

Article

Co-Treatment of Municipal Landfill Leachate in Sewage Treatment Plants: A Model Based on a Literature Review

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Abstract

The management of landfill leachate remains a persistent environmental issue for municipalities globally. Although dedicated treatment in engineered landfills mitigates environmental contamination, it is often cost-prohibitive. Co-treatment of landfill leachates in sewage treatment plants has been broadly studied, but there are a lot of issues associated with it. Sewage treatment plants apply physical, chemical, and biological processes, and the co-treatment of leachates—contaminated with metals, pesticides, emerging contaminants, and other toxic compounds—can impair the biological equilibrium of the system and compromise the quality of effluents and sludges. In the present research, the processes leading to the formation of landfill leachates and the processes that promote the removal of contaminants in sewage treatment plants were discussed. A theoretical, early screening level mixing model, incorporating removal rates and leachate concentrations from the literature, was employed to simulate effluent concentrations from a co-treatment process involving sequential decantation and an upflow anaerobic sludge blanket (UASB). Under a conservative worst-case scenario obtained from the literature, the model predicts that adsorption of contaminants onto the particulate phase enables removal of metals from the solution. However, considering the volumes of sludge involved, the predictions indicate that concentrations should be lower than naturally occurring in the sediments. It is proposed that continuous monitoring follow-up is a mandatory safeguard for any co-treatment operation.

Keywords: treatment procedures; contaminant; organic matter; biochemical oxygen demand; BOD; humic substance; modeling; sludge; effluent



Academic Editor: Eva Pongrácz

Received: 24 September 2025

Revised: 26 December 2025

Accepted: 4 January 2026

Published: 7 January 2026

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1. Introduction

Municipal waste landfill leachates (MWLL) are liquid substances generated primarily from rainwater percolation, which mobilizes contaminants from decomposing food, pesticides, detergents, personal care products, waste oil, metal-containing batteries, wood treatment dangerous substances, pharmaceuticals, electronic and electrical equipment components, and CFC-containing equipment, plastics, papers, paints, and cleaners [1]. This leachate is a hazardous mixture of organic and inorganic chemicals that can contaminate groundwater and subsequently reach rivers, coastal and marine waters, and sediments [2]. As these contaminants disperse through environmental compartments, they can undergo

biomagnification within trophic chains, ultimately posing risks to human health [3,4]. Despite these risks, leachate production from municipal landfills continues to increase with urban expansion, particularly in underdeveloped countries [5].

Dedicated, large-scale leachate treatment plants at municipal waste landfills represent an effective solution to prevent environmental contamination. However, leachate treatment is technologically challenging, due to the variable composition and high concentrations of refractory organic and inorganic compounds in these effluents [6]. Numerous procedures for leachate treatment have been proposed, including membrane bioreactors (MBR), upflow anaerobic sludge blanket–membrane bioreactors (UASB-MBR), aerobic granular sludge (AGS), bioelectrochemical systems, and activated charcoal [7–14]. However, these technologies often suffer from declining efficiency over time due to process decay and involve elevated costs, which can hinder the sustainable operation of sanitary landfills.

Co-treatment of MWLL with other waste streams at existing treatment plants has been extensively discussed as a potential alternative, provided certain limitations are addressed. Some studies indicate that organic sludges can serve as inoculants for bacteria that degrade toxic residues, such as stillage [15]. Furthermore, the use of discarded materials that modify the physical and chemical conditions enhances the degradation process. For instance, Parra-Orobio et al. [16] adjusted pH in an experimental set to improve the efficiency of the digestion process of a refractory municipal bio-waste through co-treatment with a domestic sewage sludge. In this case, the chemistry of each residue improved the treatability of the ensemble.

Given that co-treatment can be beneficial by improving efficiency and reducing costs for various waste streams, a key question arises: Can MWLL be safely managed in a sewage treatment plant (STP)? Hua et al. [7] tested an anaerobic sewage treatment procedure (UASB, upflow anaerobic sludge blanket), coupled with the MBR (membrane bioreactor) and observed effective reductions in chemical oxygen demand (COD), biochemical oxygen demand (BOD), ammonium (NH_4^+), and suspended solids (SS). These authors attributed the observed improvements to substances that interact in the sewage. Conversely, Campos et al. [17] applied an integrated fixed film in activated sludge (IFAS) system to co-treat sewage sludges and MWLL at rates varying between 1:20 and 1:5, noting good performance for nitrogen removal. Nonetheless, these results are often applicable to very specific conditions, where varying influent residues may yield different outcomes.

When sanitation companies must construct treatment or co-treatment systems, technology selection is frequently driven by cost rather than predicted performance. This often leads to the choice of inexpensive but ineffective systems, ultimately incurring higher costs for retrofitting and upgrades. This preference stems from the fact that cost models are considerably simpler than performance models, which depend on numerous unstable variables. A simplified model capable of predicting contaminant behavior and fate in co-treatment systems would enable more straightforward and informed decision-making.

The present research aims to develop a simplified model to evaluate the chemical behavior of MWLL in STPs. The model predicts concentrations of contaminants in the effluents of a co-treatment plant, as a first approach to decision-making. First, a background on co-treatment is provided, detailing the chemical characteristics of leachates and sewage treatment systems (part 2). This is followed by a theoretical evaluation of co-treatment (part 3), which informs the development of a simplified mass balance model for performance at various mixing ratios of these residues (part 4).

2. Background of Co-Treatment

2.1. Leachates from Municipal Waste Landfill (MWLL)

MWLL differs from other anthropic originated organic materials because its composition is associated with two basic processes [18]. The first is leaching from percolating water (rainfall or moisture from the wastes) in the residue column, where chemicals dissolve, and solids are carried downward by advection. The second process is determined by the permanence of the liquid phase within the residues of the landfill, leading to different levels of biochemical maturation of the organic matter. In this process, the role of bacteria is relevant, considering that in recent landfills, aerobic fermentation is dominant, producing higher concentrations of biodegradable fatty acids [19]. In older landfills, the anaerobic process dominates, producing methane, and the organic molecules tend to humidify [20]. Both processes will determine the basic composition of the leachate, including concentrations of metals, pH, alkalinity, Eh (oxy-redox potential), conductivity, COD (chemical oxygen demand), BOD (biochemical oxygen demand), and concentration of non-refractory or persistent organic compounds [21].

Metals and metalloids can be present in batteries and electronic waste. For instance, lithium, nickel, cadmium, and zinc can be released from spent batteries (Fu, 2019#6266), while LED bulbs contain variable concentrations of Ag, Au, Ga, Mg, Co, Sb, Ba, As, Pb, Ce, and Y, according to Cenci et al. [22]. Most of these metals and metalloids are recyclable [23,24], but recovery from garbage is an issue, leading to their disposal in Municipal Waste Landfills. Organic soluble compounds like pesticides, dioxins [25–27], medicines, and pathogens [28], and many other emerging pollutants contaminate Municipal Waste Landfills. While some organic pollutants such as PCBs and furans do not leach directly due to their oily nature (aromatic hydrocarbons; Haarstad et al. [26]), they are still carried into the leachate by advection and, at some point, may emulsify. Fine-grained particulate matter also enters the composition of the leachate, including micro and nanoplastics [29,30].

