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Some topics of current practical relevance in environmental geotechnics

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Article

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Abstract
Environmental Geotechnics has been an established branch of Geotechnical Engineering for about 40 years. The contribution from Brazilian practitioners and researchers is meaningful in the many activities in this field. This paper proposes to discuss three topics of relevance to modern sustainability in Brazil, in which Geotechnicians could have an even greater involvement: expansions in MSW landfills, geotechnical confinement and other geotechnical solutions for remediation of contaminated land, and reuse of wastes as geomaterials. First, important aspects of the environmental protection system recommended for landfill expansions are described through examples, as well as the possibility of immersing geogrids to reinforce the MSW mass and increase storage capacity. Secondly, an industrial-site case study is presented to point out the additional challenges associated with site remediation at an urban region of past industrial land use and the importance of a joint regional investigation and remediation plan. The possibility of benefiting from geotechnical confinement and *in situ* passive remediation to treat the area also is highlighted. Finally, on the third topic, preparedness to accept working with wastes in geotechnical works is encouraged, and two investigation examples on the reuse of construction and demolition waste and water treatment sludge are presented and discussed.

1. Introduction

Environmental Geotechnics is the branch of Geotechnical Engineering that deals with environmental conservation in the face of impacts from anthropic activities and natural disasters. The term environmental conservation expresses the intent to both preserve and benefit from Nature for social-economic and technological development, safeguarding natural resources for future generations. Geotechnics may help reduce the extraction of natural resources for new developments, dispose of waste, control water, soil and atmospheric contamination, and recover degraded areas of the planet. Environmental Geotechnics may provide new spaces for human use by recovering areas degraded by desertification, erosion, salinization, pollution, and neglect after termination of industrial activities. The UNEP (UN Environment Programme) estimates that 15 % to more than 30 % of the soils of the planet are degraded by human activities, and the proportion of degraded rangelands, which cover about 50 % of the global land area, is around 23 % (Thenkabail, 2016). The recovery of degraded areas may use traditional Geotechnical techniques, such as earthwork, dredging, drainage, erosion control works, as well as techniques for remediation of contaminated land. Continuous development in Environmental Geotechnics is in great demand due to the increasing generation of waste, wastes of greater complexity, reuse of contaminated areas due to scarcity of space in urban conglomerates, ever more stringent environmental standards, and growing sustainability awareness and requirements in all human activities.

1.1 A brief historical perspective

Environmental issues have become a significant component of Geotechnical Engineering since *circa* 1980, although for long Geotechnical engineers have been involved with such themes (Shackelford, 2005). A first technical session on Environmental Geotechnics took place in the IX ICSMFE (International Conference on Soil Mechanics and Foundation Engineering) in 1977. In 1992, TC5, the Technical Committee on Environmental Geotechnics, presently

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TC215, was created in the scope of ISSMGE (International Society for Soil Mechanics and Geotechnical Engineering). In 2012, came the time for creating the Technical Committee on Sustainability in Geotechnical Engineering, TC307. The 1st International Congress on Environmental Geotechnics occurred in 1994, in Canada, and since then, the congress occurs every four years. In 2002, the ICEG took place in Brazil, and the latest congress was held in 2018 in China. Most Geotechnical journals have been covering the subject area, explicitly the prestigious ASCE's Journal of Geotechnical Engineering altered the name to Journal of Geotechnical and Geoenvironmental Engineering, in 1996, and ISSMGE TC215 launched in 2014 the journal Environmental Geotechnics.

In Brazil, a state-of-the-art report on Environmental Geotechnics was conveyed in the VIII COBRAMSEF (Brazilian Conference on Soil Mechanics and Foundations Engineering), in 1986. A Symposium on Tailings Dams and Waste Disposal, sponsored by the Brazilian Societies for Soil Mechanics and Geotechnical Engineering (ABMS) and Engineering Geology (ABGE), took place in 1987. This technical-scientific meeting turned eventually into a regular congress, the Brazilian Congress on Environmental Geotechnics, taking place every fourth year after 1991, and since 2003 occurring together with the Brazilian Congress on Geosynthetics. In 1994, the Technical Committee on Environmental Geotechnics (CTGA) was founded at ABMS. Presently, there are more than 20 research groups on Environmental Geotechnics registered at CNPq (National Research Council).

1.2 Branches of activity in Environmental Geotechnics

The main branches of activity in Environmental Geotechnics include waste disposal (site selection, design, operation and monitoring of MSW landfills, industrial landfills and other waste disposal facilities); use of soils and geosynthetics as construction materials for environmental protection works; use of geotechnical techniques for environmental protection; use of waste as geomaterial; monitoring and prevention of, and recovery from, accidents and natural disasters; prevention of contamination of superficial soil, subsoil, and surface- and groundwater; recovery of degraded areas; remediation of contaminated land; environmental impact assessment of civil works; risk analyses; investigation, instrumentation, monitoring and sampling of water and soil; environmental licensing and elaboration of environmental impact studies; environmental diagnosis and risk management of urban slopes; among others. Such applications of Environmental Geotechnics could be divided in three main groups: soils as receptors of contamination; soils as construction material in geoenvironmental works; and use of waste as geotechnical materials.

Not all of these activities are exclusive to Environmental Geotechnicians, and interaction with other fields is the key to meet the challenges with relevant solutions based on up-to-date knowledge. Basic knowledge of other disciplines to develop a common language is required, as well as a capacity to move away from the problem to acquire a wider perspective, and then move back to contribute in the specific scope of Geotechnics. Yet, multidisciplinarity is not a stranger to Geotechnics. From the rheology of polymers for injections in dam foundations or special concretes for tunnels to the survey of the geological history of a region to understand the behavior of a particular soil, Geotechnical Engineers have frequently worked together with professionals from other fields of knowledge. Nonetheless, in Environmental Geotechnics multidisciplinarity is a marked characteristic; the Engineer works with colleagues from Geology, Pedology, Chemistry, Hydrology, Microbiology and, more recently, Rheology, Thermodynamics, Biology and Nanotechnology.

1.3 Objectives

The objectives of this paper are to provide firstly a brief critical overview, and then a discussion on selected topics of practical relevance, in each of the following themes:

- Municipal solid waste landfills,
- Site remediation, and
- Geotechnical reuse of waste.

2. Municipal solid waste landfills

2.1 Overview

A municipal solid waste (MSW) landfill is a facility to contain the waste collected in households, small businesses and urban public spaces (roads, streets, parks, squares, public buildings, etc.) designed and built according to well-defined environmental and engineering concepts so as to guarantee structural and environmental safety. The demand for landfill storage capacity depends on waste generation, waste management and alternatives to landfilling, societal practices, and legislation, varying from place to place. Nonetheless, increasing landfill storage capacity remains a necessity around urban areas in the majority of countries. Landfill piggyback expansions and the possibility to reinforce slopes of municipal solid waste for increasing storage capacity are exemplified and discussed.

2.1.1 Destination of MSW in Brazil

In Brazil, an estimate of the average generation of MSW is 1.039 kg/inhabitant/day: in 2018, approximately 199×10^3 tons of waste were collected daily in Brazil, 59.5 % being disposed in landfills, 23.0 % in controlled dumps and 17.5 % in uncontrolled dumps (ABRELPE, 2020). Between 2000 and 2018, the percentage of MSW destined to landfills increased significantly, from 35.4 % to 59.5 % (ABRELPE, 2020). Estimates of MSW generation growth and disposal over the years can also be found in BNDES (2014).

In terms of number of municipalities, in 2015, 40.2 % of the Brazilian municipalities disposed MSW in landfills, 31.8 % in controlled dumps and 27.9 % in uncontrolled dumps (ABRELPE, 2016). In contrast, in 2008, only 13 % of municipalities disposed waste in landfills, while 59 % still used uncontrolled dumps (IBGE, 2010). The numbers vary according to region and population; for instance, 301 out of 399 municipalities in Paraná state disposed waste in landfills in 2017 (IAP, 2017, Oliveira, 2019), whereas the proportion was only 43 out of 417 municipalities in Bahia state. According to BNDES (2014), in 2012, the northeast and southeast Brazil generated, together, 75 % of the total MSW; however, NE destined only 35.4 % of MSW to landfills that year, whereas in the SE the percentage was 72.2 %.

The Federal Law 12,305 - National Policy on Solid Waste (BRASIL, 2010) - established that by 2014 the total generated MSW in the country should be adequately disposed of and, subsequently, waste dumps should be recovered and remediated. BNDES (2014) estimated financial investments on the order of US$ 1 billion between 2015 and 2019 to build the necessary landfills for Brazil to comply with the National Policy on Solid Waste, based on consolidated data from 2012. The requirement, however, was not met, and the deadline has been extended.

The demand for landfills continues to grow, due to the growth of cities, consortia formed among small municipalities to share the costs of implementation and operation, and implementation of large private MSW landfills to serve several neighboring municipalities. The increase in height, maintaining design concepts and construction methods, has led to slides, such as, for example, at Aterro São João, 2007, and Taiaçupeba, 2011.

A study by ABLP (2019), published in the journal *Limpeza Pública*, aimed at updating waste-disposal site numbers in Brazil. The study compared 2016 data from the National Data System on Solid Waste Management (SINIR) to a 2018-2019 survey by ABLP. The SINIR data are the official data of the Environmental Ministry, based on an annual survey on state-level environmental agencies. According to the SINIR data from 5,393 municipalities, 2,692 municipalities deposited waste in uncontrolled dumps, 427 in controlled dumps, and 2,274 in landfills. Based on these data, there were 1,803 uncontrolled dumps in Brazil, 40 controlled dumps, and 801 landfills in 2016.

The survey by ABLP, also from direct consultation to the state-level environmental agencies (ended January 2019), comprised 25 states and the Federal District, resulting in 792 landfills in the country, and 308 more landfills under licensing process. The numbers on a per-state basis are shown in Fig. 1.

Since the approval of the National Policy on Solid Waste (BRASIL, 2010), the country faces the challenge of implementing planned collection, selection, treatment and adequate disposal of MSW, domestic, commercial and industrial. Law 12,305 establishes shared responsibility for integrated management of solid wastes. The National Policy on Solid Waste rests on the principles of public-health and environmental protection, promoting non-generation, reduction, reuse, recycling, treatment and environmentally-adequate disposal of waste, as well as fostering industrial recycling, clean technologies, integrated management, and continued technical capacitation.

Decrees for the implementation of the law enforce the development of municipal-, state- and national-level management plans. Municipalities and states are enforced to

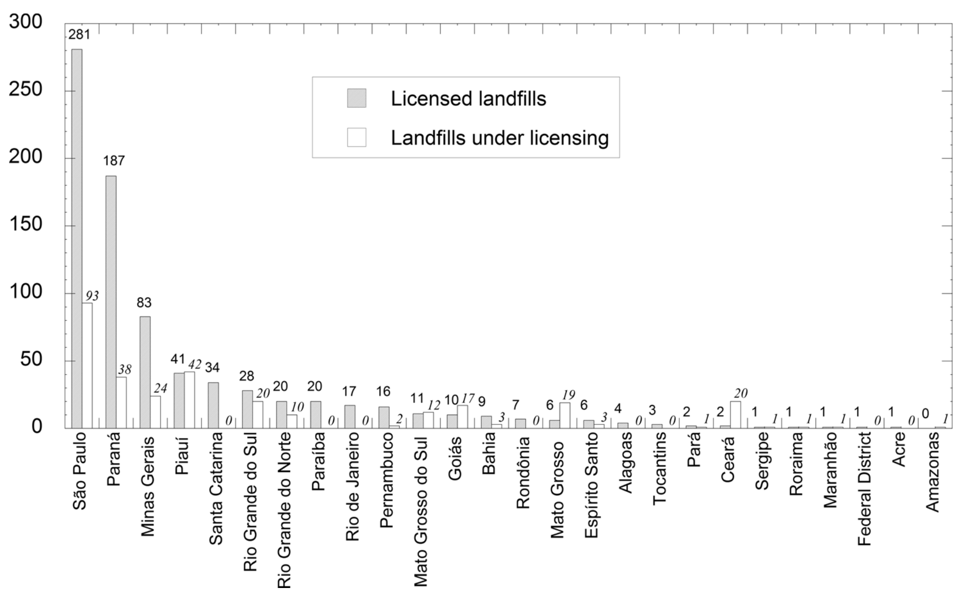


Figure 1. Number of licensed and under licensing landfills in 25 Brazilian states and the Federal District (Based on data from ABLP, 2019).

prepare a Plan for Integrated Solid Waste Management (PGIRS) as a condition for receiving federal sanitation funds. In 2011, a preliminary version of the National Plan for Solid Waste was prepared, containing the following targets (BNDES, 2014, van Elk & Boscov, 2016):

- Eradication of open uncontrolled dumps in Brazil, originally by August 2014; all such dumps should be decommissioned or converted into sanitary landfills, and the possibly contaminated area remediated;
- Reduction of the amount of waste generated, from around 1.1 kg/inhabitant/day to 0.6 kg/inhabitant/day;
- Implementation of organic matter composting (recycling), since organic matter should no longer be disposed in sanitary landfills;
- Differentiation between "solid waste" and "refuse" (the latter without any usefulness), with solid waste being selected/sorted and processed for reusable and recyclable materials, with a reduction of up to 70 % in the amount of waste going to landfills;
- Implementation of selective collection, with the insertion of 600,000 collectors;
- Implementation of energy from MSW biogas based on a viability study supported by gas monitoring, and
- Establishment of directives and responsibilities over the integrated management of solid waste, with reverse logistics and shared responsibility.

These targets, environment-friendly and up-to-date with most developed countries, are however very far from being robustly implemented, so that demand for MSW landfills is still on the agenda in the vast majority of the country.

Additionally, a Brazilian technical standard specifying the minimum requirements for location, design, implantation and operation of low volume sanitary landfills was enacted in 2010 (ABNT, 2010 - NBR 15,849). According to the coordinator of the group that developed the standard, the standard would allow the adoption of solutions adequate for the geographical reality of each municipality, making construction of landfills easier and therefore avoiding the proliferation of dump sites; else, requirements for a large city (*e.g.*, São Paulo) would be the same as for little towns. Other public managers also believed that the standard would allow the sustainability of MSW landfills for small municipalities, with lower costs of implantation and operation.

Unfortunately, the standard used very limited contribution from Geotechnical Engineers, and was prepared by a group majorly composed of public managers without an engineering background. In order to simplify licensing procedures, despite the insistence of Geotechnicians in the group, the requirements of engineered design, stability analyses, surface drainage, groundwater flow, among others, were oversimplified, regardless of the fact that the municipalities might be located over vulnerable subsoil profiles. The basis for adhering to the standard was simply daily generation of MSW (< 20 ton/day), and not landfill geometry and height. Also, a single compacted-clay layer as bottom liner may indeed adequately protect the subsoil and groundwater from leachate release in many cases (depending on climate, subsoil and compacted clay), and is a feasible solution even for small and poor municipalities, since compaction equipment is generally available. However, in order to keep distance from the requirement for a complex environmental protection system, statements on the need for designing a case-specific bottom liner and drainage systems as the adequate engineered solution were avoided in the standard. This was a strong example of the lack of Geotechnicians involvement and participation in environmental legislation, where they would have an important contribution.

Nowadays, the design of MSW landfills, mostly in large urban areas or shared MSW landfills, is carried out by Geotechnicians, with relevant technical and scientific contributions to the understanding of MSW hydro-mechanical properties and to the design, operation, monitoring and closure of landfills. However, there is still little contribution in standardization and regulation, as well as lack of an efficient channel of communication with society.

2.1.2 Main geotechnical issues in landfills

The main geotechnical issues affecting MSW landfills are geomechanical behavior, structural stability and waste compressibility, liquid and gas pore pressures, and design of the bottom liner and cover. The stability assessment remains generally based on limit-equilibrium, with Mohr-Coulomb strength parameters. Since the first Brazilian values of 13.5 kPa for cohesion and 22° for effective friction angle, obtained from back-analyzing the Bandeirantes landfill failure (Benvenuto & Cunha, 1991), national research has been developed (Machado *et al.*, 2002; Mahler & Lamare Neto, 2003; Campi & Boscov, 2011; Norberto *et al.*, 2020; Daciolo, 2020; among others), including more advanced constitutive models (Machado *et al.*, 2002; Mahler & Lamare Neto, 2005; Malavoglia, 2016; among others). Since landfills undergo large settlements, modeling of compression time evolution remains important. From adapting Terzaghi's consolidation theory (Sowers, 1973) to creating new models including biodegradation and creep (Machado *et al.*, 2009; Simões & Catapreta, 2010; Alcântara & Jucá, 2010; among others), advances have been made. Two approaches have been developed in parallel: estimating MSW parameters for geotechnical models, and developing specific models for MSW. Pore pressures in MSW are difficult to predict, measure and interpret, and are related to composition, age, biodegradation, compaction, and drainage conditions, factors that are correlated (Benvenuto & Cipriano, 2010; Coelho, 2005; Miguel *et al.*, 2018, among others). There is also an extensive Brazilian literature on the performance of bottom liners, including hydraulic conductivity and pollutant retention capacity issues. Recently, much research has focused on the release of biogas through the landfill cover (Teixeira *et al.*, 2009; Bridi *et al.*, 2015; Borba *et al.*, 2017; Costa *et al.*, 2018; among others).

2.1.3 Alternatives to landfilling

With socio-economic development, two opposite trends in terms of the quantity of MSW destined to landfills occur: increase in the generation of MSW, as an indicator of economic progress, and, on the other hand, adherence to policies of reduction, reuse, recycling and stabilization of wastes before landfilling, an indicator of social progress. In the European Union (EU), the quantities of waste sent to landfill sites are decreasing, as waste management must include differentiation and recycling, composting and waste incineration. Finland, Sweden, Denmark, Poland, Germany, the Netherlands, Belgium, Austria and Slovenia have laws banning or severely restricting the disposal of household waste in landfills (Conte & Carrubba, 2013). Figure 2 illustrates the waste disposal distribution percentages for countries in the EU based on 2018 data.

