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A review of blasting waste generation and management in the ship repair industry

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ABSTRACT

Solid blasting waste generated from coating removal during ship repair and maintenance poses environmental challenges. This paper presents a review of the generation, characterization, and treatment of blasting waste in the ship repair industry. The quantities, properties, and environmental impacts of the generated blasting waste are summarized and analyzed, and the results indicate that blasting waste has a high generation rate and/or high toxicity. As alternatives to landfill, available blasting waste treatment methods include hydrometallurgical leaching, physical/physicochemical separation, thermal treatment, and direct utilization in the production of building materials. The advantages and disadvantages of these treatments are reviewed and compared. The production of building materials from blasting waste is currently attractive owing to its economic benefits and technical simplicity, whereas recycling blasting waste for high-value applications is gradually gaining research interest. The high dependence of the choice and performance of treatment methods on the waste type and characteristics is highlighted. The results emphasize the interest in conducting more research on physicochemical and thermal properties and the environmental effects of blasting waste. This review suggests that future research should focus more on paint waste management because of the restricted application of dry abrasive blasting and the development of nonabrasive blasting methods in the ship repair industry.

1. Introduction

1.1. Marine coatings and coating removal in shipyard

The shipping industry accounts for transporting 90% of global trade. The current world fleet comprises approximately 95,000 merchant vessels, including bulk carriers, tankers, containers, general cargo, ferries, and passenger ships (Asariotis et al., 2019). As illustrated in Fig. 1, a typical merchant ship comprises distinct areas such as underwater hull, boot top area, decks, ballast tanks, topsides and superstructures, and vessel interiors (Samrat et al., 2018). Different marine coatings are vital and tailored to protect the ship surfaces from corrosion, heat or fire, and fouling. A coating system typically includes several layers of coatings: a primer coat, one or more intermediate coats, and a topcoat. For instance, the coating system commonly applied in underwater hulls and boottops comprises two layers of epoxy anticorrosive coatings and three layers of antifouling coatings (Roberts, 2018). The basic components in each coating layer are pigments (including

extender/filler), binder, solvent and additives (such as defoamer, wetting agent, rheological agent) (Sørensen et al., 2009). For this study, only nonvolatile components (mainly pigments and binder matrix) are of interest, because it is presumed that the volatile solvents have essentially dissipated once the maintenance cycle of the vehicle reaches the stage where the hull is repainted. Typical pigment types used in conventional marine coatings and their functions are listed in Table S1 in Supplementary Materials.

Under the protection of the coatings, a ship can operate for a service lifetime of 20–30 years. However, coating degradation and rusting of the vessel surfaces occur during marine transportation, which requires the ship to be docked for repair and maintenance at intervals of 3–5 years (Ashari et al., 2018). During ship repair, foreign matter on the ship coating surfaces such as oil, grease, salts, attached marine organisms, and slime, are washed down with high-pressure water, followed by the removal of rust and coatings via spot or full blast cleaning.

Current coating removal methods applied in shipyards can be classified into abrasive and nonabrasive blasting methods. Abrasive blasting (i.e., grit blasting) uses air pressure, water pressure, or centrifugal force

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Abbreviations

ABM	Abrasive blast media
CNT(s)	Carbon nanotube(s)
C&D	Construction & demolition
FBSG	Fluidized-bed sloped-grid
MSW	Municipal solid waste
SLP	Synthetic precipitation leaching procedure
TBT	Tributyltin
TCLP	Toxicity characteristic leaching procedure
UHP	Ultrahigh pressure
VOC	Volatile organic compound

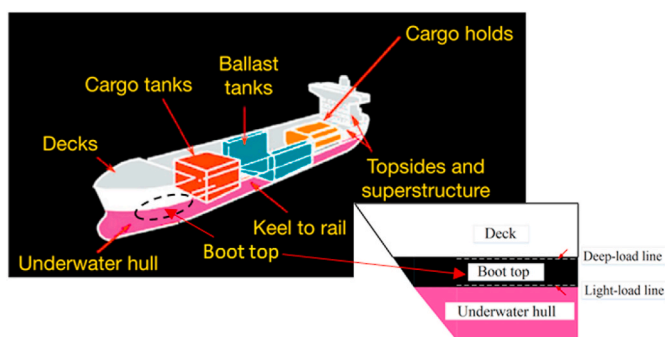


Fig. 1. Different areas on a ship (Samrat et al., 2018).

to propel a high-velocity stream of an abrasive against a surface to remove rust, mill scale, dirt, and old paints, and to create a rough surface profile. Nonabrasive blasting removes surface contaminants and coatings without the use of abrasives. However, it cannot create a surface profile, and therefore, it is primarily used for an old profiled surface rather than new steel surfaces.

For many years, dry abrasive blasting has been the most efficient and economical method applied on large surfaces to remove old paints, rust, and other impurities. Fig. 2 (a) illustrates the simple working mechanism of the most used air pressure blasting, wherein compressed air is used to propel abrasive materials to the workpiece. Air pressure blasting is used in an open-air environment because the vessels are too large for an indoor facility. Abrasive media such as natural minerals (e.g., garnet and olivine), metallic grits, coal slag, copper slag, and other metallurgical slags have been commonly used after the ban of silica sand because of silicosis (Flynn and Susi, 2004). In this process, a large amount of solid waste is generated, which consists of contaminated abrasives and paint chips. Further, open-air dry abrasive blasting has an increasing

challenge in terms of compliance with state and local health and environment regulations. To this end, efforts have been invested to reduce dust emissions, including the application of vacuum blasting (Appleman, 1992), use of dust suppressants (Flynn and Susi, 2004), and development of (semi-)automatic systems (Iborra et al., 2002; Li et al., 2018). Government and local regulations increasingly restrict the use of open-air dry blasting, thereby motivating the development of new technological solutions such as finding alternative blast media and techniques.

Wet abrasive blasting methods were developed to reduce dust emissions and waste generation. Wet abrasive methods used in the ship repair industry can be divided into two categories: air abrasive blasting with water addition (i.e., vapor blasting or slurry blasting), and water blasting with abrasive addition (i.e., hydraulic blasting) (Appleman and Bruno, 1985). In hydraulic blasting (Fig. 2 (b)), high-pressure water (200–700 bar) is used to drive the abrasives to the surface. In contrast, in slurry blasting (Fig. 2 (c)), fine abrasives suspended in a liquid are projected at a high velocity by a jet of compressed air, or less commonly, a high-pressure centrifugal pump (Zhu and Chang, 2021). Compared to hydraulic blasting, slurry blasting is ‘gentler’, achieves a finer finish, and has lower water consumption (Andronikos et al., 2004). However, compared with the dry abrasive method, both techniques generate an extra waste stream, i.e., wastewater.

Another popular wet blasting method is water blasting without any abrasives, which is called water jetting. Ultrahigh pressure (UHP) water jetting is one of the fastest growing surface preparation methods used in ship repair yards. In UHP water jetting (Fig. 2 (d)), the UHP pump pressurizes fresh water into an ultrahigh pressure (typically 2000 bar as a minimum) stream and then passes it through rotary nozzles with small orifices, forming an intensive blasting stream to remove old paints, rust, and other surface contaminants. The system is usually equipped with a vacuum suction system to collect wastewater and waste paint chips. Without the use of abrasive media, the amount of waste generated is greatly reduced. Nevertheless, an on-site water recycling system is important for the sustainable utilization of fresh water.

Other techniques such as dry ice blasting (Alankaya and Celebi, 2015; Kohli, 2019), cryogenic N₂ jetting (Hajji et al., 2018), plasma depainting (Aliofkhazraei and Rouhaghdam, 2011; Bybin et al., 2016; Merati et al., 2017, 2019), and laser depainting (Chen et al., 2010; Li et al., 2017; Marimuthu et al., 2014; Shamsujjoha et al., 2015) have been developed and are being increasingly applied in the coating removal process to minimize waste generation.