Besides contaminants that are incorporated in the MWLL, the physical and chemical characteristics of the liquid phase control biological and chemical reactions [31]. pH, alkalinity, hardness, Eh, conductivity, temperature, and dissolved oxygen determine different processes that may occur in the interstitial spaces of the landfill, or later in the landfill leachate reservoir.

The rainfall in municipal landfills is slightly acidic, even in areas without significant atmospheric pollution, but as it percolates through residues, there is an increase in salt concentrations that buffers pH, which becomes relatively neutral [18,32], reducing its reactivity and preventing intense leaching of contaminants [33]. On the other hand, dissolved oxygen in the MWLL is controlled by the extent of the contact surface between the liquid and air interface, and temperature and salinity determine saturation of the gas in the solution [34,35]. On the other hand, the elevated values of biochemical oxygen demand (BOD) imply a rapid consumption of the gas in the solution [36,37], evolving to anoxic conditions, which are predominant in municipal waste landfills.

The concept of residence time [38] applied to semi-restricted systems describes the time leachate is in contact with the residues, determining the intensity of biochemical processes. Figure 1 shows a generic scheme of a municipal landfill.

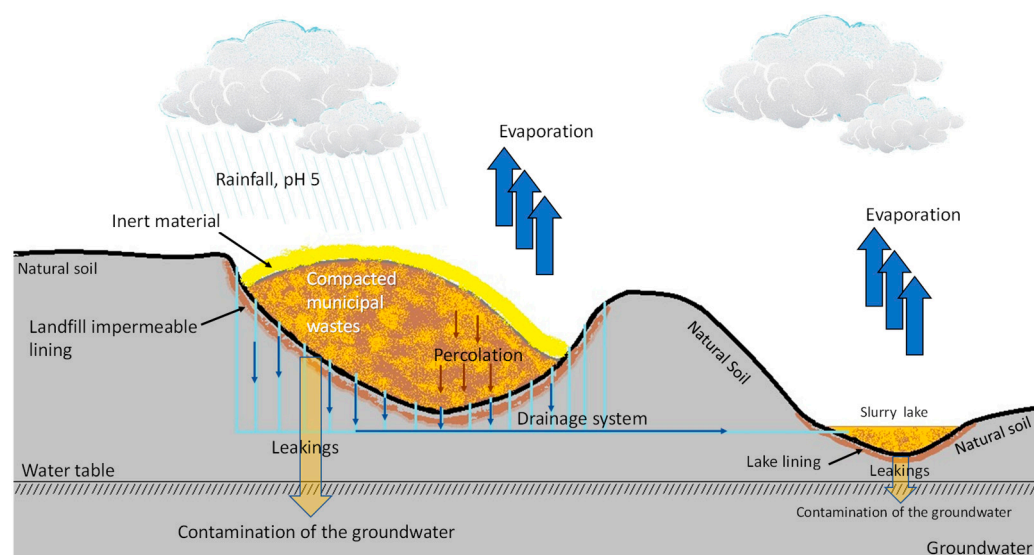


Figure 1. Generic scheme of a municipal landfill, showing residue retention and lining structures, as well as the constructed drainage leading to the leachate lake. In some landfills (uncontrolled landfills), the leachate drains directly into rivers, causing severe contamination.

In the reducing environmental conditions of the landfill, severe modifications of the physical and chemical forms are expected [39,40]. The chemical form of the element and its speciation or fractionation will determine its mobility, bioavailability, and chemical reactivity [41]. For instance, iron and manganese are known for their redox sensitivity [42]. In reducing environments, they remain in solution for a while and then react with sulfides to precipitate [43]. In leachates, the concentration of sulfides may be elevated, but not enough to precipitate all iron and manganese, which can remain in solution for long periods, until oxidation processes lead them to precipitate [44]. Although unreactive in its metallic form, mercury is hazardous in its ionic form (Hg^{+2}), due to methylation by sulfate-reducing bacteria [45,46]. Other elements like copper have a large affinity to organic matter, whether refractory or not [47,48]. These interactions are intensified in the MWLL because metals accumulate, resulting in increased toxicity, which may affect the performance of co-treatment in STPs.

Concerning nutrients, the most studied is nitrogen. In the case of MWLL, the most important is ammonium (NH_4^+) [18], which is responsible for enhancing bacterial activity, intensifying biochemical transformations in the leachate. Although phosphate is an important nutrient, concentrations in the leachate are reduced [49] due to precipitation with iron in reducing conditions [50,51].

Processes involving organic matter in MWLL are more complex because its origin is variable and its residence time in the landfill is longer, allowing intense evolution [52,53]. The most degradable organic matter is basically composed of proteins, lipids, and sugars that are immediately colonized by anaerobic bacteria [54]. Hydrocarbons are also colonized by specialized groups of anaerobic bacteria [55], but due to their smaller nutritive potential, degradation of these products is slower [56]. The degradation of pharmaceuticals can be differential because the excipient is composed of sugars, proteins, and amino acids, flavoring agents, and sweeteners, while the active substance may undergo slower degradation and is frequently released into the leachate solution, constituting an important pollutant [57].

In landfills, degradation of organic matter is essentially anaerobic, leading to the formation of low molecular weight compounds (amino acids, fatty acids, and carbohydrates) that support specialized bacterial chains, producing large amounts of methane [12]. These molecules endure a slow degradation process, followed by polymerization, leading to the

formation of larger humic molecules that are refractory and characterize organic matter in leachates of older municipal landfills [58].

The formation of humic substances is a long-term process and, therefore, older landfills should present higher concentrations than more recent ones [21,59]. It is noteworthy that the residence time of leachate in the landfill does not determine the concentration of humic substances because the solid organic matter converts into humic substances before it is incorporated into the liquid phase. Furthermore, landfill leachate presents a non-refractory organic phase, similar to BOD (biochemical oxygen demand), micro and nanoplastics, nanoparticles, and other contaminants [60,61].

2.2. Biochemistry of the Domestic Sewage Treatment Plants (STP)

STPs are sequential systems that apply different contaminant removal techniques. When introducing MWLL in STP, adsorption of contaminants into particles may lead to removal by decantation or sedimentation. Chemical and biochemical oxidation may remove part of the COD, and most of the BOD, and anaerobic bacterial decomposition may promote further degradation of the organic matter and consumption of nutrients, or denitrification.

The oxidation of organic matter with oxygen is broadly applied in STPs because the main pollutant in these influents is biochemical oxygen demand (BOD), which is a labile organic matter oxidized in a mild oxygen flow [62].

Aerated treatment systems [63] oxidize sewage with air diffusers (fine bubble diffusers), which optimize the process. It is important to note that oversaturation is desirable because oxygen remains in the solution for longer periods and oxidizes organic matter faster. Processes applying supersaturation of oxygen aeration are more efficient than fine bubbles alone [64]. Although the process can become quite efficient, removal of BOD may reach a rate not higher than 86% [65]. The remaining BOD has to be removed with other techniques, mainly biological. Although oxygenation alone is not quite efficient to degrade COD, associated with activated sludge techniques [66], or associated with post-treatment phytoremediation [67] may remove almost 90% of this contaminant. There are also other BOD/COD removal processes like advanced oxidation (with H_2O_2), Fenton systems, photo Fenton, heterogeneous photocatalysis, and ozone-based systems [68], which can be quite efficient.

