Research Advances on the Use of Solid Wastes in Concrete and Asphalt

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LIST OF ABBREVIATIONS, ACRONYMS & UNITS OF MEASUREMENT

AAR	Alkali Aggregate Reaction			
AASHTO	American Association of State Highway and Transportation Officials			
ACR	Alkali Carbonate Reaction			
AES	Atomic Emission Spectrometry			
AMR	Advanced Metal Recovery			
APA	Asphalt Pavement Analyzer			
ASR	Alkali-Silica Reaction			
ASTM	American Society for Testing and Materials			
BA	Bottom Ash			
CFA	Coal Fly Ash			
ECS	Eddy Current Separator			
EP	Extraction Procedure			
ESAL	Equivalent Single Axial Loads			
FA	Fly Ash			
FDEP	Florida Department of Environmental Protection			
FDOT	Florida Department of Transportation			
FHWA	Federal Highway Administration			
FM	Florida Method			
GHG	Greenhouse Gas			
GP	Glass Powder			
HDPE	High Density Polyethylene			
НМА	Hot-Mix Asphalt Concrete			
HPY	Hours Per Year			
ICP	Inductively Coupled Plasma			
IDT	Indirect tensile strength test			
JMF	Job Mix Formula			
LCA	Life Cycle Assessment			
LEAF	Leaching Environmental Assessment Framework			
LOI	Loss on Ignition			
L/S	Liquid-to-Solid			
MB	Mass Burn			
MSW	Municipal Solid Waste			
MSWI	Municipal Solid Waste Incinerator			
NMAS	Nominal Maximum Aggregate Size			
PCC	Portland Cement Concrete			
PG	Performance Grade			
PVC	Polyvinyl Chloride			
QA/QC	Quality Assurance / Quality Control			

RAP	Recycled Asphalt Pavement		
RCRA	Resource Conservation and Recovery Act		
RDF	Refuse-Derived Fuel		
RSL	Regional Screening Levels		
SCM	Supplementary Cementitious Material		
SCTL	Soil Cleanup Target Level		
SEM	Scanning Electron Microscopy		
SHRP	Strategic Highway Research Program		
SMA	Stone Matrix Asphalt		
SMI	Strategic Materials Inc.		
SPLP	Synthetic Precipitation Leaching Procedure		
S/S	Solidification/Stabilization		
SSD	Saturated Surface Dry		
Std. Dev.	Standard Deviation		
Superpave	Superior Performing Asphalt Pavements		
ТС	Toxicity Characteristic		
TCLP	Toxicity Characteristic Leaching Procedure		
TPY	Tonne Per Year		
TSR	Tensile strength ratio		
USCS	Unified Soil Classification System		
UF	University of Florida		
US	United States		
US EPA	United States Environmental Protection Agency		
VFA	Voids Filled with Asphalt		
VMA	Voids in Mineral Aggregate		
VOC	Volatile Organic Compounds		
WTE	Waste-to-Energy		
WTA BA	Waste-to-Energy Bottom Ash		
\$PT	Price Per Ton		
cm	centimeter		
cm ²	square centimeters		
cm ³	cubic centimeters		
mg/m ²	milligrams per meter squared		
ft3	cubic feet		
g	gram		
kg	kilogram		
L	liter		
lb/ft ³	pounds per cubic foot		
mg	milligrams		

mg/kg	milligrams per kilogram
cm/s	centimeters per second
mL	milliliter
mL/g	milliliter per gram
mg/L	milligram per liter
pH measurement of the hydrogen ion activity or measureme	
	the acidity of water.
μg	microgram
µg/L	microgram per liter
μm	micrometer

ABSTRACT

The use of solid wastes in civil engineering materials, such as portland cement concrete (PCC) and hot-mix asphalt (HMA) have been explored around the world for decades with varying degrees of success. Previous work supported by the Hinkley Center, "Use of Solid Waste in Asphalt and Concrete in Florida", demonstrated that utilizing waste to energy (WTE) ash in PCC and HMA is feasible from a physical performance and environmental health standpoint. However, it was acknowledged both by the researchers and in the literature that WTE ash may be processed by a method such as washing or co-utilized with another waste to develop a more marketable product. Currently, waste glass in Florida is experiencing low market value for recycling causing recycled glass to often be disposed of rather than reused. Previous literature shows that ground glass can act as a pozzolan which may limit deleterious effects associated with WTE ash reuse in PCC such as alkali-silica reactions (ASR). A literature review of the effects of glass powder (GP), or pozzolan, in PCC applications has been assembled along with a review of ash treatments utilized in various studies. The most common and economical method for WTE ash treatment was washing, which was noted to remove contaminates such as chloride, alkalis (i.e., Na, K), and to some extent trace metals. For HMA, utilizing coarse WTE ash (> No. 4, 4.76 mm) typically results in lower voids in mineral aggregate (VMA), tensile strength, and moisture susceptibility as evaluated using Modified Lottman Test (FM 1-T283). However, when a washed coarse WTE ash aggregate was utilized, there were improvements to the asphalt mix volumetrics and physical performance such as strength, albeit slightly. To evaluate the infrastructure and economic feasibility of GP recycling and WTE ash recycling, several models were developed to account for capital and maintenance costs along with material transport and current market values of aggregates and cementitious materials already used. This analysis showed that adequate throughput of waste glass and BA is crucial for making these products economical. Based on the results, several recommendations and options for future work were proposed including developing technical specifications for waste glass as a GP and for WTE ash in PCC and HMA applications, exploring innovative washing systems to limit contact time necessary for yielding a high quality aggregate product, developing infrastructure to handle ash wash wastewater along with the filter cake that is generated, investigating the quality of metals from the WTE ash stream for maximum removal for materials conservation and to increase the economic feasibility of ash recycling, and increased source separation of glass from the waste stream.

Key Words: WTE ash, concrete, asphalt, beneficial use, washing, glass, pozzolan

EXECUTIVE SUMMARY

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PROJECT TITLE: Research Advances on the Use of Solid Wastes in Concrete and Asphalt

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PROJECT SUMMARY

The use of solid wastes in civil engineering materials, such as concrete and asphalt, has been explored around the world for decades with varying degrees of success. Previous work supported by the Hinkley Center, "Use of Solid Waste in Asphalt and Concrete in Florida", demonstrated that utilizing waste-to-energy (WTE) ash in portland cement concrete (PCC) and hot-mix asphalt concrete (HMA) is feasible from a physical performance and environmental health standpoint. Researchers from the University of Florida observed that using WTE ash in these materials resulted in a product that had limited environmental impact from a leachability perspective and met minimum specifications (e.g., strength, durability), but not to the degree of concrete and asphalt made with virgin materials. These observations matched the trend documented in the extensive literature review conducted by UF. However, this research revealed that other waste materials, such as ground glass, may limit these deleterious effects when used in conjunction with WTE ash. Currently, market demand for waste glass is low resulting in much of the glass recovered at materials recovery facilities to be landfilled and cessation of glass recycling for many communities across Florida. The purpose of this research study was to investigate the material properties, treatments, and the infrastructure and economic considerations necessary for promoting WTE ash as an aggregate and glass recycling as a supplementary cementitious material (SCM) in PCC.

This report begins by presenting a literature review of the role glass powder pozzolan plays in PCC by describing its chemical composition, the methods used to evaluate its suitability as a pozzolan, and its performance in physical properties such as strength and

durability. The second part of this review centers on WTE ash and describes the various treatments used in the literature to produce a material that is more inert and/or more suitable as an aggregate replacement in beneficial use applications. Predominantly, washing has been explored for treatment of WTE fly ashes as these are typically considered hazardous wastes, while the washing of bottom ashes is not emphasized in the literature to date.

Following the literature review, the combined use of glass powder and WTE ash in PCC was explored using WTE ash from several Florida facilities. These facilities were named Facility A, B, and C. The ash from these facilities were processed to create a coarse aggregate (No. 4 - 3/4") for use in PCC. Several PCC mixtures were created to study the effect that a glass powder (GP) pozzolan would have on mitigating alkali-silica reaction (ASR). This reaction involves the formation of an expansive gel due to interactions between alkalis (i.e., Na, K), amphorous silica (e.g., bottle glass), and water. For comparison, mixtures without a pozzolan (i.e., only using portland cement) and coal fly ash (CFA), the industry standard for pozzolans, were also created and studied. The combined use of GP and BA in PCC resulted in relatively low expansion for ash from Facility C, while it was still unable to prevent deleterious expansion for Facilities A and B; although, it was still able to delay these reactions when compared to control mixtures (i.e., no GP or other SCMs).

Washing was the primary treatment method for WTE ash studied in this report. Washing was previously noted in the literature as an economical and practical method to remove soluble contaminants, such as alkalis, chlorides, and to a lesser extent, trace heavy metals, which have a negative impact on ash-amended PCC and HMA both from a physical and environmental standpoint. Washing was also hypothesized to be an effective means of removing the finer particles that remain on the surface of the coarser ash particles even with a dedicated screening step. This was performed on the coarse ash-derived aggregate from Facility A and the combined ash-derived aggregate from Facility D. The objective in this study was to observe how different liquid-to-solid (LS) ratio and contact times for washing may affect the quality of a treated ash product. The quality of a treated ash product was evaluated on the basis of fines (< No.4) removal, heavy metal removal, alkali removal, chlorides removal, and sulfate removal. The importance of washing is to create a product that is not only more environmentally inert for reuse but also to improve the physical properties of the ash for its intended usage. For example, the surface of coarse WTE ash contains fine dust which may interfere with asphalt binder adhesion and may also be a source of deleterious contaminates such as heavy metals and alkalis. After examining the effect of L/S and contact time, a trend was observed that treatment was more dictated by contact time rather than L/S ratio with significant increases in removal efficiency with increasing contact time. The physical effects (e.g., gradation, specific gravity, absorption)

that this washed ash-derived aggregate could have on the physical properties of PCC and HMA were then examined.

As noted in previous works by UF for HMA, suitable volumetrics were more difficult to acquire when utilizing WTE ash in the design but were feasible. However, washing produced mixtures that could meet all volumetrics for a Superpave mix design at a lower design asphalt content, albeit slightly. Meanwhile, rutting susceptibility for HMA showed limited change with washing, except for one mixture which could be attributed to its large proportion of 3/8" (9.5 mm) to No. 4 (4.76 mm) material. The most significant changes resulting from washing ash-derived aggregate included a decrease in optimum asphalt binder content and an increase in moisture resistance.

The final part of this work investigated the infrastructure needed for GP and WTE ash recycling and its economic feasibility. This involved understanding the capital, operational, and labor costs necessary for these types of facilities along with understanding the importance of economies of scale for decreasing the cost of production for these materials. For GP, a hypothetical waste glass recycling facility was simulated accounting for contaminate removal (e.g., non-glass materials such as paper products, metals, plastics) and milling to the necessary particle size for glass pozzolan (i.e., average particle size of 10 μm). It was found that GP produced from waste glass could become economically competitive with coal fly ash at throughputs exceeding 45 metric tonnes (t) per hour (TPH). As for WTE ash recycling, two systems were examined and compared: System #1, where only screening is considered and System #2, where ash is screened, and a metals recovery operation is utilized to recover valuable ferrous and nonferrous metals. The results show that ash recycling can be economically feasible at throughputs above 85,500 tonne per year (TPY) for System #1, and 76,500 TPY for System #2. Transportation of ashaggregate was found to have a notable impact on ash recycling feasibility. Regarding System #1, if the stockyard is 5 miles away (10 miles roundtrip) then the TPY shifts from 85,500 TPY to 90,000 TPY, while 25 miles away (50 miles roundtrip) shifts the TPY to 108,000 TPY and at 50 miles (100 miles roundtrip) the TPY shifts to 139,500. Regarding System #2, 10-mile, 50 mile and 100-mile roundtrips adjust the TPY from 76,500 TPY to 81,000 TPY, 90,000 TPY, and 103,500 TPY, respectively. The significant implications of these findings are that ash recycling using systems as shown in Systems #1 and #2 may only be feasible for average sized facilities, while smaller facilities may have difficulties recycling WTE ash economically based on this analysis.

Understanding the results gathered from the literature and UF's laboratory and modeling analysis, solutions and future work are proposed for increasing the beneficial use of waste glass and WTE ash in Florida. These solutions include developing technical specifications for reusing waste glass and WTE ash, examining washing systems to limit contact time for yielding a high quality aggregate product, developing infrastructure to handle ash wash wastewater along with the filter cake that is generated, investigating the quality of metals from the WTE ash stream for maximum removal for materials conservation and to increase the economic feasibility of ash recycling, and exploring how to limit the presence of waste glass in the municipal solid waste (MSW) stream to create a more desirable ash product and reduce resource loss for this material.

METRICS

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1.0 INTRODUCTION

1.1 Use of Solid Wastes in Portland Cement Concrete and Hot-Mix Asphalt Pavement

The beneficial use of various solid waste materials as replacements for natural aggregates, portland cement and asphalt binder in the production of portland cement concrete (PCC) and hot-mix asphalt (HMA) have been investigated in previous studies. The most effective beneficial reuse applications currently in practice for PCC are for supplementary cementitious materials (SCM), which includes coal fly ash, rice husk ash, and sugar cane baggase ash. University of Florida (UF) researchers have recently examined additional beneficial use opportunities, including the use of glass powder (GP) as a SCM (Paris et al. 2015) and the use of waste-to-energy bottom ash (WTE BA) as an aggregate replacement in PCC and HMA (Ferraro et al. 2016, Roessler et al. 2016).

The use of recycled glass as a SCM in PCC is a timely topic since the availability of coal fly ash (a traditional SCM that functions as a pozzolan) used in PCC is declining due to decreasing use of coal in electricity production (Gray, 2019; USEIA, 2019). Previous studies by UF and other researchers has demonstrated that recycled glass that is size reduced to GP behaves as a pozzolan in PCC. The use of recycled glass in this application is of particular interest since additional markets for this material are needed and other studies have shown clear environmental benefits associated with using GP as a pozzolan in comparison to coal fly ash or ordinary portland cement (Jiang et al., 2014). There are still many technical issues that must be resolved before glass can be sustainably recycled as a SCM. Perhaps the most challenging aspect of this recycling approach is the development of large-scale processes and processing infrastructure that will produce a product acceptable to the PCC industry.

Previous studies by UF and other researchers have demonstrated that WTE ash has properties that are similar to natural aggregates and has the potential to be used as a replacement for them in the production of PCC and HMA. Research has focused on screening the ash to useful size fractions and using treatments to remove deleterious contaminants. These treatments include solidification/stabilization, thermal treatment, and washing. Washing, which appears to be one of the most cost-effective treatments, can remove contaminates such as chlorides, alkalis, and sulfates, which can cause issues with the physical performance of PCC and HMA, while removing trace metals that pose environmental concerns. Screening operations alone cannot remove the fines or dust that remains on the surface of coarser ash particles. It is hypothesized that the finer fractions of WTE ash are heavily contaminated with the deleterious constituents mentioned previously that negatively impact PCC and HMA performance. Also, the highly absorptive nature of these finer particulates is suspected to be the cause for higher asphalt binder demands in HMA observed in the literature and in previous UF research. These constituents can also interfere with the adhesion of asphalt to the aggregate surface and may be the explanation for the lower moisture susceptibility (i.e., resistance to water degradation) that ashamended mixtures typically experience in the literature.

This study includes an analysis of the processes and infrastructure required to implement large-scale beneficial use of GP and WTE ash in PCC and HMA production. It also includes an estimate of the capital, operation, maintenance, and transportation costs of these activities, and a comparison of these costs to the market values of the materials they are replacing.

1.2 Objectives

The overall objective of this research is to build upon existing work on the use of WTE ash in PCC and HMA pavement and to provide data that will be helpful for the beneficial use of this and other waste materials. Three specific objectives were established for this project:

- Expand on previous research regarding use of GP as a replacement for portland cement by examining the combined use of GP and WTE ash in concrete mixes.
- Conduct research on the benefits of WTE ash washing as a pretreatment step to create products to be used as concrete aggregate and asphalt pavement aggregate.
- Examine the infrastructure needs and associated costs for the implementation of glass recycling to SCM and WTE ash recycling for aggregate.

1.3 Organization of Document

This report is composed of eight sections, with the first one being this introduction. Section 2 discusses background information and reviews the literature pertinent to this project. Section 3 includes results on the combined use of GP and WTE ash aggregate. Data on the washing pretreatment for WTE ash is included in Section 4. Sections 5 presents data on the performance of washed ash on WTE ash-amended HMA. Section 6 presents the infrastructure needs and economic analysis for implementing a glass recycling program for SCM and WTE ash for aggregate. Section 7 identifies research needs and suggested initiatives as a result of this project, and a summary and conclusions are presented in Section 8. Lastly, references are provided in Section 9.

2.0 BACKGROUND & LITERATURE REVIEW

2.1 Glass Powder in PCC

For the past 40 years, the concrete industry has used coal fly ash as a SCM in the production of concrete (Manz, 1999; Meyer, 2009). Coal fly ash is a waste by-product resulting from the incineration of coal and is now used to enhance properties of concrete such as strength, workability, and durability (Manz, 1999; Canpolat et al., 2004; Meyer, 2009). However, due to the decline in use of coal for the production of electricity, the production of fly ash has also started to decline in the United States (USEIA, 2019c; Gray, 2019) while consumption has remained relatively consistent, leading to a deficiency in the supply of fly ash. As shown in Figure 2-1, the percentage of fly ash consumption has increased significantly in recent years from about 40% to 60% of the fly ash that is produced (American Coal Ash Association, 2017).