1.2. Blasting waste management in shipyard

Conventional dry abrasive blasting, though efficient and economical, produces a large amount of dust emission and abrasive blasting media (ABM) waste, which contains abrasive media, paint chips, and rust (Kura et al., 2006). Currently, the ABM waste produced in shipyards is usually landfilled or processed in cement kilns. Alternative treatment methods

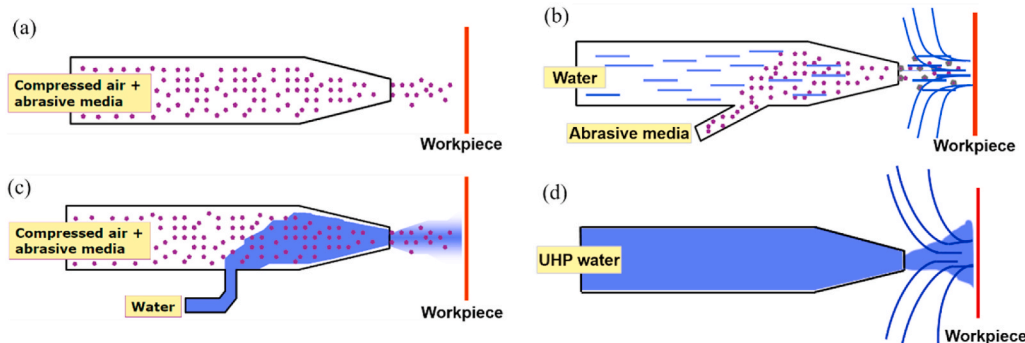


Fig. 2. Schematics of (a) air pressure blasting, (b) hydraulic blasting, (c) slurry blasting, and (d) UHP water jetting.

of ABM waste such as thermal detoxification (Bryan et al., 1991), stabilization/solidification (Means et al., 1992; Zain et al., 2004), physical/physiochemical separation (Katsikaris et al., 2002), leaching (Yang and Kravets, 2000), and reuse in the production of concrete, asphalt, and cement (Madany et al., 1991; Sua-iam and Makul, 2017; Tajunnisa et al., 2019), have been investigated through research and demonstration projects. In contrast, wet blasting, especially UHP water jetting, generates a smaller quantity but usually more hazardous solid paint waste. The paint waste is enriched in heavy metals such as Zn and Cu, which are toxic but also valuable metal sources. However, the recycling and treatment of paint waste has not been extensively explored.

1.3. Objectives of the present review

Extensive research has been conducted for decades to minimize the generation of blasting waste and to treat blasting waste to reduce toxicities and/or produce valuable products. However, to the best of our knowledge, besides being partly covered in a book (Momber, 2008) and a review article (Kotrikla, 2009), dedicated review articles on blasting waste from the ship repair industry are unavailable. Further, with the rapid development of novel nonabrasive blasting methods such as UHP water jetting and new blasting waste treatment technologies such as leaching and electrochemical precipitation in the last decade, a dedicated and updated review on solid blasting waste from the ship repair industry is needed.

This paper aims to provide an up-to-date review on the generation, characterization, and treatment technologies of solid blasting waste from the ship repair industry. The characteristics of the generated blasting waste from the ship repair industry, including the ABM waste from dry abrasive blasting and the solid paint waste from UPH water jetting, are summarized and evaluated. A spectrum of technologies for the treatment of blasting waste is introduced and compared in terms of removal/reuse efficiency, economic cost and environmental impact. The prospects for further studies on solid blasting waste including emerging blasting waste streams and their management are suggested, and we aim to provide suggestions on future research technological development for the effective utilization/treatment of blasting waste.

2. Quantity and characteristics of blasting waste

2.1. Generation rate

A variety of abrasive media is used in the air or wet blasting processes. The amount of abrasive required to remove the paint from a ship hull depends on the surface condition of the hull (i.e., rusting and existing paint), nozzle diameter of the hose used in abrasive blasting, number of coats to be removed, and other contract specifications. In ship repair yards, a total of 200–2000 tons of abrasive media are required for blast cleaning a 10,000-m² surface of the ship hull; this corresponds to a spent abrasive disposal rate of 20–200 kg/m² (typically 40–50 kg/m²)

Table 1
Physical properties of ABM wastes produced from dry abrasive blasting.

ABM waste	Particles size (mm)	Bulk density (kg/m ³)	Specific gravity	Water absorption (%)	pH	Reference
Copper slag	0–1.00 ^a					Dung et al. (2014)
	0.15–0.60	2260	3.85	0.33		Anjos et al. (2017)
	0–1.18 ^b	3310	3.34	0.27		Madany et al. (1991)
	4.75–12.5		3.50	0.40		Salleh et al. (2014)
	0–6.5	1990–2500	3.57		8.40	Lim and Chu (2006)
Ferrous-nickel slag	0–1.00 ^c	3320			8.03	Katsikaris et al. (2002)
Steel grit	0–0.16					Buruiana et al. (2011)
Garnet	0–0.94		3.95	2.56		Taha et al. (2001)
Alumina	0–0.30	1994 ± 45.83	3.18 ± 0.02			Sua-iam and Makul (2017)

^a particles with size >1.00 mm account for 36 ± 3.7%.

^b particles with size >1.18 mm account for 9.7%.

^c particles with size >1.00 mm account for 14.9%.

(Momber, 2008). The produced ABM waste predominantly contains used abrasives and 1–2 wt% of paint chips. The paint chips containing lead or antifouling coatings may be hazardous, however, in practice, the concentration of toxicants is significantly reduced by dilution with considerable amounts of the spent abrasives. In contrast, UHP water jetting generates slurry waste comprising wastewater, paint chips, and rust. In terms of solid waste, the amount generated is about 0.3–1.0 kg per m² of surface, with a typical value of 0.5 kg/m² (Salminen et al., 2016), which is considerably smaller than that of dry abrasive blasting (40–50 kg/m²).

2.2. Physical property

The physical properties of typical ABM wastes are listed in Table 1. Most ABM wastes have a large percentage of small particles (<1 mm). Compared with the original ABM, the particle size of ABM waste is greatly reduced by the impact of the substrate surface during blasting operations. For example, an original copper slag accounted for 40% within the particle size range of 1.25–2.5 mm. After blasting, approximately 90% of the particles were smaller than 1.18 mm (Madany et al., 1991). For ferrous-nickel slag, the percentage of large particles (>1 mm) was reduced by half after one cycle of use (Katsikaris et al., 2002). Compared to expendable abrasives, reusable abrasives such as steel grit, alumina, and garnet have smaller and narrower distributed sizes after blasting. This may be because, on the one hand, the original media is sieved and graded, and on the other hand, they have been recovered and reused many times before ending up as waste. These physical properties can be an important reference information for waste treatment, especially for physical separation; for example, size classification and gravity separation.

2.3. Chemical analysis

Chemical compositions of several typical ABM wastes from the literature are listed in Table 2. The constituents of ABM waste vary somewhat; however, the primary components are aluminum oxide, calcium oxide, silicon dioxide, and iron oxide. These four components account for up to 90% of the ABM waste composition.

Table 3 presents the composition analysis of the paint waste collected from ship repair and maintenance. Although from multiple sources, the paint waste is generally enriched in zinc (approximately 3–16 wt%) and copper (approximately 10–40 wt%), which is consistent with the composition of conventional marine coatings applied. Copper may exist in the form of cuprous oxide (or small amounts of metallic copper, copper thiocyanate, and copper acrylate) from the antifouling paints. Zinc may come from the anticorrosive paints in the form of metallic zinc, inorganic zinc silicate polymer, zinc oxides and other zinc corrosion products, or antifouling paints in the form of zinc oxide, zinc pyrrhione, and zinc acrylate. Further, small amounts of Sn and trace quantities of Pb and Cr were found in some paint waste. Although the

Table 2
Chemical compositions of ABM wastes produced from dry abrasive blasting.

ABM waste	Copper slag	Copper slag	Copper slag	Copper slag	Steel grit	Garnet	Alumina	Quartz sand	Silica sand	Grit
Reference	Dung et al. (2014)	Anjos et al. (2017)	Lim and Chu (2006)	Madany et al. (1991)	Buruiana et al. (2011)	Taha et al. (2001)	Sua-iam and Makul (2017)	Ho et al. (2019)	Tajunnisa et al. (2019)	Borucka-Lipska et al. (2019)
Al ₂ O ₃ (%)	3.77 ± 0.17	1.08	5.05	12.6	7.56	50.78 (+Fe ₂ O ₃)	85.87 ± 4.98	28.40	1.35	11.69
CaO (%)	3.02 ± 0.25	0.67	1.00	9.7	6.7	2.79	0.81 ± 0.04	1.07	1.58	27.57
K ₂ O (%)	0.42 ± 0.23	0.31	3.82	0.5			0.41 ± 0.09	0.17		4.56
MgO (%)	1.37 ± 0.13	0.41	0.81	8.3	6.57	5.25	0.03 ± 0.01			6.56
SiO ₂ (%)	2.93	11.20	33.16	27.5	39.28	34.02	4.34 ± 0.93	57.70	92.77	28.82
Fe ₂ O ₃ (%)	41.02 ± 1.54	79.5	54.19	34.5	38.28	–	0.84 ± 0.24	6.22	5.19	18.20
Na ₂ O (%)			1.08	2.0			0.23 ± 0.06			0
ZnO (%)	0.00	2.06	0.75	2.5				0.01		2.60
Cr ₂ O ₃ (%)		0.10	0.04	0.5	1.20					
TiO ₂ (%)							5.64 ± 0.17			
MnO (%)	0.17 ± 0.02		0.05		0.50					
Cu (%)	0.85 ± 0.04	1.6	0.81	0.32						
As (mg/kg)	798 ± 88		630							
Cd (mg/kg)	23 ± 1		40							
Co (mg/kg)	228 ± 12									
Cr (mg/kg)	566 ± 71		280							
Mo (mg/kg)	2054	8200								
Ni (mg/kg)	113 ± 37		2110							
Pb (mg/kg)	853 ± 61	4270		1856.7						
SO ₃ (%)	0.21 ± 0.06	0.67		<0.1				0.19		
LOI (%)			5.05			0.26	0.03 ± 0.01			

Table 3
Chemical compositions (metals) of paint waste collected from ship maintenance (unit: mg/g).