Variable metabolic pathways employed by bacteria can be harnessed to develop more effective sewage treatment procedures, significantly enhancing their performance. Santos et al. [69] evaluated aerobic hybrid reactors (AEHR), observing improved nutrient removal, thanks to a better performance of denitrifying bacteria within this type of system. The finding was supported by Li et al. [70], who observed an intensive phosphorus accumulation, generating uncontaminated effluents. Furthermore, Feng et al. [71] documented synergistic mixotrophic, autotrophic, and heterotrophic microbiota that increased both the performance and stability of the treatment process. The use of biodisks has also been shown to considerably increase the conversion of organic nitrogen-containing pollution [72]. Despite these performances, these structures may require significant investments and higher operational costs.

Aerobic biological sewage treatment systems provide more flexible conditions, where physical and chemical conditions may be controlled to drive bacterial activity (mainly applying bacterial consortia). For example, Mehrotra et al. [73] isolated 35 bacterial strains from domestic sewage to develop a consortium of *Bacillus* sp., *Achromobacter* sp., and *Comamonas* sp. that was able to efficiently degrade BOD and COD in aerobic systems. Safitri et al. [74] developed three consortia that were efficient in bioremediation processes,

and Dhall et al. [75] identified excellent domestic sewage treatment performances with the species *Bacillus pumilus*, *Brevibacterium* sp., and *Pseudomonas aeruginosa*.

While aerobic treatment systems are generally easier to control, they are often less effective than widely used anaerobic systems for removing high-strength organic loads. This performance gap is partly due to the inherent limitations of aerobic bacterial biofilms. Technologies like biodisks (rotating biological contactors) can enhance biofilm efficiency by improving oxygen and nutrient transfer [76,77]. Given aerobic systems' recognized effectiveness for intensive nutrient removal, the development of new techniques to improve their efficiency for broader applications remains an open field of research [78–80].

Due to the intrinsic anaerobic conditions of sewage, attributed to elevated BOD and COD concentrations, anoxic treatment systems are broadly used because they work well with little intervention [81]. Furthermore, the decanted sludge from the system can act as an excellent substrate for bacterial growth and biofilm formation.

The production of biofilms improves the efficiency of bacteria in sewage treatment systems because, although they have no metabolic role, they promote the stability of the colony, even under harsh starving conditions, thanks to their role as a nutrient storage system [82]. In a UASB (upflow anaerobic sludge blanket) treatment system with activated sludges, degradation of the organic matter may reach 93% BOD and 88% of COD [65], but bacteria are also able to eliminate nitrogen (denitrification) and phosphorus (60% and 35% respectively). They are also able to eliminate pathogenic bacteria.

Differently from bioremediation processes, which use artificially developed probiotics, bacteria in a sewage treatment biological system are not selected and cultivated to execute biodegradation because they are naturally present in raw sewage [83]. The indigenous or autochthonous resident bacteria are well adapted to the environmental conditions of the sewage, occupying the system to the detriment of other bacterial groups (including pathogenic), winning the ecological competition [84]. Nonetheless, regardless of the resistance of bacteria, their equilibrium may be broken through the incorporation of toxic substances, like trace-metals, surfactants, formaldehydes, and pharmaceutical compounds, which may contaminate raw sewage [85–91]. These contaminants are abundantly present in the MWLL, constituting a point of attention in co-treatment.

3. Municipal Waste Leachate Behavior in STPs

Although MWLL co-treatment in STPs has been done for a long time, the first essays were published in the 1990s (e.g., Berry and Lin [92], Niininen et al. [93]). The studies were developed because of the impact of the procedure on the quality of the effluent, and also because the toxicity of the leachate provoked disruption of the biological sewage treatment process [92,94].

In the early research, higher percentages of leachate had to be applied in batch experiments because the effects on the effluent could not be detected with smaller amounts [95,96]. Recently, Campos et al. [17] applied percentages of 5%, 10%, and 20% and were able to identify a detectable contaminant gradient in the final effluent. In large cities like Rio de Janeiro, with a population of over 6 million people, the estimated amount of landfill leachate may reach 20 L s^{-1} [97], where advanced treatment plants are necessary to dispose of these effluents. Presently, most of these effluents percolate into Guanabara Bay, a coastal semi-enclosed estuary whose water quality is severely compromised [98]. Based on the knowledge discussed in the previous item, the physical, chemical, and biological processes of landfill leachate co-treatment in sewage treatment plants will be evaluated.

During mixing of MWLL and sewage, it is expected that contaminants will preferentially associate with particulate matter, considering that they should have dense surface charges [99,100]. In the leachate, the surface charges are associated with the neutral pH

and with the chemical characteristics of the particles, leading to a partitioning of less than 30% of the metals in the dissolved phase [101]. Therefore, metals should be associated with smaller inorganic particles ($<63\ \mu\text{m}$), and more intensely with organic particles of any size [102]. It is acknowledged that further studies on the chemical partitioning of metals in MWLL are necessary.

In a co-treatment process, leachate must be combined with domestic sewage, which provides a rich source of labile organic matter. This organic fraction is subsequently mineralized, producing CO_2 in aerobic environments or a biogas mixture of CO_2 and CH_4 in anaerobic systems [103]. However, most of the particulate material should remain in the system and will settle as sludge. Considering that the pH of the solution remains relatively neutral throughout the sewage treatment process, the contaminants are expected to remain associated with the particulate matter. If the organic matter is decomposed during the process, the contaminants of the leachate will associate with other particles and will tend to decant into the sludge [104]. It is probable that very little of these contaminants will be released into the solution to be released as an effluent solution.

The contaminants present in the leachate are mainly refractory organics (pesticides, humic substances, PCBs, etc.) and trace metals. However, part of the humic substances will remain in the dissolved phase, sometimes responsible for quenching of ultraviolet radiation [105,106]. On the other hand, during co-treatment, nutrients (phosphorus and nitrogen) should remain in the solution, feeding the biological process of degradation of the sewage [17].

A point of debate concerns the impact of contaminants on biological activity in sewage treatment plants. The introduction of metals, pesticides, PCBs, and similar compounds can be toxic to the microorganisms responsible for degrading organic pollutants. This toxicity is likely the primary reason that only small amounts of municipal waste landfill leachate (MWLL)—typically less than 10%—are incorporated into co-treatment systems [107].

Although the evaluation of the impacts of these contaminants on microorganisms is not simple, a few studies have evaluated the process. De Carluccio et al. [108] evaluated changes in the abundance and dominance of bacteria in co-treatment systems with or without pre-treatment with Fenton oxidation. The net reduction in toxicity after Fenton application led to modifications in the bacterial community, initially dominated by the phylum Bacteroidota, shifted to Pseudomonadota. The performance of the system was maintained, indicating that the bacterial community adapts to newer conditions. Another study was carried out by Mannarino et al. [109], evaluating the toxicity (with ecotoxicological tests) of sewage with or without mixing of small amounts of MWLL (less than 1.5%). Although the authors identify differences in the toxicity before and after mixing, and after treatment of both solutions, the levels were considered low when compared to local environmental laws.