Figure 2-1. Coal fly ash production and consumption from 2000-2017 adapted from American Coal Ash Association (2017).

The impending fly ash shortage has motivated the concrete industry to determine potential remedies, some of which include the use of previously impounded fly ash and the use of alternative supplementary cementitious materials not traditionally used in the production of concrete (AASHTO, 2016). The acceptance of a new material for use in the construction industry is dependent upon availability, compatibility, and economy with respect to the portland cement. Therefore, a potential replacement for fly ash would require adequate reactivity in order to provide a structurally comparable product and wide availability to reduce any potential transportation costs and create a financial incentive for their use.

Recently, ground waste glass has been investigated and evaluated by many as a potential alternative supplementary cementitious material (Paris et al., 2016). In the United States, the total municipal solid glass waste in 2015 was approximately 11.5 million tons with 60% being landfilled where only 40% is recycled or used for combustion. outlines several years of glass waste generation, recycling, and landfilling (Shi & Zheng, 2007). The lack of glass recycling and prevalence of landfilling in the United States is relatively inefficient and poses detriments to the environment when compared with beneficial use and creates an abundance of waste glass that could be reused (Hemalatha & Ramaswamy, 2017; Rashidian-Dezfouli et al., 2018; Afshinnia & Rangaraju, 2015).

The evaluation of ground waste glass as a pozzolanic supplementary cementitious material has the potential to divert waste-stream glass from landfills into the construction industry as a material for beneficial reuse (Afshinnia & Rangaraju, 2016). To evaluate the performance of relatively new cement replacements, a number of test methodologies must be performed. The testing program must sufficiently evaluate and ensure performance of the concrete product in terms of plastic, mechanical, and durability performance. One of the largest concerns associated with the inclusion of waste glass into concrete is a deleterious reaction called alkali-silica reaction (ASR) (Paris et al., 2016), which has traditionally formed within the concrete matrix at the interface of the reactive siliceous aggregate and cement paste (Jian-xin Lu et al., 2017). The reaction forms a gel that expands in the presence of water and can cause cracking in the concrete and ultimately weaken it (Jian-xin Lu et al, 2017; Islam et al., 2017). In an effort to determine the potential for ASR, the concrete industry has developed several test methods that can be used to determine if there is potential for reactivity. However, first the chemistry of cementitious materials and the composition of waste glass must be discussed.

2.1.1 Chemistry of Cementitious Materials

Ordinary portland cement (PC) is comprised of four main oxides: tricalcium silicate (alite), dicalcium silicate (belite), tricalcium aluminate, and tetracalcium aluminoferrite; additionally cement contains a number of minor components which can be equally important such as gypsum, alkalis, and magnesium oxide (Weihua Jin et al., 2000; Nassar et al., 2011). The typical proportions are shown in Zheng (2016). The reaction between cement and water is called hydration. During cement hydration, the tricalcium silicate (alite) and dicalcium silicate (belite) in the cement react with water to produce calcium-silicate-hydrate (C-S-H), calcium hydroxide, and heat, which is outlined in Equations 1 and 2 (Zheng, 2016). Alite is responsible for early strength due to its high reactivity and belite is responsible for later strength due to its lower reactivity (Dyer & Dhir, 2001; Du & Tan, 2017).

$$2 \text{ Ca}_{3}\text{SiO}_{5} + 11 \text{ H}_{2}\text{O} \rightarrow 3 \text{ Ca}_{2}\text{SiO}_{2}\cdot8\text{H}_{2}\text{O} + 3 \text{ Ca}_{0}(\text{OH})_{2}$$
(1)

$$2 \operatorname{Ca}_2 \operatorname{SiO}_4 + 9 \operatorname{H}_2 O \rightarrow 3 \operatorname{Ca}_2 \operatorname{SiO}_2 \cdot 8 \operatorname{H}_2 O + \operatorname{Ca}_2 (OH)_2$$
(2)

As mentioned above, calcium hydroxide is a reaction byproduct in the formation of calcium

silicate hydrate (C-S-H). A pozzolan is a material that contains silica, which when added into a cementitious system, reacts with the calcium hydroxide created from hydration. The reaction between the silica in the pozzolan and the calcium hydroxide creates additional C-S-H and can increase strength (Garcia-Diaz et al., 2006; Thomas, 2011). For every ton of cement produced, about a ton of carbon dioxide is generated (Lothenbach et al, 2011; Ludwig & Zhang, 2015), which is a much higher carbon footprint than pozzolans. It is well known that the cement industry accounts for about 5-9% of anthropogenic global carbon dioxide emissions (Meyer, 2009).

Some well-known pozzolans in the concrete industry include silica fume, coal fly ash, ground blast furnace slag, metakaolin, and natural pozzolans. These pozzolans are byproducts of other industries (with the exception of natural pozzolans) such as the coal industry for fly ash and steel industry for slag and can enhance the performance of the concrete (Afshinnia & Rangaraju, 2015; Rashidian-Dezfouli et al, 2018). Most SCMs are typically more economical because they are waste stream sourced (Thomas, 2011; Rashidian-Dezfouli et al, 2018). Therefore, by using pozzolans as cement replacements, the cost and carbon footprint of the concrete as a whole decreases (Meyer, 2009; Paris et al., 2016). The decrease in fly ash production in the USA stems from the transition towards different energy sources such as renewables and natural gas. The decrease in the availability of fly ash could result in a price increase to the point where it is no longer economically viable in the production of concrete (Gray, 2019). The increased demand for alternative supplementary cementitious materials has led to researchers investigating a wide variety of potential alternatives (Nassar & Soroushian, 2011; Idir et al., 2015). Ground glass has been investigated as a potential option due to its abundance, availability as a waste material, and the potential reactivity it can have with portland cement (Gifford & Gillott, 1996). In addition, it has been found that the use of ground glass in concrete would reduce carbon dioxide emissions by about half a kilogram for every kilogram of glass used (Shehata & Thomas, 2010).

2.1.2 Glass Chemistry and Composition

Crystalline silica is most commonly known as quartz and is less reactive with portland cement (Thomas et al., 2008). Crystalline materials have an ordered, symmetrical, and consistent structure. The lattice structure in amorphous silica does not have long-chain order and varies throughout (Rangaraju et al., 2006); it is commonly created by heating and melting sand (silicon dioxide), soda ash (sodium carbonate), and limestone to 1400-1600°C (Cyr et al., 2009). When heated, the crystalline structure becomes amorphous and rapid cooling prevents the crystalline structure from reforming. Silica fume and ground glass are both forms of amorphous silica. In order to reduce the melting point of the silica, a soda ash flux is used to generate sodium silicates through the reactions outlined in Equations 3 and 4 (Aprianti, 2017). Glass with only silica and a flux usually has a low resistance to corrosion and is also water-soluble. Limestone (CaCO₃) is added to form Ca+Si compounds which increase the resistance to corrosion and reduce water solubility. These chemical reactions are outlined in Equations 5 and 6 (Aprianti, 2017).

$$Na_2CO_3 \rightarrow Na_2O + CO_2 \tag{3}$$

$$Na_2O + SiO_2 \rightarrow Na_2SiO_3 \tag{4}$$

$$SiO_2 + CaCO_3 \rightarrow CaSiO_3 + CO_2 \tag{5}$$

$$SiO_2 + 2CaCO_3 \rightarrow Ca_2SiO_4 + 2CO_2 \tag{6}$$

Amorphous silica is more reactive than crystalline silica due to the different dissolution rates. While crystalline silica is soluble in high pH environments such as concrete, amorphous silica is soluble regardless of the pH and has a higher dissolution rate than crystalline silica (Thomas et al., 2008). The atomic structure of crystalline and amorphous glass are compared in Figure 2-2. The atomic structure of amorphous glass has no clear orientation and varies throughout, which leads to lower bond strength between the silicon and oxygen atoms. A lower bond strength makes it easier for the calcium hydroxide from the cement hydration to react with the silica. Due to these factors, it can be concluded that glass does cause a pozzolanic reaction in a concrete system (Maheswaran et al., 2016).



Figure 2-2. Comparison between atomic structure of Quartz and Amorphous Silica (Glass) adapted from Omran & Tagnit-Hamou (2016).

It has been well established that the particle size of the cementitious component is directly related to reactivity (Guerrero et al., 2004; Hasanbeigi et al., 2010). The typical trend is for reactivity to increase as particle size decreases and surface area increases (Jani et al., 2014; Wu et al., 2018). To be used as a pozzolan, soda-lime glass must be ground down to a fine powder. Glass with a smaller particle size is more reactive (Shayan & Xu, 2004; Afshinnia & Rangaraju, 2015) and can better reduce expansion caused by ASR (Afshinnia & Rangaraju, 2015). Particle size plays a very important role when it comes to using glass in concrete. If the glass being used is not fine enough, it will not serve as a supplementary cementitious material and can exacerbate ASR rather than reduce it (Al-Ayish et al., 2018). It has been found that glass used as an aggregate with a size larger than 1mm can exacerbate ASR expansion (Moffatt et al., 2017). The size range in which glass begins to reduce ASR is ambiguous, as different studies provide different ranges. However, a general size in which reduction is begun is below 1mm (Maheswaran et al., 2016). Some studies mention a "pessimum" which outlines the nature of glass particle size and the effect it has on ASR. At larger particle sizes (greater than 1mm), the incorporation of ground glass in concrete increases ASR expansion. However, once the particle size is decreased below the pessimum (1mm), ground glass reduces ASR expansion (Aprianti et al., 2015). A potential for future research would be in determining the optimal particle size to maximize pozzolanic reactivity and ASR mitigation. While there is a general idea that a particle size lower than 50µm causes a pozzolanic reaction, it has not yet been determined which particle size is the most beneficial in terms of reactivity, economy, and reduction in expansion caused by ASR.

According to the Corning Museum of Glass, most commercial glasses can be categorized into 6 types as follows: Soda-Lime glass, Lead glass, Borosilicate glass, aluminosilicate glass, ninety-six percent silica glass, and fused silica glass (Farfan et al., 2019). The Soda-Lime composition consists of 60-75% silica, 12-18% soda, and 5-12% lime (Farfan et al., 2019). Lead glass consists of at least 20% lead oxide, Borosilicate glass contains a minimum of 5% boric oxide, and Aluminosilicate glass contains aluminum oxide. Ninety-six percent silica glass is borosilicate glass that's melted and processed in order to remove non-silicate impurities. Fused silica glass is a pure silicon dioxide in a non-crystalline state. Soda-lime glass is the most common type, accounting for 90% of glass made (Farfan et al., 2019). For the purposes of this investigation, soda-lime is the primary focus and low-alkali glass is discussed briefly.

2.1.3 Methods for Examining GP Performance in PCC

Alkali-Silica reaction involves the reaction between alkalis from the pore solution in concrete and the silica from a reactive aggregate (Paris et al., 2018; Tucker et al., 2018) Alkalis break down the aggregate's siliceous layer and react to form an expansive gel. The gel absorbs water from the pore solution and expands, inducing tensile stresses onto the concrete and cracking it (Dove et al., 2008). ASR can occur with any reactive aggregates, which includes both coarse and fine aggregates. Some examples of commonly known reactive aggregates include: Sudbury (greywacke-argilite), Spratt (siliceous limestone), and Jobe (chert). In some cases, however, individual aggregates that are potentially reactive individually can become reactive when used together in a concrete mix. In order to mitigate ASR, at least one of the reaction components (alkalis, reactive aggregate, or water) must be eliminated (Frigione & Marra, 1976). For example, cracking from ASR is less likely to occur when the aggregate is unreactive, which prevents the formation of the expansive gel. In addition, cracking from ASR is less prevalent when there is a lack of water for the expansive gel to absorb. Reducing the amount of alkalis in the pore solution can also help protect from the concrete from ASR (Paris et al., 2018). A higher concentration of alkalis in the pore solution exacerbates the reaction and produces more expansive gel (Tucker et al., 2018). Therefore, by using a pozzolan, the available alkalis in the pore solution are consumed (Paris et al., 2018).

ASTM C1778 provides a guide to standardized test methods used to evaluate aggregate susceptibility to alkali aggregate reaction (AAR), which includes both ASR and alkali carbonate reaction (ACR). Figure is the flow chart provided in ASTM C1778 and showcases which test methods are appropriate in different situations, along with recommendations based on the results of testing. ASTM C1778 can be used as an initial guide when the reactivity of an aggregate is unknown to develop a testing plan in order to determine whether it is viable.

The test methods most commonly used for ASR testing are ASTM C1260, ASTM C1567, and ASTM C1293. ASTM C1567, C1260, and C441 are accelerated mortar bar tests while ASTM C1290 is a concrete test. ASTM C227 is another test method that involves ASR but through lack of use in industry, it has been recently withdrawn and is no longer updated.

The first edition of ASTM C1260 was created in 1994. In 2004, ASTM C1567 was created and is closely related to ASTM C1260. ASTM C1260 is used to evaluate potentially reactive aggregates with relation to ASR. ASTM C1567 involves the same procedure but is meant for determining the mitigation potential of an SCM. ASTM C1567 is performed by using an already known reactive aggregate and measuring expansion. An important distinction is the mention of an alkali limit of 4% in the SCM. If the alkalis in the SCM are higher than 4%, the results provided by ASTM C1567 are inconclusive.

ASTM C1260 is used to determine the potential reactivity of an aggregate. The alkali content in the cement is disregarded because the specimens are immersed in sodium hydroxide solution. The test is not intended for evaluating supplementary cementitious materials (SCMs). The test involves casting mortar bars and measuring their length change over time. The length change must be lower than 0.20% in order for the aggregate to be considered innocuous. ASTM C1567 is used to evaluate a supplementary cementitious material's ability to mitigate ASR and involves the use of a reactive aggregate. In both cases, measurements are recorded throughout 16 days, but can be extended in order to further evaluate the expansion. ASTM C1567 specifies that it is limited to SCM's with an alkali content of 4% or less. The alkalis are supplied by the solution in order facilitate the reaction. If the SCM being examined has more than 4% alkalis, the alkalis provided from the solution won't account for the alkalis lost due to leaching. Therefore, a loss of total alkalis in the system can result in an underestimation of expansion.



Figure 2-3. Procedure given in ASTM C1778 to select appropriate measures to manage Alkali Aggregate Reaction.

The more common forms of ground glass (soda-lime glass) are notorious for having an alkali content of in excess of 10%. Because of this, ASTM C1567 is inappropriate for the evaluation of ground glass as an addition to PCC for the mitigation of ASR. However, several publications have evaluated the ability for ground glass to mitigate ASR using ASTM C1567 due to the common misconception. Some publications also erroneously refer to ASTM C1567 as ASTM C1260 due to their similarity and the recent implementation of ASTM C1567. Publications written before 2004 only had ASTM C1260 to refer to, which has no mention of an alkali limit because it was meant to evaluate aggregates. For the purposes of this paper, it is assumed that the accelerated mortar bar test method used to evaluate glass as a pozzolan is ASTM C1567, even if studies refer to using ASTM C1260. The purpose of this research is to find publications relating to ground glass and ASR mitigation and compare the results and test methods they used.

ASTM C1293 is used to evaluate the potential for an aggregate to react deleteriously and also for an SCM's ability to mitigate expansion. It involves creating concrete specimens with high alkali cement, the SCM being analyzed, one reactive aggregate and one non-reactive aggregate. The specimens are demolded and stored above water in a container at 38°C. ASTM C1293 is a one-year test if SCM's are not being used and a two-year test if SCM's are being used. Measurements are recorded and expansion is calculated. Expansion is limited to 0.04% at two years to conclude if an SCM can prevent excessive expansion. The advantage of using ASTM C1293 over ASTM C1567 is that there is no limit to the alkali content of the SCM being used. ASTM C1293 doesn't have an alkali limit because there is less leaching, and the alkalis are supplied to the specimen by adding sodium hydroxide into the mixing water. One of the drawbacks of using ASTM C1293 to determine the reactivity of an SCM in a portland cement system is the relatively long nature of the test as it requires two years to complete. Both ASTM C1567 and C1293 have been reported to have leaching issues. However, ASTM C1567 has higher levels of leaching than ASTM C1293. The results from both of these test methods are not always congruent, especially with higher levels of cement replacement (Paris et al., 2018).

ASTM C441 was first electronically published in 2002 and was last updated in 2017. The specimens are created with crushed, washed and graded borosilicate glass (Pyrex no. 7740) as the fine aggregate and a high alkali cement with 0.95-1.05% total alkalis. These materials are used in order to exacerbate the reaction and determine the potential for a pozzolan to mitigate it. The specimens are to be created with a control mix and kept in the same container. The containers must ensure that the surfaces of each specimen are evenly exposed to wicking material and are kept above water without any splashing. They are kept in a moist room or moist closet at $38^\circ \pm 2^\circ$ C for 12 days and then measured. Subsequent measurements can be made later on to further monitor long-term expansion, but the 14-day expansion is reported and compared with the control mixture.