Cu	Zn	Sn	Ba	Ni	Cr	Pb	Cd	Mn	Reference
296	6.70		1.950	0.329	0.234	0.217	<20 ^a		Momber (2003)
178	108	0.880		0.141	0.0686	0.421		0.232	Soroldoni et al. (2018a)
140	24.6			0.0665	0.0473	0.310		0.0388	
234	112			0.223	0.067	0.512		0.0560	
287	116	0.589		0.170	0.159	0.546		0.231	
311	114			0.149	0.0340	0.525		0.0750	Soroldoni et al. (2017)
234	137								
298	111			0.139	0.0320	0.518			
141	38.4			0.088	0.0370	0.534			
250	136	29.3							Turner et al. (2015)
329	110	24.6							
288	96.0								Holmes and Turner (2009)
176 ± 4.0	61.1 ± 1.5								Jessop and Turner (2011)
146 ± 2.3	33.1 ± 0.5								
311 ± 20.6 (<63 μm)	114 ± 7.66 (<63 μm)								Singh and Turner (2009)
284 ± 1.79 (63–180 μm)	108 ± 4.87 (63–180 μm)								
376.0 ± 8.4	158.5 ± 2.5	0.16 ± 0.01							Parks et al. (2010)
94.5 ± 110.5	53.8 ± 67.3	0.03 ± 0.05							
227.0 ± 6.7	138.2 ± 3.3	0.21 ± 0.11							
191.9 ± 50.6	74.7 ± 25.6	0.21 ± 0.03							
279.7 ± 4.7	77.1 ± 1.1	2.09 ± 0.11							
234.7 ± 10.6	78.0 ± 3.6	0.17 ± 0.02							
326.7 ± 4.1	113.3 ± 0.6	0.04 ± 0.01							
105.5 ± 6.3	34.0 ± 2.5	1.03 ± 0.20							
109.6 ± 28.9	29.1 ± 2.5	0.74 ± 0.21							

^a Concentrations of Cd, Se, Ag, and As were determined to be below 20 mg/kg.

application of organotin in antifouling coatings has been banned, it may still exist in some old ships; further, it might be used as a stabilizer. Similarly, chromate used to be added as a pigment and drier to the primer to prevent corrosion and enhance drying (Turner et al., 2015). Lead was previously used in anticorrosive paints, which has now been replaced by zinc and aluminum, and in antifouling paints as a stabilizer, pigment, and biocide at a typical concentration of 1–5 wt% (Johnsen and Engoy, 2000).

2.4. Thermal degradation property

Thermal degradation property is important for the thermal treatment of waste. The temperatures required for decomposition are dependent on the type of waste being processed. Commonly used mineral or slag abrasives have high melting points and decomposition temperatures. Therefore, the ABM waste itself is relatively heat-stable. The processing of ABM waste by thermal technologies is mainly aimed at destroying paint chips, and therefore, the thermal degradation behavior of paint chips is of concern. Most binders (such as epoxy, acrylic, and polyurethane resins) and other organic ingredients (such as coalescents, defoamers, and thickeners) in the coating can be decomposed at temperatures below 500 °C, whereas major inorganic pigments and extenders remain unaffected (Allahverdi et al., 2012; Duquesne et al., 2001; Dyakonov et al., 1996). Extenders such as dolomite and calcite usually decompose to metal oxides at temperatures above 700 °C. Metallic pigments (e.g., zinc dust and zinc/aluminum flakes) are heat-stable up to their melting point, and metal oxide pigments such as titanium dioxide and micaceous iron oxide are nonreactive with most chemicals and heat-stable up to melting point of over 1500 °C without transformation into different crystalline and amorphous phases (Karls-son et al., 2015).

3. Treatment methods for blasting waste

The considerable amount of ABM waste and paint waste generated from the ship repair industry is not a listed hazardous waste in most areas; however, it is still an environmental concern because of the requirement of landfill space and the presence of toxicants. Therefore, recycling or reusing blasting waste becomes an attractive option for reducing the amount/toxicity of waste; and/or generating valuable products. Fig. 3 and Fig. 4 summarize respectively the treatment methods of ABM waste and solid paint waste that have been reported in the literature. The treatment methods are described and evaluated in detail below.

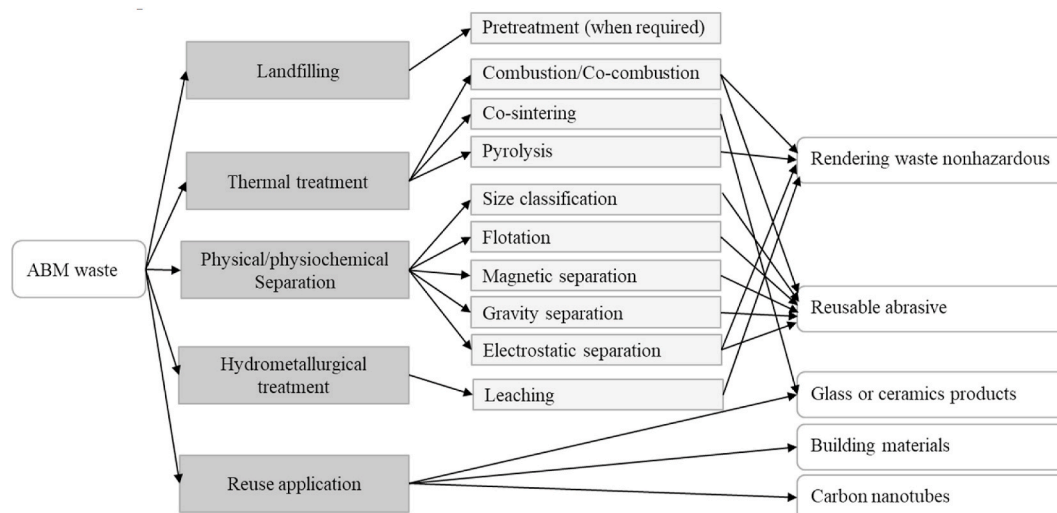


Fig. 3. ABM waste treatment methods reported in the literature.

3.1. Landfilling

The disposal of waste for nonhazardous or hazardous landfills is assessed according to the local, state or federal regulations. Waste is considered nonhazardous when it does not exceed the threshold limits, and no further actions must be taken before landfill disposal.

3.1.1. Landfilling of blasting waste

Traditionally, ABM waste has been stockpiled on-site, disposed of in landfills, or directly into the sea as a usual practice for small shipyards (Means et al., 1996; Taha et al., 2001). Although spent abrasive is not a listed hazardous waste in most areas, disposal by landfilling is costly and environmentally unfriendly. The landfill cost in the USA is \$50–\$250 per ton of ABM waste (Mattei and Khanfar, 2008); in Germany, the landfill cost for nonhazardous waste ranges from €60 to €220 per ton of waste, and it is 73–158 €/ton in Denmark (European Environment Agency, 2013). Paint waste, which is usually classified as hazardous, requires government-licensed transporters; and needs to be disposed of in permitted hazardous waste landfills, which leads to higher transportation and disposal costs. The costs of incineration and/or landfill disposal of paint waste range from 250 to 350 €/ton in Italy (Ruffino and Zanetti, 2010).

Some paint waste cannot be directly landfilled without pretreatment. For example, the direct landfilling of zinc-rich paint waste can easily produce a leachate of zinc above 500 mg/L, and this has a critical pollution impact on the environment (Momber, 2008). Therefore, research has been performed to assess the environmental impact of landfill disposal of blasting waste by investigating the long-term potential release of hazardous substances under landfilling conditions (Dung et al., 2014; Vaajasaari et al., 2004).