However, the use of landfills is still widespread, as nine out of 27 EU countries dispose more than 80 % of the MSW in landfills. The trend is stabilizing, as the use of recycling and pretreatment has increased (in fact, six countries dispose < 10 % of the MSW in landfills). The practice of waste incineration, involving partial recovery of energy, is widespread in the Nordic countries, such as Sweden and Denmark. Germany and Italy mainly use recycling (48 % and 37 %, respectively), while Austria is the main user of composting and anaerobic digestion (about 40 %) (Conte & Carrubba, 2013).

Nonetheless, the ashes from waste incineration must be, at least partly, taken to landfills. As a by-product of the treatment of municipal solid waste in waste-to-energy plants, roughly 230-280 kg of ashes are generated per ton of waste incinerated, bottom ash being the major stream (ISWA, 2006). Fly ash is regarded as a hazardous material due to the high content of heavy metals, whereas incineration bottom ash (IBA) can be either landfilled or utilized. Since IBA contains toxic heavy metals, not only the geotechnical properties, but also the environmental leaching properties must be studied. In China, for example, where incineration is widely used for MSW, ashes are submitted to solidification/stabilization treatment and then landfilled (Chen *et al.*, 2019). In Japan, where 5 million tons of IBA are generated every year, and landfilling space is scarce, IBA is being considered as a construction geomaterial (Fujikawa *et al.*, 2019). Almost 500 municipal solid waste incineration plants in the EU, Norway and Switzerland generate about 17.6 Mt/year of IBA. Since there is no uniform regulation for IBA utilization at EU level, countries developed their own rules with varying requirements. Metals are mostly separated and sold to the scrap market and minerals are either disposed of in landfills or utilized in the construction sector (Blasenbauer *et al.*, 2020). In France, a dedicated national legislation for IBA exists since 1994 (which has been improved along the years), which provides a detailed regu-

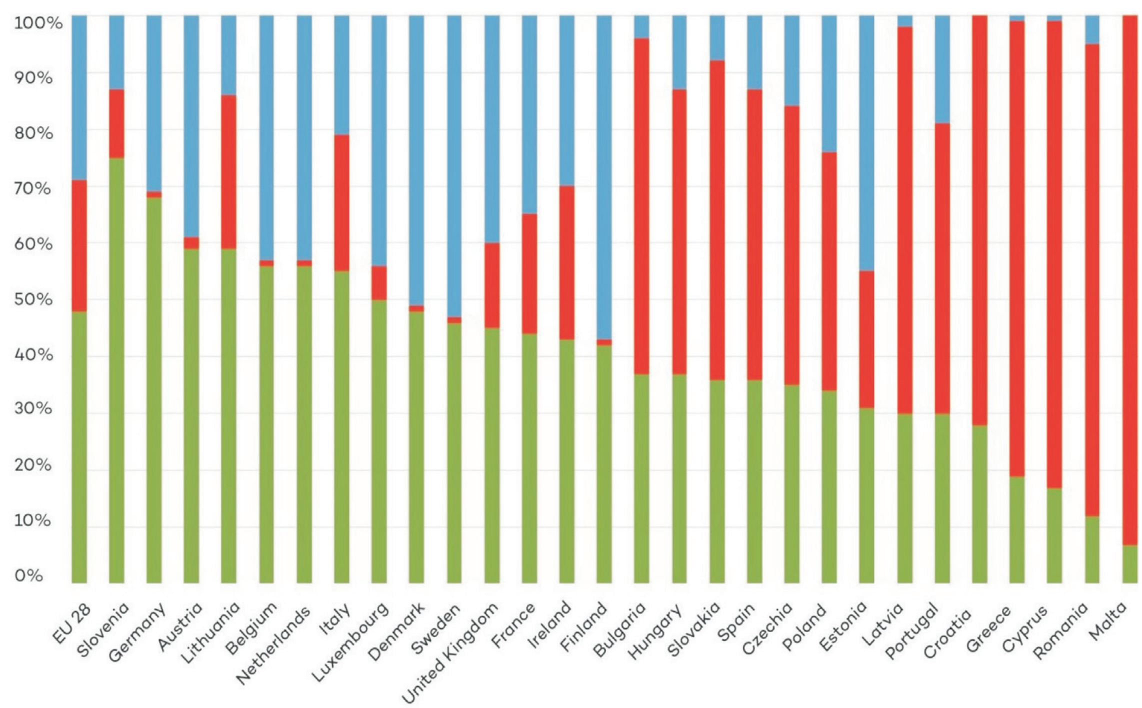


Figure 2. Percentages of MSW disposal practices in the European Union, including recycling (green), waste-to-energy (blue), and landfilling (red). Waste-to-energy includes incineration, composting and anaerobic digestion (Eurostat, 2018).

latory framework to facilitate management, with a view to reuse in road construction (ISWA, 2006).

It is important to mention that when waste-to-energy alternatives are implemented in Brazil, a trend to be expected for metropolitan regions with high MSW generation and lack of space for landfilling, Geotechnicians will have two new challenges and opportunities: design and operation of IBA landfills, and reuse of IBA in geotechnical works.

2.2 Landfill expansions

Due to the increased difficulty in finding and licensing new areas for landfills near cities, the option of expanding existing landfills becomes most attractive. The capacity increase in existing facilities may involve Geotechnical Engineering solutions such as the construction of a high peripheral reinforced-soil dike for verticalization of the landfill, rising of the landfill with geogrid-reinforcement of MSW slopes, or the so-called piggyback expansions. The two latter cases will be further discussed.

In Brazil, vertical and/or lateral expansions in landfills near urban areas, generally called amplifications, are much frequent. Brazilian landfill designers point out that layers for leachate drainage and impermeable barriers are applied in landfill expansions, but there is, however, a lack of technical guidance on the issue. The burden of the project rests entirely with the designer since there are no specific technical standards or recommendations. Geotechnicians are aware of the technical challenges imposed by the expansion foundations being constituted of a highly compressible and heterogeneous waste mass, where gas and leachate are still being generated, but most feel technically prepared to deal with this challenge.

The use of geogrid-reinforcement at the contact between the old and new landfills is generally never adopted. As shown by experience in other countries, there is a possibility for damage of the emplaced environmental protection systems, caused by large and differential settlements occurring in the old underlying landfill, indicating the need for additional measures aimed at reducing strains on the mineral and geosynthetic components.

Possibly, some mistrust relative to the maintenance of the geogrid properties for a long time inside the waste mass is sensed amongst Brazilian landfill designers. However, there is strong evidence of PVA-geogrid compatibility in caustic environment (Huesker, 2017; Nishyama *et al.*, 2006) and HPDE geomembrane compatibility in acidic environment (Renken *et al.*, 2007). On this subject, the academia could collaborate with designers, investigating geogrid performance specifically under MSW-leachate conditions: the pH range of MSW leachate in Brazil is reported as 5.7-8.6 (Souto & Povinelli, 2007, based on data from 25 Brazilian landfills; more recent papers corroborate this range), whereas temperatures may easily reach 60 °C (Carvalho, 1999).

2.2.1 Piggyback expansions

The terminology "piggyback" is used in the international literature to describe a new landfill (expansion) constructed on top of an existing one that has been either closed or scheduled to be closed, or when the new landfill uses the side slope of an old landfill as part of the support. Figure 3 illustrates some examples of possible geometries.

The reasons for adopting a piggyback expansion are maximizing the landfill utilization factor, economy in construction, sharing infrastructure, rationalizing use of equipment and facilitating authorization processes, among others. The main concerns involve safeguarding the integrity and maintenance of an adequate geometry for the environmental protection systems of both the extension and the old landfill amidst large differential settlements, enabling

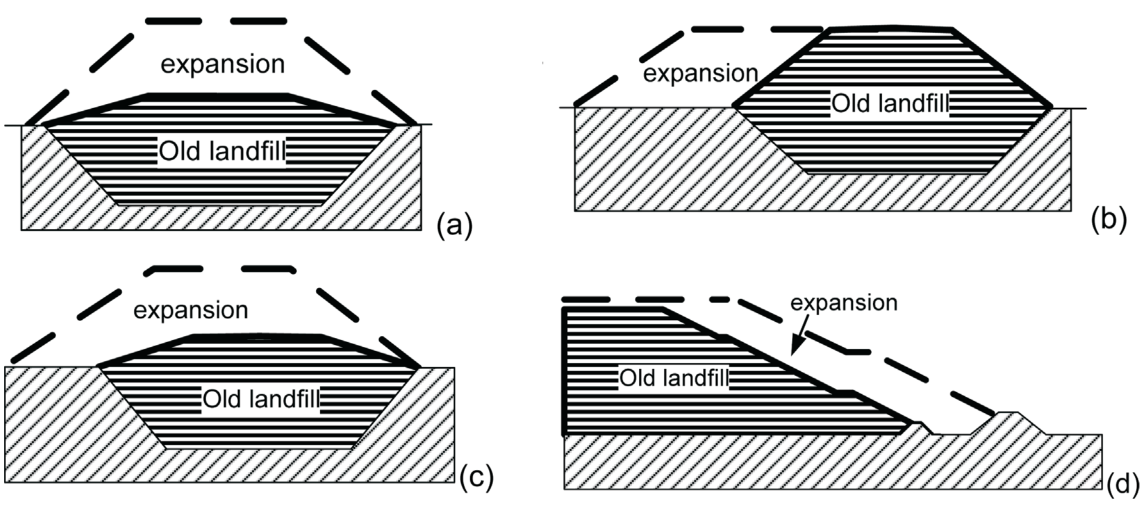


Figure 3. Geometric configurations in piggyback expansions: (a) vertical, (b) lateral, (c) mixed, and (d) veneer (Based on Qian *et al.*, 2001, Tano & Olivier, 2014, Bonaparte, 2018).

gas drainage from the old landfill and leachate drainage from the new one, and ensuring local and overall stability.

The least-desirable solution is the placement of the new landfill directly over the closed one, *i.e.*, without any new environmental protection layers. Bonaparte (2018) describes, in the 54th ASCE Karl Terzaghi Lecture, the forensic investigation carried out for a veneer piggyback sliding failure that occurred in 2011 at a MSW landfill located in the Eastern USA, designed by a third party. In this case, the slide was found to have occurred along the interface between the intermediate cover soil of the old landfill and the expansion, when the expansion achieved a height of 55 m supported on a lateral slope of the old landfill. The investigation also revealed that, at the time of failure, the waste placed in the expansion was very wet, due to leachate recirculation, introduction of municipal sludge, and rainfall. In addition, the intermediate cover soil layer of the old landfill was found to have low hydraulic conductivity, thus, causing leachate to accumulate in the expansion. With leachate accumulation in an excessively wet landfill, gas drainage efficiency was greatly reduced. Liquid accumulation and high pore-water and gas pressures were the main factors leading to the expansion failure; ultimately, failure was due to the lack of a leachate drainage system that should have been installed between the new and old landfills, after, at least, partial removal of the intermediate cover soil.

The design considerations are well presented in Qian *et al.* (2001), and have been adequately addressed decades ago, as shown in Tieman *et al.* (1990), who described the first piggyback extension (mixed configuration) in 1987 at the Blydenburgh landfill in New York state. The knowledgeable design already included environmental protection layers and geogrid reinforcement equivalent to the current paradigm (*e.g.*, Tano *et al.*, 2015) (Figs. 4a and b). The role of the geogrid is to limit the deformation of other components, such as drainage system and the geomembrane, amidst overall and differential settlements of the old cell. At Blydenburgh, for instance, the reinforcement was dimensioned for ensuring the integrity of the geomembrane liner under the conservative assumption of bridging a 2.4-m diameter cavity in the refuse beneath the expansion.

An additional concern with landfill expansions relates to the fact that often the new landfill is placed over an old unlined controlled dump. Contaminant hydrogeology studies are required to investigate the presence of underground contamination, as well as enable differentiating future contamination coming from the old or new landfills (*e.g.*, Brome-Missisquoi Landfill in Canada, Bouthot *et al.*, 2003).

2.2.2 Reinforcement of MSW

The concept of using high tensile strength, high stiffness geosynthetics, such as geogrids, to reinforce MSW slopes in landfills allowing higher and steeper MSW slopes appears natural given the accumulated experience with reinforced earth walls. Even though geogrids are used in veneer reinforcement of landfill cover soils, geogrids embedded directly in the MSW mass are not commonplace (Hettiarachchi & Ge, 2009).

In Brazil, landfills often receive high-organic content MSW, which may have lower shear strength than low organic MSW. Thus, reinforcement may help safely attain steeper slopes. A similar consideration applies to bioreactor landfills and landfills disposing shredded MSW, which exhibits lower interlocking, as a result of the shredding of the original MSW. Also, in the context of landfill mining, when re-landfilling the remaining waste after removing the usable MSW fractions, use of reinforcement may be interesting.

The embedment of geogrids in the waste mass requires consideration of durability issues, in particular the long-term environmental damage factor, which depends on the waste characteristics and the geosynthetic polymer. In

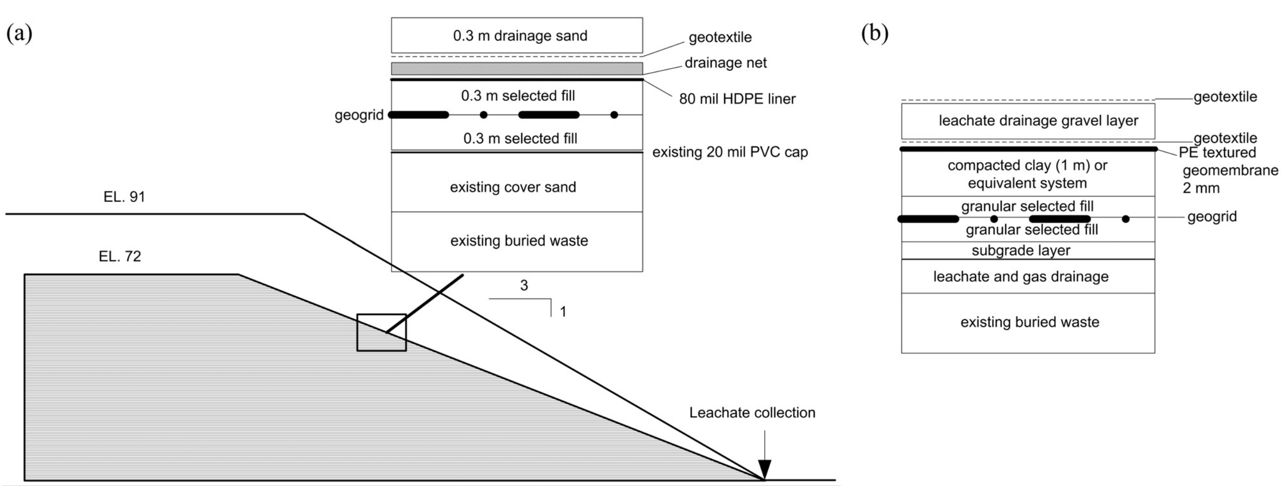


Figure 4. Geosynthetic and mineral layers to use for a piggyback expansion, (a) based on Tieman *et al.* (1990), and (b) based on Tano *et al.* (2015).

fresh MSW, temperatures may reach 50 to 70 °C, and leachate pH and chemicals may be aggressive. Other relevant concerns related to reinforcing MSW may include strain compatibility between geogrid and surrounding MSW, long-term interface strength amidst MSW degradation, effect of creep phenomena of both geogrid and MSW, mechanical damage during installation, and the aforementioned environmental concerns.

Carieri *et al.* (1999) describe one of the first times MSW reinforcement with geogrids was used, at the hillside landfill in Sesti Levante, Italy. The solution allowed the landfill storage capacity to be more than doubled, since the MSW slope increased, from gentler than 2H:1V to 1H:1V. The primary geogrids were horizontal layers spaced every 1.0 m in the vertical direction and with ultimate tensile strength of 400 kN/m (design strength = 157 kN/m). Lighter geogrids were placed near the face of the MSW slope. The selected geogrid materials were made of composite geosynthetic strips, with a core of high tenacity polyester (PET) tendons encased in a polyethylene (PE) sheath. The face of the reinforced-MSW slope was finished with a wrap-around method (Fig. 5), which included segments of 1-mm HDPE geomembrane.

Alexiew *et al.* (2015) discuss design concepts for reinforcing a 50-m rising of a hillside landfill receiving construction waste and IBA. The horizontal geogrids were considered for preventing a critical polygonal slip surface crossing the waste and emerging at the toe. The geogrids were made of high tenacity PET of ultimate tensile strength 1,600 kN/m at approximately 9 % strain.

Figure 5. Photograph from the hillside landfill in Sesti Levante, Italy (From Carieri *et al.*, 1999).

Ma *et al.* (2019) present an approach allowing a 20-m vertical expansion at Xingfeng landfill in China (Fig. 6). The approach consisted of reinforcing an entire existing critical MSW slope before the expansion. The reinforcement was based on 33-m long HDPE geogrids at a vertical spacing of 1.0 m from the toe to the top of the slope. Thus, the project involved excavating the old MSW, and re-constructing the slope with the geogrid inclusions. At the face of the slope, geogrids were wrapped around geotextile gravel bags.

Boscov *et al.* (2020) performed limit-equilibrium and stress-strain analyses to verify the possibility of raising a landfill geometry based on reinforced soil dikes built in successive steps and geogrid reinforcement inside the MSW mass (Fig. 7), with the total landfill height reaching 48 m, after a number of successive stages of waste placement, each stage with 6.0 m in height. The soil dikes had a crest width of 5.0 m, slopes of 1H:1V, and were reinforced with geogrids; the mean slope of the landfill resulted equal to 1.8H:1V. The waste mass was reinforced with geogrids every 6.0 m (vertical distance), *i.e.*, every construction stage.