Summarizing, the principal physical process during co-treatment of landfill leachates in sewage treatment plants is the adsorption of contaminants to particles that settle in the sewage sludge, which are removed from the system. On the other hand, the effect of contaminants on treating bacteria is clear; however, adaptations in the community profile avoid a severe reduction in the performance of the system. Although further research is needed in this subject, apparently, co-treatment seems a feasible solution for the destination of toxic MWLL. Subsequently, a mass balance model based on literature data for sewage and municipal waste landfill leachate (MWLL) is presented. This model proposes a simplified calculation to evaluate the performance of co-treatment systems.

4. Mass Balance Model for Co-Treatment Systems

4.1. Simulated Concentrations in the Effluents

Aiming to demonstrate numerically the processes that occur with contaminants during co-treatment of MWLL in sewage treatment plants, a simple mathematical simulation is presented, based on literature data.

Input data of the following model were obtained from the literature on the description of the characteristics of landfill leachates. There is a large number of studies that report characteristics of MWLL, and the most relevant review article on the subject is the work of Renou et al. [18], which is not quite recent, but reports over 1500 published works on the subject, including patents, since 1973. Table 1 (physical and chemical parameters) and Table 2 (metals and metalloids), obtained from those authors' publication, show a survey of MWLL concentrations from a total of 32 works carried out in 16 countries worldwide. The data were obtained from the authors, but averages and standard deviations were calculated for recent, mature, and old landfills.

Table 1. Physical and chemical parameters of MWLL sampled in landfills all over the world. Data were assembled by Renou et al. [18], but here the average, standard deviation, minimum, and maximum values were calculated.

	COD (mg L ⁻¹)	BOD (mg L ⁻¹)	BOD/COD	pH	TSS (mg L ⁻¹)	N-Kjeldal (mg L ⁻¹)	NH ₄ ⁺ -N (mg L ⁻¹)
Recent sanitary landfills							
Average	21,809	10,477	0.43	7.5	2313	1381	1663
Standard deviation	18,507	7986	0.17	1.0	1278	1480	1501
Maximum	70,900	26,000	0.70	9.1	5000	3400	5210
Minimum	1870	90	0.05	5.6	950	11	3
N	18	18	18	21	10	8	17
Mature sanitary landfills							
Average	5353	897	0.21	8.0	632	1339	1638
Standard deviation	2500	400	0.09	0.5	215	271	1718
Maximum	9500	1436	0.33	9.0	784	1670	5500
Minimum	1180	331	0.07	6.9	480	1100	743
N	12	9	9	10	2	4	7
Old sanitary landfills							
Average	2125	147	0.10	8.2	574	586	632
Standard deviation	2676	249	0.12	1.3	671	637	571
Maximum	10,000	800	0.37	11.5	1600	1680	1590
Minimum	100	3	0.01	7.0	13	5	0.2
N	12	9	10	11	7	6	9

TSS = Total suspended solids.

Recent landfills have larger rates of BOD/COD, but their rates fall with aging (Table 1). This can be attributed to the fact that with aging, more refractory organic matter is preserved. It is also possible to observe that variation in pH is quite small, corroborating that contaminants tend to adsorb into particles as discussed above. As the landfill ages, the capacity of filtering particulate matter increases, thereby reducing TSS in leachates. The concentrations of nitrogen indicate that the ammoniacal form is the only possible form in the reducing environment of the landfill. Ammonium will be utilized by denitrifying bacteria in aerobic sewage treatment systems [107].

Table 2. Metals in leachates sampled in landfills all over the world. Data were assembled by Renou et al. [18], but here the average, standard deviation, minimum, and maximum values were calculated.

	Fe (mg L ⁻¹)	Mn (mg L ⁻¹)	Ba (mg L ⁻¹)	Cu (mg L ⁻¹)	Al (mg L ⁻¹)	Si (mg L ⁻¹)
Recent sanitary landfills						
Value	2.7	0.04	–	–	–	–
n	1	1	0	0	0	0
Mature sanitary landfills						
Average	780.1	3.66	0.09	0.39	0.47	7.1
Standard deviation	1694.6	7.15	0.11	0.35	0.64	4.8
Maximum	3811.0	16.40	0.20	0.78	0.92	10.5
Minimum	1.3	0.03	0.01	0.12	0.02	3.7
n	5	5	2	3	2	2
Old sanitary landfills						
Average	13.8	4.03	0.15	0.04	1.50	5.0
Standard deviation	10.7	7.65		0.03	0.71	
Maximum	26.0	15.50	0.15	0.08	2.00	5.0
Minimum	4.1	0.13	0.15	0.01	1.00	5.0
n	4	4	1	4	2	1

Concerning metals and metalloids (Table 2), concentrations tend to reduce with aging (except for barium and aluminum). In the work of Renou et al. [18], the authors did not consider concentrations of some harmful contaminants like lead, arsenic, chromium, and mercury, which are frequently present in municipal landfills. These data were obtained from the work of Yang et al. [110] and are presented in Table 3. Table 4 shows some data from Norwegian landfills [111], including concentrations of cadmium, nickel, and zinc.

Table 3. Concentrations of some metals and metalloids in municipal landfills in China. Data were obtained from Yang et al. [110], but here the average, standard deviation, minimum, and maximum values were calculated.

Landfill	Pb (mg L ⁻¹)	As (mg L ⁻¹)	Cr (mg L ⁻¹)	Hg (µg L ⁻¹)
Pingwang	0.195	0.113	0.086	0.467
Badu	0.034	0.078	0.132	0.093
Taochuashan	0.098	0.033	0.064	4.670
Xinhu	0.098	0.065	0.110	0.930
Miaogang	0.179	0.065	0.110	0.561
Average	0.121	0.071	0.100	1.344
Standard deviation	0.066	0.029	0.026	1.883
Maximum	0.195	0.113	0.132	4.670
Minimum	0.034	0.033	0.064	0.093
N	5	5	5	5

Table 4. Concentrations of some harmful metals in Norwegian municipal landfills. Data were obtained from Øygard et al. [111], but here the average, standard deviation, minimum, and maximum values were calculated.

Landfill	Fe (mg L ⁻¹)	Zn (µg L ⁻¹)	Cr (µg L ⁻¹)	Cu (µg L ⁻¹)	Pb (µg L ⁻¹)	Cd (µg L ⁻¹)	Ni (µg L ⁻¹)
Landfill 1	22	150	63	31	11	0.44	–
Landfill 2	15	220	7.9	8.4	3.4	0.16	–
Landfill 3	35	550	44	190	–	0.32	–
Landfill 4	110	240	58	16	4	0.15	31
Landfill 5	24	52	30	11	3.1	0.09	13
Landfill 6	10	320	36	24	7.1	1.4	20
Landfill 7	20	110	31	21	4.3	0.65	12
Landfill 8	59	50	12	5.3	1.5	0.09	8.5
Average	36.9	211.5	35.2	38.3	4.9	0.4	16.9
Standard deviation	33.2	166.0	19.6	61.9	3.2	0.4	8.9
Maximum	110.0	550.0	63.0	190.0	11.0	1.4	31
Minimum	10.0	50.0	7.9	5.3	1.5	0.1	8.5
n	8	8	8	8	7	8	5

As noted earlier, Baun et al. [101] highlighted the importance of metal speciation in landfill leachates. Their analysis found that while up to 30% of metals and metalloids remain dissolved, only about 10% are present in forms that pose a significant risk to microorganisms in the treatment process. These data were corroborated by Øygard et al. [111], who identified a soluble phase between 2% and 10% for chromium and between 10 and 28% for copper in Norwegian landfills. Although metals are associated with oxyhydroxides of iron and manganese in landfills, they can be released in the leachate whenever anaerobic conditions develop in these systems.