3.1.2. Potential release of heavy metals from abrasive blast media waste

The mobility and leachability of toxic metals have important environmental and biological implications for the landfill disposal of blasting waste. In general, the construction of landfills (such as bottom liner and top cover systems) and leachate characteristics such as pH, ionic strength, and total or dissolved organic carbon content, are likely to influence the release of metals. To the best of our knowledge, no study has been published on the in-situ monitoring of metal release from blasting waste in landfill sites. However, at the laboratory scale, the potential release of metals from blasting waste has been studied using a series of methods. For example, toxicity characteristic leaching procedure (TCLP) and synthetic precipitation leaching procedure (SPLP) have been used to simulate the leaching test under landfill conditions

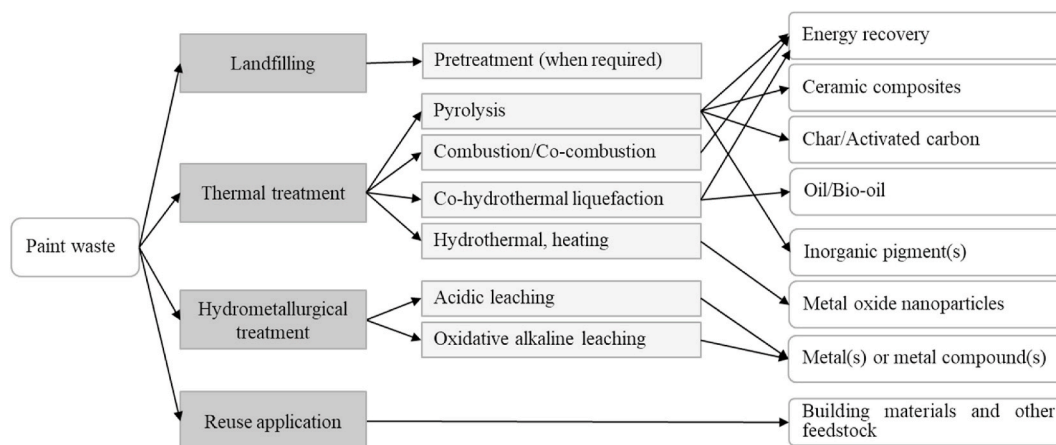


Fig. 4. Solid paint waste treatment methods reported in the literature.

(Wadanambi et al., 2008).

Dung et al. (2014) investigated the release of As and heavy metals (i. e., Cd, Pb, Zn, Cu, Ni, and Cr) from spent copper slag produced from ship blasting using a cascade leaching test (NEN 7349) and pH_{stat} leaching test (CEN/TS 14429). The leaching results and EU limits for non-hazardous waste are listed in Table 4. Most elements except Cd and Pb in the fraction of small particles (<1 mm) showed a higher release compared to those in the bulk sample because of the higher specific surface areas. Furthermore, the composition analysis of soil and sediment close to the dump site, 3 km from the shipyard, revealed that Cu and Cd were characterized by medium to high mobility under the influence of acidification in the solid samples (e.g., sediments/soils). Cu usually exhibited the highest release among all determined metals in the spent copper slag during leaching tests, which exceeded the EU regulatory limit for nonhazardous waste.

Madany and Raveendran (1992) investigated the leachability of heavy metals from spent copper slag produced by sandblasting tankers in a dry dock using three leaching media simulating different environmental conditions. As indicated in Table 4, the leaching of Cu, Zn, and Pb from the spent copper slag under acidification conditions (acid solution or synthetic landfill leachate) exceeded the EU limit for nonhazardous waste. Thus, if landfilling (or recycling) of this slag is considered, the release of Cu, Zn, and Pb needs to be focused on, especially at lower pH values.

3.1.3. Potential release of heavy metals from paint waste

The primary toxicity of paint waste is from antifouling paints, wherein copper, zinc, and biocides are the dominant contaminants. Therefore, the toxicity of paint waste and the safe disposal options are closely related to the leachability of Cu, Zn, and biocides (Amara et al.,

2018).

Vaajasaari et al. (2004) found that harmful compounds that remain in solid paint debris may be leached into the environment when the paint waste comes in contact with water at the landfill sites. A study suggested that the leaching of Cu and Zn from discarded paint particles was greater in rain water (pH = 4.7) than in tap water (pH = 7.3) because of the lower pH; further, after a period of 120 h, up to 3% of total Cu and 30% of total Zn was released (Jessop and Turner, 2011). The leaching of metals was also influenced by the salinity of the leachate. The spent antifouling paints have a greater Zn release into the water (pH = 7–8) at lower salinities, while greater Cu release was observed at higher salinities (Soroldoni et al., 2018b). This result was consistent with the study conducted by Singh and Turner (2009), in which an increased leaching of Cu and decreased leaching of Zn were found with increasing salinity. The difference in the leaching of Cu and Zn caused by salinity was consistent with the mechanisms governing the dissolution of Cu_2O in the presence of chloride and zinc acrylates in the presence of seawater cations. Further, the leaching of Zn and Cu is influenced by their existing forms (e.g., hydrophobic/hydrophilic, microsized/nanosized) in the discarded antifouling paints (Holmes and Turner, 2009). The leaching of hydrophobic Cu and Zn (normally organometallic booster biocides and potential complexes formed through interactions with hydrophobic ligands co-leached from the paint matrix) was sensitive to the presence of light. The absence of light resulted in a higher release of hydrophobic Cu, whereas the release of hydrophobic Zn was detected only in the presence of light. This was attributed to the zinc pyrithione released from the paint matrix being relatively rapidly transchelated into copper pyrithione in the dark. However, the more rapid degradation of copper pyrithione in light inhibited the transchelation. Therefore, copper pyrithione was predicted

Table 4

Leaching assessments of spent copper slag (unit: mg/kg).

Leaching test	Cd	Pb	Cu	Zn	Cr	As	Ni	Reference
Cascade leaching test ^a	<1 mm	0.23	0.2	10–85	4–10			Dung et al. (2014)
	<125 μm	0.04	0.4	10–20	2–5	0.5–1.5		
pH_{stat} leaching test ^b	<1 mm	0.24	2.3	173	16	<0.2	0.34	
	Bulk	0.26	2.3	146	12		0.26	
Distilled water (1 h)		0.010	0.050	1.065	0.285	0.01		Madany and Raveendran (1992)
Distilled water (24 h)		0.100	0.200	6.500	1.600	0.100		
Acid solution		0.120	10.00	1985	590.0	2.600		
Synthetic landfill leachate		0.160	26.60	2568	740.0	8.800		
EU non-hazardous landfill limits ^c		1	10	50	50	10	2	10
								EU (2003)

^a Leaching was performed with distilled water acidified to pH 4 and without any pH adjustment during the test (24 h, L/S ratios of 20, 40, 60, 80, and 100 (L/kg).

^b Leaching was performed with an L/S ratio of 10 L/kg, pH = 4 and was continually adjusted by the automatic addition of 1 mol/L HNO_3 solution, as a function of leaching time (up to 168 h).

^c Values given in mg/L and mg/kg are determined from the percolation test at the L/S ratio of 0.1 L/kg (EN 14405) and batch test at the L/S ratio of 10 L/kg (EN 12457/1–4), respectively.

to be more long-lasting and intrinsically more toxic than zinc pyrrithione; and it was known to affect microbial diversity and processes.

The potential release of other toxicants such as Cu_2O nanoparticles, SiO_2 nanoparticles, and lead has been studied (Adeleye et al., 2016; Wadanambi et al., 2008; Zuin et al., 2014). The European standard EN 12457-3 aimed at compliance testing of waste destined for landfills was used to study the leaching properties of SiO_2 nanoparticles from the removed paint residues. The results suggested that the pigment volume fraction could be a crucial factor for the release of SiO_2 nanoparticles, and a high portion of the binder formed a suitable matrix to hold SiO_2 nanoparticles in the paint (Zuin et al., 2014). Lead leaching from lead-based paint was investigated using the leachate from simulated (TCLP, SPLP) and actual landfill environments (three municipal solid waste (MSW) landfills and three construction & demolition (C&D) waste landfills). The pH and presence of other materials (e.g., removed paint substrates and spent abrasives) were proved to have a significant influence on lead leaching (Wadanambi et al., 2008). The leaching of Pb and Cr from paint waste was reduced in the presence of steel grit. This may be because metallic iron could reduce metal compounds in the paint waste to less soluble forms, or the metals were adsorbed to iron oxides formed on steel surfaces (Shu et al., 2015).

Nevertheless, simply equating laboratory-scale tests (e.g., TCLP and SPLP) to conditions in an actual landfill may not be appropriate for ship blasting waste. Additional work is required to gain an insight into phenomena such as sorption, entrainment, precipitation, and biological reduction occurring in actual landfill sites.