The analyses were performed considering adopted soil and MSW parameters, both for limit-equilibrium and stress-strain analyses. Also, ranges for the pore pressures due to leachate and gas generation within the MSW were varied. The geogrids in the soil dikes were assumed to have a tensile strength of 100 kN/m. The geogrids in the MSW were assumed to have 400 kN/m of maximum tensile strength, placed every 6.0 m, and anchored at both ends. Considering a safety factor of 1.5 and the geometry and parameters adopted in the study, the use of geogrid reinforcement allowed the landfill height to be increased from 10-15 m to 30-45 m, these ranges depending on the pore-pressure ratios, r_u, considered. The concept of reinforcing the MSW with geogrids in landfills may be considered relevant in the future, due to the need to increase capacity. However, impact on the operation of the landfill is expected. Not only technical, but also operational aspects must be taken into consideration, such as interference of the geogrids on the geometry of the landfill cells.

3. Site remediation

3.1 Overview

Remediation is generally defined as the process of restoring land that has been contaminated. Shackelford & Jefferis (2000) point out that, although the words ‘remediation’ and ‘reclamation’ often are used interchangeably in terms of environmental contamination, arguably the words have slightly different meanings: the goal of reclamation may be inferred as reuse of the land, whereas the

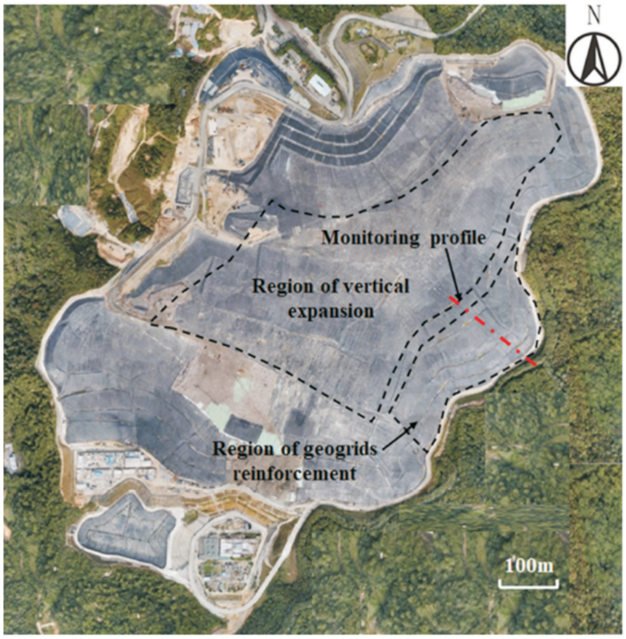


Figure 6. Region of vertical expansion and reinforced slope at Xingfeng landfill (From Ma *et al.*, 2019).

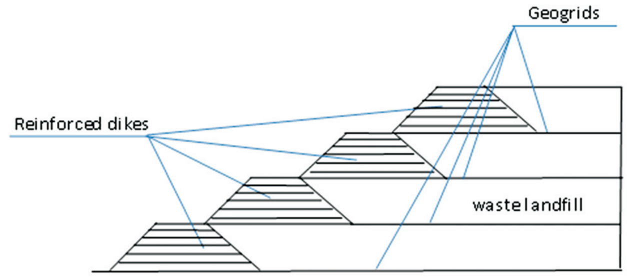


Figure 7. First four stages of construction in the reinforced landfill configuration proposed in Boscov *et al.* (2020).

goal of remediation may be inferred as a process to prevent or minimize a real or perceived risk of harm to humans. However, there are many situations where reclamation involves remediation, and remediation is often related to new uses of the land.

Site remediation engineering knowingly must be based on a sound site conceptual model, which includes characterization of pollutants, source zones, spatial distributions and phases (solids, water, gas) involved, hydrogeological and geochemical characterization of the physical medium, flow and transport modeling, and risk analyses defining pollutant target concentrations. Not so conspicuous, on the other hand, is the need for geotechnical characterization of the site, which must be added to all this knowledge, to properly select and dimension the remediation system. Site monitoring, finally, allows adjustments to be made to remediation operation, as well as site closure.

3.1.1 Management of contaminated sites (CETESB)

The Environmental Agency of São Paulo state (CETESB), which is a reference for the whole country, classifies registered contaminated sites, according to Decree 59,263/2013, as contaminated site under investigation, contaminated site with confirmed risk, contaminated site under remediation, contaminated site under process of reutilization (new use for the site after elimination or reduction of risk to acceptable levels), site under monitoring for closure (site where risk was not confirmed, or site where remediation targets were achieved but are still under monitoring to verify maintenance of concentrations at acceptable levels), and site rehabilitated for declared use. Last year, the number of rehabilitated sites (1,775) increased remarkably (23 %) as compared to 2018 (1,441) (Fig. 8). Adding the sites under monitoring for closure (1,375), half of the registered sites are no longer classified as contaminated (Table 1).

Considering sites under remediation and sites where remediation was completed (3,710), the mostly employed remediation techniques for the treatment of subterranean water (saturated zone) were multiphase extraction, pump-and-treat and free phase recovery, while removal (excavation) and vapor extraction were mostly used for soils (unsaturated zone), as shown in Fig. 9.

In addition, among the rehabilitated sites, a total of 942 sites are being reused, or reutilization is planned. This information is relevant to show the trend of changing the use of industrial sites, now usually destined to the construction of commercial and residential real estate developments, or even construction of parks and leisure areas. This trend is bringing forth the revitalization of former industrial areas, mainly in the metropolitan region of São Paulo. Decree 59,263/2013, that regulated Law 13,577/2009, established that reutilization of rehabilitated sites, as well as the revitalization of regions, must be encouraged by government.

The main groups of contaminants in the registered sites reflect the influence of the activity of distribution of automotive fuels: aromatic solvents (benzene, toluene, ethylbenzene and xylene), Polycyclic Aromatic Hydrocarbons (PAHs) and total petroleum hydrocarbons (TPH). Following, metals and halogenated organic compounds are also frequently found, according to Fig. 10.

Automotive fuels and halogenated organic compounds are scarcely water-miscible liquids, or non-aqueous phase liquids (NAPLs). The higher solubility compounds, also toxic, and often carcinogenic (*e.g.*, in automotive fuels: benzene, toluene, ethylbenzene, xylene, naphthalene) form a considerable groundwater plume that can migrate in the direction of groundwater flow. Halogenated organic compounds are some of the most recalcitrant pollutants in sites and present low to moderate solubilities, high volatilities, low to moderate soil partition coefficients, high mobility, and densities greater than water.

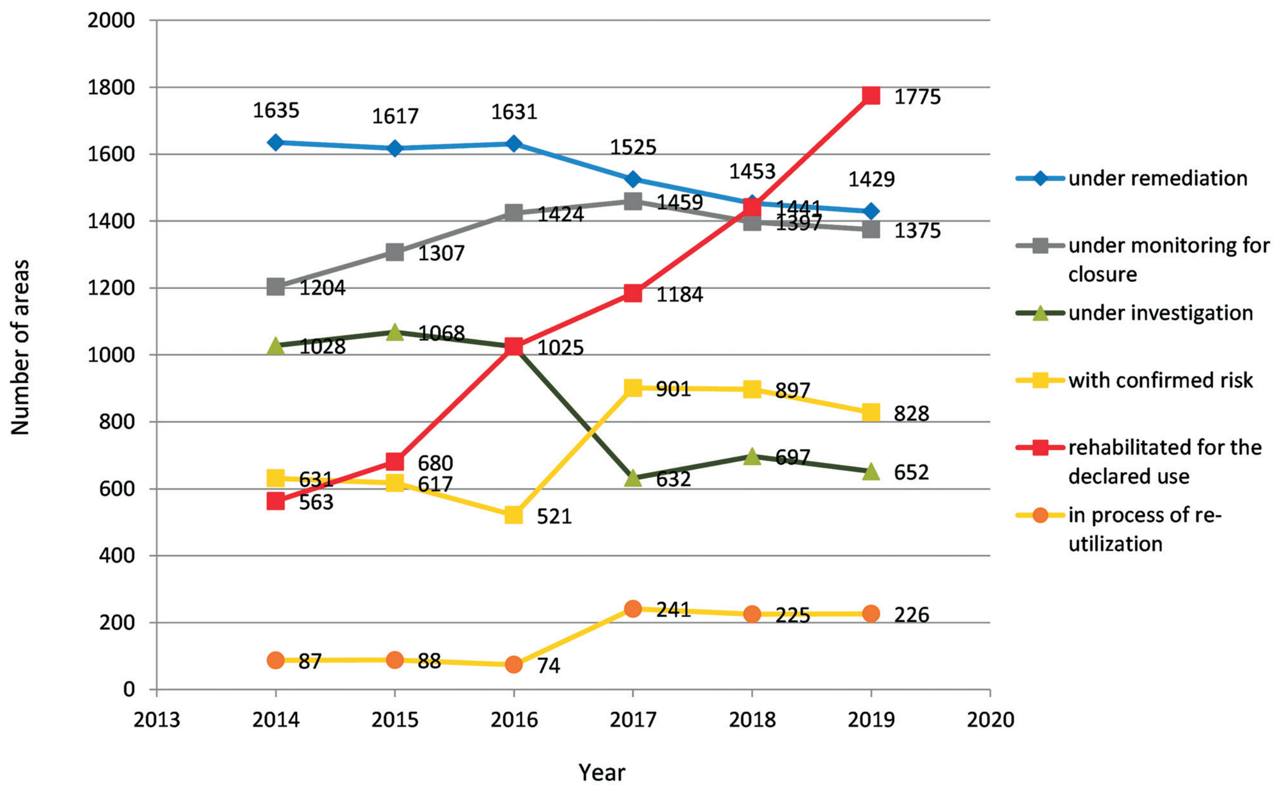


Figure 8. Time evolution of registered contaminated sites in São Paulo state by category (Based on CETESB, 2019).

3.1.2 Geotechnical confinement

Interestingly, in Brazil, differently from many other countries (*e.g.*, USA, Canada, Japan and EU countries), geotechnical confinement is practically never used as a rehabilitation solution. Geotechnical confinement may be achieved by impermeable vertical barriers (trenches or diaphragm walls with different fillings such as soil-bentonite and cement-bentonite, geomembrane panels, jet-grouting, sheet pile curtains), impermeable covers and, in some cases, also a bottom impermeabilization. The goal is to isolate the contaminated soil or buried waste, avoiding release of contaminants to the environment and contact with living beings. The solution is acceptable when the extension or volume of soil to be treated is very large, when there is a mixture of different pollutants that would require an association of different remediation techniques (this is not rare in remediation projects, but there is a practical limitation to the number of concurrent techniques in the field), or the current available techniques are still not efficient for the pollutants found at the site.

The risk is minimized by limiting the release of contaminants (liquids and/or gases) to groundwater, surrounding subsoil or the atmosphere to acceptable levels. However, there still is in Brazil the perception that pollution is being "buried" or "hidden from the public". Also, passive reactive barriers are seldom used (excavated permeable curtains through which groundwater flows and is treated by the filling material). There is still much more reliance on pump-and-treat or injection of reagents, even when these techniques are inadequate for subsoils with low permeability, preferential flow channeling and specific adsorption of contaminants, which are not uncommon in tropical subsoil profiles. These techniques may not deliver the reagent to the desired targets or demand a long time for remediation, and may not be environmentally sustainable when operational (*e.g.*, energy for pumping or injecting) and social (hindrance of use of the area) costs are taken into consideration.

Table 1. Occurrence numbers and percentages by category of registered contaminated sites in São Paulo state (CETESB, 2019).

Classification	Number of areas	Percentage (%)
Area under investigation	652	10
Contaminated area with confirmed risk	828	13
Contaminated area under remediation	1,429	23
Contaminated area under monitoring for closure	1,375	22
Contaminated area rehabilitated for declared use	1,775	28
Contaminated area under process of reutilization	226	4

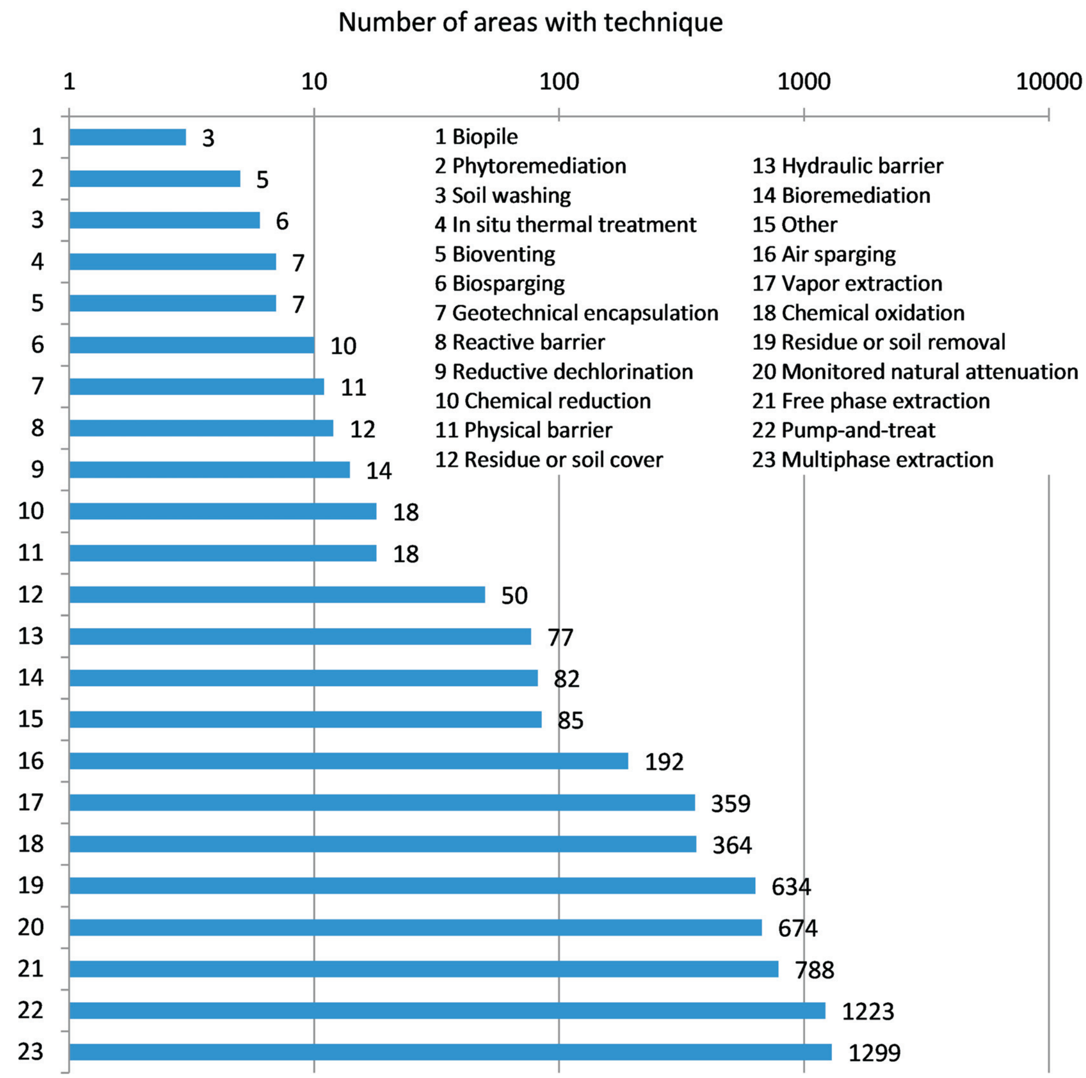


Figure 9. Occurrence (number of implementations) of each remediation technique (Based on CETESB, 2019).

Remediation time can be long and usually measured in decades (Stroo & Ward, 2010). This finding should stimulate the use and improvement of confinement techniques and monitored natural attenuation. While the former has not been internalized as a trustworthy rehabilitation technique by Brazilian professionals, the latter has been increasingly used (674 out of 3,710 sites, Fig. 9). Conferences on Environmental Geotechnics usually bring new research and practical aspects of geotechnical confinement, while CETESB's list of contaminated sites shows only 11 cases of geotechnical confinement out of 3,710 treated areas.

Another particularity of remediation of contaminated sites in Brazil, which probably helps understand the aforementioned trends, is that remediation design often underutilizes Geotechnical knowledge on local soils. Also, improvement or development of techniques for the unsaturated zone, which may be thick in tropical climates and retain a significant portion of the contamination, is very restricted except for gas-phase pollutants.

Technical developments are necessary and constantly under way in the area of remediation, although other aspects also have to be addressed, such as public perception (as mentioned) and problems related to complex urban areas, as will be exemplified by the case study.

3.2 Complex urban areas

Contaminated areas under investigation and remediation become more complex when located in urban regions with industrial past and recent change of land use. The conceptual model for site contamination must consider the regional scale, rather than be restricted to the study area. However, the involvement of all stakeholders for a joint regional plan for investigation and remediation is rarely brought about by the environmental agencies.