ASTM C227 was last updated in 2010 and was withdrawn in 2018 due to its limited use by industry. A minimum of four mortar specimens were to be prepared, demolded after 24 ± 2 hours in a moist room/cabinet, measured for initial length, and stored above but not in contact with water in a sealed container at 38°C. The specimens are measured after removing them from the 38° storage and storing them at 23°C for 16 hours. Measurements

are to be made at 14 days of age and later ages if necessary. After making a measurement, they are to be returned to 38°C storage until the next measurement is made.

2.1.4 GP Performance in the Literature

In order to properly evaluate the performance of ground glass in concrete, the plastic properties, mechanical performance, and durability performance must be compared to that of regular portland cement concrete. In each category, there are several test methods used to determine how the addition of the material affects the performance of the concrete. There are some publications which are mentioned in this review, but not in the summary table due to different testing parameters within those investigations.

Slump and flow are used to evaluate the effect of a cementitious material on the workability of concrete or mortar. Several studies have evaluated the performance of ground glass on slump (Garcia-Diaz, 2006; Afshinnia & Rangaraju, 2016) and mortar flow (Shayan & Xu, 2004). Several studies report that incorporating ground glass into concrete reduces workability (Garcia-Diaz et al., 2006). However, there are also reports that state that workability is increased (Shayan & Xu, 2006; Afshinnia & Rangaraju, 2016) and some that conclude that workability is comparable (Shyan & Xu, 2004, Dhir et al., 2009). It is clear that there is no established consensus on the effects that ground glass has on workability. A potential reason for why the slump and flow values differ between publications is that the particle size of the glass being used differs. While the particle size of the glasses that were studied were small enough to be considered pozzolans, the variations could still have an effect.

Concrete density measurement is a form of quality control used to compare unit weight of the concrete mix to an expected unit weight. If the density is significantly different from what is expected, the concrete mix could have potentially been mixed incorrectly. Some have measured the effect ground glass has on concrete density (Corning Museum of Glass; Neithalath et al., 2009). It was concluded by Neithalath et al. (2009) that the density is comparable to control, while (Corning Museum of Glass) determined that the density decreased when compared to control.

Air content is another form of quality control to ensure that the concrete was mixed appropriately. Air content can also indicate wither or not a material can promote or hinder the incorporation of air into a mix, which would be useful for certain applications such as freeze-thaw (Hasanbeigi et al., 2010). Some studies have evaluated the effect ground glass has on air content in concrete (Corning Museum of Glass; Neithalath et al., 2009). Generally, the air content is not affected by the incorporation of glass into concrete (Neithalath et al., 2009), however some have reported an increase in the air content (Corning Museum of Glass).

Compression is one of the most common forms used to evaluate a material's performance as a pozzolan. Several studies have been conducted on the effects of ground glass has on the compression of concrete (Afshinnia & Rangaraju, 2015, 2016) and mortar (Shayan & Xu, 2006). The results of the strength performance of glass have been divided into different ages and replacements in order to better summarize the general consensuses. The general consensus is that an increase in ground glass replacement reduces the compressive strength when compared to control at earlier ages. However, at later ages, ground glass mixes can eventually reach a comparable strength to control. It is also shown that an increase in replacement increases the amount of time needed for the mix to reach a comparable strength to concrete (i.e. a 10% replacement reaches similar strength to control at 28 days, but a 20% replacement reaches control strength at 56 days.) (Afshinnia & Rangaraju, 2015; Shayan & Xu, 2006). The optimal replacement levels for strength seem to be between 10-20% (Shi et al., 2005). This behavior is similar to that of class F fly ash (Shayan & Xu, 2006).

Splitting tension is another test method used to evaluate the performance of a pozzolan in a concrete system and can also be used as a quality control measure. Concrete specimens that use ground glass as a partial replacement exhibit a statistically comparable splitting tensile strength at most ages and replacements (Shi et al., 2005; Paris et al., 2016).

Modulus of elasticity affects the design of concrete sections due to deflection limits. A higher modulus of elasticity could result in a smaller section size and therefore reduce cost through labor, material, or both. In the case of using ground waste glass, Afshinnia & Rangaraju (2016) report that the modulus of elasticity decreases as the percent replacement increases.

The only test available for the rapid assessment of ASR in mortar with pozzolans is prescribed by ASTM C1567. Accordingly, there has been a great deal of research performed evaluating the effectiveness of ground waste glass as a pozzolan using the rapid mortar bar test using ASTM C1567. However, ASTM C1567 requires the pozzolan has a maximum alkali content of 4%. Glass has more than that.

Therefore, conclusive statements based on that test alone are premature and technically invalid. The test method can be used as a form of determining if a material is inadequate in mitigating ASR, but it cannot be used to conclude that a material can mitigate ASR, which results in false positives. However, the findings using this test method might still give initial indication of the validity of ground glass as a pozzolan. If the results in ASTM C1567 indicate that there is potential for a material to mitigate ASR, additional testing is encouraged such as ASTM C1293.

Shi et al. (2005) reported that a cement replacement of 20%, Al-Ayish (2018) reported that a replacement of 30%, and Paris et al. (2016) reported that a replacement of 50% with ground glass can mitigate ASR according to the limits set by ASTM C1567. However, others report that glass powder can reduce expansion, but it cannot reduce ASR expansion to below the expansion limits of 0.1% in order to be classified as adequate in mitigating ASR (Neithalath et al., 2009). There is a much smaller amount of research evaluating the potential for ASR mitigation with glass powder using ASTM C1293 and ASTM C227. According to Maheswaran et al. (2016), who evaluated glass with ASTM C227, ground glass powder reduced expansion. Al-Ayish et al. (2018) evaluated ground glass with both ASTM C1267 and ASTM C1293 in order to compare their results. As mentioned previously, it was

reported that a 30% replacement of ground glass was needed to mitigate ASR according to ASTM C1567. However, when a 40% replacement was used for ASTM C1293, it failed to reduce expansion to below the threshold to be determined as a mitigation method.

2.1.5 Environmental and Economic Impacts of Waste Glass Reuse in PCC

The reuse of waste glass limits environmental impacts by diverting the material from landfills and by replacing the use of other raw materials which require energy and other additional resource inputs (Disfani et al., 2012; Islam et al., 2017; Heriyanto et al., 2018; Tucker et al., 2018; Patel et al., 2019). Disposed glass is a non-biodegradable material which means it does not decompose and hence consume valuable landfill space permanently (Jani et al., 2014; Mohajerani et al., 2017). These actions also prevent greenhouse gas (GHGs) emissions such CO₂ by avoiding management as a waste material (e.g., transport to and compaction at landfill working face) and the high energy costs associated with extracting and processing virgin materials to create new products (Jani et al., 2014; Tucker et al., 2018).

As for the specific environmental impacts of waste glass as a pozzolan in PCC, these have been mostly studied by life cycle assessments (LCAs) with impact categories focusing on air emissions while also considering resource use (Jiang et al., 2014; Tucker et al., 2018; Patel et al., 2019). The production of portland cement is known to be a major source of GHG emissions such as CO₂ along with nitrogen oxides (NOx), sulfur oxides (SOx), and particulates while simultaneously being energy intensive (Cattaneo, 2008; Jiang et al., 2014; Islam et al., 2017; Tucker et al., 2018). The LCAs from these studies suggest that the use of glass powder in place of portland cement results in reductions in emissions and energy usage with increasing replacement; however, it should be noted that these analyses may include variables that result in higher impacts for portland cement use depending on factors such as different cement and PCC production technology and their locations (Jiang et al., 2014). These reductions in emissions and energy use are attributed to the replacement of portland cement, whose production process involves calcination, an energy intensive process that yields approximately 50-60% of the greenhouse gas emissions associated with portland cement production (Jani et al., 2014; Jiang et al., 2014). The use of waste glass as a pozzolan in PCC may also provide greater reductions in greenhouse gas emissions and energy use beyond traditional glass recycling (Jiang et al., 2014; Tucker et al., 2018).

Reusing waste glass as a glass powder in PCC may also yield benefits from an economic standpoint by competing with portland cement and other pozzolans from a cost perspective. The cost of GP manufacturing, and by extension waste glass processing itself, is heavily influenced by the quality of the waste input and scale. Tucker et al. (2018) investigated the development of a GP processing facility using waste glass as received by a materials recovery facility (MRF) and involved assessing fixed capital costs, operating costs, maintenance costs, and labor costs incurred by a facility during production.

Fixed capital costs consider initial costs of procuring the land for the facility along with construction of the facility itself. Presumably, this would be an extension to an existing

MRF, as these operations have the logistical and processing infrastructure to remove contaminants from the incoming waste glass stream (Tucker et al., 2018). Metals are a common contaminant in a waste glass stream; to remove all metals, both a conventional magnet is required along with an eddy current separator (ECS). Some other common processing equipment include: a rolling stock (i.e., front-end loader), conveyor, drum feeder, vacuum, crusher, trommel screen, air classifier, and ball mill (Tucker et al., 2018). Operation, maintenance, and labor costs must also be considered in an economic feasibility analysis. Vehicles and machinery run on fuel and electricity, prices of which can fluctuate over time and geographically. These fluctuating costs also apply to estimating facility electricity and natural gas costs for operation. Additionally, costs are accrued for building and machinery maintenance. One potential barrier is that the size reduction process for GP is energy intensive (Shao et al. 2000). In particular, size reduction media for the ball mill can be a large expense due to energy requirements for operation in addition to breakdown and replacement of the media over time (Tucker et al., 2018). As for labor costs, employees must be paid to operate vehicles and equipment at the facility, for supervisors, and for administrative staff. Other considerations include dust generation, as production of GP (especially during the milling phase) will likely create high amounts of dust that must be properly handled for human and environmental health. A facility will likely need to apply for an air permit to produce GP, which has additional costs beyond initial capital and continued maintenance of air pollution control systems (e.g., baghouse) and more frequent equipment cleaning (Tucker et al., 2018).

Waste glass stream quality can be a major factor. More contaminates translate to higher processing costs or may result in the material being rejected to landfill. Contamination is common in waste glass collected from residential and commercial recycling programs (Wartman et al., 2004; Landris, 2007; Younus Ali et al., 2011), which are typically containers (e.g., bottles) but plastic containers are often found along with metals (e.g., cans) or even paper products. Residues remaining on the glass containers are also a source of contamination (Wartman et al., 2004; Landris, 2007; Younus Ali et al., 2011). These factors are exacerbated when considering that waste glass is often "negatively-sorted" at MRFs in which other recyclables (i.e., plastics, metals) are removed and the glass being classified as an "contaminant" rather than the resource (Dubanowitz, 2000).

Scale is also a driver in economically feasible waste glass recycling. Tucker et al. (2018) modeled the total processing cost of GP with annual throughput (TPY) of glass from the facility and observed a continuous decrease in cost with increasing throughput. This amount of glass may be unfeasible for locales with smaller populations without importing waste glass externally. Strategies for increasing this supply rate include shipping waste glass from surrounding MRFs while incurring nominal transportation costs and incentivizing other MRFs to send their glass if costs are less than private glass recycling facilities (Tucker et al., 2018). Understanding regional generation, costs, and current practices in glass recycling and management may be the pathway for more efficient, cost-effective waste glass recycling into GP. As of 2013, the price of portland cement and fly ash were approximately \$120/ton and \$40/ton respectively (USGS, 2016; Tucker et al. 2018). The fly ash shortage has resulted in a price of approximately \$75/ton as of 2017 and from a cost perspective, has the potential to make GP more economically feasible as a pozzolan.

2.2 Properties of WTE Ash and their influence on PCC and HMA

2.2.1 Properties of WTE Ash

Municipal solid waste incineration reduces the volume of MSW by about 90% and mass by about 70% while yielding a new product: WTE ash (Kumar & Samadder, 2017). This process along with its major inputs and outputs are outlined in Figure 2-4.



Figure 2-4. Typical WTE process in the United States.

In this process, MSW is combusted in a furnace at temperatures between 850-1,200°C (Kumar & Samadder, 2017), which volatilizes and combusts virtually all organic material such as plastics, paper products, food waste, and yard wastes. Heat from the combustion process is used to produce steam, which powers a turbine within the plant to generate electricity. This electricity is used by the facility for its operations and the excess is sold to local power utility companies. All unburned materials such as metals, glass, slags, and unburned organics end up as BA. Many incineration facilities use magnets to remove ferrous metals from the BA immediately after combustion, but often smaller sized ferrous and nonferrous metals (e.g., aluminum) remain. Figure 2-5 shows the typical oxide composition of BA (Lam et al., 2010).


Figure 2-5. Example oxide composition of BA by weight (Lam et al., 2010).

As a result of burning waste at high temperatures, gases are formed which must be controlled using air pollution control systems prior to being released into the environment. These gases can contain hazardous air pollutants such as acidic compounds (e.g., sulfur oxides, hydrogen chloride), heavy metals, and other contaminants such as polychlorinated biphenyls (PCBs) and volatile organic compounds (VOCs) that can have a serious impact on human health and the environment (Lam et al., 2010). Lime (CaO) scrubbers are used to remove acidic gases, and activated carbon is used to remove heavy metals, PCBs, and VOCs (Lam et al., 2010). Fabric filters are used to filter and remove fine particulates and dust particles (Lam et al., 2010). This output from treatment is a fine material known as WTE fly ash (FA), which contains the fine particulates from the incineration process along with the lime residue from the air pollution control system. WTE BA and FA are shown side-by-side for comparison in Figure 2-6.



Figure 2-6. WTE BA and FA shown side-by-side for comparison.

Figure 2-6 shows that WTE ash, particularly BA, is a highly heterogeneous material consisting of multiple components (e.g., slags, glass, ceramics) with varied morphology (i.e., highly angular and irregular) at a range of particle sizes (e.g., 2 in. to 0.0117 in, or No. 50 sieve, and even lower). WTE FA is arguably more homogenous than BA and is typically below 1/4" in diameter. A typical oxide composition for FA is shown in Figure 2-7 (Lam et al., 2010).



Figure 2-7. Example oxide composition of FA by weight (Lam et al., 2010).

While these oxides make up a majority of the BA and FA, trace elements exist within WTE ash that can hinder beneficial use from both a performance and environmental perspective. Examples of these trace elements include arsenic, alkalis, such as sodium and potassium, chlorides and sulfates, and heavy metals including lead and antimony. To beneficially reuse a waste material in applications such as PCC and HMA, it must be shown to not cause a significant risk to human health and the environment but also prove to produce a functional material. For the purposes of this report, this involves PCC and HMA which are discussed in the following subsections.

2.2.2 PCC Review and Effects of WTE Ash Utilization

PCC consists of four main ingredients: coarse aggregate (granite, limestone), fine aggregate (sand), portland cement (often in combination with other cementitious materials), and water. When mixed together in the proper proportions, the cement and water in this mixture create a paste that coats the aggregates and undergoes complex hydration reactions (described in section 2.1.1.) that lead to the hardening of the paste and subsequent formation of concrete. The proportions of the core components of a typical PCC mix are shown in Figure 2-8 below (Kosmatka et al., 2002).



Figure 2-8. Component percentages of a typical portland cement concrete mix by mass.

PCC, the most widely used construction material worldwide, has a wide variety of uses that range from roadway pavements to large mass-pour elements such as bridge pilings. It is strong, durable, and relatively inexpensive to manufacture, causing it to become a common component of many building and infrastructure projects.

Two main forms of concrete that are widely manufactured; ready-mixed concrete and precast concrete products. Ready-mixed concrete is the most common method for concrete manufacture. At ready-mixed plants, concrete is batched into the appropriate proportions and placed into a concrete mixing truck with a revolving drum and is then driven to the job site to pour the concrete. Precast products are made in factories and then shipped to a job site to be assembled into structures. A wide variety of products, from small pavers to large structural components for bridges, can be manufactured at precast plants. These products are subject to more strenuous quality control specifications than ready-mixed concrete. There will need to be institutional and infrastructural changes (such as updated mix designs and new aggregate bunkers) at ready-mixed plants if there is to be widespread usage of a new product, WTE ash, in ready-mixed concrete applications.

Concrete performance is typically evaluated based on several physical, chemical, and durability related parameters. From a physical performance standpoint, it is important to make sure that any concrete product manufactured will meet required specifications for a specific job.

Concrete will often need to meet compressive strength and similar physical performance requirements. Concrete for a low traffic rural roadway pavement, for instance, may have lower strength requirements than concrete used for a bridge span or a high traffic

roadway. When evaluating WTE ash as a replacement for coarse aggregate in concrete, it is important to note the specific application that the concrete will be used for and include this in the evaluation of construction feasibility. The specifications a concrete structure must meet depends on local and state laws.

Concrete durability is also of special importance, as any concrete mix needs to be well suited for its environment and expected service life. The Portland Cement Association reports that concrete is durable if it can resist weathering action, chemical attack, and abrasion while maintaining desired engineering properties, a concrete that lasts a long time without significant deterioration ("Durability", 2017). Several physical and chemical processes can affect concrete durability, and when attempting to introduce a new material, such as WTE ash, into a traditional concrete mix, it is important to ensure that new material does not have any negative effect.