In addition to the potential environmental impact, the main challenges for landfill disposal are rising costs, restrictions or bans for landfill disposal, and growing emphasis on waste minimization. Therefore, environmentally friendly alternatives have been investigated for the treatment of blasting waste.

3.2. Thermal treatment

Thermal treatment aims to destroy the toxic organic portions in the waste and reduce the volume of solid waste to be disposed. Pyrolysis, combustion, and hydrothermal processes are the main thermal processes available for treating blasting waste.

3.2.1. Abrasive blast media waste

The main purposes of the thermal treatment of ABM waste are to 1) destroy the toxic organic components and 2) vitrify hazardous components by incorporating them into glass or ceramic products.

The destruction of toxic organic components is often achieved by heating ABM waste to a temperature of 340–1200 °C under an air atmosphere. Under these conditions, the paint chips in the ABM waste, which contain antifouling agents (e.g., organotin compounds and other organometallic booster biocides), are converted to carbon dioxide, water vapor, and metal oxides, which can be further recycled or safely disposed (Ticker, 1976). Song et al. (2005) found that an ABM waste containing organotin compounds (including tributyltin (TBT), dibutyltin, and monobutyltin) could be effectively detoxified when heated to 1000 °C. Hazardous substances including tin oxides were collected by filters during flue gas treatment and the removal efficiency of organotin compounds was up to 99%. The produced tin-free product can be potentially reused as a building material or as cover soil at a landfill site without environmental concerns.

The recycling of ABM waste was investigated by introducing fluidized-bed sloped-grid (FBSG) technology (Bryan et al., 1991). The developed FBSG reclamation system (Fig. 5) contains a screw feeder, a fluidized bed reactor with an operation temperature of 650–820 °C, and a cyclone for the collection of fine particles that are carried out of the reclaimers with the effluent gas for further treatment and/or disposal. The system provides drying, thermal treatment, and size classification of contaminated solids in a single-stage fluidized-bed reactor. The performance of the thermal reclamation system for organics destruction and

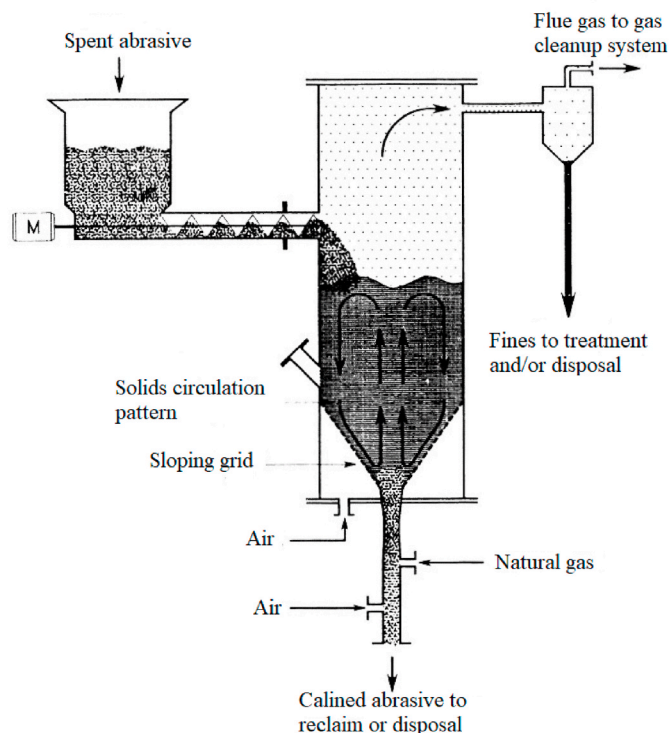


Fig. 5. Fluidized-bed sloped-grid reclaimer (Bryan et al., 1991).

fines separation was tested in a 0.9-m-diameter, pilot-scale FBSG reclaimer with a feeding rate of approximately 0.7 ton/h. The pilot-scale test results showed that both organic contaminants and fines were effectively removed from the spent abrasive. Approximately 80% of the spent abrasive could be reused, which meets the required specifications including California Title 22 Hazardous Waste Tests and standard industry practice that all particles should be larger than 80-mesh. The reclaimed abrasives provided comparable coating removal efficiencies and surface preparation to new abrasives. Economic analysis indicated that the system would be cost-effective in processing both hazardous and nonhazardous spent abrasives.

ABM waste containing predominantly silica and/or alumina with low levels of metal contaminants can be processed thermally to form glass or ceramic products, or be used as raw materials in ceramic manufacturing (Mymrin et al., 2014; Rodriguez et al., 2016). A sandblasting waste, mainly composed of Al_2O_3 , was recycled into the water-retention porous ceramics by co-sintering with waste diatomite at 1000–1270 °C with a sandblasting waste replacement level of up to 20% (Ho et al., 2019). The resulting porous ceramic product displayed good comprehensive strength, water absorption, and retention capacities, and it was favorable in urban road pavements to counteract heat island effects (Ho et al., 2017).

3.2.2. Paint waste

Paint waste is thermally treated 1) for energy recovery; 2) to produce chemical products such as activated carbon, oil, and ceramic composites; and 3) for the recovery of inorganic pigments (e.g., TiO_2).

Co-processing in a cement kiln was reported to be an effective, environmentally friendly, and safe waste management technology. The high temperature and long residence during the cement process facilitated the complete destruction of the waste. The co-processing trial of paint sludge conducted by a cement company proved that the paint waste could be safely co-processed in cement kilns in terms of harmful emissions such as dioxins, furans, particulate matter, acidic gases, and heavy metals (Gautam et al., 2010).

Dried paint solids have been investigated for energy recovery

through co-combustion with coal in an electrically heated alumina tubular reactor at 1200 °C (Suriyawong et al., 2009). Besides, pyrolysis has been reported as a reclamation method to convert paint waste into gaseous (e.g., carbon dioxide, methane, ethane, propene, or their isomers), liquid (e.g., benzene, amides, melamine, aliphatic esters, and hydrocarbons from C6 to C13) and solid products (e.g., titanium dioxide, alumina, quartz) (Alves et al., 2018). Product yields are influenced by processing conditions such as reaction time and temperature (Muniz et al., 2003). The pyrolysis of a dried paint sludge waste from overspray was conducted in a quartz tube for 2 h at 600 °C under a N₂ atmosphere. About 36.5 wt% of the paint sludge was evolved as offgases that can be reused as fuel for pyrolysis, while the solid residues containing mainly CaTiO₃, BaTiO₃, and TiO₂ were further sintered at 1000 °C for 4 h under N₂ and NH₃ atmospheres, respectively, to produce ceramic composites that can be used as reinforcing fillers in applications such as the production of representative metal matrix composites and reinforced plastic components (Nakouzi et al., 1998).

Compared with conventional pyrolysis, microwave-assisted pyrolysis can deal with high moisture content and facilitate hydrocarbon chain breakdown. An automobile paint sludge waste was characterized and showed a potential for chemical and energy recovery via microwave-assisted pyrolysis (under temperatures from 260 °C to 500 °C) using a wet paint waste sample in water which acts as an absorber to microwave radiation (Januri et al., 2015a). With the use of a modified conventional kitchen microwave oven, the optimum yield of liquid oil was achieved at 200 g of sample loading, 1000 W of microwave power, and 30 min of radiation time with 8.9%, 0.27%, 26.13%, and 64.7% for solid char, liquid oil, aqueous, and gas products, respectively (Januri et al., 2015b). In another study, the same authors reported that, at a sample loading of 500 g and microwave power of 1000 W, up to 40% of the liquid product (consisting of aqueous and liquid oil) could be recovered from automobile paint sludge waste via microwave pyrolysis. The liquid product could potentially be used as fuel owing to its high energy content (22.6 MJ/kg) (Januri et al., 2014). Microwave absorbers, such as activated carbon, may be used for paint waste with low moisture content. Rahman et al. (2017) applied activated carbon as a microwave absorber in the microwave pyrolysis of automotive paint at 600 W of microwave power and 30-min radiation time. The results indicated that the application of the microwave absorber switched the major pyrolysis product from the gas phase to the liquid phase (total of aqueous and oil), and the highest oil yield was obtained with the addition of 10% activated carbon (from paint waste), whereas slightly different char yields were observed.

Hydrocarbon-rich paint waste can be used to produce high-surface-area activated carbon and oil. Kim et al. (1996) investigated the pyrolysis of dried paint sludge (mixed with 50 wt% potassium hydroxide) at 600 °C for 30 min to prepare activated carbon using an electrically heated furnace. The produced char was useful for volatile organic compound (VOC) absorption during the spray-painting process. This pyrolysis process for converting the paint sludge to activated char recovered approximately 10% of the total mass of the dried paint waste.