Based on data obtained from the literature and presented so far, quantitative estimates of concentrations affecting sewage treatment systems as a function of co-treatment of leachate mixing rates of 1%, 3%, and 5% were calculated after Equation (1). To achieve a worst-case scenario, from Tables 1–4, the maximum concentration values were picked for each parameter, creating the worst leachate possible. Table 5 presents a summary of the model inputs and the references where the data were obtained.

Table 5. Summary of the model inputs and the references where the data were obtained.

Parameter	Description	References	From Equation
[Max_{MWLL}]	Maximum concentration in MWLL (values from Tables 1–4)	[18,110,111] *	Equation (1)
% _M	Percentages of mixture sewage and MWLL (corresponds to 1, 3 or 5%)	Attributed by the authors	Equation (1)
% _{dec}	Percentage of removal from the decantation system	[65]	Equation (2)
% _{UASB}	Percentage of removal from the UASB system	[65]	Equation (2)
[Mix_{raw}]	Concentration of the mixture of sewage and MWLL (Equation (1)). Different values for 1, 3, and 5%	Obtained from Equation (1)	Equation (3)
% _{dec/UASB}	Efficiency from the sequential treatment in the decanter and in the UASB	[65]	Equation (3)
[$Susp$]	Concentration of total suspended matter in the sewage MWLL mixture	[112]	Equation (4)
RA	Rate of adsorption. Conservatively assumed to be 70%	[101,113]	Equation (4)

* Table 6 includes references for each one of the parameters.

Table 6. Concentrations of various parameters from the leachate and sewage mixture before any treatment. Maximum values in the landfill leachate were obtained from the literature (Tables 1–4).

Parameter	Maximum Values in the Landfill Leachate	Expected Concentrations from the Mixture (with no Treatment)		
		1%	3%	5%
COD (mg L ^{−1})	70,900 ¹	709	2127	3545
BOD (mg L ^{−1})	26,800 ¹	268	804	1340
Suspended solids (mg L ^{−1})	5000 ²	50	150	250
Ammonium (mg L ^{−1})	5500 ¹	55	165	275
Al (mg L ^{−1})	2.00 ¹	0.02	0.06	0.1
As (mg L ^{−1})	0.113 ³	0.00113	0.00339	0.00565
Ba (mg L ^{−1})	0.20 ¹	0.0020	0.0060	0.010
Cd (μg L ^{−1})	1.4 ⁴	0.014	0.042	0.07
Cr (μg L ^{−1})	132.00 ³	1.32	3.96	6.60
Cu (mg L ^{−1})	0.78 ¹	0.0078	0.0234	0.039
Fe (mg L ^{−1})	3811.0 ¹	38.11	114.33	190.55
Hg (μg L ^{−1})	4.67 ³	0.0467	0.1401	0.2335
Mn (mg L ^{−1})	16.40 ¹	0.164	0.492	0.82
Ni (μg L ^{−1})	31 ⁴	0.31	0.93	1.55
Pb (mg L ^{−1})	0.195 ³	0.00195	0.00585	0.00975
Si (mg L ^{−1})	10.5 ¹	0.1048	0.3144	0.524
Zn (μg L ^{−1})	550 ⁴	5.5	16.5	27.5

μg L^{−1} units are presented in bold to call attention to the difference. ¹ [18], ² [112], ³ [110], ⁴ [111].

In Table 6, expected concentrations from the simple mixture of 1%, 3% and 5% are presented. The model simulates processes in a simple sewage treatment plant, containing a primary decanter, with the capacity to remove 40% of the particulate material and a UASB (upflow anaerobic sludge blanket) system with a complementary capacity to remove BOD, COD, and particles. In the simulated system, there is no dedicated process to remove metals, metalloids or other contaminants, and their removal was only based on adsorption to particulate matter that is incorporated into sewage sludges. Table 7 shows the total removal capacities of a composition of both systems calculated after Equation (2), considering extreme cases of smaller efficiency, as established by Von Sperling [65]. Although the removal percentages were considered constant, the performance of a sewage treatment plant may vary with time, underestimating or overestimating values. After DelaPaz-Ruiz et al. [114], variations in the sewage characteristics and in the performance of domestic wastewater treatment plants may reach 25%. A seasonal variation is also expected, but in tropical systems with little temperature variation along the year, a small variation is expected in the physicochemical parameters [115]. Therefore, in the calculations of expected performances, a variation of 25% (−12.5% underestimation and +12.5% overestimation) was included to indicate the limits that concentrations may attain with the instability of the system.

$$[Mix_{raw}] = [Max_{MWLL}] \times \%_M \quad (1)$$

where

$[Mix_{raw}]$ —Concentration in the sewage after mixture with MWLL.

$[Max_{MWLL}]$ —Maximum concentration in MWLL (values from Tables 1–4).

$\%_M$ —Percentages of mixture sewage and MWLL (corresponds to 1, 3, or 5%).

Table 7. Expected efficiency conditions of co-treatment systems for sewage and landfill leachate. A simple decantation system, followed by a UASB, was considered. Values after Von Sperling [65]. A variability of 25% in the removal capacity was included as a function of estimates from DelaPaz-Ruiz et al. [114]. Values after “±” indicate the % oscillation of the process with respect to the removal percentage.

Parameter	Removal Capacity (%)		
	Primary Decanter	UASB System	Total
COD	10.0 ± 1.3	93.0 ± 7.0 *	93.7 ± 6.3 *
BOD	25.0 ± 3.1	88.0 ± 11.0	91.0 ± 9.0 *
Suspended solids	60.0 ± 7.5	30.0 ± 3.8	72.0 ± 9.0
Ammonium	10.0 ± 1.3	60.0 ± 7.5	64.0 ± 8.0
Aluminum	42.0 ± 5.3	25.2 ± 3.2	56.6 ± 7.1
Arsenic	42.0 ± 5.3	25.2 ± 3.2	56.6 ± 7.1
Barium	42.0 ± 5.3	25.2 ± 3.2	56.6 ± 7.1
Cadmium	42.0 ± 5.3	25.2 ± 3.2	56.6 ± 7.1
Chromium	42.0 ± 5.3	25.2 ± 3.2	56.6 ± 7.1
Copper	42.0 ± 5.3	25.2 ± 3.2	56.6 ± 7.1
Iron	42.0 ± 5.3	25.2 ± 3.2	56.6 ± 7.1
Mercury	42.0 ± 5.3	25.2 ± 3.2	56.6 ± 7.1
Manganese	42.0 ± 5.3	25.2 ± 3.2	56.6 ± 7.1
Nickel	42.0 ± 5.3	25.2 ± 3.2	56.6 ± 7.1
Lead	42.0 ± 5.3	25.2 ± 3.2	56.6 ± 7.1
Silicium	42.0 ± 5.3	25.2 ± 3.2	56.6 ± 7.1
Zinc	42.0 ± 5.3	25.2 ± 3.2	56.6 ± 7.1

* Variations were reduced because they exceeded 100%.

$$T_{\%} = 100 - ((1 - (\%_{dec} \div 100)) \times (1 - (\%_{UASB} \div 100)) \times 100) \quad (2)$$

where

$T_{\%}$ —Total removal capacity in a system of decantation followed by UASB.