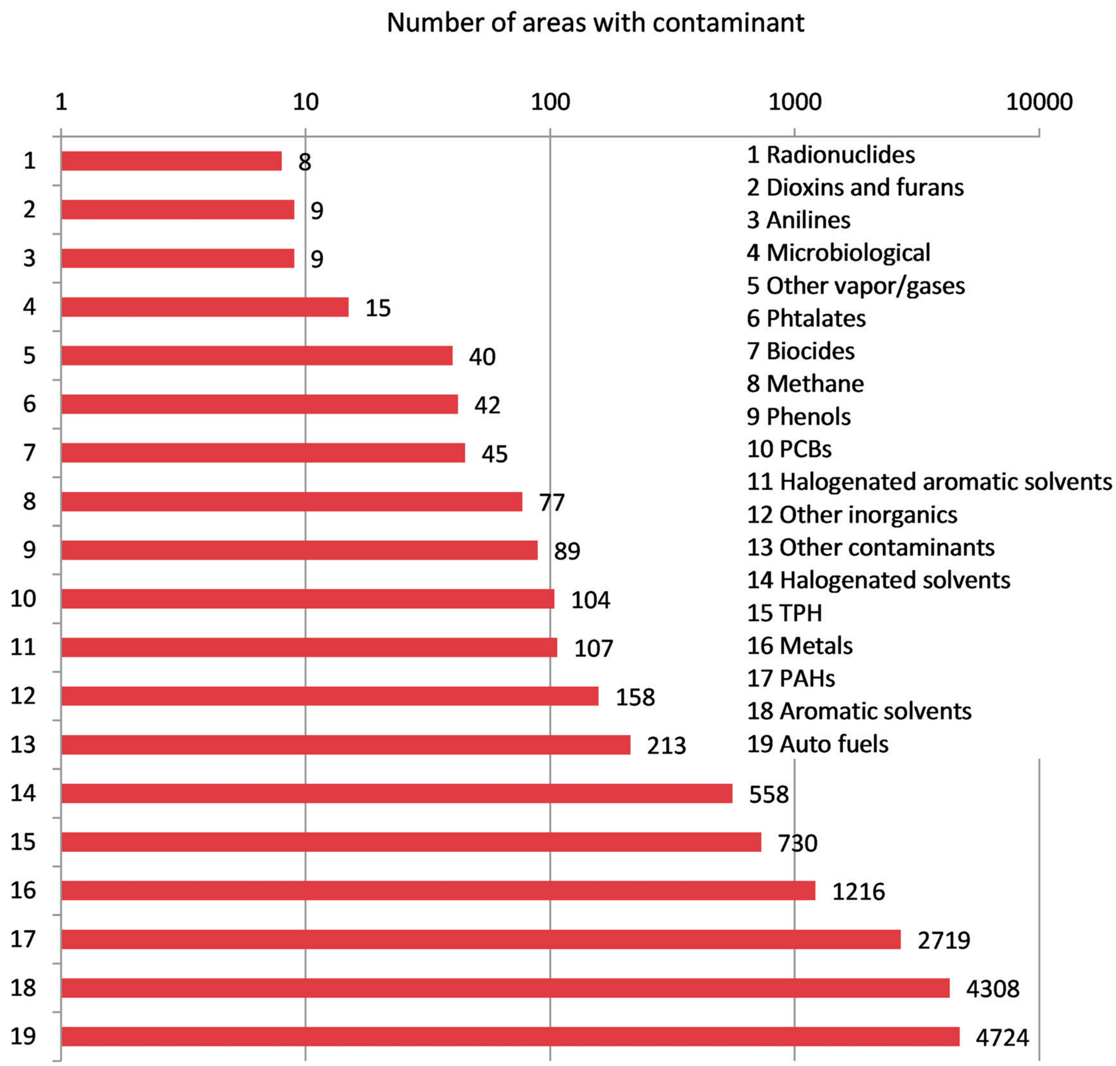


Figure 10. Occurrence of contaminant groups in contaminated areas (Based on CETESB, 2019).

3.2.1 Industrial site case study in São Paulo (Caram, 2019; Caram & Boscov, 2019)

The case study refers to an old industrial area in the north part of the city of São Paulo, where an automobile industry operated in the 1950s and 1960s (before the existence of CETESB, created in 1968), and a construction deposit from 2001 to 2008. The area is located in a district historically marked by hosting a number of heavy industries.

At the site, the main groundwater contaminant was VC (vinyl chloride). Collected soil samples showed that halogenated organic compounds were not present in the soil matrix. During excavation works, two masonry oil tanks were discovered near the northwest boundary of the area. The tanks were removed in 2009, and the whole area was covered with a compacted-soil layer. Subsequently, investigation of groundwater plumes started. Hot spots detected upstream of the area could not be related to the oil tanks. Since the CETESB process began, five study campaigns were carried out, the area has been investigated with 86 monitoring wells reaching different depths (4 to 30 m), however still the plumes could not be totally defined.

The region is located over the Tertiary sediments of the São Paulo Basin and modern alluvial deposits. The stratigraphic profile indicated a top layer of 3 to 5 m of a clayey fill overlying alternating plastic clay, sandy clay and fine sand layers down to the depth of 15 m. The water table was found at a depth of 4 m. Groundwater flow directions are mainly northwest, north and northeast in shallow (6-m deep) and intermediate (9-m deep) monitoring wells. The potentiometric levels in the area agree with the regional groundwater flow pattern, which was oriented to northwest, discharging into River Tamanduateí. Slug tests, based on U.S. EPA standard, were performed in seven multilevel monitoring wells, to yield hydraulic conductivity values. The geometric means of conductivity values were 6.6×10^{-5} m/s (shallow range), 1.2×10^{-3} m/s (intermediate range, from 7 to 9 m) and 2.6×10^{-5} m/s (deep range). Thus, the more conductive layer was located between 7 and 9 m.

3.2.1.1 The contaminant

Vinyl Chloride (VC) poses high human toxicity and is known to be a human carcinogen. VC does not occur naturally, and anthropogenic sources are related to PVC production or formation by degradation of organochlorides (WHO, 1999). In this case, there was no nearby production of PVC. Under anaerobic conditions, VC is formed by the reduction of chloroethylenes - PCE, TCE and dichloroethylene isomers (cis-1,2-DCE, trans-1,2-DCE and 1,1-DCE) - and under aerobic conditions by a direct or co-metabolic oxidation of DCE. Since PCE and TCE are the chlorinated solvents used in industry, and VC is a product of the slow natural degradation of PCE and TCE, the predominance of VC at the site indicates that the contamination is old. VC can be released to the environment through air, water or soil, however VC is most commonly found in air and groundwater. VC solubility in water is relatively low but can be raised by the presence of salts. When released to air, VC is expected to exist almost exclusively in the vapor phase, but VC half-life in air is limited by reaction with OH radicals photochemically produced (WHO, 1999). Volatilization is a significant transport mechanism and the risk assessment indicated vapor inhalation as the major exposure pathway.

3.2.1.2 Remediation technique

The remediation system adopted for the area was installed in early 2015 and operated continuously for 2.5 years. The system comprised 31 vapor extraction wells (in the vadose zone, above the water table) and 33 ozone injection wells in the saturated zone, in a technique known as ozone sparging. Also, several monitoring wells were included for control.

Ozone sparging attacks VC through oxidation and volatilization, and the vapor extraction system recuperates the volatized contaminant mass. Ozone can also dissolve in the aqueous phase and react with the organic compounds in water (Henry & Warner, 2002). Therefore, ozone would also oxidize VC in the dissolved phase.

The selection of ozone sparging for remediating the VC plume may be justified by the combined effects of gas-phase extraction and *in situ* oxidation by ozone. In particular, vapor extraction is applicable when the contaminant has vapor pressure higher than 1.0 mmHg (20 °C) and Henry's law constant higher than 0.001 atm × m^3/mol. As revealed by the parameters for VC in Table 2, favorable strippability and volatility are expected. In terms of oxidation-reduction state, VC is the most reduced compound amongst the chlorinated compounds, thus prone to oxidation. Also, VC has a low adsorption coefficient, indicating a small tendency to remain retained in the soil. Comparing the parameters for VC with the general guidelines in Table 3 (U.S. EPA), the contaminant may be considered very weakly sorbed (water-soil organic carbon partitioning coefficient, $K_{oc} < 10$), with high mobility in the aqueous phase, and high volatility (Table 3).

Ideal conditions for the application of gas sparging in the field occur when the soil layer is a homogenous coarse-grained material, with a saturated hydraulic conductivity on the order of 10^{-5} m/s. The injection of gas beneath the water table inevitably causes mounding of the phreatic level and may laterally spread contaminated groundwater. Complex hydrogeologic and contaminant-distribution settings may be challenging; the occurrence of low-permeability clay lenses, or very high permeability layers, above the point of gas injection may further spread the contamination plume. The subsoil heterogeneity causes preferential gas flow, such that the contaminant outside the preferential flow is poorly exposed to the reagent gas. Air channeling may occur, short circuiting the path of gas between the injection point and a monitoring well, as shown in Fig. 11.

In addition, as for any contaminant-extraction technique, gas sparging is challenged by the existence of contaminant mass stored in the free phase and in immobile compartments, such as the residual pure liquid phase, the adsorbed phase and contaminant diffused into low-permeability layers, prone to reverse matrix diffusion. The complexity of mass transfer among vapor, aqueous, free (NAPL) and sorbed phases in a subsoil composed of transmissive and low permeability zones has been discussed by Vanderkooy *et al.* (2014), which presented a compartment model of mass transfer of organochlorides.

3.2.1.3 Monitoring and plumes

Several monitoring wells reaching different depths were used for the investigation of contaminant levels in groundwater and for measuring geochemical parameters. The physical-chemical parameters indicate whether the oxidant was reaching the subsoil layers in the whole area, while VC concentrations showed whether ozone degraded VC in the subsoil. Before the remediation, the following

Table 2. Phase-partitioning and other parameters for vinyl chloride (Based on Suthersam, 1999; CETESB, 2016).

	Molecular weight (g/mol)	Henry's law constant (atm × m^3/mol)	Vapor pressure (mmHg)	Solubility (mg/L) (25 °C)	K_{oc} (mL/g)	U.S. MCL /CETESB 2016 (mg/L)
VC	62.5	2.78	2,660 (25 °C)	1,100	2.5	0.002/0.002

K_{oc} = organic carbon partitioning coefficient; MCL = maximum contaminant level.

Table 3. Physical-chemical parameter ranges and classification according to sorption, mobility and volatility for organic compounds (U.S. EPA, 1997).

Property	Range	Description
Sorption Soil adsorption coefficient, K_{oc} (mL/g)	< 10	Very weakly sorbed
	10-100	Weakly sorbed
	100-1,000	Moderately sorbed
	1,000-10,000	Moderately to strongly sorbed
	10,000-100,000	Strongly sorbed
	> 100,000	Very strongly sorbed
Mobility Based on a combination of solubility (S) (mg/L) and soil adsorption (K_{oc})	$S > 3{,}500$ and $K_{oc} < 50$	Very high mobility
	$850 < S < 3{,}500$, and $50 < K_{oc} < 500$	High mobility
	$150 < S < 850$, and $150 < K_{oc} < 2{,}000$	Moderate mobility
	$15 < S < 150$, and $500 < K_{oc} < 20{,}000$	Low mobility
	$0.2 < S < 15$, and $2{,}000 < K_{oc} < 20{,}000$	Slight mobility
	$S < 0.2$, and $K_{oc} > 20{,}000$	Immobile
Volatility Henry's law constant (H), atm m^3/mol	$H < 3 \times 10^{-7}$	Nonvolatile
	$3 \times 10^{-7} < H < 10^{-5}$	Low volatility
	$10^{-6} < H < 10^{-3}$	Moderate volatility
	$H > 10^{-3}$	High volatility

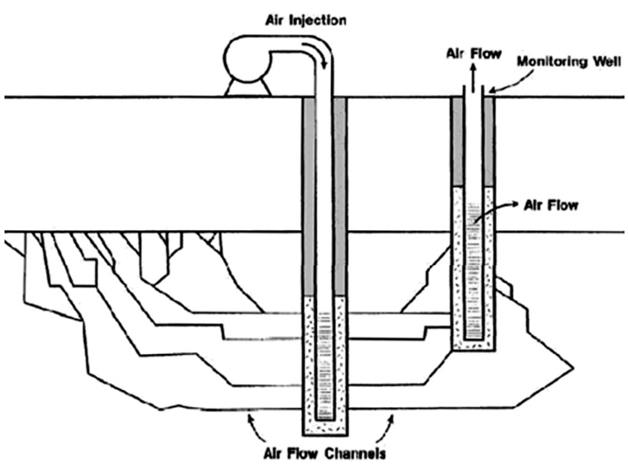


Figure 11. Illustration of gas channeling to a monitoring well (U.S. EPA, 1997).

were the average geochemical parameters for the intermediate level: *ORP* (oxidation/reduction potential) = -110 mV, *DO* (dissolved oxygen) = 0.48 mg/L, and *EC* (electrical conductivity) = 577 µS/cm. After 16 months of remediation, the parameters were measured, respectively, as 103.5 mV, 0.63 mg/L and 326.4 µS/cm. The variation of parameters along time and the final values indicate that the oxidant (ozone) reached subsurface in desired depths, since there was a general increase in *ORP* and *DO*, and a reduction in *EC* in the groundwater. As expected, the intermediate level presented the highest VC concentrations, which can be explained based on hydraulic conductivity.

The VC plumes obtained before remediation and for the five campaigns at the intermediate level are shown in Fig. 12a to f. Also, Fig. 13 presents the dissolved-phase mass of VC, calculated based on the monitored plumes, as a function of the monitoring time.

Figure 12 shows an initial decrease in VC concentrations and plume width (Aug. and Nov. 2015), probably in response to remediation, followed by a substantial increase in VC concentrations (Feb. and May 2016) with new hotspots, and again a decrease of VC concentrations with time (Aug. 2016). However, there was no guarantee that concentrations would not rise again. Additional VC mass, originated from the upstream area or VC transfer among subsoil phases, apparently is being carried by water flow in the downstream direction. Before remediation, the VC plume showed high concentrations (*i.e.*, average of 5.3 mg/L, maximum of 37.3 mg/L) and was located near the SE border and outside the study area. The location and behavior of the plume raised important concerns. Large portions of the VC plume located outside the area of interest, and the contaminant possibly migrating from upstream adjacent areas, indicated the need to study groundwater contamination at the regional scale. For instance, the original industrial plant may have been divided, such that the location of the source is outside the study area.

3.2.1.4 Discussion

The results from the case study bring some points to be considered:

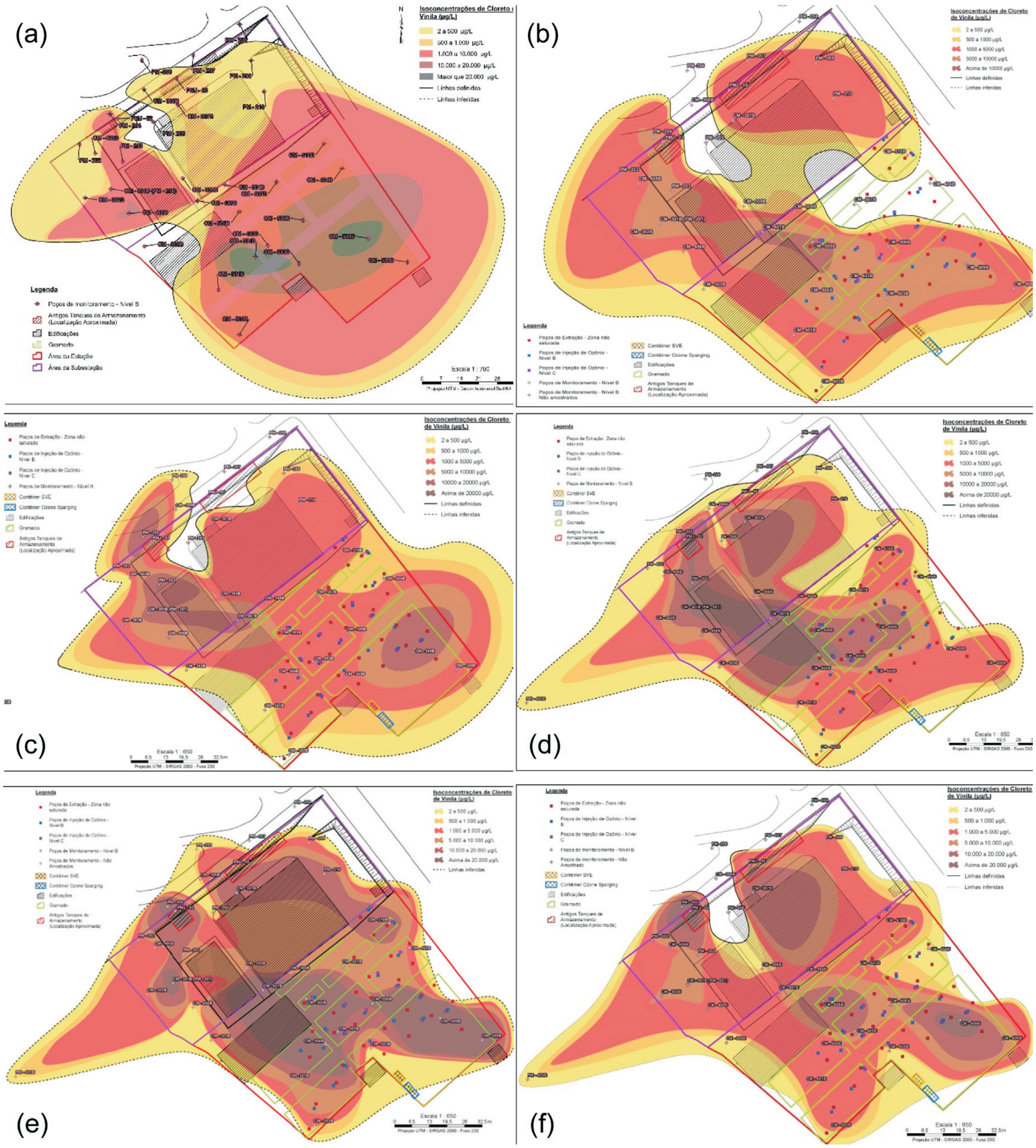


Figure 12. VC plumes at the intermediate level wells: a) 2014; b) Aug/2015; c) Nov/2015; d) Feb/2016; e) May/2016; and f) Aug/2016 (Caram & Boscov, 2019).

