas a $p \geq 0,05$.

ANEXO C – RESULTADOS DOS ENSAIOS DE TOMADA DA TAXA DE CONSUMO DE OXIGÊNIO

Tabela C1: Taxa de consumo de oxigênio no esgoto bruto (EB) e em misturas esgoto-lixiviado ($L_{\%}EB$) nas proporções de 5, 10 e 20% v/v, e sólidos suspensos voláteis do lodo.

	Taxa de consumo de oxigênio mg O_2/L.h				**SSV lodo**
Amostra	EB	L_5EB	$L_{10}EB$	$L_{20}EB$	**g/L**
Janeiro	0,13	0,17	0,18	0,20	4,1
Fevereiro	0,13	0,16	0,23	0,28	4,1
Março	0,11	0,07	0,09	0,33	4,7
Abril	0,39	0,56	0,47	0,43	16,5
Maio	1,92	2,12	2,16	1,92	12,1
Junho	0,60	0,68	0,61	0,64	15,8

Tabela C2. Correlações entre a taxa específica de consumo de oxigênio (TECO) do esgoto bruto (EB) e misturas lixiviado-esgoto ($L_{\%}EB$) e os resultados dos parâmetros físicos e químicos do esgoto bruto.

	TECO			
Parâmetros (EB)	***EB***	***L_5EB***	***$L_{10}EB$***	***$L_{20}EB$***
pH	0,55	0,73	0,43	-0,46
Cond.	0,05	-0,25	-0,08	0,08
DQO	-0,31	-0,38	0,05	0,69
DBO	-0,74	-0,60	-0,37	0,36
NTK	0,10	-0,14	-0,30	-0,45
$N\text{-}NH_3$	0,17	0,04	-0,18	-0,59
N_{OX}^-	-0,51	-0,33	-0,16	0,28
P	-0,18	-0,46	-0,20	0,72
ST	-0,13	-0,25	0,32	**0,91**
SST	-0,05	0,00	0,56	0,58
SF	-0,27	-0,31	0,28	**0,83**
SV	-0,06	-0,22	0,33	**0,93**
Pb	0,28	0,29	-0,21	**-0,84**
Cd	0,32	0,29	-0,18	**-0,82**
Cu	-0,55	-0,05	-0,33	-0,57
Cr	0,49	0,38	-0,09	-0,56
As	-0,05	0,19	-0,30	**-0,91**
Hg	-0,29	-0,43	-0,60	-0,39

Valores em negrito: correlações significativas a $p \geq 0{,}05$.

Tabela C3: Correlações entre a Taxa específica de consume de oxigênio (TECO) do esgoto bruto (EB) e misturas lixiviado-esgoto ($L_{\%}EB$) e os resultados dos parâmetros físicos e químicos do lixiviado (L).

	TECO			
Parâmetros (L)	***EB***	***L_5EB***	***$L_{10}EB$***	***$L_{20}EB$***
pH	0,80	0,53	0,33	-0,31
Cond.	0,36	0,13	-0,25	-0,61
DQO	-0,18	-0,07	-0,57	-0,55
DBO	-0,60	-0,38	-0,64	-0,19
NTK	0,49	0,36	-0,17	-0,74
$N\text{-}NH_3$	0,44	0,23	-0,20	-0,64
N_{OX}^-	0,45	0,10	-0,24	-0,39
P	0,52	0,42	0,21	-0,43
ST	0,20	0,34	0,11	-0,11
SST	0,16	0,34	0,22	0,02
SF	0,16	0,29	0,03	-0,15
SV	0,22	0,36	0,15	-0,09
Pb	0,24	0,30	-0,22	**-0,85**
Cd	-0,19	0,14	-0,33	**-0,85**
Cu	0,17	0,34	0,23	0,03
Cr	0,42	0,61	0,10	-0,54
As	0,43	-0,08	0,25	0,66
Hg	-0,26	-0,38	0,12	**0,90**
CE_{50}	-0,80	-0,57	-0,27	0,48

Valores em negrito: correlações significativas a $p \geq 0,05$.

ANEXO D – RESULTADOS DOS ENSAIOS DE CRESCIMENTO DE BIOMASSA

Tabela D1: resultados do monitoramento da DQO nos reatores inoculados com esgoto bruto (R1 e R2) e mistura esgoto-lixiviado 20% v/v (R3 e R4), por um período de 13 dias.

	Reatores			
Datas	**R1**	**R2**	**R3**	**R4**
14/mai	86	64	3707	3524
15/mai	111	108	2648	2352
16/mai	63	111	2027	1968
17/mai	34	35	1585	1677
18/mai	80	94	1548	1710
19/mai	80	95	1510	1277
20/mai	114	101	1598	1585
21/mai	89	91	718	685
22/mai	96	98	460	710
23/mai	39	64	1099	1224
24/mai	104	73	1118	1230
25/mai	23	28	1338	1351
26/mai	54	67	1930	1263

Tabela D2: Resultados do monitoramento de SSV nos reatores inoculados com esgoto bruto (R1 e R2) e mistura esgoto-lixiviado 20% v/v (R3 e R4), por um período de 13 dias.

	Reatores			
Datas	**R1**	**R2**	**R3**	**R4**
14/mai	39	41	51	45
15/mai	44	43	61	51
16/mai	34	37	56	87
17/mai	31	32	80	82
18/mai	35	31	185	192
19/mai	37	36	170	198
20/mai	42	35	172	184
21/mai	38	45	145	134
22/mai	43	47	235	146
23/mai	39	42	244	320
24/mai	44	33	401	411
25/mai	46	42	408	300
26/mai	47	43	614	437