Jayakishan et al. (2019) investigated the production of bio-oil through the hydrothermal liquefaction of paint waste with *Prosopis juliflora* (biomass) at different temperatures and at biomass-to-paint waste ratios under N₂ (5 MPa) atmosphere for a holding time of 60 min. A maximum bio-oil yield of approximately 50 wt% was achieved at 420 °C, biomass-to-paint waste ratio of 2:1 and 4 wt% of bentonite catalyst. The energy and carbon recovery of the bio-oil was approximately 70% and 96%, respectively.

Furthermore, it is possible to recycle valuable components such as titanium dioxide from paint waste. A thermal recycling process involving pyrolysis, oxidation, and purification (Fig. S1 in Supplementary Materials) was developed to recover TiO₂ pigments from the decorative paint waste (Karlsson et al., 2018). A model waste paint with approximately 30.5 wt% TiO₂ was subjected to a pyrolysis retort with inner dimensions of 300 × 300 × 150 cm. After drying, the dried paint was pyrolyzed under atmospheric pressure at 500 °C under a N₂ flow

rate of 0.85–0.95 L/min. The subsequent oxidation of the pyrolysis char was conducted to remove the carbon residues and nonvolatilized organic materials. The recycled pigment was further purified by washing with ion-exchanger resins to remove unwanted salt residues, and the purified pigment was separated from the ionic exchangers via sieving. After drying and homogenization, the recycled TiO₂ pigment was incorporated into fresh paint production. The evaluation of the fresh paint showed a small decrease in paint quality; however, it was acceptable in applications with lower demand for surface finish and gloss (Karlsson et al., 2019a,b).

Bharti et al. (2014) developed a synthesis method (Fig. 6) of TiO₂ nanoparticles from automobile paint sludge waste (containing TiO₂ > 35 wt% with several surfactants, organic polymers, and ~2–10 wt% inorganic matter depending on the type of paint used) for producing sustainable organic polymer solar cells. In this method, a hydrothermal treatment was conducted to convert micro-sized TiO₂ particles into nano-sized particles. All surfactants and other organic impurities were removed by heating in a specially designed reaction vessel, followed by acid treatment to remove inorganic impurities. Recycled TiO₂ nanoparticles with a size of ~100 nm were incorporated as an electron transport layer, thereby increasing the power conversion efficiency of the fabricated polymer solar cells.

3.3. Physical and physicochemical separation

Paint chips are the main source of contaminants in ABM waste. Physical and physicochemical methods, including sieving, air classification, magnetic separation, flotation, and electrostatic separation, have been developed to separate paint chips from ABM waste to reduce the hazardous waste to be disposed of and/or to recycle reusable abrasives.

Sieving is based on particle size differences; air classification and flotation are based on the difference in physical properties such as density, size, and shape of particles. Magnetic separation removes iron-bearing magnetic materials, such as steel particles, from the spent abrasive. Electrostatic separation separates nonmetallic abrasives from fine nonconducting paint particles based on their different surface electrical conductivities. The aforementioned methods are often

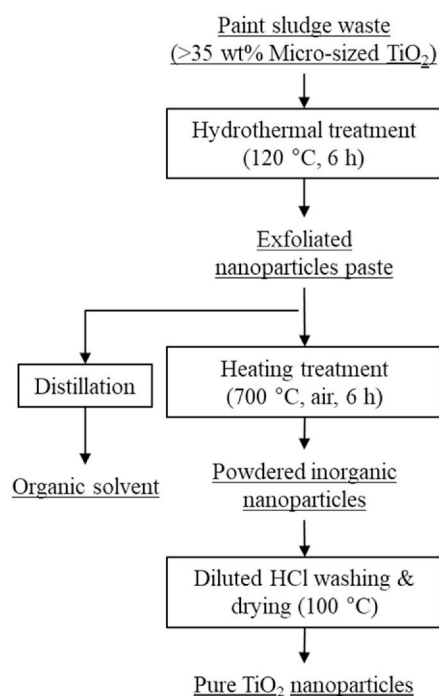


Fig. 6. Recycling flow chart of TiO₂ nanoparticles from paint sludge waste (Bharti et al., 2014).

combined to separate fine particles and paint chips from reusable abrasives.

Katsikaris et al. (2002) recycled spent ferrous-nickel slag generated from blast cleaning in a shipyard by sieving it with a two-deck vibrating screen, i.e., a 12-mesh top screen frame (1.4 mm) and an 80-mesh bottom screen (0.17 mm), to separate reusable fractions from the disposable material. Laboratory tests showed that the recycling of ferrous-nickel slag was feasible for three life cycles (one life cycle was determined as a sequence comprising four steps: blast cleaning → collection of used ferrous-nickel slag → sieving–separation of reusable material from spent slag → blasting with reclaimed slag.). A reclamation yield of over 80% of the reusable ferrous-nickel slag was achieved in each cycle. The recycled slag was incorporated with fresh slag for blasting and showed satisfactory performance. A semi-industrial-scale study further supported that it is a profitable recycling method for shipyards.

Hwang and Jeong (2001) investigated the separation of paint chips and other contaminants from ABM so that a majority of the ABM waste can be rendered nonhazardous. The recycling process (Fig. S2 in Supplementary Materials) that includes mainly magnetic separation, desliming, and acid washing, was developed to remove the paint contaminants (Ba, Cd, Cr, and Pb existing in the finest fractions (325 mesh)) effectively from a glass bead ABM waste. The initial magnetic separation was conducted to remove and minimize the metal content, thereby reducing the subsequent acid consumption. Desliming is a wet screening process for the separation of coarse and fine particles. The deslimed material was further washed with a 0.01% H₂SO₄ solution to reduce the Cd content. The pilot test showed that 82% of the ABM waste was rendered nonhazardous.

A patented process (Fig. S3 in Supplementary Materials) involving size classification and flotation can separate paint contaminants from spent alumina abrasives (Jackson, 1974). After the sequential processes of size classification, washing/cleaning with liquid detergent, flotation, and heating (at approximately 650–700 °C), the reclaimed alumina abrasive was finally restored to a high level of purity and can be reused in metal cleaning or blasting processes where low contamination is required. A similar separation process that includes size classification, ferromagnetic separation, and electrostatic separation was disclosed for the recovery of nonmetallic ABM waste (such as slag, sand, garnet, and steel shot) contaminated with paint chips (Gilbert and Weyand, 1990). After the entire separation process, approximately 72% of the spent abrasive was rendered nonhazardous, and could be recycled, reused, or disposed of as a nonhazardous material. The remaining hazardous paint particles (approximately 28% of the original spent abrasive) were then packaged for storage, disposal, or possible further treatment.

3.4. Hydrometallurgical treatment

Hydrometallurgical technology uses acids, bases, aqueous solutions or other solvents to selectively dissolve metal(s), and recover or remove the desired metal(s) from the waste through precipitation, electrolysis, solvent extraction or ion exchange. The hydrometallurgical treatment of blasting waste is conducted to recover valuable metals or metal compounds or remove hazardous species and render waste nonhazardous.

The spent ABM is contaminated by paint chips containing toxic metals such as Pb, Cr, Cu, and Zn. Yang and Kravets (2000) reported that more than 85% of Cr⁶⁺ was effectively leached out from spent abrasives by aqueous solutions of HCl, H₂SO₄ or HNO₃ at pH 2. Three stages were required to leach all of the Cr⁶⁺; it took 3–5 min to reach the equilibrium concentration in each stage. Subsequently, Cr⁶⁺ was successfully removed via electrochemical precipitation from the leachate. The consumable Fe electrodes generated ferrous ions to reduce Cr⁶⁺ to Cr³⁺, which was removed by precipitation as Cr(OH)₃.

In addition to the removal of toxic metals from ABM waste, hydrometallurgical processes have been applied to extract inorganic pigments from paint waste. A study reported that TiO₂, which is the main pigment in an automobile paint sludge, was extracted using acidic digestion,

followed by purification via centrifugation and heating (Khezri et al., 2013). The water-based paint sludge waste was initially heated at 850 °C to remove organic materials. Further, it led to changes in the nature and combinations of some harmful elements; for example, Mg₃O(CO₃)₂ and MgCO₃ were converted to MgO during heating. Subsequently, chemical digestion with sulfuric acid was conducted to remove heavy metals such as Ba, Sr, Cr, Mg, Cu, and Pb, and the percentage of Ti increased from 35 wt% to 59 wt%. After filtration, the powdered deposit (containing mostly Ti) was further purified by centrifugation because TiO₂ has a higher density than the main inorganic impurities Al₂O₃ and Al₂(SO₄)₃. The recycled rutile TiO₂ pigment had a high purity degree of 67.4% TiO₂, and the fresh paint produced with this recycled TiO₂ pigment showed equal gloss, hardness, and washability as commercial industrial paint.