(1) The physical-chemical parameters at the control points indicated that ozone reached the depths where the contaminant was present throughout the area, however VC concentrations did not decrease effectively along time. Two explanations are more likely: continuous contribution of upstream contamination and mass transfer between phases. Ozone sparging can volatilize VC dissolved in the pore water and induce back diffusion from the low permeability layers, so that additional VC mass is brought to the dissolved plume. The conceptual site model did not primarily consider the importance of the presence of alternating soil layers with different hydraulic conductivities and should be reviewed. Unintentional VC mass transfer from groundwater to the vadose zone by the ozone sparging into the subsurface should also be considered (Chong & Mayer, 2017). VC concentrations in the vapor phase in the vadose zone should also be investigated. VC contribution from neighboring areas is also a plausible hypothesis. The primary sources may be located upstream, external to the studied area, but still feeding and contributing to the dissolved phase plumes.

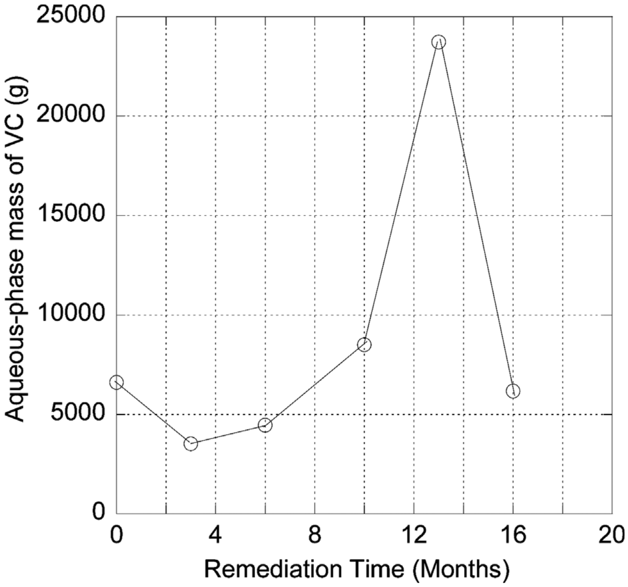


Figure 13. VC dissolved mass *vs.* time (Based on Caram & Boscov, 2019).

(2) The results and consideration lead to the conclusion that a joint regional plan for investigation and remediation is essential for urban complex regions with past industrial land use and recent change of land use. The difficulties to assess a conceptual model for site contamination based on a restricted part of the potentially contaminated region often result in long-term disputes or legal action between relevant stakeholders. Remediation/containment actions at individual areas can be expensive and long-term, representing a cost to society in energy, inputs, and land use itself.

(3) The risk assessment is based on receptors. In this case, similarly to other central and densely built urban areas with complete sanitary infrastructure, the contact of living beings with subterranean water is not a hypothesis. However, contaminated groundwater not susceptible to reach or be used by human beings may still contaminate water bodies. The very polluted and unusable rivers in the city of São Paulo are expected to undergo clean-up and rehabilitation in the coming decade, therefore remediation targets will have to be reviewed. This calls for remediation techniques that confine contamination or treat contamination in a new scenario.

(4) The three former points would benefit from Geotechnical expertise that should be ever more used in this field. An example of alternative or complementary measures in this case study follows to make this point.

3.2.2 Other possible techniques at the industrial site

The location of the original plume relative to the groundwater flow pattern indicated that contamination was likely to come from outside the study area. Also, monitoring results, *i.e.*, dissolved-phase VC concentrations *vs.* time, indicated concentration increase with time since the start of remediation.

An important measure to be implemented at the site is the construction of containment barriers to isolate the area from the inflow of pollutants from upstream neighbors. A viable option would be to build a soil-bentonite cut-off wall along the southeast boundary, also extending to the sides, provided that local soils are adequate for backfilling. As shown in Fig. 14, this classic cut-off wall is built by excavating a trench with a backhoe (maximum depth ~ 10 m) or clamshell (maximum depth ~30 m), using bentonite or polymer slurry for temporary support and to form a filter cake, and backfilling the trench with a mixture of local soils and bentonite (~3-5 % dry weight). Soil-cement, soil-bentonite or soil-cement-bentonite mixtures are possible backfilling alternatives. An important consideration is the resulting hydraulic conductivity of the soil-bentonite back-

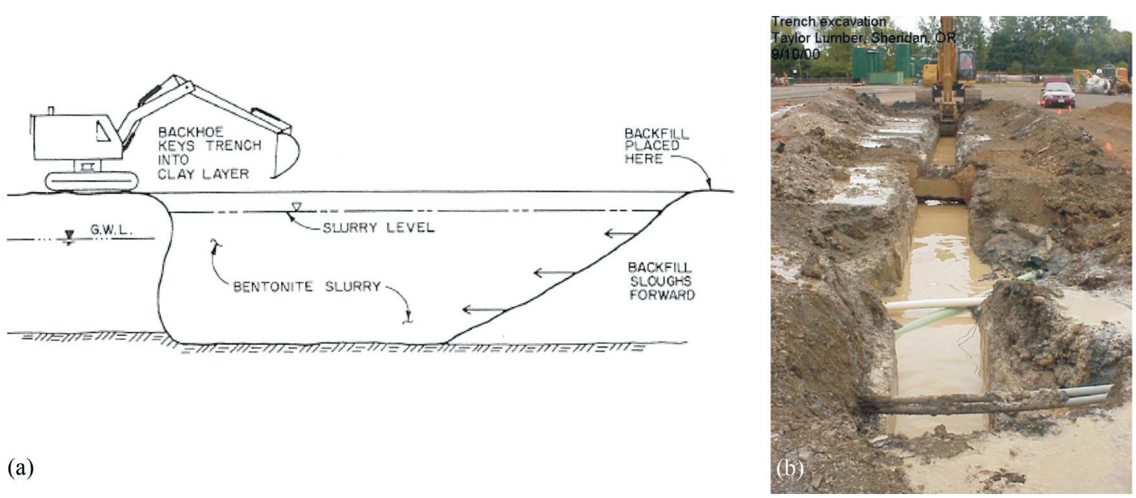


Figure 14. Soil-bentonite cut-off wall, (a) construction method schematic (Ryan, 1987), and (b) photograph during construction (From McKnight & Owaidat, 2001).

fill, which must be lower than 1×10^{-9} m/s, requiring the testing of mixtures of the available soils with different dosages of bentonite, to choose an adequate backfill, as shown in the classical work by D'Appolonia (1980), and described in Benson & Dwyer (2006). This is a low-permeability curtain to physically block the inflow of contaminated groundwater into the site.

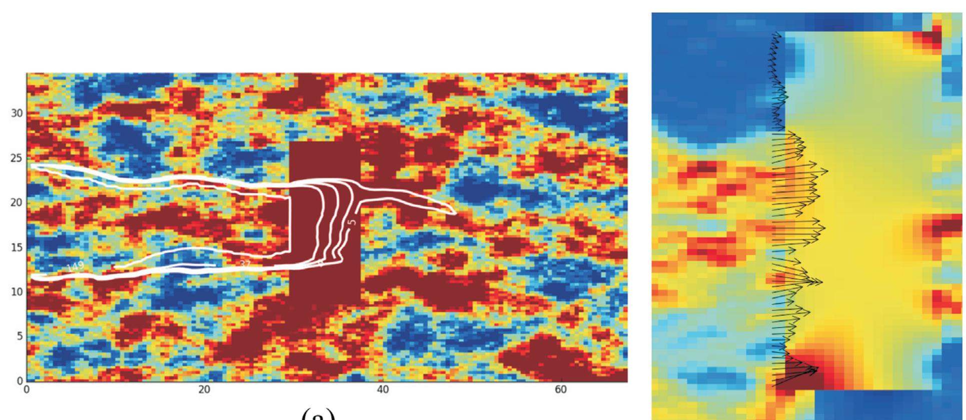
The performance of the soil-bentonite barrier is due to the high swelling ability of the bentonite in water, which results in low hydraulic conductivity. Saline solutions and organic compounds cause a permeability increase due to chemical incompatibility with bentonite, as extensively studied for geosynthetic clay liners (*e.g.*, Shackelford *et al.*, 2000). Different polymer modified bentonites, multiswellable bentonite and HYPER clay have been developed to improve the chemical resistance in aggressive environments (*e.g.*, De Camillis *et al.*, 2019).

Another possibility would be to build one or more permeable reactive barriers (PRBs) in the area, to intercept the outside contaminant inflow, as well as promote chemical and/or biological destruction of contaminants within the site. PRBs are critically affected by (1) hydraulic performance (contaminants are routed through the reactive medium with an appropriate residence time, and should not bypass the medium), and (2) chemical/biological performance (contaminants are involved in reactions when in contact with the medium, and concentration goals must be achieved downgradient from the barrier) (Naidu & Birke, 2015). The classical continuous-trench PRB is built by the supported excavation of a trench (without bentonite), filled with a mixture of gravel, sand and reactive materials (Fig. 15).

The performance of a continuous-trench PRB may be significantly affected by flow channeling due to aquifer heterogeneity and complexity in the hydraulic conductivity (k) structure of the medium (*e.g.*, Hemsi & Shackelford, 2006). Preferential pathways of flow and contaminant transport expose the PRB to spatially variable groundwater seepage velocities (v). Where contaminant residence times are shorter, the PRB effluent concentrations may locally surpass the prescribed limit. Results from numerical modeling of reactive multi-component transport in heterogeneous aquifers generated with geostatistical methods are exemplified in Fig. 16.

Figure 15. Filling of a supported trench with a mixture containing ZVI (zero-valent iron) for building a continuous-trench PRB (From ITRC, 2011).

A classical alternative to the continuous-trench configuration is the so-called funnel-and-gate. The groundwater flow is directed to the permeable reactive "gate" by insertion of impermeable barriers in the subsoil (Gavaskar *et al.*, 1998, Naidu & Birke, 2015). An interesting case of a funnel-and-gate PRB in Brazil to treat mercury contaminated water was designed by Nobre *et al.* (2006), presented in Fig. 17.



(a)

(b)

Figure 16. Plan views of (a) contaminant transport through a PRB with a formed effluent plume (5 mg/L) due to flow channeling (red: high k and blue: low k), and (b) seepage velocities map and vector representation of seepage velocities at the influent side of a PRB (red: high v and blue: low v) (Machado & Hemsi, 2016).

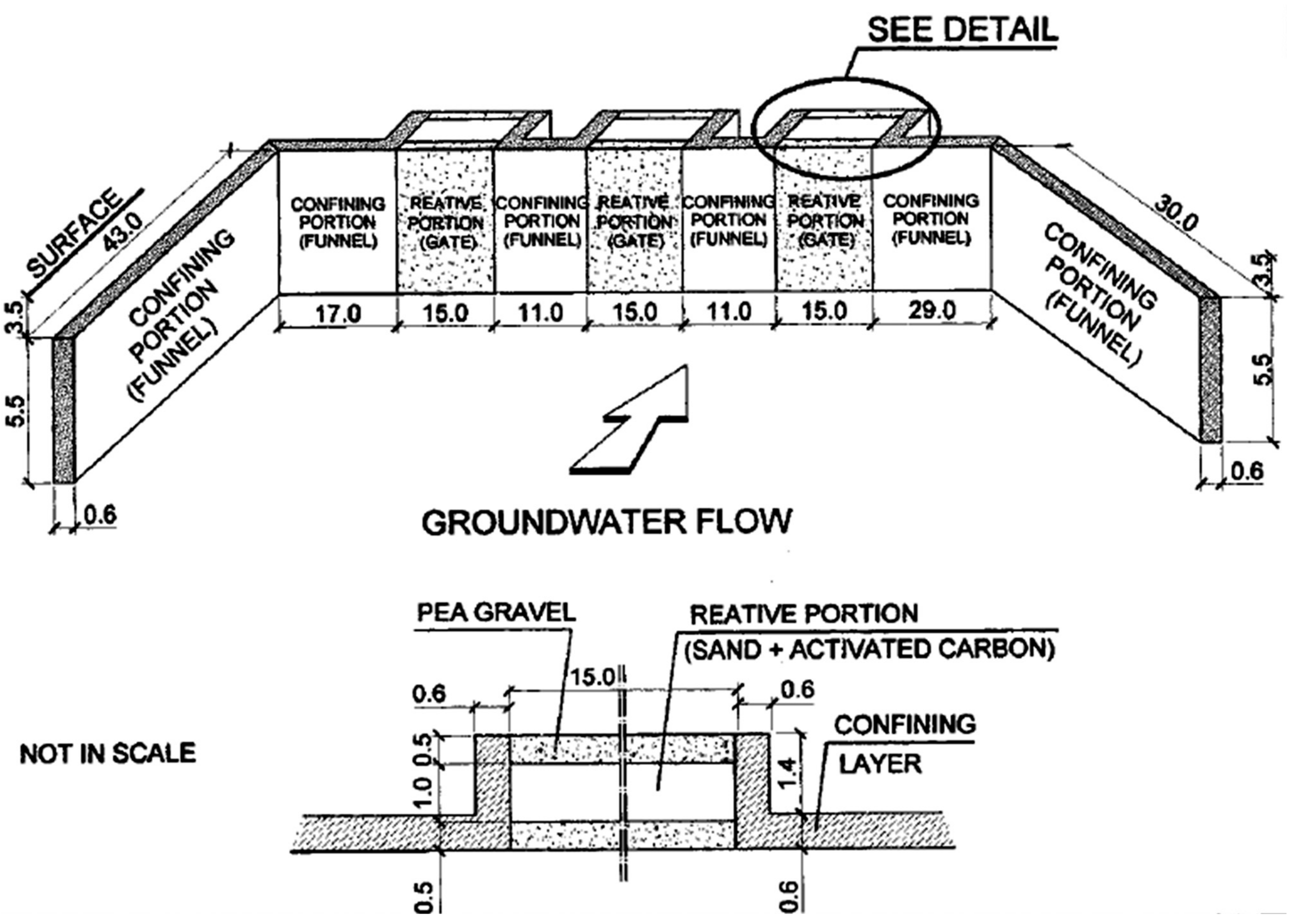


Figure 17. Funnel-and-gate PRB to treat mercury contaminated groundwater (Nobre *et al.*, 2006).

Several classes of reactive materials have been tested, as well as used in full-scale implementations, aiming at different groundwater contaminants (Table 4). From the first PRBs used for dechlorination of halogenated compounds by zero-valent iron (ZVI) (Gillham & O'Hannesin, 1994, Di Molfetta & Sethi, 2003), there has been significant innovation in the reactive materials used, including biobarriers, combination of organic materials and ZVI, and nano-scale ZVI, among others. Biobarriers contain organic materials as the major reactive component. Several organic materials have been tested, both for organic contaminant compounds and metals in acid mine drainage. For example, Mattos *et al.* (2014) and Trindade *et al.* (2018) performed tests on the use of sugarcane bagasse for removing metals and sulfate from synthetic acid mine drainage solutions. Trindade *et al.* (2018) performed column tests (triplicate) for precipitating nickel and zinc under the anaerobic (sulfate reducing) conditions that may occur in an organic PRB. The organic reactive medium used was sugarcane bagasse. The results indicated satisfactory rates of sulfate reduction and metals precipitation. Assumpção *et al.* (2020) performed batch-equilibrium tests to remove dissolved nickel using a biogenic-apatite char. The results for the fine-grained char were very satisfactory and suggested the Ni removal mechanism to be Ni substitution for Ca in the structure of the hydroxyapatite.

Alterative configurations include vertical-flow reactors that can be filled with different reactive media, such as adsorbents, organic materials, ZVI, etc. Such configuration was used in two PRBs in the UK. The Belfast PRB described by Birke *et al.* (2007) used a reactor consisting of a 12-m height by 1.2-m diameter steel reactor filled with ZVI (Fig. 18). Cox *et al.* (2009) describe the funnel-and-gate

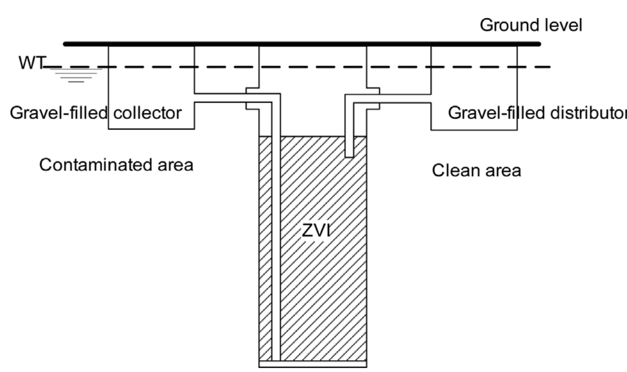


Figure 18. Passive treatment achieved by groundwater flow through steel reactor filled with ZVI (Birke e*t al.*, 2007).

Table 4. Classes of contaminants treated and types of reactive media used in PRBs. Symbols: F: full-scale application, L: laboratory evaluation, and P: pilot-scale application (From ITRC, 2011).

Contaminant	ZVI	Biobarriers	Apatite	Zeolite	Slag	ZVI-carbon combinations	Organophilic clay
Chlorinated ethenes, ethanes	F	F			L	F	
Chlorinated methanes, propanes						F	
Chlorinated pesticides						P	
Freons						L	
Nitrobenzene	P						
Benzene, toluene, ethylbenzene, xylenes (BTEX)		F					
Polycyclic aromatic hydrocarbons (PAH)							L
Energetics	P	F				P	
Perchlorate		F	F	L		L	
NAPL							F
Creosote							F
Cation Metals (*e.g.*, Cu, Ni, Zn)	L	F	F		L	F	
Arsenic	F			L	F	F	
Chromium VI	F			L	L	F	
Uranium	F	P	F			T	
Strontium-90			F	F			
Selenium	L					L	
Phosphate					P		
Nitrate		F	F			F	
Ammonium				L			
Sulfate		F				L	
Methyl tertiary butyl ether (MTBE)		F					

adopted at a site in Manchester, comprising long cement bentonite slurry walls and two reactive gates, as indicated in Fig. 19a. Each individual gate consisted of two parallel treatment trains. Each treatment train consisted of two in-line reactor vessels (Fig. 19b), with the inlet reactor vessel having downward flow and the outlet reactor vessel having upward flow. The reactors were prefabricated steel vessels of 5 m height by 3 m diameter. Since one treatment train could be taken off-line for maintenance, this allowed for future exchange of the reactive medium, when necessary.