Regulatory agencies such as the EPA also set stringent standards for allowable contaminant and hazard leaching levels. An extensive suite of leaching tests (i.e., EPA LEAF) has been developed for determining the dangers associated with any material in the environment. These standards should be met by ash-amended PCC, the same way they should be met for any other material placed in the environment.

WTE ash incorporation into construction materials such as portland cement concrete mixes has been widely used in many European and Asian countries. These countries have further advanced ash recycling programs because they have been developing systematic processes for years to develop and evaluate reuse options involving consistent physical and environmental testing criteria. In the US, there has been a significant effort to implement widespread beneficial use of WTE ash. Significant controversy has surrounded how to properly manage WTE ash. As such, there is no clear regulatory classification of the hazardous or nonhazardous nature of WTE ash, and inconsistent management requirements that have led to uncertainty about the beneficial use status of WTE ash in the US. Starting in the early 90s, multiple research and demonstration projects nationwide have been undertaken, but less than 5% of ash is beneficially used in the US. There is a clear need to develop proven uses for WTE ash. One of the largest issues being the difficulty in producing a uniform and consistent product (Wiles & Shepherd, 1999).

One focus of attempting to use WTE ash in PCC is on the replacement of the virgin coarse aggregate portion of PCC with WTE combustion ash. Coarse aggregate is commonly within the size range of about 1/4 to 1.5 inches in diameter and function most effectively when they are hard and strong, free of contaminants, and unreactive with the cement matrix. The coarse aggregates used in a concrete mix are generally what is regionally available; in Florida this would mean a limestone. This limestone would be mined from a large quarry, crushed, and screened to the appropriate size to be used in concrete. Concrete aggregates must function from two main standpoints; performance and environmental. The coarse aggregate has an integral role in influencing the proportioning, mixing, and subsequent plastic and hardened properties of the concrete. For instance, if a weak aggregate is used a higher cementitious materials content may be required to make up for the weakness in the

aggregate. These particles must satisfy several different factors like an appropriate shape and texture, absorption, and durability to be discussed in the following sections.

With aggregates accounting for up to 75% of the total volume of a concrete mix, this accounts for a significant amount of material when considering the amount of concrete that may be poured at a traditional job site. It is apparent that WTE ash as a coarse aggregate replacement for traditional materials presents itself as an ideal opportunity for large-scale beneficial use of a material that would traditionally be landfilled. Though there may be extensive challenges facing the widespread implementation of WTE ash as a coarse aggregate replacement in concrete. If the research can prove that WTE ash can function as well or even better than traditional aggregates, cradle to cradle ash management can become increasingly mainstream. The literature illuminates useful applications in the wide range of beneficial use opportunities for WTE ash.

The body of literature concerning WTE ash as a construction aggregate has evolved quite dramatically since the inception of the idea to beneficially reuse WTE ash. There has been extensive research on the general characterization of WTE ash. When WTE plants were still an emerging technology in the early 1990s, there was no focus on beneficial use applications for WTE ash. Instead, researchers and legislators were excited about the prospect of the volume reduction afforded by burning municipal solid waste and were focused on disposal strategies. The widely cited paper *Use of incinerator bottom ash in concrete* by Pera et al. (1997) was one of the earliest and most popular papers introducing the feasibility of using WTE BA as an aggregate replacement in concrete. This paper focused on very simple break strength experiments and even outlined potential durability issues and resolutions. The field has since grown. The general trend of research regarding WTE ash as an aggregate in concrete is illustrated in Figure 2-9. A listing of key WTE ash beneficial use publications is provided in Table 2-1.

Year	Торіс	Papers	Description/Major Findings
1996	Characterization and disposal, utilization of WTE ash	Disposal strategies for municipal solid waste incineration ash residues, Ole Hjelmar.	Leachate behavior is the most important issue associated with WTE ash disposal. Leachate treatment and pretreatment are important to avoid environmental issues associated with disposal.
		Municipal solid waste combustion ash: State-of-the- knowledge, Carlton C. Wiles.	Some ashes fail TCLP, primarily for Pb and Cd but do not account for dilution/attenuation. Ash is not normally used in the US but are technically suitable for beneficial use applications. Fly ash typically contains the most heavy metals. Proper technical and engineering testing can prove these ashes to be suitable for use in the environment.
2017		Leaching behavior of municipal solid waste incineration bottom ash: from granular material to monolithic concrete, Sorlini et al.	pH is the most important factor affecting pollutant mobility. Ageing of concrete specimens aids in reducing salt leaching. Granular materials leach more contaminants than monolithic elements.

Table 2-1. Sampling of publications relating to WTE ash incorporation into PCC mixes.

Table 2-1	Continued	(2).
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1997	WTE ash Characterization and use in concrete including structural, durability, and environmental concerns	Use of incinerator bottom ash in concrete, Pera et al.	Ash-amended concretes can perform at a satisfactory level, but steps should be taken to avoid problems with swelling and cracking due to reactions between metallic aluminum and cement by reacting away the aluminum in a sodium hydroxide bath.
1998		Municipal solid waste bottom ash as portland cement concrete ingredient, Berg et al.	MSW ash residues have potential applications as an aggregate. Angularity, chloride, and sulfate content can present issues in PCC. Strength could present itself as a limiting factor. If it were to be used as an aggregate, ash may need to be treated or enhanced with processes such as metals removal, washing, and blending.
2003		Replacement of raw mix in cement by municipal solid waste incineration ash, Shih et al.	Replacement of cement raw mix with ash has been proven to work. Compressive strength at higher replacement percentages could be an issue.
2006		Utilization of municipal solid waste bottom ash and recycled aggregate in concrete, Jurič et al.	WTE ash can be used effectively in low strength requirement PCC mixes. Plastic properties were affected with introduction of ash.
		The microstructure of concrete made with municipal waste incinerator bottom ash as an aggregate component, Müller & Rübner.	The reaction of metallic aluminum with cement paste is the main factor affecting WTE ash-amended concrete durability. ASR is also a huge issue associated with amorphous silica components in the ash. Additional processing will likely be necessary to made ash a suitable aggregate product.
2009		Combined use of MSWI bottom ash and fly ash as aggregate in concrete formulation: Environmental and mechanical considerations, Ginés et al.	WTE ash in low replacement percentages is suitable to meet low compressive strength requirements such as a nonstructural precast concrete. Leaching can also be minimized with low replacement
2014		On the reliability of reusing bottom ash from municipal solid waste incineration as aggregate in concrete, Abba et al.	percentages. Suitable amounts of WTE ash replacement in PCC allows for strength values comparable
2015		Use of leaching tests to quantify trace element release from waste to energy bottom ash-amended pavements, Roessler et al.	Encapsulation in concrete mixes has proven to be effective in reducing leached contaminant concentrations of WTE ash- amended pavements.
2016		Sustainable high-quality recycling of aggregates from waste-to-energy, treated in a wet bottom ash processing installation, for use in concrete products, Van et al.	Fine and coarse aggregates can be replaced effectively with WTE ash. It is important that preprocessing of bottom ash be done to be avoid gradation and reactivity issues.
		rrom trasn to treasure, Ferraro et al.	WTE ash was successfully used as a coarse aggregate replacement in an in-use roadway with no significant issues with workability or long-term performance.

Table 2-1 Continued (3).

	Торіс	Papers	Description/Major Findings
2000	WTE treatment technology and use in concrete	Short term natural weathering of MSWI bottom ash, Chimenos et al.	90-day natural weathering of bottom ash reduced leach of heavy metals by stabilizing pH.
2006		Carbon dioxide sequestration in municipal solid waste incinerator (MSWI) bottom ash, Rendek et al.	Accelerated carbonation of bottom ash can lower hazardous nature of bottom and chemically block leaching of certain heavy metals.
2011		Recovery of MSWI and soil washing residues as concrete aggregates, Sorlini et al.	MSWI bottom ash can be reused after a washing pretreatment and pretreated ash show good characteristics for use in concrete mixtures.
2013		Upgraded MSWI bottom ash as aggregate in concrete, Van et al.	Bottom ash upgraded through metals recovery and washing can be suitable for concrete aggregate and shows similar performance to present day recycled concrete.
2016		Use of waste to energy bottom ash as an aggregate in portland cement concrete: Impacts of size fractionation and carbonation, Roessler et al.	WTE bottom ash incorporation in PCC mixes negatively impacted compressive strength. Mixes do perform well at low replacement percentages. Aluminum content is an issue with regards to hydrogen gas production.
		High performance of treated and washed MSWI bottom ash granulates as natural aggregate replacement within earth- moist concrete, Keulen et al.	Wet treatment of bottom ash results in increased removal of soluble salts, heavy metals, and organic structures. Bottom ash is in compliance with many aggregate specifications already.
2017		Hydrogen gas generation from metal aluminum-water interaction in municipal solid waste incineration (MSWI) bottom ash, Nithiya et al.	Metal aluminum in bottom ash is a huge issue with regards to hydrogen gas generation. Treatment processes to react away aluminum could be created.

Researchers main concern when WTE technology was first implemented was characterization of bottom ash components and attempting to understand the general composition of WTE ash.



Having created a body of literature that describes the chemical makeup of the ash, research focuses shift towards environmental hazards and proper disposal techniques with leaching tests and environmental toxicity characterizations.



Focus shifts to potential reuse applications of waste to energy residuals such as road base, asphalt, and concrete mixes. Basic examinations of the suitability of WTE ash as an aggregate in concrete are performed and ash is found to be generally okay for use in concrete from a performance perspective.



Research moves towards identifying and solving common physical, environmental and durability issues associated with using WTE ash as a concrete material as the scope of research narrows in on very specific problems associated with WTE ash-amended concrete.



In its current state, novel discoveries in the literature are focused around processes such as washing that can be used to create a consistent, homogenous product to be used in concrete and other construction applications. Limited landfill space, population growth, and the inevitability of WTE technology means that the body of research surrounding WTE technology will only grow.

Figure 2-9. The progression of PCC research utilizing WTE ash over time.

To achieve a suitable product for use as an aggregate in PCC production, it is important to first evaluate basic properties of the WTE ash itself. When evaluating the effectiveness of BA an aggregate in concrete, there a number of potential areas of concern beyond the typical factors that PCC aggregates are evaluated on, including particle density, metallic aluminum content, loss on ignition, chloride, sulfate, and alkali content (Van d.W. et al., 2013). WTE BA is different from traditional natural aggregates since it consists of a wide range of constituents due to the heterogeneity of MSW inputs to WTE facilities. From UF's own research seeking to incorporate WTE BA as a coarse aggregate replacement in PCC, a cross-sectional photo of a concrete cylinder cast with 50% by mass replacement of coarse aggregate with WTE BA is shown in Figure 2-10.



Figure 2-10. Cross section view of a PCC specimen cast using WTE ash as a 50% by mass aggregate replacement. Several components of traditional concrete and WTE ash are visible including limestone, cement matrix, slag, copper, metallic aluminum, and glass.

Figure 2-10 shows silver colored aluminum and black slag is visible, as well as pieces of ceramic and copper. Chemically, there are large amounts of oxides of silicon, aluminum, iron, magnesium, and sodium (Abba et al., 2014). The aim with attempting to use WTE ash as a coarse aggregate replacement is not to necessarily create a PCC product that performs better than traditional aggregates. Rather, the goal is to create an ash-amended product that performs at a satisfactory level. It is important to note that, in general, WTE incinerator BA has a lower density, higher water absorption, and lower strength than a traditional limestone or granite (Pera et al., 1997).

The negative traits of WTE ash means that often times there is a requirement for an admixture, such as a super plasticizer, in the concrete mix due to the lack of workability associated with particles that have a high porosity absorbing too much water (Van d.H. et al., 2016). When comparing to a traditional mix that does not often require the addition of any admixture because of an already satisfactory workability, thus the use of WTE ash as a coarse aggregate replacement may be significantly costlier than a traditional mix.

Of special concern in WTE ash-amended concrete is the high content of metallic aluminum. Müller & Rübner (2006) discovered that when using municipal waste incinerator BA as an aggregate component, elongated voids developed that followed the contours of aluminum grains. These voids are not found in a normal concrete mix (Müller & Rübner, 2006). They also discovered that concrete spalling can occur due to aluminum hydroxide production with aluminum grains near the surface of the concrete, this reaction is expected to proceed long after the hardening of the concrete (Müller & Rübner, 2006).

High aluminum content of WTE ash can have detrimental effects on the durability of PCC mixes. Concrete specimens created with WTE BA as a coarse aggregate replacement react in the highly alkaline environment of cement paste to create hydrogen gas. The reaction between metal aluminum and a water solution can occur spontaneously and at low temperatures and standard pressure (Nithiya et al., 2017). This hydrogen gas causes expansion and subsequent cracking of concrete specimens that has been confirmed by x-ray diffraction to be due to an aluminate reaction and production of hydrogen gas (Pera et al., 1997; Van d.H. et al., 2016; Nithiya et al., 2017). This problem can be compounded by the delay in setting time brought on by the use of super plasticizer, resulting in significantly more time for voids to form in the concrete due to hydrogen gas production.

The presence of metallic aluminum may not always be detrimental. When creating PCC concrete specimens using BA as a coarse aggregate replacement in concrete, past research indicates no negative effect on freeze-thaw durability and some have even shown a positive impact on freeze-thaw durability (Van d.W. et al., 2013; Van d.H et al., 2016). It has been postured that an improved resistance against freeze-thaw could be due to dihydrogen gas formation from metallic aluminum and zinc reactions, these hydrogen gas bubbles may tend to act as surrogate air entraining agents (Van d.W. et al., 2013).

Based on the existing literature, using a virgin WTE BA, without any modifications to account for excess metallic aluminum content, could lead to potential issues. Efforts have been made to treat metallic aluminum with a sodium hydroxide wash. The following reaction shows the production of hydrogen gas that treatment with sodium hydroxide hopes to avoid in the cement paste (Pera et al., 1997):

$$NaOH + Al + H_2O \rightarrow NaAlO_2 + \frac{3n}{2}H_2$$

Furthermore, Pera et al. (1997) found that BA has no negative effects on concrete durability if one can react away and avoid all hydrogen gas generation due to metallic aluminum content. Beyond chemical treatment, metallic aluminum can be physically

removed by means of nonferrous metal separation technology, such as an eddy current separator.

Of further concern when considering WTE ash as a coarse aggregate replacement in concrete is the potential for deleterious reactions with the aggregate, specifically the alkalisilica reaction (ASR). As previously discussed in Section 2.1, ASR occurs when there is a high alkali presence in concrete, forming alkali silicate gels that expand in the presence of moisture and induce cracking in the concrete specimen. In this case, the alkali content is clearly linked to bottle glass and other glassy amorphous silica components present in WTE BA (Müller & Rübner, 2006). WTE ash can consist of 80% of these components; thus, using WTE ash as a coarse aggregate replacement in concrete can lead to extensive damage if left unprocessed (Müller & Rübner, 2006). Furthermore, length changes due to ASR reactions have been shown to be around twice the value for a reference concrete mixture with no presence of WTE ash (Van d.H. et al., 2016). This same study found that the length change values in concrete cast with WTE bottom indicated a "serious sensitivity" to ASR but noted that the presence of an ASR is largely dependent on the specific composition of any one BA sample because of the inherent heterogeneity of WTE ash (Van d.H. et al., 2016). Traditionally, ASR can be mitigated with the use of alternative pozzolans, such as class F fly ash, at low replacement percentages of cement (Wieker et al., 1994; Islam, 2014).

Researchers have also shown that reacting the BA in a wash of sodium hydroxide is very effective in counteracting the ASR. On a large scale, this process could be quite cost prohibitive when considering the tons of coarse aggregate material required for a large-scale concrete construction project (Van d.H. et al., 2016). A visual schematic representation of the ASR is provided in Figure 2-11.



Figure 2-11. Development of ASR leading from alkali diffusion to expansion and cracking.

Despite the drawbacks due to reactions within the amended concrete, it has been shown that from a strictly performance standpoint, WTE ash can act as a suitable substitute for traditional aggregates. Pera et al. (1997) noted that BA is of average quality for a concrete aggregate.

It has been shown experimentally that WTE ash can replace a traditional concrete aggregate at low percentages and yield a satisfactory compressive strength, but with

increasing addition of waste product comes a decrease in strength (Pera et al., 1997; Sorlini et al., 2011; Van d.W. et al., 2013; Van d.H. et al., 2016; Nithiya et al., 2017). Concrete cylinders made with low percentages of WTE BA still meet design strength standards determined by a control group, and thus concrete made with recycled aggregates is suitable for applications with low compressive strength requirements such as low traffic highways and concrete bunker "Lego" blocks (Abba et al., 2014; Roessler et al., 2016). Van et al. (2016) showed that even at a water to cement ratio of 0.65 a standard compressive strength required for a prefabricated "Lego" block can still be achieved that is made with a treated WTE ash from a wet BA processing installation as a coarse aggregate replacement.

An important relationship in concrete is also that porosity of a concrete is inversely proportional to the strength of the concrete. It has been shown that concrete made with WTE BA replacement exhibits a significant increase in porosity when compared to a reference concrete, exhibiting even as much as twice the porosity of a reference concrete mixture (Müller & Rübner, 2006; Van d.H, et al., 2016). There can be some attempt at manual separation or separation based on density to get rid of low density, porous particles that would contribute to weaker concrete and improve the overall suitability of WTE ash as a coarse aggregate replacement. This is an extremely tedious and labor-intensive process.