Zinc-rich paint is widely used for corrosion protection of industrial constructions and marine applications, and the paint waste produced is considered a potential source of zinc extraction because of its high content of zinc (both metallic zinc and zinc oxide). For instance, a dried zinc-rich paint sludge from a spraying factory may contain approximately 46 wt% Zn. A hydrometallurgical route through oxidative alkaline leaching and electrowinning (Fig. 7) was developed to recover zinc from dried zinc-rich paint residue (Xing et al., 2018). In this method, zinc and zinc oxide were almost completely dissolved during the oxidative alkaline leaching process. Other impurities such as calcium, magnesium, silica, titanium, and iron, remained in the leaching residue. Following leaching, electrowinning was conducted to recover the zinc powder from the zinc-bearing leachate. A leaching rate of 98% and metallic zinc powder with a purity of 99.8% were obtained under optimal conditions. Moreover, the depleted electrolyte could be returned to the leaching process for zinc extraction. The resulting leaching residue, which contains mainly organic compounds and some metal impurities, may be properly disposed by co-processing in a cement kiln, which not only reduces the secondary environmental pollution but also recovers energy.

3.5. Direct utilization of blasting waste

ABM waste contains approximately 90% aluminum oxide, calcium oxide, silicon dioxide, and iron oxide, which is less toxic and can potentially be utilized directly as an alternative to landfill disposal. Previously, ABM waste was shipped to a cement company that would reuse the waste in its products, but only if the waste was not classified as hazardous. Shipyard could thus save substantially on disposal costs for

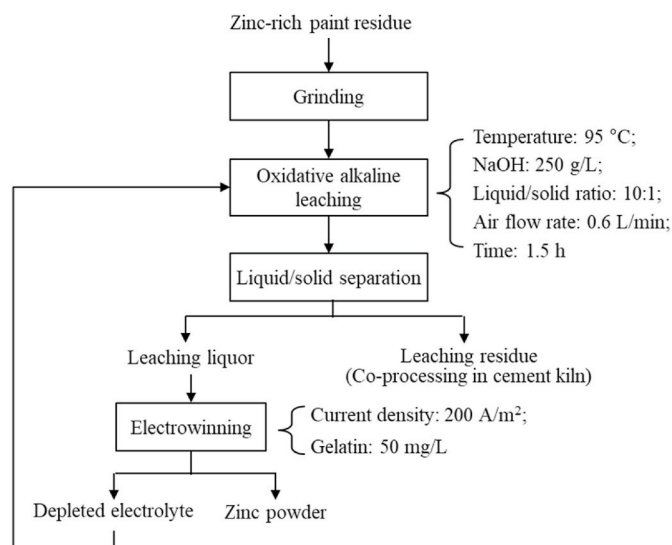


Fig. 7. Flowsheet for recycling zinc-rich paint residue (Xing et al., 2018).

the waste by treating the waste with iron filings so that it no longer exhibited the toxicity characteristics (Webster, 1994). However, this is gradually restricted because of more stringent waste management regulations and heightened awareness of environmental contamination.

Lim et al. (Lim and Chu, 2006) directly utilized spent copper slag as a fill material for land reclamation. The material displayed good physical and geotechnical properties without environmental concerns, and areas with limited land could benefit a lot from land reclamation.

In addition, ABM waste has been widely explored for reuse as a full/partial fine aggregate substitute in the production of building materials such as concrete, asphalt, mortars, and bricks (Means et al., 1996). This alternative has two advantages: 1) it is economically cheaper than landfill, and 2) it is environmentally safe, with toxic ingredients being immobilized by an asphalt system or Portland cement within the final product of building materials (Brabrand and Loehr, 1993; Buruiana et al., 2013; Webster and Loehr, 1996).

When reusing ABM waste as a substitute for normal fine aggregates, the precondition is that the waste must comply with both performance and environmental standards, and it is important that the waste exhibits physical and chemical properties comparable to fine aggregates. Table 5 summarizes the reuse applications of ABM waste in the production of building materials. The final products are always safe for use as building materials in terms of their level of toxicity, regardless of the total and leachable hazardous metal concentrations, because toxic metals are immobilized into the cement/concrete matrix. However, only a limited replacement content of ABM waste is optimized to obtain the desired product quality. The composition and impurities of the ABM waste has a significant effect on the product quality, and the addition of ABM waste results in increased setting time and decreased strength (Borucka-Lipska et al., 2019; Muttashar et al., 2018). In addition to the specified requirements/limitations summarized in Table 5, some common influences/limitations need to be considered when recycling ABM waste into the production of building materials (Khanfar, 2007; Means and Heath, 1996; Peride et al., 2000):

- Very fine and round-shaped ABM waste particles are undesirable because they contribute to poor mixing/wetting capabilities and/or do not add considerably to the structural integrity of the final product.

- Cu, Pb and Zn compounds in the blasting waste may act as a retarder in the hydration reactions due to their set-inhibiting properties, which causes longer setting times and lower strengths.
- Hydration of the ferrous oxides could cause unacceptable levels of shrinkage of the concrete.
- Al in the blasting waste reacts with alkalis in the cement to produce hydrogen gas, and the elevated Al content can lead to swelling and cracking, which results in lower strengths and more permeable products.
- Anions such as chloride, sulfate, or metallic iron are undesirable in asphaltic concrete because they can lead to swelling and premature cracking upon oxidation. Additives may be needed to solve the problem; for example, silica fume and calcium nitrite are effective for reducing the set times and increasing compressive strength.

In addition to the abovementioned reuse applications, Sasai et al. (2009) proposed the possibility of recycling spent SiC abrasive powders as a substrate resource in the synthesis of carbon nanotube (CNT) particles using the surface decomposition method. However, pretreatment such as acid rinsing and preheating is required to remove impurities such as Fe species and lattice defects on the surface; this can affect the formation of CNTs.

Paint sludge waste, which is often a hazardous waste, has been reported to be reused in the production of building materials (Burande, 2017; Salihoglu and Salihoglu, 2016; Souza et al., 2014). Avci et al. (2017) utilized paint sludge waste as an aggregate replacement for sand and lime admixtures in cement and lime, and the unpleasant odor was quickly removed by mixing paint sludge with cement/lime. The final structure showed potential characteristics for lightweight residential construction materials, although it exhibited a relative decrease in strength. Zanetti et al. (2018) recycled paint sludge as a modifying agent for the production of bituminous binders for paving applications. A comprehensive study of the associated economic costs and environmental effects showed that paint waste is an attractive low-cost substitute for virgin bitumen (400 €/t), and the life cycle assessment revealed that the CO₂ emissions were reduced by 39% when hot mix asphalt was produced with a modified binder containing 20% paint waste (Ruffino et al., 2020).

Table 5

Re-use applications of ABM wastes in the production of building materials.

Application	Product	ABM Waste	Requirements/Limitations	Reference
Fine aggregate substitute	Asphalt concrete/mixes	Spent steel grit, spent copper slag, spent beach sand, spent coal slag, spent silica sand	Necessary grit granulation and Si and iron oxides concentrations. A maximum ABM concentration of (7–10 wt%) of the final asphalt concrete is used based on quality control.	(Al-Sayed and Madany, 1992; Buruiana et al., 2011; Khanfar, 2007; Mattei and Khanfar, 2008; Means et al., 1991, 1996)
Iron ore substitute and providing additional silica	Portland cement	Contaminated ABM waste, spent copper slag	High iron content (advantage of spent copper slag). Typical percentage of spent ABM in the final cement product is ~0.3–2.0%. Metal emissions during cement production should be well within permitted levels.	(Salt et al., 1994, 1996, 1994; Sua-Iam and Makul, 2013)
Fine aggregate substitute	Portland cement concrete	Spent copper slag	Low total and leachable hazardous metal concentrations in the final concrete product.	Anjos et al. (2017)
Cement/Fine aggregate substitute	Concrete mortar	Spent copper slag, contaminated ABM waste	Reduction in the strength of mortar and works for only small quantities of the waste (e.g., about 5% of spent copper slag). Product is for unreinforced applications such as concrete kerbs for road works and concrete blocks in general.	(Borucka-Lipska et al., 2019; Madany et al., 1991; Zain et al., 2004)
Fly ash substitute	Geopolymer mortar	ABM waste	Workability of geopolymer is improved due to the high Si content of the waste; however, the substitution reduces compressive strength.	Tajunnisa et al. (2019)
Fine aggregate substitute	Self-compacting concrete	Spent alumina, spent garnet	Addition of waste no more than 75 wt%.	(Muttashar et al., 2018; Sua-Iam and Makul, 2017; Sua-Iam and Makul, 2013)
Fine aggregate substitute	Bricks	Spent copper slag	Meet the specifications for strength and absorption, and the metals are immobilized chemically stable.	Salleh et al. (2014)

4. Comparative assessment of different treatment methods and future perspectives

Although landfills have been a common option for treating blasting waste for a long time, the rising cost of landfills, limited land resources, and stringent legislation for solid waste recycling have resulted in technological developments for aimed at blasting waste treatment. Over the past decades, various blasting waste treatment methods have been developed. However, none of these is versatile because of the inconsistent waste characteristics caused by the different coating removal techniques and applied coating systems. Table 6 summarizes the advantages and disadvantages of the blasting waste treatment methods reported in the literature.