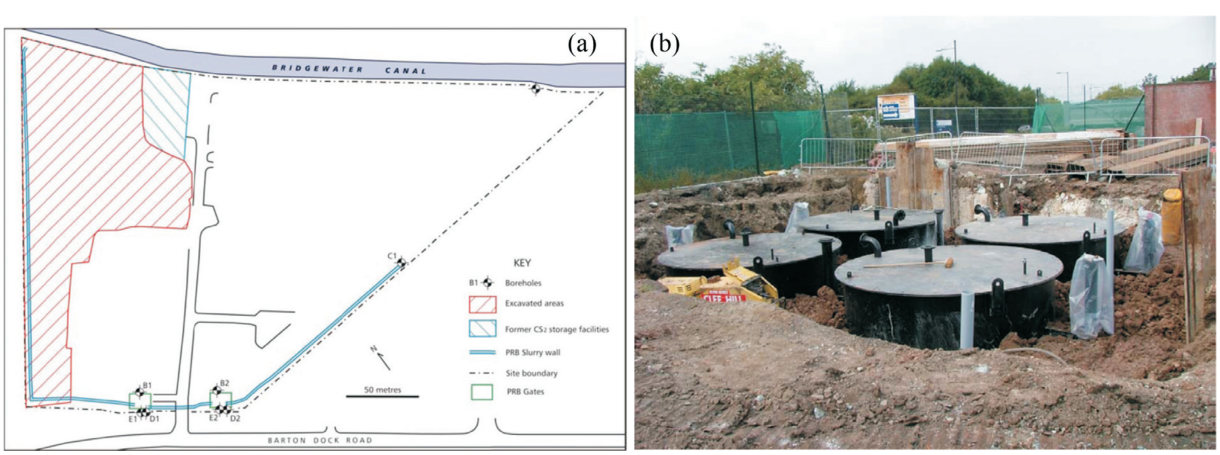


Figure 19. Reactor PRB described in Cox *et al.* (2009), (a) location of the slurry walls and gates, and (b) photograph of reactors.

4. Geotechnical reuse of waste

4.1 Overview

The use of waste and recycled materials in Geotechnical Engineering, differently from pavement and buildings construction, is still mostly limited to academic studies. Examples of reuse of shredded tires, foundry sand and fly ash are described in Aydilek & Wartman (2005). The suitability of different types of wastes as geomaterials has also been investigated: construction and demolition waste, mine tailings (sand, red mud, phosphogypsum), sewage and water treatment sludge, tires, sugarcane bagasse, coconut fibers, rice husk, coal ash, MSW IBA, fly ash, rock powder, PET bottles, crushed glass, etc.

The main technical challenges facing waste reuse for geotechnical purposes are dealing with variability and obtaining representative samples, and adapting the geomechanical models of behavior to the new materials. Once the waste is considered suitable as a geomaterial, additional tests must be conducted, in accordance with the foreseen application, in order to ensure environmental safety. Statistics and probability, as well as chemistry, are mandatory knowledge. Such extension of technical-scientific scope is welcome, anyway, since the need for probabilistic approaches in Geotechnics has been long underscored. However, the problem is not confined to solving the technical aspects. In order for waste to be used as a geomaterial, acceptance and preparedness from the part of environmental agencies, designers, contractors, and society as a whole must be achieved.

Environmental agencies are ultimately responsible for environmental damage upon official permission. The society must be convinced that earthworks built with waste will not display poor performance, neither be hazardous. Designers must learn to design with unknown materials with different properties, and contractors also must adapt long established procedures. Therefore, to move from laboratory to the full scale, much work remains to be done, concerning public policies, standardization, and networking. These are fields for which engineering courses do not prepare professionals yet. Without the incorporation of such perspectives into the Geotechnical mindset, however, reuse of wastes will never become widespread. Two research studies on waste-to-geomaterial perspectives are presented next.

4.2 Examples of waste-to-geomaterial research

4.2.1 Construction and demolition waste

In Brazil, construction and demolition waste (CDW) has been largely employed in civil and pavement construction. Regulations and standardization involve, among others, technical standards ABNT (2004a, 2004b, 2004c and 2004d) and ABNT (2011), and federal environmental regulations CONAMA Resolutions n. 307/2002, n. 420/2009 and n. 431/2011. States and municipalities also have regulations such as DD CETESB 045/2014/E/C/I for São Paulo state, and Decree n. 48,075/2006 for São Paulo city. This decree, for instance, declares mandatory the utilization of recycled aggregates generated from solid waste of civil construction in paving services and works for public roads in the São Paulo municipality.

CONAMA Resolution n. 307/2002 divides CDW into four categories: Class A - waste reusable or recyclable as aggregate for construction, renovation and repair of buildings, pavements and other infrastructure: bricks, blocks, roof tiles, cladding plates, mortar, concrete, pipes, curbs and soils from earthworks; Class B - waste recyclable for other destinations: plastics, paper, cardboard, metals, glass, wood, and others; Class C - waste for which technologies or feasible economic applications that permit recycling/recuperation have not yet been developed (*e.g.*, plaster products), and Class D - hazardous waste generated during construction processes: paints, solvents, oils and others, or contaminated waste generated during demolition, renovation and repair of radiology clinics, industrial installations and others.

It is important to remember that in Brazil the amount of CDW generated is high, according to ABRELPE (2020), 0.585 kg/inhabitant/day of CDW were collected in Brazilian municipalities in 2019. Despite the generated volume of CDW being significantly lower than that of MSW, the weight of CDW is very significant.

CDW could well be used as backfilling for geosynthetic-reinforced soil retaining walls (Santos, 2011). Other options could be for drains (drainage of natural water courses before landfill implantation, leachate drains, gas recovery drains) and pavements (access roads, storage platforms, parking areas) in landfills, where there could be greater acceptance of the use of waste by the environmental agency.

In addition, the use of recovered soils and CDW fines in geotechnical works should be promoted. Table 5 shows the percentages of excavated soils present in CDW (by weight) in different countries, demonstrating that excavated soils are an important portion of CDW and should be specifically studied.

In densely urbanized areas, large quantities of excavation soils can be generated due to the construction of underground garages of multi-story buildings and urban infrastructure such as subway lines, flood prevention reservoirs, and energy, gas, and water supply networks. Kataguiri *et al.* (2019), based on studies carried out in many countries, reported that excavation soils are generally disposed of in landfills, or dumped illegally, which is also the case in the Metropolitan Area of São Paulo.

When not segregated at the source, excavated soils turn into waste and must be dealt with as such. Generation of CDW increases 3-5 times when excavated soils are included (Monier *et al.*, 2011), and only around 6 % of excavated soils are recycled worldwide. Since excavated soils

Table 5. Excavated soils relative to total CDW in several countries (Kataguiri *et al.*, 2019).

Country	Excavated soils % total CDW (by weight)	Reference year
Austria	50	2011
Australia	65	2012
Brazil	32	2011
Denmark	53	2012
Finland	75	2011
France	69	2012
Germany	55	2011
Hong Kong	70	2013
Italy	24	2012
Norway	44	2008
United Kingdom	40	2012

are mixed with other types of waste, these soils have to meet construction and environmental requirements to be used in earthworks. Such soils could be used on site or redirected as daily or final covers for landfills, for backfilling of trenches or walls, for earth dams, for pavement sub-bases or bases, and for vegetation replacement.

Besides segregation of excavated soils at the source, the potential use of CDW fines should also be highlighted. The use of the coarse fraction of recycled CDW is regulated for road construction and for non-structural concrete. However, the current processes of CDW recycling mostly produce fine-grained recycled aggregates (< 4.8 mm) (Ulsen *et al.*, 2013), for which reuse strategies are still required (Magnusson *et al.*, 2015). These materials are mostly composed of mineral grains and cementitious materials (cement and lime), with good potential for earthworks. Stankevicius *et al.* (2019) reported promising results from own investigations and those of Kataguiri *et al.* (2019), Nomachi & Boscov (2016), Sharma & Hymavathi (2016), Amorim (2013), and Santos (2007) aiming at the geotechnical reuse of CDW recycled fine aggregates.

4.2.1.1 Reuse of excavated soils (Kataguiri, 2017; Kataguiri *et al.*, 2019; Nomachi & Boscov, 2016)

The investigation on the reuse of excavated soils aimed at delineating a flowchart to support screening excavated materials for different reuse options, based on current geotechnical and environmental characterization. To appeal to the end users, CDW recycling facilities and local municipalities, the flowchart was based on very simple tests and parameters. As excavated soils are still rarely segregated at the source (construction site), potential materials are assumed to be found in CDW landfills and recycling plants. Thus, the methodology includes procedures for adequate sampling of materials at these locations.

From thirty-five representative samples collected at a CDW landfill in São Paulo city, following Sampling Theory (Petersen *et al.*, 2005), eight samples were randomly selected and visually separated as either “CDW” (mixtures of excavation soils and other types of CDW) or “soil” (predominantly excavation soil). Three samples (B-5, B-19 and B-22) were defined as “CDW” and five as “soil” (B-4, B-7, B-12, B-15 and B-23), as illustrated in Figs. 20a and b. The grain-size distributions of the samples are presented in Fig. 21 and the geotechnical characterization in Table 6.

The samples were separated by sieving (0.1-0.4, 0.4-0.6, 0.6-1.2 and 1.2-2.0 mm), to estimate the percentages of cementitious and mineral grains in each fraction using image resources. The results indicated that soil grains and cementitious material are present in all fractions of all samples. “Soil” samples had higher contents of kaolinite, gibbsite, hematite, and goethite than the “CDW” samples. “CDW” samples, on the other hand, had a higher content of CaO and SO_3 than soil, due to the presence of cement, mortars, and gypsum. Calcite (CaO) is related to calcareous and cementitious materials, and the higher the content of cementitious materials, the higher the content of calcite.

The three “soil” samples were also classified according to the MCT Classification system (Nogami & Villibor, 1995). The fines of the five “CDW” samples were mixed and treated as a single sample, “CDW-composite”, as would be the case in a recycling plant, where collected

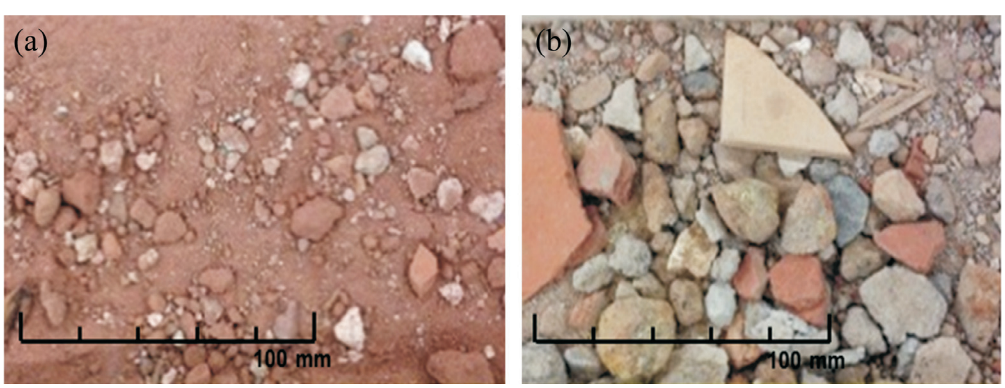


Figure 20. Visual classification of samples: (a) “soil”; (b) “CDW” (From Kataguiri, 2017).

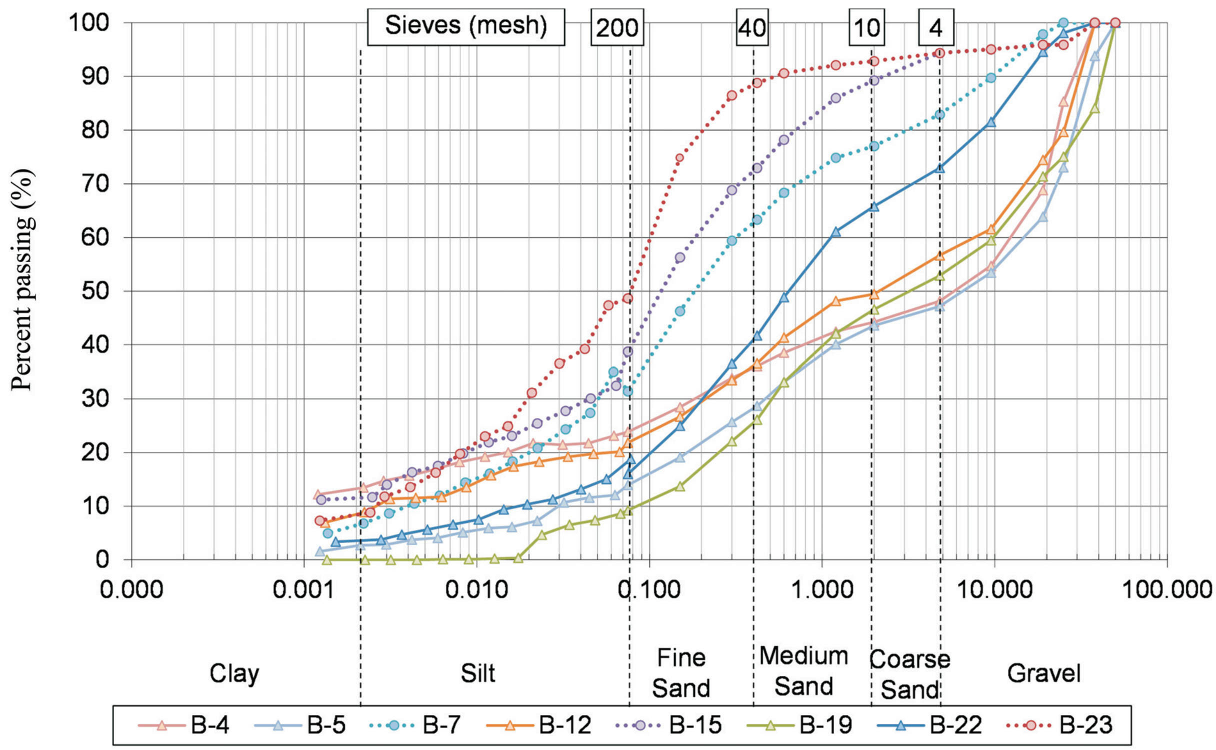


Figure 21. Grain-size distributions of the samples, including "soil" and "CDW" (From Kataguiri, 2017).

Table 6. Geotechnical characterization of "soil" and "CDW" samples (From Kataguiri, 2017).

Sample	Visual classification	Grain size distribution			Specific gravity	Atterberg limits			USCS classification
		Fines (%)	Sand (%)	Gravel (%)		Liquid limit (%)	Plastic limit (%)	Plasticity index (%)	
B-7	Soil	31.4	51.5	17.1	2.692	*	*	*	SM
B-15	Soil	38.7	55.6	5.7	2.703	30	22	8	SC
B-23	Soil	48.6	45.7	5.7	2.852	33	27	6	SM
B-4	CDW	23.9	24.3	51.8	2.637	28	25	3	GC
B-5	CDW	13.9	33.2	52.9	2.784	29	21	8	GM
B-12	CDW	21.8	34.8	43.4	2.567	32	19	13	GC
B-19	CDW	9.2	43.7	47.1	2.709	*	*	*	GP-GM
B-22	CDW	16.0	56.9	27.1	2.222	27	18	9	GC

*fine fraction with no plasticity.

"CDW" usually would be disposed and managed in heaps without segregation. The results from compaction, direct shear, and mini-CBR tests are presented in Table 7. Comparisons between mini-CBR results before and after immersion in water would provide an estimate of the loss of bearing capacity due to saturation. However, due to material scarcity, tests were carried out on saturated samples without surcharge in order to investigate the most unfavorable condition.

The classification and parameters of the "soil" samples are in accordance with typical soils from the outskirts of São Paulo city, saprolitic or young tropical soils derived from granite, gneiss, phyllite and other acidic rocks. These soils swell and lose strength remarkably when saturated; however, usually they show a low swelling pressure. On the other hand, "CDW-composite", *i.e.*, the fines from CDW samples, composed of soil and cementitious materials, were non plastic and not sensitive to water, and presented a high friction angle.

Environmental characterization showed that all samples, except for B-23, presented at least one of the contaminants sulfate and nitrate at a concentration above the respective maximum allowable value established by waste regulations (ABNT 2004a, 2004b) and drinking water standards (CONAMA n. 357/2005). The presence of nitrate and sulfate may be related to the degradation of organic matter, from sulfide oxidation in soils and rocks, and from crushed concrete and gypsum materials. The highest sulfate concentrations were found in the "CDW" samples and in "soil" sample B-7, which had a higher fraction of cementitious materials than the other "soil" samples, while nitrate correlated to organic matter content.

Kataguiri *et al.* (2019) point out that pH values measured for the eight tested samples were higher than 7.0 (pH range = 7.3-9.4). Alkalinity decreases leaching of nitrate and sulfate by infiltration or water seepage. Nitrate in water supply has been associated with the "blue-baby syndrome", a gastrointestinal disturbance and infant poisoning related to high levels of methemoglobin in infants, not reported in areas where nitrate concentration in drinking water is consistently lower than 50 mg /L (WHO, 2017). All studied samples had concentrations of dissolved nitrate below 50 mg/L. Sulfate concentrations in the studied samples, except for "CDW" samples B-4 and B-19, were below 1,000 mg/L, which is adequate for human health; for higher sulfate concentrations, site-specific risk assessment must be carried out. However, sulfate in water or soil may attack concrete foundation structures (Neville, 2004; WHO, 2017), requiring concrete with characteristic strength above 40 MPa for nearby foundations. The segregation of gypsum panels at the source (construction site) may reduce the concentration of dissolved sulfate in CDW. Finally, the flowchart considering visual classification as "soil" or "CDW", fines content (diameter < 0.075 mm), swelling at optimum water content, mini-CBR, and strength parameters was proposed, allowing to select the destination alternative as reuse as backfill for trenches and retaining walls, reuse in paving, or landfill disposal.