$\%_{dec}$ —Percentage of removal from decantation system [65].

$\%_{UASB}$ —Percentage of removal from UASB system [65].

Table 8 shows expected concentrations in the outlet of the primary decanter, and values after “±” indicate the oscillation of 25% (±12.5%) in the performance of the process. In this Table, the values were calculated exclusively considering the performance of the primary decanter. Table 9 shows expected concentrations in the outlet of the UASB (after it also passed through the primary decanter), applying Equation (3). Values after “±” indicate the oscillation of 25% (±12.5%) in the performance of the entire process, including primary and UASB treatment.

$$[Tr_{level}] = [Mix_{raw}] \times (\%_{dec/UASB} \div 100) \quad (3)$$

where

$[Tr_{level}]$ —Expected concentration after a defined level of treatment (decanter or UASB).

$[Mix_{raw}]$ —Concentration of the mixture sewage and MWLL (Equation (1)). Different values for 1, 3 and 5%.

$\%_{dec/UASB}$ —Efficiency from the sequential treatment in decanter and in UASB (from Table 6).

The results in Table 9 are presented as $\mu\text{g L}^{-1}$ (ppb) to make them easily readable. The Operational Standard 45 values from the State Environmental Agency (INEA—Rio de Janeiro), which are the legal concentration limits for effluent disposal in Rio de Janeiro (Brazil), are presented in the same units for comparison. It was determined that there would be no other depuration process, like an aeration station or other polishing process, and the presented values would correspond to the concentrations expected in the final effluent.

The concentrations presented in these tables did not consider contaminants present in the sewage itself. This is due to the fact that ammonium, BOD, and COD (presenting elevated concentrations in the sewages) may considerably improve the performance of degrading bacteria in the UASB, at an unknown level, generating a large uncertainty. It is assumed that metal concentrations in the sewage and its sludge are small [116].

Table 8. Expected concentration in the effluent of the primary decanter. Values after “±” indicate the oscillation of 25% (±12.5%) in the performance of the process.

Primary Decanter				
Level of Treatment (%)		1%	3%	5%
COD (mg L ⁻¹)	10.0 ± 1.3	638.10 ± 8.86	1914.30 ± 26.59	3190.50 ± 44.10
BOD (mg L ⁻¹)	25.0 ± 3.1	201.00 ± 8.38	603.00 ± 25.13	1005.00 ± 41.88
Suspended solids (mg L ⁻¹)	60.0 ± 7.5	20.00 ± 3.75	60.00 ± 11.25	100.00 ± 18.75
Ammonium (mg L ⁻¹)	10.0 ± 1.3	49.50 ± 0.69	148.50 ± 11.25	247.50 ± 3.44
Al (µg L ⁻¹)	42.0 ± 5.3	11.60 ± 0.00	34.80 ± 2.06	58.00 ± 0.01
As (µg L ⁻¹)	42.0 ± 5.3	0.66 ± 0.06	1.97 ± 0.18	3.28 ± 0.30
Ba (µg L ⁻¹)	42.0 ± 5.3	1.16 ± 0.11	3.48 ± 0.32	5.80 ± 0.53
Cd (µg L ⁻¹)	42.0 ± 5.3	0.01 ± 0.00	0.02 ± 0.00	0.04 ± 0.00
Cr (µg L ⁻¹)	42.0 ± 5.3	0.77 ± 0.07	2.30 ± 0.21	3.83 ± 0.35
Cu (µg L ⁻¹)	42.0 ± 5.3	4.52 ± 0.41	13.57 ± 1.23	22.62 ± 2.05
Fe (µg L ⁻¹)	42.0 ± 5.3	22,104.00 ± 2000.00	66,311.00 ± 6002.33	110,519.00 ± 10,003.88
Hg (µg L ⁻¹)	42.0 ± 5.3	0.03 ± 0.00	0.08 ± 0.01	0.14 ± 0.01
Mn (µg L ⁻¹)	42.0 ± 5.3	95.12 ± 8.61	285.36 ± 25.83	475.60 ± 43.05
Ni (µg L ⁻¹)	42.0 ± 5.3	0.18 ± 0.02	0.54 ± 0.05	0.90 ± 0.08
Pb (µg L ⁻¹)	42.0 ± 5.3	1.13 ± 0.10	3.39 ± 0.31	5.66 ± 0.51
Si (µg L ⁻¹)	42.0 ± 5.3	60.78 ± 5.50	182.35 ± 16.51	303.92 ± 27.51
Zn (µg L ⁻¹)	42.0 ± 5.3	3.20 ± 0.29	9.60 ± 0.87	15.95 ± 1.44

µg L⁻¹ units are presented in bold to highlight the difference.

Table 9. Expected concentration in the effluent after the primary decanter and the UASB. Values can be compared with the State of Rio de Janeiro regulation (NOP-INEA-45). All metal concentrations given in µg L⁻¹. Values after “±” indicate the oscillation of 25% (±12.5%) in the performance of the process.

Primary Decanter + UASB					
Level of Treatment		1%	3%	5%	NOP-45
COD (mg L ⁻¹)	93.7 ± 6.3 *	44.67 ± 44.67	134.00 ± 134.00	223.30 ± 223.30	180
BOD (mg L ⁻¹)	91.0 ± 9.0 *	24.12 ± 24.12	72.36 ± 72.36	120.60 ± 120.60	40
Suspended solids (mg L ⁻¹)	72.0 ± 9.0	14.00 ± 4.50	42.0 ± 13.50	70.00 ± 22.50	40
Ammonium (mg, L ⁻¹),	64.0 ± 8.0	19.80 ± 4.40	59.4 ± 13.20	99.00 ± 22.00	20/10
Al (µg L ⁻¹)	56.6 ± 7.1	8.70 ± 1.42	26.0 ± 4.26	43.40 ± 7.10	3000
As (µg L ⁻¹)	56.6 ± 7.1	0.49 ± 0.08	1.47 ± 0.24	2.45 ± 0.40	100
Ba (µg L ⁻¹)	56.6 ± 7.1	0.87 ± 0.14	2.60 ± 0.43	4.34 ± 0.71	5000
Cd (µg L ⁻¹)	56.6 ± 7.1	0.01 ± 0.00	0.02 ± 0.00	0.03 ± 0.00	100
Cr (µg L ⁻¹)	56.6 ± 7.1	0.57 ± 0.09	1.72 ± 0.28	2.86 ± 0.47	100
Cu (µg L ⁻¹)	56.6 ± 7.1	3.38 ± 0.55	10.15 ± 1.66	16.92 ± 2.76	1000
Fe (µg L ⁻¹)	56.6 ± 7.1	16,534.00 ± 2697.04	49,601.00 ± 8091.13	82,668.00 ± 13,485.22	15,000
Hg (µg L ⁻¹)	56.6 ± 7.1	0.02 ± 3.30	0.06 ± 9.91	0.10 ± 16.52	10
Mn (µg L ⁻¹)	56.6 ± 7.1	71.15 ± 11.61	213.45 ± 34.82	355.75 ± 58.03	1000
Ni (µg L ⁻¹)	56.6 ± 7.1	0.13 ± 21.94	0.40 ± 64.82	0.67 ± 109.69	1000
Pb (µg L ⁻¹)	56.6 ± 7.1	0.85 ± 0.14	2.54 ± 0.41	4.23 ± 0.69	500
Si (µg L ⁻¹)	56.6 ± 7.1	45.47 ± 7.42	136.40 ± 22.25	227.33 ± 37.08	
Zn (µg L ⁻¹)	56.6 ± 7.1	2.39 ± 0.39	7.16 ± 1.17	11.93 ± 1.95	1000

µg L⁻¹ units are presented in bold to highlight the difference. * Variations were reduced because they exceeded 100%.