Experiments have shown that sintering of WTE ash between 1,000 and 1,050 degrees Celsius can produce a higher quality aggregate. This modified aggregate produced "pellets with density, water absorption, and crushing strength properties comparable to commercially available lightweight aggregates" (Sorlini et al., 2011). Based on the temperature required to create this product though, it can be seen as extremely energy intensive and thus expensive to modify the WTE ash in such a way. Wet ground WTE ash has the bonus of behaving as a surrogate pozzolan leading to strength improvements and reductions in concrete permeability. The connection can be made that fine, cement sized particles attached to larger coarse aggregate sized particles can function as a pozzolanic material (Sorlini et al., 2011).

Past pilot projects have shown WTE ash as a coarse aggregate in concrete to be feasible. In Pasco County, Florida, an approximately 200 ft. long section of roadway was made from concrete pavement that had 19% by mass of its coarse aggregate component replaced by WTE BA from the Pasco County Resource Recovery Facility (Roessler et al., 2015). This roadway exhibited no significant differences in workability as reported by the construction crew, and met FDOT requirements for roadway pavements, showing late age compressive strength values at or near the control value. Since being constructed in May of 2014, the roadway has experienced significant heavy vehicle traffic and has performed well. The performance of this road has paved the way for future beneficial reuse of ash products in Pasco County (Ferraro et al., 2016).

Table 2-2 has been prepared to highlight some of the major issues associated with WTE ash as a coarse aggregate replacement in PCC. This table breaks down descriptions of the five main issues, as described by the literature, associated with WTE ash incorporation in PCC

mixes that have been discussed above. These issues include decline in workability, poor aggregate properties, hydrogen gas production, ASR, and decreasing mechanical strength.

Issue	Description
Decline in concrete	High porosity and absorptivity of WTE ash means that as an
workability	aggregate in concrete, these particles absorb a large amount
	of water and thus lead to a decline in workability. Often times
	this leads to the need for admixtures such as
	superplasticizers, that can drive the cost of a mix-up, for a
	satisfactory workability.
Poor aggregate properties	WTE ash typically has low density, high absorptivity and
	porosity, and low strength. This can lead to weak or
	unworkable concrete. Important to note as well is the high
	likelihood of environmental contaminants or reactive
	elements that can cause issues such as ASR or hydrogen gas
	formation. Rapid sintering or proper homogenization of
	aggregates, as well as post-production upgrades such as
	removal of ferrous/nonferrous metals and particles with
	poor aggregate characteristics can remedy this problem.
	Washing also presents itself as a solution.
Hydrogen gas production	This becomes an issue because of the high metallic
	aluminum content within WTE ash. This aluminum reacts in
	the alkaline cement mixture, creating hydrogen gas that
	causes voids and other deformations and reduces the
	strength and durability of concrete. This can be remedied by
	washing in a reactive solution hydroxide solution, or physical
Allrali cilico reaction	ASD can be caused by a high allrali presence in concrete that
Alkali-silica reaction	ASK can be caused by a high alkall presence in concrete that
	which loads to grading. Washing has been shown to
	mitigate ASP as well as use of different pozzelans as
	supplementary comentitious materials
Decreasing mechanical	Posoarch has shown that concrete with increasing
strength	replacement percentages of its coarse aggregate content
Strength	with WTF ash leads to a decrease in a mechanical strength
	This is due to the inherently weaker strength of the
	aggregate and can be remedied with time and labor
	intensive aggregate upgrades but is largely unavoidable.

Table 2-2. Overview of performance deficiencies due to WTE ash coarse aggregate replacement.

2.2.3 HMA Review and Effects of WTE Ash Utilization

AC pavement consists of three principle phases: aggregates, asphalt binder, and air. It is the properties and ratios of these components which determine AC's physical properties such as strength, durability (e.g., resistance to rutting/deformation), and resistance to water damage (i.e., moisture susceptibility). Aggregates form the skeleton of HMA and dictate the type of mix that it is (e.g., dense/open/gap-graded) and is typically controlled to specified limits, such as those used in the Superpave mixture design. Using WTE ash in HMA involves

blending this material in to meet certain gradation requirements while also producing a mixture that still meets the necessary asphalt volumetrics and physical properties. Asphalt volumetrics have a significant influence on the physical properties of an AC mixture. Meeting volumetrics for HMA concrete mixtures requires attaining the appropriate air void content (which is approximately 4.0%) along with voids in mineral aggregate (VMA) and voids filled with asphalt (VFA). VMA is the percentage of the mixture (by volume) that consists of voids between the mineral aggregates, while VFA is a percentage of how much VMA is filled with asphalt binder. Figure 2-12 shows a volumetric phase diagram of a typical HMA mixture.



Figure 2-12. Volumetric phase diagram of a typical HMA mixture.

While the properties of WTE ash are similar to conventionally used aggregates, their unique properties must be considered when designing an asphalt mixture. Most studies in the literature utilize WTE ash in dense graded mixtures but a few consider open-graded (Luo et al., 2017) and stone matrix (Xue et al., 2009) mixtures. For these types of mixtures, WTE ash has been utilized more so as a fine aggregate or filler rather than a significant aggregate replacement (i.e., >10% of total aggregate mixture).

The most common method of manufacturing AC pavement is known as hot-mix asphalt (HMA). This process involves heating aggregates and asphalt binder to temperatures of 300-350°F (149-177°C). There are other methods used in producing AC pavement such as

warm mix asphalt (VMA) and cold mix asphalt which use lower temperatures (< 300°F, or 149°C), but few works to date has been done with these methods for utilizing WTE ash nor are these methods nearly as common as HMA; hence, this report only discusses the effects of WTE ash in HMA mixtures. These two heated materials are then mixed together in a specified ratio previously determined through mixture design to create a suitable mix. This mix is then poured onto a base road layer and compacted using rollers as shown in Figure 2-13.



Figure 2-13. Construction of an HMA ash-amended pavement.

Just as there are multiple ways to construct roadways, there are also multiple ways of developing an HMA mix design. This section discusses two major methods of developing a mix design for HMA: Marshall and Superpave. These methods are the most widely used methods for asphalt mixture design around the world and also were used to develop ashamended HMA designs for studies in the literature to date. Each of these mixture designs follow a similar pattern: several stockpiles of aggregates are combined at specific ratios to produce a well graded particle size distribution (or gradation). Several trial aggregateasphalt blends are then made with each having a different asphalt content to determine the optimal asphalt content. Where the Marshall and Superpave mixture designs can differ is the monitoring of the composite gradation (i.e., what control points must be met), how is an asphalt binder selected, how is the optimal asphalt binder content determined (i.e., what parameters must be met, such as volumetrics and strength), and compaction method. Nearly all ash-amended HMA mixtures studied in the literature are based on a Marshall design for two primary reasons: 1) the time that these experiments were carried out was before the development and implementation of the Superpave method, and 2) a vast majority of these experiments are performed in countries where Marshall mix is the standard for asphalt mixture design. Table 2-3 briefly highlights the key differences between these two mixture designs.

Property	Marshall	Superpave
Aggregate	Agency/owner	Gradation and aggregate
	requirements	property restrictions
		(e.g., angularity,
		durability, soundness)
Binder Selection	Agency/owner	Expected traffic and
	requirements, stiffness,	climate conditions
	expected traffic and	
	climate conditions	
Asphalt Binder Content	Ideally 4.0% air voids if	Approximately 4.0% air
	minimum Marshall	voids (accepted at 3-5%
	stability (strength) and	air voids) and other
	flow (deformation) is met	volumetric ranges met.
Compaction Method	Marshall Hammer	Gyratory Compactor

Table 2-3. Comparison between Marshall mix and Superpave mix designs.

Interestingly, Xue et al. (2009) compared volumetric properties and physical testing properties of two identical ash-amended mixtures and compacted them using the Marshall hammer and the gyratory compactor used for Superpave designs and found significant differences in volumetric and physical properties, such as optimum asphalt binder content and tensile strength. The Marshall hammer compacts using an impact force (hammer) for a specified number of blows per side of an asphalt specimen, while the gyratory compactor "kneads" the mixture as it compacts at an angle in an attempt to have particles slide into a denser arrangement.

Besides the test pavements constructed by the US FHWA in the 1970's, one of the earliest studies reported in the literature utilizing WTE BA in HMA was by Walter et al. (1976). Their work showed that BA could be utilized in replacements up to 50% and achieve acceptable Marshall mix properties, such as stability and flow and volumetrics while also noted as an economically and environmentally viable process. While a significant portion of WTE ash was successfully utilized as aggregate replacement in this study, more recent studies have shown optimal volumetrics at lower total replacements (< 40%) (Hassan, 2005; Huang et al., 2006; Chen et al., 2008; Toraldo et al., 2013; An et al., 2014).

Aside from volumetrics, examining the physical properties of an ash-amended HMA mixture is necessary to understand how the addition of ash may change their behavior. Several important physical properties for HMA include rutting or degradation susceptibility, tensile strength, and moisture susceptibility. These properties along with why they are important and typical testing protocols are shown below in Table 2-4.

Physical Property	Rationale	Procedure	Common Testing Methods
Rutting/degradation susceptibility	Identify an HMA pavement's resistance to deformation	Compacted HMA specimens are subjected to physical wear and stresses, typically at elevated temperatures, to attempt to breakdown its structure	Empirical testing methods such as the Hamburg Wheel Tracking test and the Asphalt Pavement Analyzer (AASHTO TP 63) and Cantabro test
Tensile strength	Identify cracking potential (higher tensile strengths correlate to higher resistance to cracking and vice versa)	Measuring HMA strength in tensile (across its vertical diametral plane)	ASTM D6931
Moisture susceptibility	Identify how moisture may infiltrate and damage HMA pavement based on aggregate blend and asphalt binder content	Compare tensile strength before and after a series of weathering cycles to simulate long-term field conditions	Boiling test (ASTM D3625), Lottman test, Modified Lottman test (AASHTO T283), Lottman (Tunnicliff procedure)

Table 2-4. Physical properties of interest for HMA along with testing methods.

Rutting, or deformation, is a frequent form of degradation on HMA roadways. This type of failure is caused by a thin asphalt film around aggregates, an insufficient aggregate gradation that does not allow for stress loading absorption, and a lack of interlocking angular particles. Rutting susceptibility and deformation capacity of HMA are commonly determined using empirical tests with predetermined tolerances (e.g., no more than 5 mm deformation). One set of methods include the use of wheel tracking devices such as the Asphalt Pavement Analyzer (APA) test. Generally, these devices heat HMA specimens to elevated temperatures representative of field conditions (e.g., hot summer day) and then subject them to rutting via a wheel apparatus that puts stresses on the HMA surface over a number of cycles. The amount of deformation measured at the end of these tests allows for an assessment of how well an HMA mixture can be expected to resist rutting under real-world conditions. An example of the APA rutting test is shown in Figure 2-14.



Figure 2-14. Example of an APA rutting test.

Wheel rutting testing done in the literature has shown an increase in rutting susceptibility with increasing ash replacement (Huang et al., 2006; Chen et al., 2008). Furthermore, it is important to consider that while rutting may increase it may still perform to federal and state requirements for rutting resistance. For example, the Florida Department of Transportation (FDOT) allows no more than 4.5 mm of deformation after 8,000 cycles.

Tensile strength of HMA is measured by performing an indirect tensile (IDT) strength test, which involves applying pressure to the diametral dimension of an HMA specimen until cracking failure occurs. An example of this procedure shown in Figure 2-15.



Figure 2-15. An IDT testing apparatus with HMA specimen.

IDT strength is influenced by aggregate gradation, individual aggregate strength, and aggregate surface chemistry. Particles that interlock with one another can resist deformation, with aggregates with rough surfaces and high angularity create more interlocking. Traditional aggregates such as limestone and granite are relatively uniform from a known stockpile and their physical and chemical properties typically remain constant from a source. On the other hand, WTE ash can be a highly variable product with materials of varying strengths from glass to slag along with various morphology, including

surface chemistries. Yuan et al. (2017) explored this concept by studying the strength of the area in HMA formed between an aggregate and the asphalt binder (known as an interfacial zone). This zone is formed when chemical reactions with the surface of the aggregate occur with the asphalt binder and develop a different microstructure than the rest of the aggregate and asphalt bitumen. Larger interfacial zones correspond to lower IDT strengths as these areas are weaker and allow cracks to propagate under excessive tensile stresses. A study performed by Yuan et al. (2017) suggests that particular components of WTE BA (e.g., ceramics) may create weaker interfacial zones relative to traditionally used aggregates while other components of WTE BA (e.g., slag) may actually increase the strength of these interfacial zones. Surface functional groups, or chemistry, differs from aggregate to aggregate. For instance, limestone is found to better absorb asphalt bitumen compared to granite due to the fact that granite surface functional groups are more acidic while limestone contains more basic functional groups (Yuan et al., 2017).

Because IDT strength is dependent on several aggregate properties, studies have shown conflicting results on whether increasing WTE ash replacement in HMA mixtures affects IDT strength. Trends have varied from a general increase in strength with increasing ash replacement (Huang et al., 2006; Luo et al., 2017) to a general decrease in strength with increasing ash replacement (Garrick & Chan, 1993; Hassan, 2005; Chen et al., 2008; Toraldo et al., 2013). Meanwhile, nonlinear relationships where peak values are derived at arbitrary ash replacements have been observed. An et al. (2014) found that they achieved peak tensile strengths at a 20% ash replacement while observing lower strengths at 10 and 30% replacements. Hassan & Khalid (2010) observed increased tensile strengths with increasing BA replacement until 80% where a sharp decrease occurred.

Arguably, the most significant parameter that ash-amended HMA has difficulty attaining in the literature when physical performance is considered is moisture susceptibility. As mentioned previously, moisture susceptibility provides an indication on how water may damage HMA under in-service conditions and is also an indicator of the material's durability over time. There are a number of methods used to evaluate a mixture's moisture susceptibility, with the tensile strength ratio (TSR) test as per AASHTO T-283 being one of the most common and widely utilized by highway and transportation agencies (Hicks, 1991). This type of moisture susceptibility testing is done by preparing several HMA specimens to approximately 7% air voids (to reflect initial compaction in the field) and conducting two IDT tests, one on an unconditioned (dry) HMA specimen and one on a conditioned, (one freeze-thaw cycle) HMA specimen and calculating the strength ratio between the conditioned and unconditioned specimens. This conditioning process is outlined in Figure 2-16 below.



Figure 2-16. Moisture susceptibility testing as per AASHTO T-283.

Dividing the conditioned specimen's strength by the unconditioned sample's strength vields the tensile strength ratio (TSR). Similar to the IDT results, studies have observed that increasing the rate of ash substitution decreases TSR (Garrick & Chan, 1993; Hassan, 2005; Chen et al., 2008; Xue et al., 2009; Hassan & Khalid, 2010; Toraldo et al., 2013), but the opposite effect has also been observed (Huang et al., 2006; Luo et al., 2017). Again, nonlinear relationships have also been observed with peak values observed between minimum and maximum replacement values (An et al., 2014). Moisture susceptibility relates to the bonding between asphalt binder and the aggregate along with asphalt film thickness. High moisture susceptibility (loss of adhesion) may be the result of glass and ceramic content in WTE ash. Glass and ceramic, as present in WTE ash is often in glass bottle pieces or broken ceramic pieces which are smooth and nonporous which does not support adhesion of the binder to the glass/ceramic aggregate. In a few studies, decreasing moisture susceptibility for ash-amended HMA has been attributed to increased lime (CaO) content within the HMA sample (Xue et al., 2009; Huang et al., 2006; Luo et al., 2017). Conversely, an increase in moisture susceptibility has been attributed to salts such as sodium and potassium on the surface of ash-derived aggregates. The interaction between asphalt binder and these salts result in a water-soluble bond that is readily displaced in water causing mixtures to have higher moisture susceptibility (Hicks, 1991).

One consequence of utilizing WTE ash in an HMA aggregate blend commonly report includes an increase in asphalt binder demand (Ogunro et al., 2004, Hassan, 2005; Huang et al., 2006; Chen et al., 2008; Hassan & Khalid, 2010; Toraldo et al., 2013; An et al., 2014). Lynn et al. (2017) estimated this increase to be approximately 0.1%, by mass, for every 1%,

by mass, addition of WTE ash aggregate. This phenomenon is attributed to the highly absorptive nature of WTE ash relative to other natural aggregates (Ogurno et al., 2004; Hassan, 2005; Huang et al., 2006) and the elevated amount of dust (i.e., < No. 50) material present on the surfaces of coarser ash particles. Because WTE ash tends to be a highly porous material and has a high surface area, more asphalt bitumen is absorbed into the ash aggregate which leaves less binder to fill in the voids in between the mineral aggregate (VMA). Not only does this have a significant economic impact, as asphalt binder comprises about 95% of the cost of HMA, but there is also less binder to form a thick asphalt binder film on the surface of aggregates which can have a negative impact on HMA physical performance. In the literature, virtually all mixtures can meet air voids and VFA if designed correctly. The most difficult parameter to meet is typically VMA, as it is usually lower than the specified requirement. This issue can be attributed to WTE ash's lower specific gravity which is considered when calculating VMA; however, it is important to point out that even mixtures using conventional virgin aggregates can struggle to meet VMA requirements (Kandhal et al., 1998). The purpose of VMA is to ensure that mixtures with adequate performance qualities are developed. However, that does not automatically preclude that a mixture with lower VMA will have poor performance properties.