4.2.2 Water treatment sludge

Water treatment sludge (WTS) is the residue generated during the production of potable water from raw water. In Brazil, water treatment plants (WTPs) usually employ the conventional treatment method, which comprises coagulation, flocculation, sedimentation, filtration, and disinfection processes. Chemicals are added to the water for coagulation (coagulants, such as aluminum and ferric sulfate, ferric chloride, lime and polymers), disinfection (chlorine), dental protection (fluorosilicic acid) and pH correction (lime). WTS is generated during the periodic washings of the sedimentation tanks and filters, which generate, respectively, 60-95 % and 5-40 % of the total WTS by weight (Yuzhu, 1996).

WTS is composed of water (approximately 96-99 % by weight), suspended solids and chemical compounds (chlorine, aluminum sulfate, and/or ferric chloride, lime and fluorine). Sludge solids include organic (organic mat-

Table 7. Mechanical properties of soil and "CDW" samples (From Kataguiri, 2017).

Test	Compaction		MCT		Direct shear strength		Mini-CBR	
Parameter	Optimum water content (%)	Maximum dry density (g/cm^3)	Mass loss PI **(%)	Class	Cohesion (kPa)	Friction angle (°)	Swelling* (%)	CBR (%)
B-7 ("soil")	18.5	1.66	290	NS'	9.6	29	1.1	4
B-15 ("soil")	18.3	1.72	110	NA'	0	34	0.4	11
B-23 ("soil")	17.8	1.69	285	NS'	0	30	2.5	2
CDW-composite	19.5	1.67	-	-	0	38	0.0	16

MCT: miniature, compacted, tropical classification; CBR: California Bearing Ratio.
*The presented values refer to the optimum moisture content.
**Mass Loss PI: mass loss after immersion, relative to the dry mass of 20-cm^3 compacted soil.

ter, algae, bacteria and viruses) and inorganic substances (colloids, sand, silt, clay, calcium, magnesium, iron, manganese, aluminum hydroxides and polymers).

In Brazil, WTS ends being predominantly discharged irregularly into rivers, or disposed of in landfills or sent to sewage treatment plants (STPs). WTS discharged in rivers results in serious environmental impacts, mainly silting and degradation of water quality and the aquatic environment. WTS landfilling may cause instability of the waste mass, besides increasing the demand for landfill space. WTS sent to STPs may clog the sewer-system pipelines and overload the STP system (which is insufficient in Brazil, and in most countries, as 80 % of the global wastewater is released without treatment), with a material of composition very different from sewage sludge.

The search for alternatives for the reuse of WTS is an important environmental concern for the sustainability of the life cycle of water production. Different uses have been investigated, such as replacement for raw materials in the production of precast concrete elements, cement, asphalt, ceramics and steel, as well as applications such as composting, coagulant recovery, phosphorous removal from residual waters, and citrus production (Tsugawa *et al.*, 2017, Montalvan & Boscov, 2018). Geotechnical investigations also are being conducted. Despite promising results, case studies of practical applications are almost inexistent in the literature.

Even after dewatering by centrifuging, or on drying beds, at the WTP, WTS presents a solids content of only 20-25 %, being still inadequate for geotechnical applications. When air-dried or oven-dried, WTS usually turns into a granular material that can be useful as a construction aggregate. However, considerable amounts of time and energy are demanded. Two approaches can be considered for the geotechnical use of fresh dewatered WTS (at the "as-collected" or *in natura* water content at the WTP) as a material for embankments, filling of trenches and retaining walls, or as landfill covers and bottom liners: mixing with soils and mixing with additives. The reuse allows a beneficial destination of WTS as opposed to release to the environment, as well as economy of mineral resources by partially substituting soils in earthworks. These two approaches were investigated for a WTS collected at one of the largest WTPs of São Paulo state, Cubatão WTP.

A remarkable characteristic of WTS is the great variation in composition and properties associated with source and seasonality, *i.e.*, the WTP location, climate, season, raw water composition, treatment process and introduced chemicals, and dewatering process. Thus, the reuse of WTS requires a thorough case-specific investigation, until adequate indicators of geomechanical behavior of WTS and mixtures based on simpler tests are available. During the investigations with Cubatão WTS, a method to obtain representative monthly samples was developed using the Theory of Sampling (Tsugawa *et al.*, 2019).

Silva (2017) determined some geotechnical properties for different mixtures of Cubatão WTS with lime. For a batch sample of the Cubatão WTS, Silva (2017) determined grain-size distribution, Atterberg limits, compaction parameters and undrained shear strength, based on unconfined compression. For the fresh WTS, the liquid limit (w_L) and plastic limit (w_P) values resulted 228 % and 75 %, respectively. For determining the undrained shear strength of fresh WTS, the material was tested at different solids content ($S_c = 1/(1 + w)$, where S_c is the solids content and w is the gravimetric water content). As shown in Fig. 22a, the S_u of the pure WTS was found to increase exponentially with solids content, as previously shown for other WTS, including a ferric-chloride sludge tested by Wang *et al.* (1992). The S_u values near the w_L and w_P were found to be 1.13 kPa

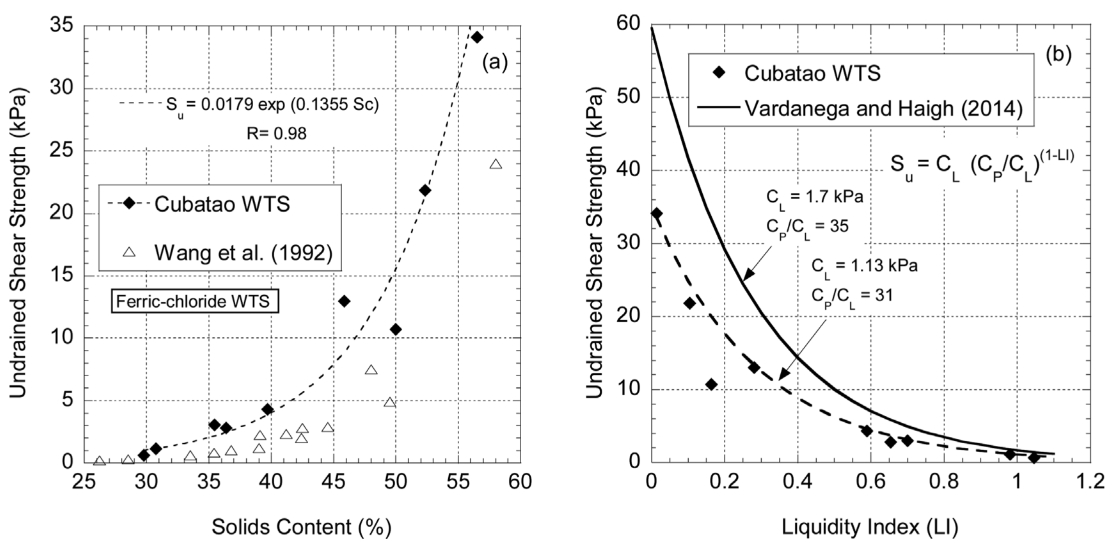


Figure 22. Undrained shear strength of the pure WTS as a function of: (a) the solids content, and (b) liquidity index, with comparison to the literature (From Silva, 2017).

and 35 kPa, respectively. Plotting the S_u values against the liquidity index (*LI*), defined as the ratio $(w - w_P)/(w_L - w_P)$, the obtained trend followed approximately the power (exponential) function proposed for soft clays by Vardanega & Haigh (2014). For soft clays, Vardanega & Haigh (2014) indicate C_L (defined as the S_u value at the w_L) to be equal to 1.7 kPa, and $C_P/C_L = 35$ (ratio between the strengths at the w_P and w_L). Based on the undrained shear strength experimental results for the WTS, C_L was 1.13 kPa, and $C_P/C_L = 31$ (Fig. 22b).

Montalvan & Boscov (2018) characterized and tested mixtures of Cubatão WTS with a lateritic clayey sand, representative of significant areas of São Paulo state and largely used as base material for low-traffic roads. The objective was to define the maximum WTS content that could be added without impairing the good geotechnical properties of the tropical soil.

WTS is similar to clayey soils, except for the high concentration of chemicals in the pore fluid, which displays an important role in geotechnical behavior (particles dispersion-agglomeration, water retention, among others). The pore liquid of Cubatão WTS has pH 7 and the grains contain a large amount of ferric chloride from the treatment process (iron concentration of 47.5 %, XRF). Major components of Cubatão WTS are quartz, goethite, muscovite, kaolinite, and amorphous phases. The particle size distribution by sedimentation indicated that about 70 %, by weight, of the solid particles were smaller than 0.005 mm. Specific gravity of grains varied from 2.9 to 3.2, the w_L was high (170-240 %), the specific surface area 52 m^2/g, the cation exchange capacity was 252 mmolc/kg, organic content 2.6 % and the organic carbon content was equal to 1.5 %.

Grain-size distribution (GSD) curves of the soil, WTS and soil-WTS mixtures are presented in Fig. 23. The mixtures present GSD curves similar to the soil, since the added percentage of solids is very small, due to the high water content of WTS. The difference in the percentage of fines with and without dispersing agent indicates flocculation caused by the ferric chloride in the WTS. Table 8 displays the geotechnical characterization and USCS classification of the materials.

Compaction curves of the soil and the mixtures are presented in Fig. 24. WTS at the *in natura* water content is impossible to compact. The maximum dry unit weight (γ_{dmax}) for the soil compacted under standard effort resulted equal to 19.1 kN/m^3 and the optimum water content (w_{opt}), 12.4 %. Mixtures of soil at $w = 1$ % (hygroscopic water content) with WTS (at the *in natura* water content of ~ 350 %) presented water contents of 15.3, 19.2 and 24.5 % for the proportions 5:1, 4:1 and 3:1, respectively. These mixtures were air-dried to a certain water content w_i to initiate the compaction tests. The tests were conducted at different values of w_i for each mixture, since the wet preparation method was a tentative trial to bracket the estimated w_{opt}, and also to investigate the influence of air-drying on the compaction parameters. When feasible, w_i was used as the first point of the compaction curve; otherwise, water was added to reach the first point of the compaction curve.

WTS addition decreased γ_{dmax} and increased w_{opt}. On the other hand, air-drying of the mixtures caused increase of γ_{dmax} and decrease of w_{opt}, and the lower w_i, the more markedly the effect was. Figure 25 shows that the compaction parameters resulted linearly correlated to the initial water content w_i for the three mixtures. The change in behavior, from that of a soft clay to a coarse-grained material

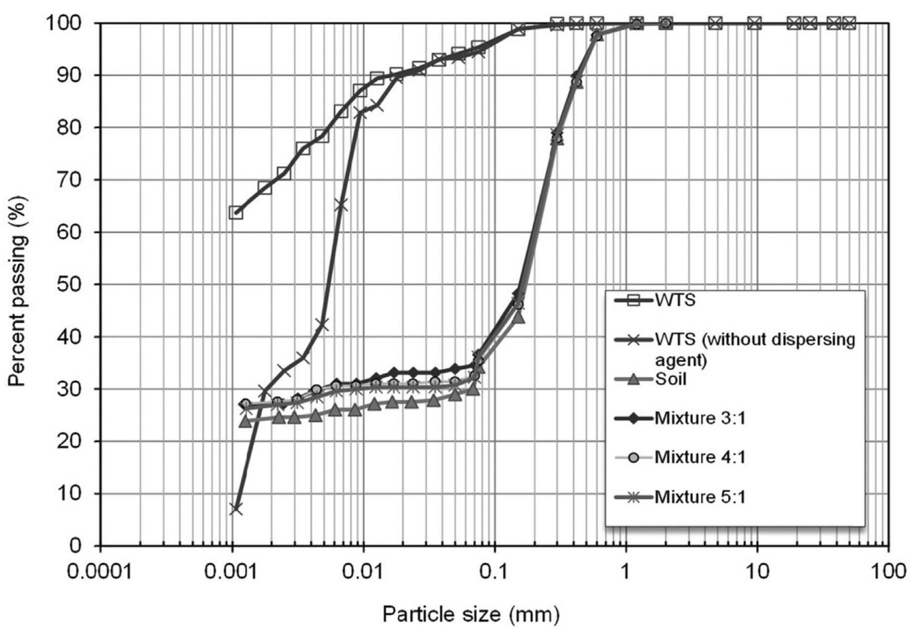


Figure 23. Grain-size distributions of the soil and soil-WTS mixtures (From Montalvan & Boscov, 2018).

Table 8. Geotechnical characterization of the materials (From Montalvan & Boscov, 2018).

Parameter	Soil	WTS	Mixture 5:1	Mixture 4:1	Mixture 3:1
Fine fraction (%)	34	95	36	36	37
Sand fraction (%)	66	5	64	64	63
Liquid limit (%)	25	239	32	29	33
Plasticity index (%)	11	158	14	12	16
Specific gravity of solids	2.69	2.85-2.95	2.69	2.70	2.71
Soil classification (USCS)	SC	MH	SC	SC	SC

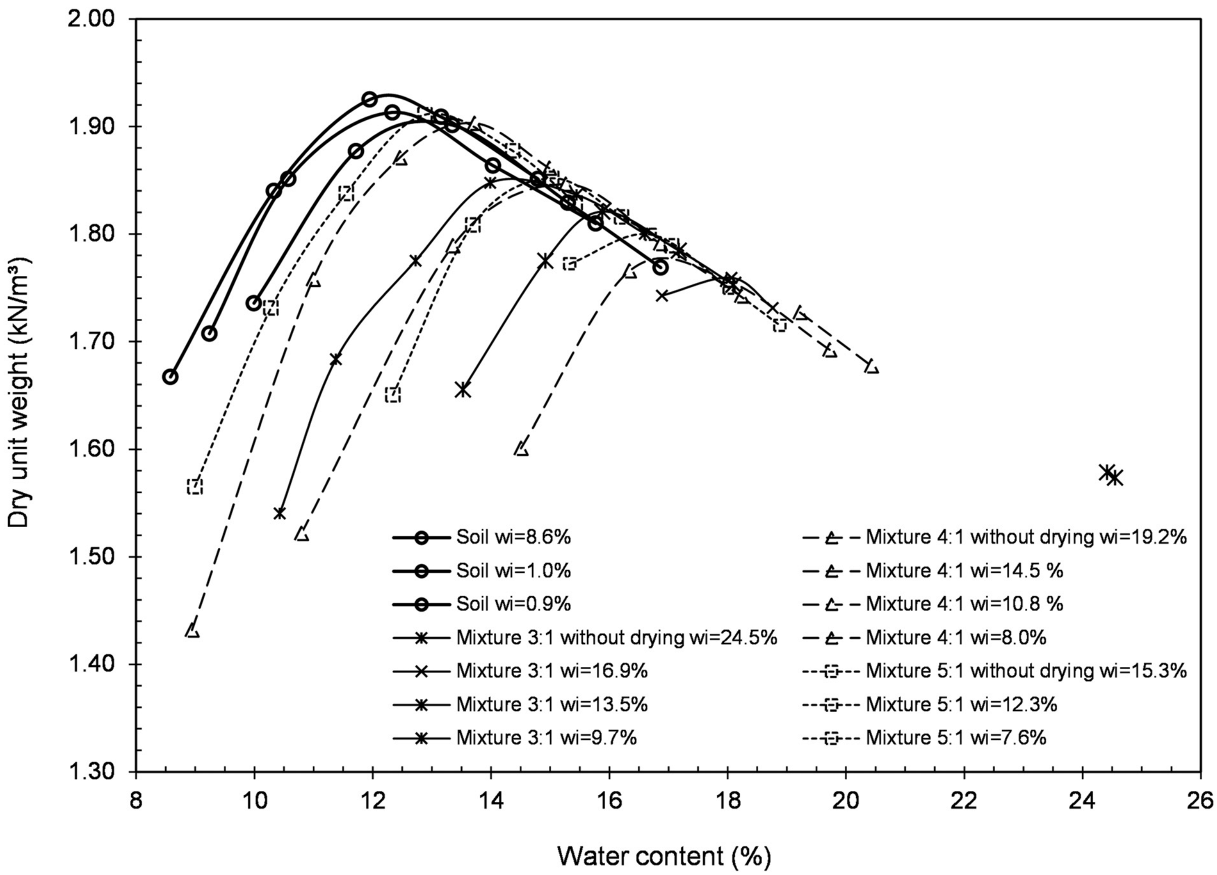


Figure 24. Compaction curves of the soil and soil-WTS mixtures (From Montalvan & Boscov, 2018).

may be attributed to WTS and the effect of coagulants, since the compaction curve of the soil did not change substantially with air-drying.

One-dimensional consolidation tests were carried out to examine whether WTS addition would increase prohibitively the compressibility of the soil. The compression indexes C_c for the 5:1, 4:1 and 3:1 soil-WTS mixtures, of 0.14, 0.13, and 0.19, respectively, were higher than that for the soil, 0.07, but mixtures 5:1 and 4:1 could be considered as still acceptable for geotechnical works. The expansion and recompression indexes of soil and mixtures were practically equal ($C_e = C_r = 0.02$). Note that the soil was compacted at w_{opt}, whereas the mixtures were compacted without drying (w_i equal to the water content after mixing the materials). Mixtures 5:1, 4:1 and 3:1 were compacted, respectively, dry of optimum, slightly wet of optimum and wet of optimum.