The analysis of Table 9 shows that the physical and chemical parameters, including ammonium, surpassed the standards with a mixture of 3% for BOD, suspended solids, and ammonium, while a mixture of 5% would yield values above threshold limits for these four parameters. Concerning metals and metalloids, although iron surpassed the threshold limits at all mixtures, none of them had their concentrations above such levels for any modeled percentage of mixture.

Regarding COD, while applied values (Table 7) were considered high, reliable comparative removal rates from the literature were unavailable. This difficulty stems from the fact that COD is quite hard to degrade by both bacterial and chemical processes. An extremely conservative removal estimate of 20% was also calculated, projecting COD concentrations should reach 574.9, 1722.9, and 2871.5 mg L⁻¹ for 1, 3, and 5% mixtures, respectively. It is important to note that a significant portion of this COD may be removed via precipitation and adsorption into the sludge, which could account for these higher removal efficiencies.

Although effluents of sewage treatment plants are intensely monitored as a mandatory procedure worldwide, data on contaminant concentrations in effluents are scarce in the literature. Most of the papers report concentrations in effluents of batch experiments of new techniques for the co-treatment of sewage and MWLL [117–119]. Ismail et al. [120] developed a leachate depuration system named SAS (self-aerated sponge), which reached removal rates for ammonium and various metals between 43 and 94%. Final concentrations of Cu, Zn, and Cd fall within the same range of concentrations found in the present study. For other elements, concentrations were different because the leachates presented different initial concentrations.

Before any further conclusion concerning the viability of co-treatment, it is necessary to consider that the modeled system is quite simple. For instance, if an aeration system were chosen instead of the UASB, the removal of iron would be considerably higher, because oxidation would lead to iron precipitation as oxy-hydroxides, which would be incorporated in the sludge [121]. An aeration system would also have a significant impact on BOD, COD, and ammonium, with oxidation and denitrification (with loss of gaseous nitrogen to the atmosphere).

Although the values selected to develop the present model were the most elevated in the literature (Tables 1–4), a brief analysis of the standard deviations and minimum and maximum values shows a broad variation, and for every percentage of applied mixture, a thorough monitoring program is necessary. A solution that would mitigate variability would be the use of equalization tanks, where large volumes of MWLL would be stored, integrating large variations in time and from different landfills. This procedure would be beneficial for the maintenance of the bacterial activity in the biological treatment systems because organisms adapt to constant conditions [108].

4.2. Simulated Concentrations in the Sludges

Another aspect of concern is the concentration and accumulation of contaminants in the activated sewage sludge. Table 10 presents the results for expected concentrations of metals and metalloids in the sludge after co-treatment of sewage and leachate, as calculated using Equation (4). In these calculations, it was assumed that the concentration of solids in the leachate and in the sewage is around 5 g L⁻¹ [110], considering that 70% of the metals and metalloids were associated with this particulate phase [101,113]. For a general reference of the concentrations, the values of the mean shale reported by Reimann et al. [122] were used as natural levels.

$$[Me_{sludge}] = [Mix_{raw}] \div [Susp] * RA \quad (4)$$

where

$[Me_{sludge}]$ —Metallic concentration in the sludge (mg kg^{-1}).

$[Mix_{raw}]$ —Metallic concentration in sewage MWLL mixture (Equation (1)).

$[Susp]$ —Concentration of suspended matter in the sewage MWLL mixture (5 g L^{-1} [110], Table 6).

RA—Rate of adsorption. Conservatively assumed to be 70% (0.7) [101,113].

Table 10. Calculated metallic concentrations in the sewage sludge after the co-treatment process with landfill leachates. The mean shale concentrations reported by Reimann et al. [122] were considered as reference natural values for metals and metalloids in the sediments.

	Concentrations in the Sewage Sludge			Mean Shale
	1%	3%	5%	
Suspended solids (mg L^{-1})	5000			
Al (mg kg^{-1})	2.8	8.4	14.0	91,000
As (mg kg^{-1})	0.16	0.47	0.79	13
Ba (mg kg^{-1})	0.28	0.84	1.40	550
Cd (mg kg^{-1})	2.0×10^{-6}	5.9×10^{-6}	9.8×10^{-6}	0.25
Cr (mg kg^{-1})	0.18	0.55	0.92	100
Cu (mg kg^{-1})	1.09	3.28	5.46	45
Fe (mg kg^{-1})	5335	16,006	26,677	55,000
Hg (mg kg^{-1})	0.00	0.00	0.00	0.18
Mn (mg kg^{-1})	22.96	68.88	114.80	850
Ni (mg kg^{-1})	0.00	0.00	0.00	70
Pb (mg kg^{-1})	0.27	0.82	1.37	22
Si (mg kg^{-1})	14.67	44.02	73.36	288,000
Zn (mg kg^{-1})	0.00	0.00	0.00	100

From Table 10, it is possible to observe that even with higher mixtures of 5% of leachate in the sewage, none of the metals and metalloids constitute any environmental risk or threat for any subsequent use of the activated sludge. Particularly, mercury, cadmium, and lead, which constitute extremely toxic elements, presented very low concentrations. Concentrations in sediments of a mildly contaminated environment, Sepetiba Bay (Rio de Janeiro), are considerably higher than observed in the sludge [123,124]. The concentrations are many orders of magnitude smaller than in the contaminated Guanabara Bay (also in Rio de Janeiro) [125–128]. The incorporation of leachates in sewage treatment plants does not constitute a danger for the management of sewage sludges.

4.3. Sensitivity Analysis

There is a significant uncertainty regarding the removal rates of contaminants from MWLL during co-treatment operations. Therefore, a brief sensitivity analysis was conducted, considering removal rates of pollutants smaller than those suggested in the mentioned literature. Table 11 compares calculations using removal rates equivalent to 100%, 50%, and 20% of the values applied in the previous sections. Table 12 applies the same approach (100%, 50%, and 20%) for calculations of concentrations in the sludge, considering the adsorption rates of the values applied in the previous sections. For these experiments, a mixing rate of sewage/leachate of 5% was chosen.

The sensitivity analysis reveals considerable variation in the final effluent concentration (Table 11). Despite this, all modeled concentrations for metals and metalloids remain far below the regulatory limits of Rio de Janeiro's NOP 45. In contrast, levels of COD, BOD, suspended solids, and ammonium are of significant concern. The MWLL would increase these contaminants in the sewage stream—an effect not considered in the present model.