One solution proposed for improving the volumetric and physical properties of WTE ash for use in HMA involves treating, or washing, the aggregate beforehand. Washing is a common treatment for mined aggregates (Tarrar & Wagh, 1992). The effect of washing WTE ash-derived aggregate for use in HMA has not been examined directly in reuse studies, but two studies have explored how a washing pretreatment step may change the properties of WTE ash in context with HMA mixtures. Still, Chen et al. (2008) was the only study that described the washing process, which involved using a liquid to solid (LS) ratio of 2 for a duration of 30 minutes to simulate how rainfall may alter an unprocessed WTE ash stockpile over time during storage. The washing processes in these studies changed the overall gradation, specific gravity, absorption, and surface pH of WTE ash as shown in Table 2-6.

Interestingly, Huang et al. (2006) decided against using washed ash in their HMA mixtures due to the quality of wastewater that was generated during the washing process, while for Chen et al. (2008) the properties of their washed ash was the motivation for only testing mixtures utilizing washed ash. These studies did not directly test mixtures made with unwashed and washed ash for volumetric and physical properties such as change in optimum asphalt binder demand and moisture susceptibility to make a conclusion on how a washed ash may behave differently compared to its unwashed counterpart. This impact can have a significant effect on future beneficial reuse efforts of WTE ash in asphalt pavements. Nevertheless, the effects that washing has on the ash-derived aggregate are clearly shown in Table 4 and from these the potential impacts that it may have on an HMA mixture can be investigated.

As shown previously in Table 2-5, the gradation is much coarser for the washed ashderived aggregate used in Huang et al. (2006) suggesting that the washing process in this study also involves a screening step to eliminate fine aggregate materials (<No. 4, 4.76mm). Meanwhile, the change in gradation in Chen et al. (2008) reflects a gradual reduction in finer particulates. As previously mentioned, the gradation of an asphalt mixture is arguably the most important parameter for HMA as it dictates the type of mixture it is and its physical properties.

	% Passing (by Mass)				
Gradation	Huang e	t al. 2006	Chen et al. 2008		
Gradation		After		After	
	Untreated	Washing	Untreated	Washing	
19 mm (3/4 in.)	100	100	100	100	
12.5 mm (1/2 in.)	100	75.1	93	92	
9.5 mm (3/8 in.)	96.7	45.3	84	81	
4.76 mm (No. 4)	72.5	5.1	62	51	
2.38 mm (No. 8)	51.8	2.4	49	47	
1.19 mm (No. 16)	34.9	2.0	30	29	
0.6 mm (No. 30)	22.2	1.7	21	19	
0.3 mm (No. 50)	14.8	1.5	12	11	
0.15 mm (No. 100)	9.7	1.2	8	7	
0.075 mm (No. 200)	6.3	0.6	5	2	
	Huang et al. 2006 Chen et al. 2008				
Other Aggregate Properties		After		After	
	Untreated	Washing	Untreated	Washing	
Specific Gravity	1.851	2.283	1.856	2.293	
Absorption (%)	15.7	6.0	14.22	10.78	
Angularity (%)	41.4	54.4	-	-	
Abrasion (%)	-	-	33	29	
Soundness (%)	-	-	7.8	6.2	
Surface pH	12.1	10.4	11.2	9.9	

Table 2-5. Properties of WTE ash-derived aggregates for HMA before and after washing pretreatment (Huang et al., 2006; Chen et al., 2008).

The specific gravity of WTE ash increased after a washing treatment. This is speculated to be the result of finer, lightweight particulates being removed during washing leaving coarser, denser aggregates. The specific gravity of a mixture dictates its density with respect to water, with a higher value indicating a greater density. WTE ash still has a lower specific gravity after a washing treatment (~2.2) when compared to other traditionally used aggregates such as granites (~2.6) and limestones (~2.35). Specific gravity plays a role when determining an asphalt mixture's volumetric properties, such as VMA, which has a set minimum depending on the maximum aggregate size used in a mixture. Voids filled with asphalt (VFA) change with traffic level or anticipated loading over its service life. VMA is then calculated by the following:

$$VMA = \left(1 - \frac{G_{mb}(1 - P_b)}{G_{sb}}\right) * 100$$

Where,

- G_{mb} = Bulk specific gravity of the compacted HMA mixture
- P_b = The asphalt binder content of the compacted HMA mixture
- G_{sb} = The bulk specific gravity of the aggregate blend
- VMA = Voids in the mineral aggregate, expressed as percent (%)

As mentioned previously, even mixtures using conventional virgin aggregates can struggle to meet VMA requirements and these limits are proposed to ensure that mixtures with adequate performance properties are consistently prepared (Kandhal et al., 1998). Interestingly, Chen et al. (2008) noticed an increase in VMA with increasing ash replacement, which is opposite of what is typically observed in the literature for unwashed ash (Ogunro et al., 2004). This trend was attributed to increases in binder content with increasing ash replacement, although it is unclear what the influence of washing the ash may be on this observation, if any. It is possible that washing could reduce the dust residing on the coarse ash particles and thereby increase the amount of space between each aggregate particle (i.e., the VMA).

These washing processes also reduced absorptivity by a significant fraction. While Huang et al. (2006) shows a greater reduction in absorptivity (9.7%) compared to Chen et al. (2008), which showed a 3.44% reduction, the Huang et al. study also eliminated much of the finer aggregate. The absorptivity reduction observed in Chen et al. (2008) is likely attributed to the wash off of finer, more absorptive ash aggregate and dust on the surface of coarser ash particles. The high proportion of finer aggregates and dust that remain on coarser ash aggregates, even with extensive dry screening, can remain relatively high when compared to other conventional aggregates. As previously mentioned, the presence of these fines are believed to be a contributing factor to the need for higher asphalt binder demands for asphalt mixtures using WTE ash, even if the gradation of the mixture remains consistent (An et al., 2014). Not to mention, fine aggregates typically exhibit a higher surface area. Since asphalt binder constitutes the majority of the material costs for asphalt pavement, a reduction in binder content can significantly increase the attractiveness of reusing WTE ash in this material along with limiting the use of binder which also contributes to various environmental issues.

Aside from volumetrics, washing WTE ash before utilizing it in HMA can have significant effects on asphalt-aggregate adhesion, which strongly influence the physical properties of HMA mixtures, particularly moisture susceptibility of ash-amended mixtures. The high absorptivity (porosity), alkali content, and dust content can have detrimental effects on asphalt-aggregate adhesion. Interference with adhesion or complete loss is significant, as it one of the properties that gives an HMA mixture it's strength and durability besides its gradation. The strength of an asphalt mixture is the sum of aggregate particle interlocking and cohesion, or the aggregate-binder bonding. There are multiple theories that attempt to explain asphalt-aggregate adhesion, with each theory contributing to the overall explanation of this phenomena (Heffer & Little, 2004). These theories boil down to the

notion that asphalt adhesion to aggregates are both physically (e.g., particle porosity, surface texture) and chemically (e.g., functional groups, aggregate composition) based. The adhesion theories and brief explanations for each are provided in Table 2-6.

Theory	Mechanisms
(Weak) Boundary Layer Theory	Contaminants on aggregate surfaces such as organics and dust, or even dissolution of aggregate surface by pH of water, may interfere with adhesion. Aggregate porosity may absorb low molecular fractions of asphalt vs. high molecular fractions causing a brittle interface.
Mechanical Theory	Occurs at two levels: macroscopic and microscopic. Macroscopic level includes size, angularity, gradation of aggregates, porosity (lock and key effect), and surface texture. Microscopic scale considers aggregate surface at microscopic level along with phases of different hardness/morphology.
Electrostatic Theory	Surface charges develop when aggregate surfaces come into contact with water. A thin layer of water (monolayer) can remain on aggregate surfaces below 500-1000 C. These surface charges may attract or repel components.
Chemical Bonding Theory	Aggregates are more variable in their chemistry and hence dominate adhesion in aggregate-asphalt systems. The acidic fraction of asphalt (the most polar fraction) adsorbs onto aggregate surfaces forming bonds of variable durability, which is affected by the mineralogy of the aggregate it comes into contact with. The surrounding pH of the system may also impact what bonds are stable vs. unstable (i.e., under what pH conditions is adhesion weakened/strengthened).
Thermodynamic Theory	The concept that the HMA mixture system has free energy, which must be minimized to reach equilibrium. Asphalt and aggregates contain compounds and structures that have their own free energies. Thus, the free energy (work) of adhesion, or strength of adhesion can be estimated.

Table 2-6. Theories of explaining interactions of aggregate-asphalt adhesion (Heffer & Little, 2004).

With respect to physical properties, there are a few properties characteristic of WTE ash that may interfere with asphalt adhesion. One of these properties is dust content. WTE ash

has been noted to contain more dust (e.g., < No. 50 material) than conventional aggregates. The extra dust may also reflect the breakdown that can occur doing handling and processing (Musselman et al., 1994). This higher proportion of dust on the coarser ash particles can interfere with asphalt adhesion by preventing adequate asphalt wetting onto the aggregate surface. The effect of dust content for asphalt-aggregate adhesion has been explored throughout the literature. A study by Tarrar and Wagh (1992) mentions that dust may trap air during mixing causing a weaker aggregate-binder bond. Not only does this air prevent contact between the asphalt and the aggregate, but it also creates a pathway for water intrusion (Tarrar & Wagh, 1992).

Another physical property that influences adhesion that has been mentioned previously is absorptivity of WTE ash. A washing pretreatment has been shown to reduce absorptivity (Huang et al., 2006; Chen et al., 2008). In addition, Chen et al. (2008) compared the surface area (m^2/g) of unwashed to washed ash and noted a slight reduction (9.85 to 9.41). These findings suggest that the porosity of WTE ash may be reduced by a washing pretreatment which could have its own potentially beneficial impacts on WTE ash mixtures. Heffer & Little (2004) report that while porosity can promote adhesion by providing a mechanical interlocking effect, too much surface texture affects the asphalt's ability to coat the particle. In addition, greater absorption of asphalt binder can result in a thinner and/or uneven asphalt binder thickness around ash aggregates which limits its adhesion to other aggregate particles. Previous works also suggest that the porosity of aggregate affects the molecular distribution of organic compounds within the asphalt binder itself (Jamieson, 1995; Heffer & Little, 2004; Luo & Lytton, 2013). Asphalt binder consists of low weight and high weight organic molecules with the low weight molecules migrating into the porous aggregate structure becoming segregated from the higher weight molecules, which are typically polar. This stratification results in a weaker bond surface which can lead to less adhesion strength and promote water damage in a mixture (Heffer & Little, 2004; Luo & Lytton, 2013).

From a chemical standpoint, Table 2-6 also shows that these washing pretreatments reduce the aggregate surface pH by approximately 1-2 units. The surface pH of aggregate particles is known to be a significant influence on binder adhesion with a more basic surface adhering better to the slightly acidic binder (Hicks, 1991; Yuan et al., 2017). This change in pH may also indicate changes in compounds and functionalities present on the surface of these ash particles. Stripping of asphalt binder from aggregates is dependent on the properties of the asphalt binder (i.e., functional groups) and the aggregate itself (i.e., mineralogy). Research suggests that moisture susceptibility is largely dictated by aggregate mineral composition. For instance, Jamieson (1995) found that while calcium, iron, magnesium, and aluminum based chemical sites on an aggregate surface have a high affinity for asphalt (and hence readily adheres to asphalt), the presence of sodium and potassium were observed to have a low affinity for asphalt (and hence readily strips off of asphalt) increasing a mixture's moisture susceptibility if present. WTE ash contains a significant amount of sodium and potassium (alkalis), which may play a contributing factor in the higher moisture susceptibility of ash-amended HMA.

Chen et al. (2008) also noted the effect that salts such as sodium and potassium may have on IDT strength for conditioned moisture susceptibility specimens. They suggest that salts in WTE ash could contribute to decreasing IDT strengths (and as a result, lower TSRs). Their explanation is that rather than simply dissolve these soluble salts may expand during dehydration and exert pressures within the asphalt-aggregate matrix causing weakening in the system. On the other hand, Huang et al. (2006) suggested that salts could have been the reason why with increasing ash replacement they observed increasing TSRs. It's well noted in the literature that the presence of calcium oxide (lime) decreases moisture susceptibility, while the presence of alkali salts (e.g., sodium/potassium) result in water soluble bonds that will dissolve when in contact with water thus resulting in loss of adhesion (Hicks et al., 1991; Heffer & Little, 2004).

Another factor that may influence asphalt adhesion is silica content of the aggregate. Due to the nature of WTE ash, silica (largely in the form of discarded glass/ceramic products) exists in large quantities in this material. As shown for siliceous aggregates (e.g., granites), the acidic functional groups in asphalt binder (e.g., carboxylic acids, sulfoxides, 2-quinolones) will readily adsorb onto the aggregate and form hydrogen bonds with silanol groups, which are also easily displaced by water (Heffer & Little, 2004). However, Heffer & Little (2004) report that if these functional groups are tied up by CaO, then the reactive sites on a siliceous surface may form strong water-resistant bonds with the remaining nitrogen groups in asphalt binder. It is unclear as to what reductions, if any, may occur of silica in WTE ash with a washing pretreatment. Nonetheless, it is important to consider the potential contributions from this type of interaction when assessing asphalt adhesion to ash-derived aggregates.

From examining the literature and evaluating the negative impacts that unwashed ash may have on PCC and HMA mixtures in the forms of excessive deleterious materials, a pretreatment step, such as washing, may improve the properties of ash-amended PCC and HMA. In the next section, the effects of pretreatments on WTE ash are investigated.

2.3 Treatment Methods Used for WTE Ash

Although WTE displays a potential for reuse as aggregates, the reuse applications of WTE ash are limited by some of its inherent properties. Most notably, there is a potential risk of harmful release of pollutants into the environment and the effects of the properties of WTE ash, such as dust and alkali content, that have negative impacts on ash-amended products. In order to improve ash characteristics for a better reuse opportunity as well as a safer disposal, several techniques of treating WTE ash to improve the ash characteristics have been explored in literature: solidification/stabilization processes, thermal treatment and separation processes. Solidification/stabilization (S/S) aims to chemically or physically immobilize elements in WTE ash using additives or binders. S/S includes using cement (or other cementitious material) for stabilization, while carbonation involves ash reacting with CO₂ to reach chemical equilibrium (chemically stabilize). Thermal treatments include melting and sintering ash, which oxidizes or destroys organics and stabilizes heavy metals by employing high temperatures. Separation processes aim to remove pollutants from WTE

ash using sieving, magnetic separation, and washing ash using various solutions and electrochemical process.

2.3.1 Solidification/Stabilization Using Additives

Extensive works with regards to S/S treatment have been carried out mainly for fly ashes as a de-hazardous characteristic treatment before disposal. Polymeric, bitumen, and cement-based binders have been explored in literatures. Cost is a critical factor in binder selection. Inorganic binders such as cement are relatively cheap while polymeric binders are more expensive; thus cement-stabilization is the most commonly used S/S method. However, research using the cement-stabilization process for WTE ash appears to limit the ash replacement percentage to less than 20% by mass of the final ash-cement product since the higher ash percentage will delay the final setting time of the cementitious mixtures and also create leaching concerns due to ash enrichment in chlorides, sulfate and heavy metals (Mangialardi et al., 1999). This significantly increases the volume of solidified product which increases the shipping and landfill costs when disposed. Although leachability of pollutants is reduced after S/S, leaching of soluble contaminants from these ash-cement mixtures may still pose a problem inside landfills (Deboom et al., 2005; Chiang et al., 2010).

2.3.2 Carbonation

Contact with CO₂ during natural weathering can help to decrease the pH of ash, stabilize certain elements, and may also improve physical properties of ash with respect to decreasing ash porosity (Todorovic et al., 2006; H.K. Lam et al., 2010). However, the process is slow, and the leachability decrease is usually not enough to bring it below typical utilization requirements (Dou et al., 2017). Accelerated carbonation has been proved to be efficient in decreasing some heavy metal leaching, especially lead; however, it has limited capacity in treating soluble salts (Alba et al., 2001; Duo et al., 2007).

2.3.3 Thermal Treatment

Thermal treatment was found to be effective in stabilizing heavy metals and destroying organic pollutants. The main categories for thermal treatment are vitrification, melting, and sintering (Lindberg et al., 2015). Promising results of heavy metals removal through evaporation using thermal treatment have been reported (Stuki and Jakob, 1997; Lindbeg et al., 2015). However, this technology is energy intensive and results in air emissions of chlorides and heavy metals under high temperatures that must be properly handled making this process costly when compared to other treatment methods (Duo et al 2017; Quina et al 2008).