Among these treatment methods, the utilization of blasting waste in the production of building materials is attractive owing to its economic benefits and low technical complexity. However, the utilization of blasting waste in the production of building materials is often restricted by the mechanical and physical properties of blasting waste such as the composition and particle size/shape and the required compressive strength of the target product. Further, physical/physicochemical separation technologies are economically attractive, but they are restricted for the recycling of reusable abrasives, whereas the separated paint waste and nonreusable abrasives need to be further treated by other methods. Thermal treatment effectively reduces waste volumes; however, it is associated with economic and environmental issues because of the high energy consumption and air pollutant emissions in need of control. This can change in the future when cheap renewable energy becomes more readily available. Compared to other thermal treatment methods, pyrolysis can produce a range of useful hydrocarbons that are potentially used as chemical feedstocks or fuels. In addition, microwave-assisted reactors are a good alternative to minimize operational costs by facilitating the full degradation of feedstock under mild conditions. Nevertheless, thermal treatment is not favorable for ABM and paint wastes that have a high content of inorganic components. Hydrometallurgical leaching processes are mainly performed via alkaline or acidic routes to extract toxic metal components; however, they produce secondary liquid waste and solid residual. Because ABM waste generally contains 90% aluminum oxide, calcium oxide, silicon dioxide, and iron oxide, the low solubility of these oxides in acid/alkaline solutions inhibits the metal/metal compound recovery from most ABM waste via hydrometallurgical methods. Further, nonrecyclable materials after physical/physicochemical separation, ultimate solid residual after heat treatment, and leaching residual from the hydrometallurgical process need to be managed eventually.

With UHP water jetting being increasingly used in shipyards, more

attention should be paid in the future to waste management after this process. UHP water jetting generates a smaller quantity of waste; however, it is more hazardous solid paint waste accompanied by a large volume of wastewater. Wastewater is typically contaminated with toxic metals such as Cu, Zn, Ni, and Cr depending on the applied coatings and rust inhibitors (e.g., sodium nitrite) used in the blasting process. Many conventional and novel wastewater treatment methods can be applied in on-site or off-site wastewater treatment to remove dissolved toxic species (Arevalo and Calmano, 2007; Awual, 2015, 2019, 2015; Awual et al., 2019; Walker et al., 2005; Znad et al., 2018) Further, the solid paint waste, although usually hazardous, is a valuable source of metals such as Cu and Zn according to the chemical composition of the applied coatings. If the paint waste is enriched with hydrocarbons, thermal treatments such as pyrolysis and combustion are good options that significantly reduce the volume of waste and produce valuable products such as heat, power, fuels and chemicals. If the waste contains mainly inorganic pigments, thermal and hydrometallurgical methods are promising for the recovery of metal/metal compounds. Although considerable research has been performed on the treatment of paint waste produced from overspray, little research has been conducted on paint waste that is specifically produced from the coating removal process. Despite similar characteristics and compositions, further studies on the treatment of paint waste from the coating removal process are important in the future, especially with the development of more nonabrasive blasting methods. Moreover, in the future, the development and application of sustainable marine coatings will vary the composition of paint waste produced in shipyards. For example, in the past, the worldwide application of TBT was phased out and replaced by Cu-based biocides, and biocide-free antifouling paints have been developed in recent decades (Yebra et al., 2004). The future development of waste treatment and disposal facilities should therefore be adapted to these changes.

Further, the comparison and evaluation of treatment methods and consumer demand for high-end products imply an increasing interest in recycling blasting waste for high-value products/applications. However, the profitability of most of these recycling strategies needs to be evaluated before industrialization.

In addition, the prevention or reduction of the generation of blasting waste and reduction of the toxicity of blasting waste are good management strategies. To this end, additional research is required to develop scalable, cost-effective and environmentally friendly coating removal techniques to reduce the generation of blasting waste. Improvements in paint and coating chemistry are also desirable; for instance, to develop novel marine coatings that are tolerant to poor surface profiles and can be used in conjunction with alternative non-

Table 6
Comparison of various blasting waste treatment methods.

Method category	Process	Advantages	Disadvantages
Landfilling	Pretreatment when required	Simple process, filled land may be reused for other community purposes	Environmental issues, rising cost, limited land resources, non-organic waste continues to accumulate
Thermal treatment	Pyrolysis, combustion, sintering, hydrothermal process	Reduce the volume of waste, efficient destruction of toxic organics, vitrify hazardous components by incorporating into the glass/ceramic products, energy/chemicals recovery	High energy consumption, air pollutant emissions, e.g., dioxins in-need of control
Physical/ Physicochemical separation	Size classification, magnetic separation, flotation, gravity separation, electrostatic separation	Recycle reusable abrasive during the process, remove hazardous species	Not feasible for friable slag-type abrasives and paint waste
Hydrometallurgical treatment	Acid leaching, alkaline leaching	High-purity product, recover valuable chemicals, remove hazardous species	Generation of liquid waste and solid residual
Utilization in building materials	Filler and aggregate, cement replacement, other chemical feedstocks	Economical and simple process	Requirements for product quality control and environmental standards, low-value product (value-added product is obtained only for specific feedstock)

abrasive blasting methods and to reduce the toxicity of blasting waste by developing more environmentally benign marine coatings.

5. Conclusions

Blasting is widely used in the ship repair industry, which produces a large quantity (40–50 kg/m² of the blasted surface) of blasting waste containing heavy metals and antifouling paints. Dry abrasive blasting, using natural minerals, mineral slags, or metallic grits as blasting media is the predominant technique applied in the ship repair industry because of its advantages of low cost and effective production. Wet blasting methods such as hydraulic blasting, slurry blasting, and UHP water jetting, are increasingly used to reduce dust emissions and spent abrasive generation.

The analyses of the physical properties, chemical compositions, and hazardness of the blasting waste indicate that ABM waste from abrasive blasting is usually classified as nonhazardous because of the dilution of a large amount of abrasive media, although it contains a certain amount of heavy metals. Paint waste (with rust) is the only solid waste produced from UHP water jetting. Its volume is considerably lower than that of ABM waste; however, concentrations of toxic metals such as Zn and Cu are much higher and it is usually categorized as hazardous.

The technologies for the treatment of ABM waste and paint waste were reviewed. ABM waste is generally disposed of in nonhazardous landfills. Alternative treatment methods including thermal detoxification, physical/physicochemical separation, leaching, and reuse applications in the production of bricks, mortars, asphalt, concrete, cement, and glass ceramics, have been explored to detoxify and recycle blasting waste into valuable products. The review suggests that the utilization of ABM waste in the production of building materials is a technically feasible, economically sound and environmentally friendly option, although there are some restrictions in terms of the physical and chemical properties of ABM waste. The hydrometallurgical extraction of metal(s) is a good option for pretreatment if ABM waste is toxic because of the presence of heavy metal(s). The waste can be pretreated through a thermal process if it is contaminated by the organometallic components and other organics in the paint chips.

Based on this review, we anticipate that less ABM waste but growing quantities of paint waste will be generated in the future, because of the increasing use of UHP water jetting. In addition, there is a need to develop technologies to facilitate the recycling and treatment of paint waste. The review implies that for hydrocarbon-rich paint waste, thermal treatment such as pyrolysis is a good option for energy recovery or recycling of carbon-related products (such as activated carbon and bio-oil). Hydrometallurgical and thermal methods are promising approaches for the recovery of metal/metal compounds (such as zinc, copper and titanium) from the paint waste containing mainly inorganic pigments. A lack of fundamental studies on the environmental impact and thermal degradation behavior of blasting waste has been observed. Future efforts are required to develop metal recovery processes, and the development of waste treatment and disposal facilities should be adapted to changes in marine coatings.

Finally, efforts need to be invested towards minimizing the generation and toxicity of blasting waste, including the use of less hazardous materials in the manufacture and production of marine coatings and the development of alternative blasting methods.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

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