Therefore, implementing an advanced oxidation process at the sewage treatment plant is recommended for these parameters.

Table 11. Sensitivity analysis considering removal rates of contaminants smaller than those suggested in the mentioned literature. The analyses were performed with a mixing rate of sewage/leachate of 5% and removal rates of 100% (as in the previous sections), 50%, and 20% were applied.

Expected Concentration from a 5% Mixture with Various Removal Rates				
Parameter	100% *	50% *	20% *	NOP-45
COD (mg L ⁻¹)	223	1884	2881	180
BOD (mg L ⁻¹)	120.6	730.3	1096.1	40
Suspended solids (mg L ⁻¹)	70	160	214	40
Ammonium (mg L ⁻¹)	99.0	187.0	239.8	20/10
Al (mg L ⁻¹)	43.40	71.70	88.68	3000
As (mg L ⁻¹)	2.45	4.05	5.01	100
Ba (mg L ⁻¹)	4.34	7.17	8.87	5000
Cd (mg L ⁻¹)	3.04×10^{-5}	5.02×10^{-5}	6.21×10^{-5}	100
Cr (mg L ⁻¹)	2.86	4.73	5.85	100
Cu (mg L ⁻¹)	16.93	27.96	34.59	1000
Fe (mg L ⁻¹)	82,699	136,624	168,980	15,000
Hg (mg L ⁻¹)	1.01×10^{-4}	1.67×10^{-4}	2.07×10^{-4}	10
Mn (mg L ⁻¹)	355.9	587.9	727.2	1000
Ni (mg L ⁻¹)	6.73×10^{-4}	1.11×10^{-3}	1.37×10^{-3}	1000
Pb (mg L ⁻¹)	4.23	6.99	8.65	500
Si (mg L ⁻¹)	227.4	375.7	464.7	
Zn (mg L ⁻¹)	0.012	0.020	0.024	1000

* These values represent a percentage of removal from those obtained from the literature and applied in Table 9.

Table 12. Sensitivity analysis considering adsorption rates of contaminants in the sewage sludge smaller than those suggested in the mentioned literature (Table 10). The analyses were performed with a mixing rate of sewage/leachate of 5% and removal rates of 100% (as in the previous sections), 50%, and 20% were applied.

Expected Concentrations in the Sludge from a 5% Mixture with Various Adsorption Rates				
Parameter	100% *	50% *	20% *	Mean Shale
Al (mg kg ⁻¹)	14	7	280	91,000
As (mg kg ⁻¹)	0.791	0.396	0.158	13
Ba (mg kg ⁻¹)	1.4	0.7	0.3	550
Cd (mg kg ⁻¹)	9.8×10^{-6}	4.9×10^{-6}	1.96×10^{-6}	0.25
Cr (mg kg ⁻¹)	0.924	0.462	0.185	100
Cu (mg kg ⁻¹)	5.46	2.73	1.09	45
Fe (mg kg ⁻¹)	26,677	13,339	5,335	55,000
Hg (mg kg ⁻¹)	3.27×10^{-5}	1.63×10^{-5}	6.54×10^{-6}	0.18
Mn (mg kg ⁻¹)	114.8	57.4	23.0	850
Ni (mg kg ⁻¹)	21.7×10^{-5}	10.9×10^{-5}	4.34×10^{-5}	70
Pb (mg kg ⁻¹)	1.365	0.683	0.273	22
Si (mg kg ⁻¹)	73.36	36.68	14.67	288,000
Zn (mg kg ⁻¹)	38.5×10^{-4}	19.3×10^{-4}	7.7×10^{-4}	100

* These values represent a percentage of adsorption rates on sewage sludge from those obtained from the literature and applied in Table 10.

Accumulation of metals and metalloids in the sludge, as presented in Table 12, likewise varies with adsorption rates. However, all modeled values are substantially lower than the reference mean shale concentrations. This suggests that the amounts of chemical elements introduced by co-treatment are easily removed by the large quantities of sewage sludge produced.

5. Concluding Remarks

Co-treatment of municipal waste landfill leachates (MWLL) at sewage treatment plants (STPs) offers a cost-effective disposal method for the disposal of these hazardous effluents. However, it has to be considered that the contaminants can inhibit the microorganisms responsible for organic matter degradation, potentially impairing treatment performance. It was shown that the processes occurring in a municipal landfill are complex and largely determine the characteristics of the leachates. Older landfills typically generate leachates containing a broad spectrum of contaminants, including trace metals and refractory organic compounds, whereas leachates from younger landfills are generally less toxic. Importantly, the particulate phase of the leachates tends to adsorb large quantities of these contaminants, representing an important remediation mechanism.

Several treatment processes commonly employed in STPs are theoretically suitable for simultaneous leachates and sewage treatment. This study examined physical (settling), chemical (oxidation), and biological (aerobic and anaerobic bacterial) processes, all of which enhance the removal of BOD, COD, and nutrients, particularly ammonium.

A straightforward mass balance model was developed to predict effluent quality based on varying leachate-to-sewage ratios. This is an early, screening-level model that serves as a practical tool for forecasting the performance of co-treatment systems. The results showed that effluent concentrations of the most toxic elements were considerably reduced in the effluents of the process. However, certain parameters—such as iron, BOD, COD, and ammonium—may remain elevated. These simulations were based on a simple STP configuration (decantation + UASB); the addition of an aeration system would further lower concentrations of these oxidizable contaminants.

The evaluation of contaminants adsorbed in the sludge (where most of the bacterial activity is harbored) showed lower concentration levels than in uncontaminated sediments (mean shale). Consequently, the expected effect of co-treatment on biological processes should be minor, allowing STPs to maintain their treatment efficiency.

The model proposed in this work has relevant limitations, first of all, because it was not validated by real experiments. The rates of removal applied in the model for the treatment procedures are quite variable in the real world and depend largely on the performance of each equipment. The sensitivity analysis showed that large variations may result from variations in the removal rates. However, the model is an easy and simple first approach to evaluate the feasibility of co-treatment and can be applied to many other processes.

It is concluded that although co-treatment of sewage and municipal landfill leachates is feasible, continuous monitoring of the effluents and of the sludge is necessary to warrant sustainability and environmental safety.

Author Contributions: Conceptualization, J.C.W.; investigation, J.C.W. and T.O.P.d.F.; data curation, J.C.W. and T.O.P.d.F.; writing—original draft preparation, J.C.W.; writing—review and editing, J.C.W. and T.O.P.d.F. All authors have read and agreed to the published version of the manuscript.

Funding: This research was funded by the Brazilian Council of Scientific and Technological Development (CNPq), under grant number 305374/2023-0.

Data Availability Statement: In the present research, no new data were created, and all data used for modeling were obtained from the cited literature.

Acknowledgments: The conception of the present article arose from issues faced by a sanitation company that, some four years ago, intended to treat landfill leachates in a sewage treatment plant.

Conflicts of Interest: The authors declare that the research was conducted in the absence of any commercial or financial relationships that could be construed as a potential conflict of interest. The funders had no role in the design of the study; in the collection, analyses, or interpretation of the data; in the writing of the manuscript; or in the decision to publish the results.

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