2.3.4 Separation Process

The separation process is an agglomerate of multiple treatment techniques including sieving, magnetic separation, and washing. Dry sieving and magnetic separation are applied before WTE ash is beneficially reused as ash-derived aggregates (e.g., road base aggregate or coarse/fine aggregates), which removes some deleterious material from WTE

ash and also recovers metals for reuse. These systems remove ferrous and nonferrous metals (e.g., aluminum, brass). The majority of nonferrous metals in WTE ash consists of aluminum, which is particularly problematic for PCC applications as it reacts under alkaline conditions (pH > 12) to form hydrogen gas which weakens and ruptures concrete. After screening and metals recovery, washing processes may be used which can involve various solutions such as water, acid solutions, supercritical fluid or other chelating agents to lower the total concentrations of pollutants in the ash and improve its physical properties. This type of treatment is a proven and reliable technique especially for soluble salts (Todorvic 2006), but usually requires large amount of solvents and will produce wastewater that needed to be treated afterwards. Wang et al. (2015) also found that washing pre-treatment using nitric acid for fly ash will increase the cancer risks by increasing the total heavy metal concentration. Compared to washing using other solvents, washing with water is considered to be most cost-effective method (Lam et al., 2010). A relatively low amount of water and short treatment time is usually sufficient to remove most of soluble salts in WTE ash. Kim et al. (2003) found that washing a mixture of WTE BA and fly ash with water at L/S of 2.5 and contact time of 5 minutes, about 77% of chlorides are removed. Thus, washing with water is considered as a promising, economic, and straightforward treatment to help improve WTE reuse. Hence, the focus of this review is mainly focused on washing with water (referred to as ash washing hereafter).

Studies on washing technology are global, with research reported in the USA, throughout Europe in countries including the Netherlands, Germany, Sweden, Italy, and Denmark as well in Asia in China and Japan. The objectives for ash washing in studies throughout the literature are presented in Table 2-7.

Title	Author	Country	Material	Objective
A two stage treatment for MSWI bottom ash to remove agglomerated fine particles and leachable contaminants	Qadeer Alam et al. (2017)	The Netherlands	WTE BA	Remove agglomerated fines before recycling and study leaching behavior of Cr, Mo, Sb, chloride and sulfates under different washing parameters.
Effect of water washing on removing organic residue in bottom ashes of municipal solid waste incinerators	Yen-Ching Lin et al. (2011)	Taiwan	WTE BA	Remove organic contaminants in MSWI bottom ash before recycling applications
Combining sieving and washing, a way to treat MSWI boiler fly ash	Aurore De Boom et al. (2015)	Germany	WTE FA	Valorize MSWI boiler ash to create non-hazardous material for both safe disposal and further utilization.
Chlorides removal and control through water- washing process on MSWI FA	Xiaofei Chen et al. (2016)	China	WTE FA	Remove chlorides for MSWI fly ash to be used in cement kiln
Water washing effects on metals emission reduction during MSWI FA melting process	Kung-Yuh Chiang et al. (2010)	Taiwan	WTE FA	Remove soluble chloride and their salts from fly ash to reduce the emission of hydrogen chloride and volatile metallic during the following melting process.
Removal of hazardous material from MSWI fly ash- an evaluation of ash leaching method	Karin Karlfeldt Fedje et al. (2010)	Sweden	WTE FA	Remove hazardous metals from MSWI FA by using alternative leaching media as well as mineral acids

Table 2-7. Washing objectives within the literature.

Table 2-7 (2). Continued.

Degradation characteristics of dioxin in the fly ash by washing and ball-milling	Raohua Li et al. (2017)	China	WTE FA	Reduce chlorine, dioxin and heavy metals in fly ash for environmental concerns
treatment				
The effect of washing on cement-based stabilization of MSWI fly ash	Xuexue Wang et al. (2016)	China	WTE FA	Remove chlorides before cement-stabilization process for MSWI fly ash
Characterization of heavy metals and PCDD/Fs from water washing pretreatment and a cement kiln co- processing municipal solid waste incinerator fly ash	Dahai Yan et al. (2018)	China	WTE FA	Remove alkali chlorides, soluble salts and the amphoteric heavy metal in MSWI fly ash before the cement kiln
Effect of water washing on the co-removal of chlorine and heavy metals in air pollution control residue from MSW incineration	Zhenzhou Yang et al. (2017)	China	WTE FA	Remove chloride and heavy metals in APC residue before cement kiln
Chlorides behavior in raw fly ash washing experiment	Fenfen Zhu et al. (2010)	Japan	WTE FA	Investigate the chlorides behavior during washing process
Comparison of two types of municipal solid waste incinerator fly ash with different alkaline reagents in washing experiment	Fenfen Zhu et al. (2009)	Japan	WTE FA	Determine the optimum conditions for washing to remove chlorides and toxic constituents
Heavy metal behavior in "Washing-Calcination- Changing with bottom ash" system for recycling of four types of fly ash	Fenfen Zhu et al. (2018)	China	WTE FA	Investigate the heavy metals behavior during washing process
Can washing-pretreatment eliminate the health risk of MSWI fly ash reuse?	Yao Wang et al. (2015)	China	WTE FA	Assess the effects and health risks associated with washing of MSWI fly ash
Optimizing the APC residue washing process to minimize the release of chloride and heavy metals	J.M. Chimenos et al. (2005)	Spain	WTE FA	Remove chloride and heavy metals using washing as the single treatment before final disposal and determine the optimum washing process
Evaluation of pre-treatment methods for landfill disposal of residue from municipal solid waste incineration	Sang-Yul Kim et al. (2003)	Japan	WTE CA	Evaluate the optimum processing conditions and effectiveness for different pretreatments of MSW combined ash

Generally, washing studies focus on WTE fly ash treatment rather than BA. This emphasis is influenced by the hazardous characteristic of fly ash which renders its treatment to be more pressing hazardous waste treatment and disposal is more stringent. The main purpose of fly ash washing is to remove what makes the ash a hazardous waste, typically a concentration of heavy metals over a prescribed threshold. Especially in European countries where the landfill space is limited, there is a crucial need of treating fly ash and disposing it in a nan-hazardous landfill. Washing has been applied as either a single treatment alone or combined with other techniques to treat fly ash. Chimenos et al. (2005) used washing as a single treatment before final disposal. However, other studies found that a single washing of fly ash in a washing plant in Sweden that treats 4,000 tons of fly ash per year and found that Cr and Mo leaching to slightly exceed the non-hazardous waste landfill limits. Thus, washing is usually performed as a pre-treatment before other

techniques. Counter-current washing (a process that involves two washing steps and one rinse step) has been used to produce a non-hazardous product from fly ash.

Common techniques including sieving, melting, S/S, ball-milling, and bioleaching has been combined with ash washing treatment for fly ash detoxication. Sieving is aimed to separate ash into different size fractions, since heavy metal distribution is dependent on size fractions, a size-based treatment is promoted. Deboom et al. (2015) combined sieving and washing to treat fly ash and successfully made it acceptable for a non-hazardous waste landfill. A melting process is a way to treat ash residue and recycle the slags where metals are concentrated on. However, emitting of volatile compounds and heavy metals during melting process is a serious concern. Chiang et al. performed washing before melting process and found that such water-extracted ash decreased the formation of metallic chlorides during its subsequent thermal treatment, and thus reduced the corrosive and leaching product problems. S/S has suffered from limited ash content and potential leaching product problems. Mangialardi et al. (1999) found that a simple washing process can increase incorporation of fly ash in cementitious matrices up to 90% and decrease the leaching concern. Ball milling process is an effective mechanochemical method to remove dioxins in fly ash (Sheng et al., 2012). Compared to thermal treatment, it's a simple route that requires no heating process and off-gas treatment. However, using ball-milling only is not efficient enough to treat fly ash since fly ash contains much Cl and heavy metals. Li et al. combined water washing and ball-milling and removed both chlorides and dioxin after the treatment. Bioleaching is to treat fly ash using microorganism. This is an approach with low cost and energy consumption. Traditional bioleaching process is usually long due to the effect of toxic compounds in FA to microorganisms. Wang et al. found that water extraction of WTE fly ash before bioleaching can speed up bioleaching period as well as increase metal extraction.

Meanwhile, only a few studies in the literature examine the effects of washing WTE BA. Lin et al. examined washing of WTE BA finer materials (<2mm) to remove organic pollutants and found that washing treatment removed most of 149 organic compounds down to 40. Alam et al. (2017) performed two stage-washing (washed twice) on WTE BA finer material (<4mm) to remove soluble contaminants as well as fines (\leq 125µm) that are adsorbed on the surface of ash. The two-stage washing process successfully brought the leaching of heavy metals as well as other contaminants under Dutch hazardous material legal limit. However, knowledge gaps still remain regarding improving WTE BA characteristics through washing treatment, especially for coarse BA material which makes up to about 80% of the BA (Lynn et al., 2016).

2.3.5 Results from Washing Studies

The removal of two types of pollutants during washing treatment are frequently examined in the literature: chlorides and heavy metals. Some researchers also studied the removal of organic pollutants (Lin et al., 2011; Li et al., 2017). Chemical species containing chlorides can be divided into inorganic and organic chlorides. Organic chlorides mainly result from polyvinyl chloride (PVC) in plastics and inorganic chlorides mostly exist as sodium chloride (NaCl) derived from food scrap (Chiang et al., 2010). Chloride contents were found to be as high as 156,000 mg/kg and 11,000 mg/kg in WTE BA and FA, respectively (Toshihisa et al., 1998). Inorganic chlorides are mostly represented by alkali chlorides like NaCl, KCl and CaCl₂. Alkali chlorides as well as amphoteric metals such as Pb and Zn are water extractible (Chiang et al. 2010). This means a washing process can remove most of chlorides, soluble salts and some amphoteric heavy metals.

Several key parameters in previous washing studies have been identified for developing an optimized washing process. Table 2-8 describes the washing parameters that have been explored in previous studies along with their major findings. The most frequently examined parameters are L/S and contact time (Chimenos et al., 2004; Yang et al., 2017; Zhu et al., 2010; Alam et al., 2017). The L/S ratio influences the washing results by influencing compound dissolution, due to the solubility of pollutants examined. Since washing remove the contaminants by transferring pollutants into the wash water, the wash water produced might has to be treated before discharge. In order to decrease the cost for treating wash water as well as saving water consumption, the optimum liquid to solid ratio (L/S) that minimize the amount of water used but also remove most of pollutants needed were explored in the literature. Generally, higher L/S will remove more compounds, but the influence was found to be generally notable only for L/S ratio lower than 10 L/kg in case of high soluble compounds such as chlorides and soluble salts. The dissolution of chlorides remain relatively constant for L/S above 10 (DeBoom et al., 2015; Wang et al., Jiang et al., 2009). Based on economic considerations, L/S of 3 was found to be sufficient for chlorides removal (Colangelo et al., 2012; Yang et al., 2017).

The contact time is also an important washing parameter that could influence the compound dissolutions. Longer washing time might reduce the dissolution and reabsorption of species (contaminates) in the wash water might occur (Deboom et al., 2015). Due to this and more practical reasons (i.e., efficiencies and costs) finding an optimal, shorter washing time that can maximize the removal efficiency of interested contaminants has also been explored. For soluble compounds like chloride and soluble salts, a few minutes washing time might be enough (Deboom et al., 2015; Colangelo et al., 2012; Yang et al., 2007). In a single washing process, L/S was found to be more important in removing chlorides than contact time or mixing speed. Some studies also applied multiple washing process and found washing cycles was more important than other washing parameters (Zhu et al, 2009; Zhu et al, 2010).

Many studies found that washing increased total concentration of heavy metals (Bien et al, 2007; Wang et al, 2015; Nordmark et al, 2018) due to the dissolution of a large amount of soluble compounds and the resulting mass loss. However, since there is no source that adds heavy metals into ash during the washing process, the total content of heavy metals will either remain the same or decrease normalizing mass loss. The removal efficiency for heavy metals depend on type of heavy metals. Yang et al. classified heavy metals into four groups based on the removal efficiency during washing. The first type of metals are those that have higher removal efficiency with higher contact time, indicating those metals have extracting behavior dominated by reaction kinetics, including Co, Cr, Fe, Ni, V and Cu. The second type of metals have lower removal efficiency with higher contact time, suggesting that carbonation of those metals have a significant influence on their leaching behavior;

those metals include Pb and Zn. The extraction mechanisms of the third type of metals are controlled by both reaction kinetics and the carbonation effect, represented by Ba and Mn. The extracting behavior of Cd and As can be grouped into the fourth type where the extraction efficiency kept constant with increasing contact time.

Though some WTE ash washing studies have been conducted, most studies are on WTE fly ash washing, and as a result the effects of washing on WTE BA is not well understood. With the recycling of WTE BA gaining more and more attention, laboratory-scale studies exploring optimum washing conditions to improve BA beneficial reuse opportunities are needed. These studies also primarily focus on the environmental/chemical properties of ash and do not explain the effects of washing on their physical properties (e.g., particle size, absorption, density). For washing, the characteristics of wastewater and filter cake produced during washing treatment are important to understand. A highly contaminated wastewater and filter cake residue can pose a risk to human health and the environment if not managed properly and also incur high costs in treatment and disposal. These considerations have not been discussed much in the literature. These topics are explored in further detail in the following section.

Table 2-8. Washing studies in the literature.

Title	Ash Type	L/S Ratios Tested	Contact Time Tested	Elements/Compounds Reported	Major Results
A two stage treatment for MSWI bottom ash to remove agglomerated fine particles and leachable contaminants	WTE BA	1, 2, 3, 5, 10	3min, 1h, 72h	Cŀ, SO4²-, Ĉu, Cr, Mo, Sb	Generally, the longer the contact time, the higher removal efficiency; the removal efficiency of contaminants will first increase with increasing of L/S, but will decrease with further increasing of L/S.
Effect of water washing on removing organic residue in bottom ashes of municipal solid waste incinerators	WTE BA	2	15min	149 organics	Washing removed 149 organic compounds down to 40 in MSWI BA
Combining sieving and washing, a way to treat MSWI boiler fly ash	WTE FA	7	n/a	Al, Ba, Bi, Br, Ca, Cl, Cr, Cu, F, Fe, K, Mg, Mn	The coarser material fractions leaching concentrations acceptable to landfill for non- hazardous materials
Water washing effects on metals emission reduction during MSWI FA melting process	WTE FA	10, 20, 50, 100	2h	Cŀ, SO4², Pb, Cd, Cu, Cr, Zn, Al, Na, K, Ca, Mg	Pb and Ca got maximum extraction efficiency of above 70% at L/S of 100. Removal of Na, K, Cl, and SO42- exceeded 94% of all tested L/S
Degradation characteristics of dioxin in the fly ash by washing and ball-milling treatment	WTE FA	4	1h	Cl, K, Na, Br, dioxin	Soluble salts like halogen- ide were sharply decreased while dioxin which are insoluble compound remain mainly the same
The effect of washing on cement-based stabilization of MSWI fly ash	WTE FA	10, 20, 30, 40, 50	30min, 1h, 2h, 3h	Cu, Zn, Cd, Cr, Pb, Cl	When washed at fixed contact time of 2h, the Cl removal efficiency changed from 66.21% to 77.41% with the higher L/S, the higher the removal efficiency. When washed at fixed washing time of 2hr, Cl removal efficiency for different L/S ratios differed from 64.80%-75.33% with the maximum extraction at L/S of 10
Effect of water washing on the co- removal of chlorine and heavy metals in air pollution control residue from MSW incineration	WTE FA	3, 10, 50	2min, 5min, 10min, 15min, 30min, 1h, 2h, 4h, 16h	Cl ⁻ , SO4 ²⁻ , Co, Cr, Fe, Ni, V, Ba, Cu, Mn, Pb, Zn, Cd, As,	L/S=3, contact time=5min was determined to be optimum washing condition; Bubbling of CO2 in the effluent is very efficient at removing Cu, Pb and Zn with the optimum bubbling time of 10-20s
Comparison of two types of municipal solid waste incinerator fly ash with different alkaline reagents in washing experiment	WTE FA	2, 3, 4, 5, 6, 8, 10	5min, 10min, 30min	CI	The frequency of washing was found to be more important than other washing parameters including L/S, mixing speed and contact time for chloride removal.
Heavy metal behavior in "Washing- Calcination-Changing with bottom ash" system for recycling of four types of fly ash	WTE FA	3	5min, 10min	Pb, Cd, Hg, Cr, Cu, Ni,	Except Cr, most of heavy metals significantly decreased during washing & calcination treatment
Can washing-pretreatment eliminate the health risk of MSWI fly ash reuse?	WTE FA	8	5h	Cr, Cd, Ni	Water washing increased the content of Zn, Pb, Cu while decreased the content of Cr and the effect of water washing remain uncertain.

Table 2-8. Continued (2).