The results from permeability tests are shown in Table 9. The hydraulic conductivity (k) values of soil and mixture 5:1, for confining pressures of 30 and 60 kPa and hydraulic gradients of 5 and 10, were similar, despite the soil being compacted at w_{opt} and the mixture dry of optimum (flocculated structure). Values of k for mixture 4:1 were lower, possibly due to the greater addition of fine-grained WTS and wet-of-optimum compaction. For mixture 3:1, the k value decreased with time due to clogging of the test

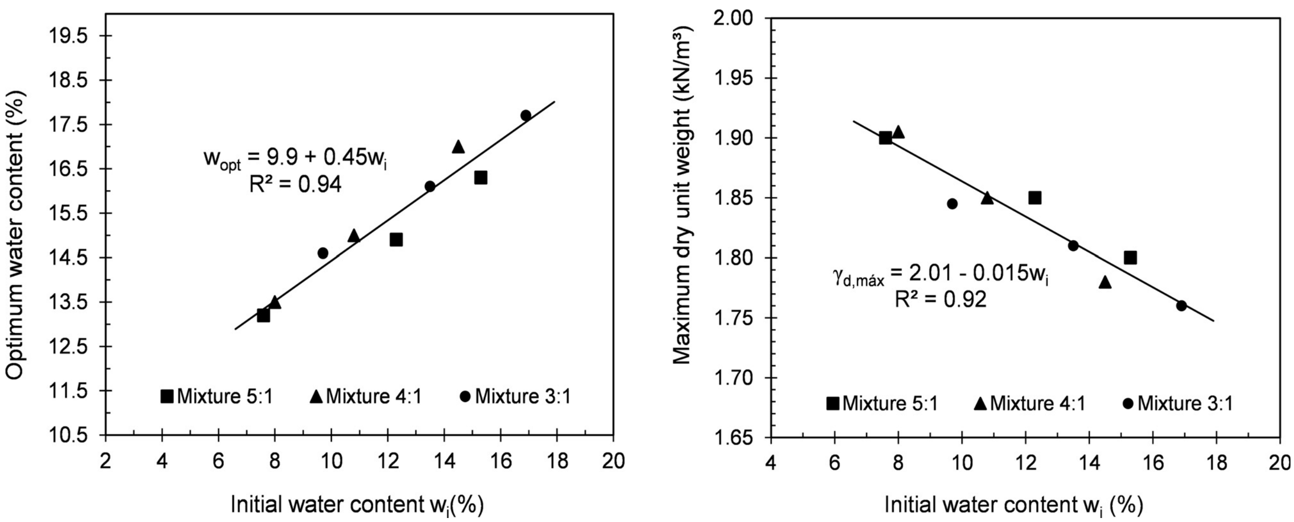


Figure 25. Variation of w_{opt} and γ_{dmax} of the mixtures as a function of initial water content (From Montalvan & Boscov, 2018).

Table 9. Hydraulic conductivity of the soil and the soil-WTS mixtures (From Montalvan & Boscov, 2018).

Confining pressure (kPa)	Hydraulic gradient	Hydraulic conductivity (m/s)			
		Soil	Mixture 5:1	Mixture 4:1	Mixture 3:1
30	5	1.3×10^{-6}	1.4×10^{-6}	4.3×10^{-7}	7.0×10^{-9}
30	10	6.9×10^{-6}	2.0×10^{-6}	3.0×10^{-8}	-
60	5	4.3×10^{-7}	1.6×10^{-7}	8.7×10^{-8}	-
60	10	3.9×10^{-7}	1.6×10^{-7}	1.3×10^{-7}	-

specimen, practically ceasing seepage after 47 days, probably due to chemical compounds reacting with the soil grains (the mixtures with lower WTS contents did not exhibit this behavior).

The stress paths obtained from CIU triaxial testing of the soil and soil-WTS mixtures are shown in Fig. 26. The effective strength parameters for the soil and the mixtures were calculated from the effective stress paths. The obtained internal friction angles for the soil and mixtures 5:1, 4:1, and 3:1 were equal to 34, 34, 35, and 37°, respectively. The increase in φ' with increasing the WTS content has been observed by other authors. The effective cohesion decreased with WTS content, from 22 kPa for the soil, to 17, 15 and 10 kPa for mixtures 5:1, 4:1 and 3:1, respectively.

Soil-WTS mixtures 5:1 and 4:1, therefore, could be considered as feasible materials for geotechnical applications, given the slight variations in geotechnical parameters caused by WTS addition. Mixtures of other soils with WTS are under study, with similar promising results, but indicating different thresholds for WTS incorporation. As already mentioned, before simple indicative tests are developed, the geotechnical behavior of soil-WTS mixtures must be extensively investigated case-by-case, since results from specific materials should not be generalized.

Ongoing research aims at confirming the environmental safety of employing soil-WTS mixtures in earthworks. This beneficial application of WTS should be stimulated by environmental policies to overcome prejudice against use. The perspective of financial compensation should also be considered, to account for the additional time and cost required for proper mixing.

Another possibility for WTS reuse as a geomaterial could be incorporating additives, such as lime and fillers, to improve WTS workability and geomechanical properties. The mechanical characteristics of fresh WTS before treatment must be known, to orient the selection and dosage of additives. However, most geotechnical tests were conceived for soils and not for fresh or *in natura* WTS, a fine-grained material with very high water content. Obtaining strength parameters for fresh WTS using standard geotechnical equipment and experimental procedures often results impracticable, even with the laboratory vane shear and fall-cone tests. For this reason, rheometry tests were explored for assessing the stress-strain behavior of fresh WTS. Samples of Cubatão WTS (w = 240 %), WTS-lime mixtures and WTS-rock powder mixtures were submitted to rotational rheometry tests. On the other hand, mixtures

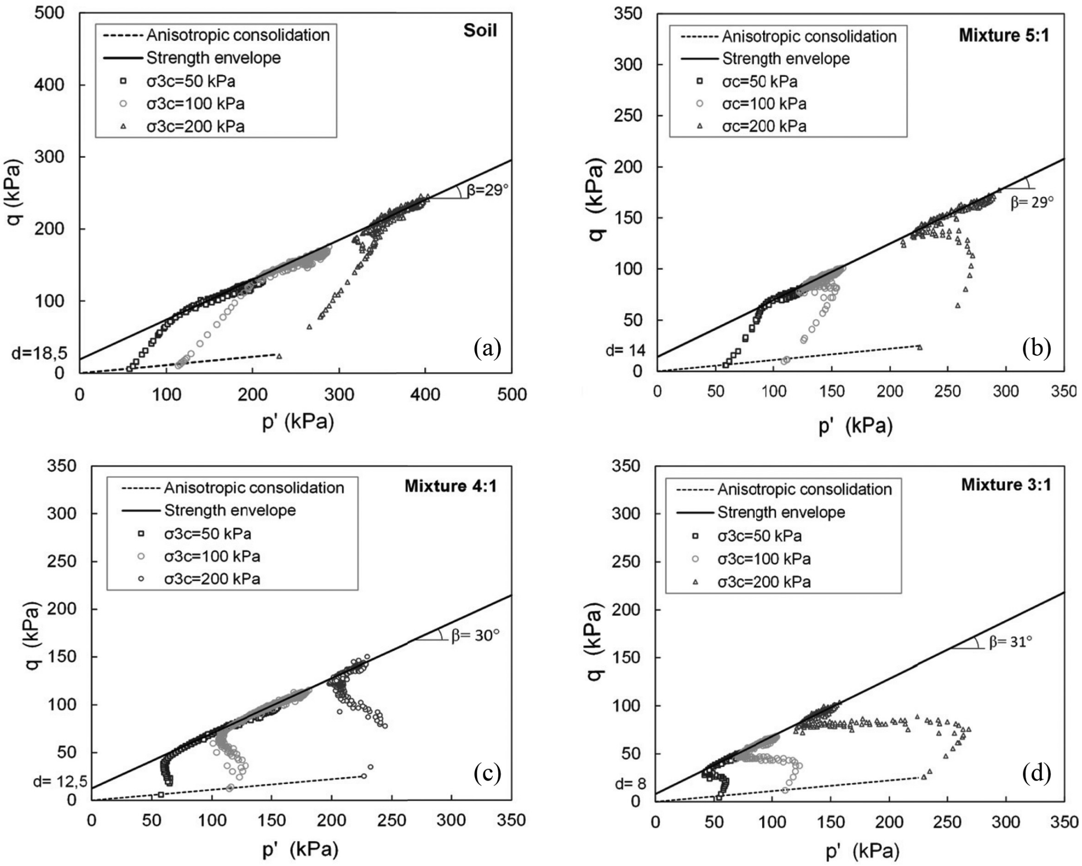


Figure 26. Effective stress paths of the soil and three different soil-WTS mixtures (From Montalvan & Boscov, 2018).

with very high additive content and, therefore, soil-like behavior were submitted to traditional geotechnical tests.

Figure 27 compares the well-known laboratory vane test with a rotational rheometry test. The vane test has a constant shear rate, while rheometry tests allow different geometries and the programmed shear rate to vary (Table 10). Measurements of rotational velocity, torque, deformation and time of response can be related to shear stress and shear strain.

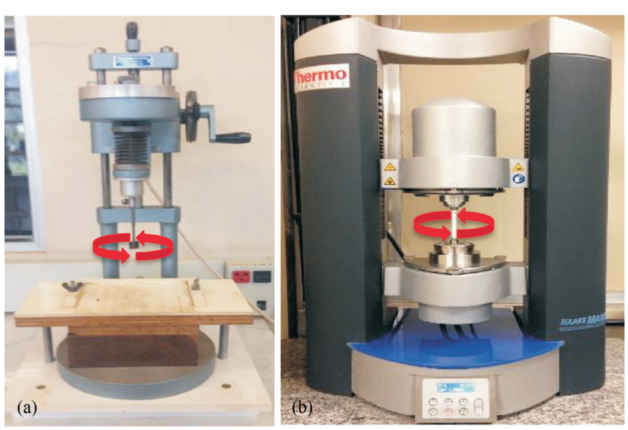


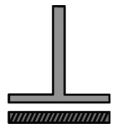
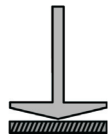
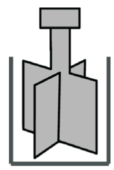
Figure 27. Comparison between experimental arrangements for: (a) laboratory vane shear test; (b) rotational rheometry test (Tsugawa *et al.*, 2018).

Stepped flow tests were performed using a steel parallel plate geometry (diameter of 35 mm, gap of 1.0 mm). Shear rate was increased (acceleration) twice and decreased (deceleration) stepwise from 0 to 50 s^{-1} (1,080,000°/min or 3,000 rpm), *i.e.*, two cycles of shear rate acceleration-deceleration were performed, totalizing 400 s of test (Fig. 28).

The output of the tests can be exemplified in Fig. 29, where results of a flow test for both cycles of acceleration-deceleration are presented. The first cycle is related to a "very early age" behavior of WTS, while the second cycle is the condition where test steady state has been reached. Results are discussed in Tsugawa *et al.* (2018) and can be summarized by the parameters in Table 11. The flow test also allowed to observe that Cubatão WTS may present thixotropic or rheopectic behaviors, depending on the applied shear rate: Cubatão WTS is thixotropic for shear rates lower than 180 rpm and rheopectic for higher shear rates, a fact that has implications in efficiency of field processes such as pumping, homogenizing using concrete mixers, removal from storage tanks, among others.

Besides mapping the stress-strain behavior of very moist WTS and WTS-additive mixtures, the results were compared to laboratory vane shear tests to characterize

Table 10. Rheometry tests: shear stress and shear rate calculations for different geometries.

Geometry	Configuration	Shear stress	Shear rate
Plate-to-plate		$\sigma = \dfrac{2\Gamma}{\pi R^3}$	$\dot{\gamma} = \dfrac{\Omega R}{h}$
Cone-plate		$\sigma = \dfrac{3\Gamma}{2\pi R^3 \sin^2\left(\dfrac{\pi}{2} - \theta\right)}$	$\dot{\gamma} = \dfrac{\Omega}{\tan\theta}$
Vane		$\sigma = \dfrac{\Gamma(R_1^2 + R_2^2)}{4\pi h R_1^2 \cdot R_2^2}$	$\dot{\gamma} = \dfrac{\Omega(R_2^2 + R_1^2)}{(R_2^2 \cdot R_1^2)}$

σ = shear stress (equivalent to τ in Soil Mechanics); $\dot{\gamma}$ = shear rate; Γ = torque, Ω = rotation velocity; R, h, θ, R_1 and R_2 = geometric characteristics.

thixotropy using different methodologies. Results are presented, and advantages and disadvantages of both methods are discussed in Tsugawa *et al.* (2018). The parameters of the laboratory miniature vane test were: constant shear rate of 50°/min (0.0024 s^{-1}), and vane blade of 12.7 × 12.7 mm. WTS at a water content of 240 % was remolded by hand, tested immediately after remolding (t = 0), and after different storage times (1, 3, 7, 14, 28, 84 and 168 days). Vane tests measured thixotropy for longer periods of time (storage times), whereas stepped flow tests indicated thixotropy

Table 11. Rheological parameters of Cubatão WTS (From Tsugawa *et al.*, 2018).

	Yield stress (Pa)	Apparent viscosity (Pa.s)	Hysteresis loop (Pa/s)
First cycle	43.4	4.53	1337.8
Second cycle	43.0	4.92	-1074.8

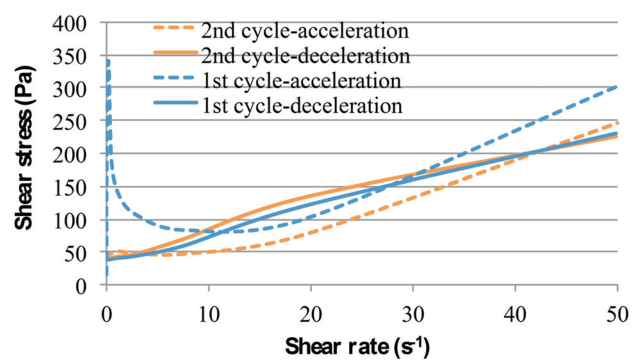


Figure 29. Example of a flow test output for fresh Cubatão WTS (w = 240 %) (From Tsugawa *et al.*, 2018).

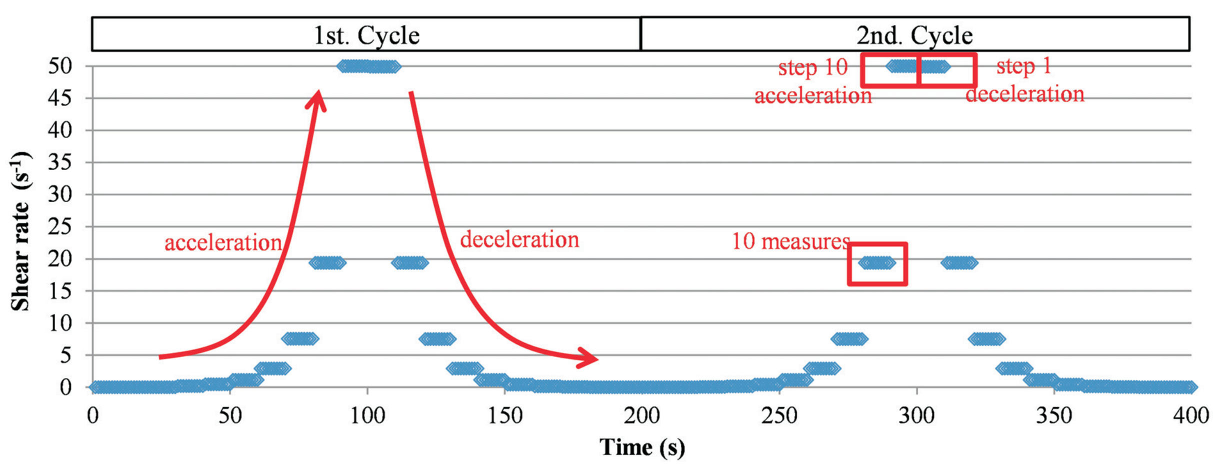


Figure 28. Applied shear rate history in rheometry tests (stepped flow type test) (From Tsugawa *et al.*, 2018).

at very early times after WTS being placed in the equipment. Thus, the results are related to different types of mobilizations and must be applied to measure thixotropy depending on the practical objective.

Once fresh WTS was characterized, geomechanical behavior of mixtures of WTS with lime and rock dust were investigated. The strength limit separating the use of rheometry and geotechnical tests is still under debate. Even though rheometry tests are quick and consume small quantities of materials, they are limited to materials with low shear strength compared to typical geomaterials. Such tests may be considered useful to define a threshold of additive content for applications such as coulis for diaphragm walls or for minimum workability in the field for spreading daily landfill covers with compaction equipment. However, to screen ranges of additive contents for road construction, backfilling of trenches or reinforced walls, and compacted embankments in general, traditional geotechnical tests remain required.

5. Conclusions

The field of Environmental Geotechnics has matured over the past decades, developing and advancing a broad repertoire of theoretical knowledge and applied techniques to deal with the challenge of building and maintaining infrastructure while safeguarding environmental conservation. This paper aimed at focusing on three topics of significant relevance to modern sustainability in Brazil in which Geotechnical Engineers contribute, but could have an even greater participation: expansions in MSW landfills, site remediation benefiting from geotechnical solutions, and reuse of wastes in geotechnical works. First, the issues associated with designing an appropriate environmental protection system at the base of new landfill expansions are highlighted, as well as the possibility of immersing geogrids to reinforce the mass of MSW, allowing increased storage capacity. Secondly, dealing with the additional challenges of site remediation at a complex urban region of past industrial land use, the importance of a joint regional plan for investigation and remediation is discussed, as well as the possibility of using geotechnical confinement and *in situ* passive remediation to treat the area. Finally, preparedness to accept working with wastes as geomaterials is pointed out, and two examples of investigation on the reuse of construction and demolition waste and water treatment sludge are discussed. Construction and demolition waste is shown to contain a significant amount of excavation soils, for which reuse options are still not in place. Water treatment sludge is a challenging material, but could be useful when mixed with local soils or stabilized with additives. The topics discussed in this paper are three examples of the many interesting challenges posed to Geotechnical Engineers facing environmental conservation. Environmental Geotechnics is permanently undergoing significant advancements, at the same time new demands require innovative solutions, making this science and engineering continuously stimulating.

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