Optimizing the APC residue washing process to minimize the release of chloride and heavy metals	WTE FA	1, 2, 4, 6, 8, 10	30min, 1h to 24h	Cl∙, SO₄²-, As, Cd, Cu, Cr, Ni, Pb, Zn, Ca	The counter-current washing process with the addition of small amounts of MgSO4 is an economically feasible treatment resulting in a non-special residue
Thermal treatment of the fly ash from municipal solid waste incinerator with rotary kiln	WTE FA	20	3h	Cl-, Cr, Cd, Cu, Pb	Water washing decreased the sintering time and temperature and time, decreased the chloride and heavy metal concentration which make the sintering product a non- hazardous waste.
Treatment process for MSW combustion fly ash laboratory and pilot plant experiments	WTE FA	0.3 to 0.7	5min, 10min, 15min	Na, K, Cl	Heavy metal concentration increased after washing; About 90% of the chloride content was removed from the ash, whereas the contents of Na, K, Ca, Cd, Pb and a number of other minor elements were removed by 10– 30%
Effects of water washing pre- treatment on bioleaching of heavy metals from MSWI fly ash	WTE FA	2	1h	Cr, Ca, K, Na, Cd, Fe, Mn, Pb	Water washing decreased the lag phase and the bioleaching period and also increased metal extraction
Soluble salts removal from MSWI fly ash and its stabilization for safer disposal and recovery as a road basement material	WTE FA	2, 3, 5, 10	30min	As, Ba, Cd, Cr, Ni, Pb, Cu, Zn, Cl ⁻ , SO₄ ²⁻	L/S Of 3 was efficient to remove above 80% of extractable chlorides and sulfate. The washing treatment was efficient to allow 2 samples to be disposed in a non- hazardous waste landfill
Effect of water-extraction on characteristics of melting and solidification of fly ash from municipal solid waste incinerator	WTE FA	2, 5, 10	2h	Ca, Si, Al, Na, K, Fe, Pb, Cr, Cu, Zn, Cd, S, Cl	L/S of 10 has the highest removal efficiency of elements tested.
3.0 Examining the Combined Use of GP and WTE Ash in PCC

3.1 Introduction

As previously mentioned, one of the major concerns for WTE ash use as an aggregate in PCC is expansion due the alkali-silica reaction (ASR). Section 2 details test methods for assessing the durability of concrete mixtures in regard to ASR potential and mitigation; ASTM C1293 effectively quantifies potential for deleterious expansion in different combinations of aggregate and cementitious material. To test the potential for deleterious ASR in WTE ash-amended mixes and assess whether alternative cementitious materials have the potential to mitigate alkali-silica expansion ASTM C1293 was performed using WTE ash from multiple different facilities.

3.2 Methods and Materials

BA samples were collected from three separate WTE facilities, samples were collected from large stockpiles of several hundred tons of material that had been staged over several weeks and thoroughly mixed via large equipment. Sampling protocols involved random grab samples from a pad of material. These samples were collected in 19 L HDPE buckets and further homogenized on a large mixing area to create a homogenous composite sample.

Samples were then sieved according to the requirements of ASTM C1293, whereby samples retained on the 12.5 mm, 9.5 mm, and 4.75 mm sieves were collected. These mass fractions were combined in equal parts by mass in mix designs specified by ASTM C1293. These mix designs incorporated traditional limestone (control) and 30 % by mass replacements of WTE ash as coarse aggregate from the three facilities mixed with normal Portland cement, as well as Portland cement with 20% by mass replacement of glass pozzolan or coal fly ash in attempts to mitigate alkali-silica reactivity. Mixes were cast in ASTM C1293 compliant molds. Length change was measured for a period of 24 (no supplementary cementitious material addition) to 36 (with supplementary cementitious material) months and compared to a deleterious threshold for alkali-silica reactivity established by the test method.

3.3 Results and Discussion

Alkali-silica expansion to date is displayed in Figures 3-1-3-3. Figure 3-1 displays the results of ASTM C1293 mixes containing no mitigation measures, the Facility B mix contains fewer measurements due to technical difficulties experienced late into the experimentation. The control (limestone) mix expectedly showed little to no expansion at all measurement ages, however results indicate that WTE addition results in expansion well beyond the deleterious thresholds at as early as one year of measurement. Measurements at approximately 500 days indicated expansions over three times that of the deleterious limit in two of three facilities. Results indicate that WTE BA is likely an alkali-

silica reactive aggregate and would result in deleterious expansion in concrete structures if there were no pre-processing controls or mitigation measures used.

Attempts to other waste materials (glass pozzolan, coal fly ash) to mitigate alkali-silica expansion were successful to a certain extent. Expansion above the deleterious limit in WTE ash-amended mixes containing glass pozzolan was delayed in Facility A and Facility B until over 500 days into the experiment. GP seems to have a considerable mitigation effect for Facility C ash, whereby expansion was maintained to slightly above the control (well below the deleterious limit) for the duration of the experiment to date. Though measurements for the F ash mixes are younger (fewer measurements to date), results are less promising for F ash use a mitigation measure. Samples are either trending towards (Facility A and C) or have already exceeded (Facility B) the deleterious limit at as early as 1 year of age. Though certainly F ash addition seems to have a positive impact on the durability of the mix, a 20% addition seems be less effective than an equivalent addition of GP. Measurements of all experiments will be taken up to 24 months of age.



Figure 3-1. ASTM 1293 expansion for normal Portland cement mixes (no GP or F ash) containing 30% by mass of WTE ash as a coarse aggregate. Data provided is the average of 3 replicates for each facility. Facility B has fewer measurements due to technical difficulties encountered during the measurement period.



Figure 3-2. ASTM C1293 expansion for normal Portland cement mixes also containing 20% mass replacement of cementitious materials with GP, containing 30% by mass of WTE ash as a coarse aggregate. Data provided is the average of 3 replicates for each facility. The deleterious limit is provided as a dashed line at 0.04% length change.



Figure 3-3. ASTM C1293 expansion for normal Portland cement mixes also containing 20% mass replacement of cementitious materials with F ash, containing 30% by mass of WTE ash as a coarse aggregate. Data provided is the average of 3 replicates for each facility. The deleterious limit is provided as a dashed line at 0.04% length change.

3.4 Conclusions

A common issue encountered with reactive aggregates, especially those with high content of reactive amorphous silica (glass) and alkali content, such as WTE ash, is expansion due to the ASR. We evaluated the potential for WTE ash to cause ASR, and further evaluated the potential for alternative cementitious materials (GP and F ash) to mitigate any observed expansions. Our results show that yes, WTE ash causes ASR as expected, but this expansion may be mitigated to a certain extent through the use of other waste materials in concrete such as GP and F ash. GP seems to be more effective at mitigating expansion than F ash, however both alternative cementitious materials at a 20% replacement did not adequately mitigate expansion below the deleterious limit; these materials only succeeded in delaying the time to deleterious expansion, as well as the magnitude of that expansion beyond the deleterious limit.

Here we tested only one combination of WTE and alternative cementitious material (30% by mass of aggregate and 20% by mass of cementitious material) and thus our results should not be used to dismiss WTE ash use in concrete or GP and F ash use as a mitigatory procedure for ASR. Real world mixes could certainly be fine-tuned so as to incorporate these waste materials and effectively mitigate ASR. As such, all three waste materials tested here have promise for use in this infrastructural application, and an optimization of the mix to allow for the safe use of these materials should be explored. These mixes were also prepared using ash that had not been subjected to any pre-processing techniques such as washing or advanced metals removal. It is expected that washing, which is known to remove surface available "wash off controlled" elements such as alkalis (necessary to have ASR), would reduce the alkali-silica reactivity of a portland cement system containing WTE BA.

4.0 Exploring Washing Pretreatment for WTE Ash

4.1 Introduction

Washing pretreatment was applied to two WTE ash samples from two facilities. The objective of the washing treatment is to find the optimum washing parameters for fines removal, heavy metals removal, chlorides and sulfate removal. Removal efficiency of each category under different washing L/S and contact time were examined for the determination of optimum washing process. This section includes experimental data collected by UF researchers for washing treatment of WTE ash. Specifically, this section discusses materials being washed, washing method and washing results for different types of pollutants along with how the process changes the physical properties of the ash.

4.2 WTE Ash Experimental Groups

WTE ash for this experiment came from two facilities. Facilities A and D are mass burn WTE facilities in west central Florida. The ash from Facility A is BA that has been aged on the working face of a landfill for a minimum of eight weeks and was then processed to produce a coarse aggregate (3/4"-No.4) and will be referred to as Sample A. The ash from Facility D was combined ash that was aged on the working face of a landfill for a minimum of four weeks and was then processed to produce a coarse aggregate (3/4"-No.4) for use in PCC and HMA which is referred to herein as Sample D. Representative ash-derived aggregates were sampled in accordance with ASTM D75 (2014) and transported to the testing laboratory in sealed, 5-gallon HDPE buckets to limit material contamination and limit interactions with the atmosphere.

4.3 Methods for Ash Washing

The experimental approach for ash washing is divided into two parts: small-scale washing and large-scale washing. Small-scale washing consists of experimenting with various liquid to solid ratios (1-20) and contact times (1 minute to 2 hours) to identify an optimum set of washing conditions as discussed in Section 4.3.1. Once this optimum was identified, a large-scale washing system was implemented to wash larger quantities of ash (approximately 25 kg at a time) for use in PCC and HMA along with testing properties of the wash water generated along with filter cake material.

4.3.1 Small-Scale Washing

An overview of washing treatment for the ash is provided in Figure 4-1. The water washing treatment was performed in a 1 L HPDE bottle on an end-over-end rotator at a speed of 30 rpm under different L/S and contact time. L/S of 1, 3, 5, 10, 20 and contact times of 1 min, 3 min, 5 min, 10 min, 30 min, 1 hour and 2 hours were explored. The volume of wash water was fixed at 500 ml for all washing scenarios while the weight of ash was changed to achieve different L/S ratios. After rotating for a specific time, the washing mixtures were

poured through a No. 4 (4.76 mm) sieve to further remove fines (< No. 4, 4.76 mm) released from the ash sample and wash water from the coarse ash material. Wash water and fines were then separated by a vacuum filtration through a 0.45μ m membrane. Fines and washed coarse were dried in an oven at 100°C overnight with the weight recorded afterward.

In order to determine the optimum washing condition for different reuse applications, the removal efficiency of fines, heavy metals, chlorides and sulfate during washing treatment was calculated by Equation 4-1.

$$\eta = \frac{C_{water} \times V_{water} + C_{fines} \times M_{fines}}{C_{unwashed ash} \times M_{unwashed ash}}$$
 (4-1)

Where,

- η is the removal efficiency of chlorides, sulfate or heavy metals in percent (%)
- C_{water} is the leached concentration of chlorides, sulfate or heavy metals concentration (mg/L) in the leachate
- V_{water} is the volume of wash water which is 0.5 L (500 mL) for all washing experiments
- C_{fines} is the total soluble chlorides, sulfate or total heavy metals concentration (mg/kg) in the fines
- M_{fines} represents mass of fines removed through washing (kg)
- Cunwashed ash is the total soluble chlorides, sulfate or total heavy metals concentration (mg/kg) in unwashed ash
- M_{unwashed} ash represents mass of ash being washed

The leached chloride and sulfate concentration in the wash water as well as the total soluble and sulfate concentration in fines and unwashed coarse were determined using an ion chromatography (Dionex 1100). The method to test the total soluble chlorides and sulfate is modified based on several methods specified in the literature (Shimaoka et al., 2011; Wu et al., 2016): 50ml of deionized water was added to 1.0 gram of fines; the mixture was then heated on a 60°C hot plate for 15 hours after being agitated for 30 min; the volume of leachate was maintained at 50 ml by adding deionized water through the heating process. The leachate produced from this procedure was filtered using a 0.45 μ m membrane for analysis.

The heavy metal concentration in wash water, fines, and unwashed coarse ash were determined by an inductively coupled plasma-atomic emission spectrometer (ICP-AES) in accordance with EPA Method 6010B after digestion process. The wash water was digested in accordance with EPA Method 3010a (for aqueous samples) whereas the fines and unwashed coarse material were digested using EPA Method 3050b (for solid samples).





4.3.2 Large-Scale Washing

After the small-scale washing treatment, the optimum washing procedure for different reuse applications were determined considering the specifically concerned contaminants for each reuse application. The large-scale washing procedure was then performed for 25 kg of ash samples from two facilities using the optimum washing parameters identified for each ash sample. Concentrations of heavy metals in the wash water were determined using the ICP to see whether wastewater needs treatment. To determine if the fines collected from washing may possibly be classified as a hazardous waste, the Toxicity Characteristic Leaching Procedure (TCLP, or EPA Method 1311) was performed. The Synthetic Precipitation Leaching Procedure (SPLP) was also conducted on unwashed ash as well as washed ash materials to determine the effect of washing on leaching characteristics of ash from a beneficial reuse standpoint.

4.3.3 TCLP

The TCLP was used as a regulatory leaching test to characterize hazardous waste. Two extraction fluids can be used for the test depending on the alkalinity of the waste: Fluid #1, which is a buffered acetic acid solution, and Fluid #2 which contains only acetic acid. The fines material (< 4.76 mm) collected after the large-scale washing process for two ash samples were tested using the TCLP. Since fines were already less than 4.76 mm, no size reduction was needed. The preliminary step was conducted to determine which of two extraction fluids would be used for the test. Then, 100 g samples of ash were added to a 2 L nonreactive high-density polyethylene (HDPE) vessel and filled with 2000 mL of extraction fluid determined previously. A lid was placed on the HDPE plastic vessels and each vessel was rotated at room temperature and 30 rpm for 18 hours. The resulting eluent was extracted by vacuum filtration through a 0.7 μ m glass microfiber filter, and a 50 mL aliquot of the leachate was preserved for elemental analysis using trace metal grade nitric acid. Samples were digested on an automated hot block digestion according to EPA Method 3010A to prepare for analysis by ICP-AES. ICP results containing concentrations of the 6 metals analyzed were compared to toxicity characteristic (TC) limits provided by the EPA.

4.3.4 SPLP

The SPLP (EPA Method 1312) was used as a screening method to identify problematic inorganic elements which may leach from ash materials and see how washing can affect the leaching characteristics. SPLP is a common batch leaching test performed on granular fractions of solid waste materials proposed for beneficial reuse. The SPLP calls for the use of an extraction solution comprised of nitric and sulfuric acids which simulate acidic rainfall contacting a solid waste material. Batch leaching tests such as the SPLP are used to identify contaminant release from size-reduced waste materials and are believed to produce a conservative estimate of trace element leaching from a large, semi-impermeable monolith (such as a pavement) that is not size-reduced.

In this analysis, washed ash as well as unwashed ash materials from two facilities were leached using the SPLP. Samples were first size reduced to pass a 9.5 mm sieve using a steel hammer and an industrial-grade blender. Then, 100 g samples from each unwashed and washed sample were placed in a 2 L nonreactive HDPE vessel and filled with 2000 mL of a dilute 60:40 nitric and sulfuric acid extraction fluid ($pH = 4.2 \pm 0.05$) to obtain the methodspecified L/S ratio of 20 L/kg-dry. The vessels were then placed on a rotary agitator at room temperature and rotated for 18 hours at a speed of 30 ± 2 rpm. The resulting eluent was extracted by vacuum filtration through a 0.7 µm glass microfiber filter, and a 50 mL aliquot of the leachate was preserved for elemental analysis using trace metal grade nitric acid. Samples were digested on an automated hot block digestion according to EPA Method 3010A to prepare for analysis by ICP-AES. Elemental concentrations were compared to USEPA regional screening levels (RSL). The RSLs were developed for use in contaminated Superfund site remediation projects and are unregulated; however, the RSLs are applicable to beneficial reuse assessments due to their reflection of the most up-to-date contaminant toxicity parameters gathered from a variety of national toxicity databases (e.g., IRIS, ATSDR). The RSLs for primary drinking water inorganic contaminants (e.g., As, Pb, Sb) are identical to the

maximum contaminant levels established by the EPA National Primary Drinking Water Standards (NPDWS) regulated under 40 CFR 141. The RSLs are based on a target cancer risk of 1:1,000,000 for carcinogenic species, and a target hazard quotient of 1.0 for non-carcinogens.

4.4 Results for Washing Pretreatment

4.4.1 Removal of Fines by Washing

After the washing treatment, adsorbed fine particles from coarse fraction were separated by sieving the washing mixture. Figure 4-2 presents the removal of fines under different L/S ratios and contact time for two samples. For both samples, the contact time showed a higher impact on fines removal than L/S ratio. The grams of fines removed per kilogram of ash increased significantly with increasing contact time which suggests that the more energy applied in washing process the more fines liberation occurs. For Sample A, the L/S ratio does not show a great impact on fines removal compared to washing time. There is a slight decrease of fines removal at larger L/S ratios, but generally for the same contact time the mass of fines removed was not highly dependent on L/S ratios. However, for Sample D the L/S ratio does not show an impact on fines removal compared to washing time. Lower L/S ratios (1-5) seem to remove more fines. This might because the larger the L/S ratios the more distant the particles are apart from each other which suggests that less abrasion will occur between particles leading to less fines being removed or produced. The difference between two samples might due to sample variation.



Figure 4-2. Removal of fines during washing process under different L/S and contact time. Each sample was tested in duplicate with the average reported. Error bars represents the minimum and maximum between duplicates.

4.4.2 Removal of Chlorides and Sulfate by Washing

Figure 4-3 shows the removal efficiencies of chlorides as a function of L/S ratios and contact time for both samples through dissolution into water versus separation of fines. In general, the removal efficiency for BA (Sample A) and combined ash (Sample D) showed a similar trend: longer contact time significantly increased chlorides extraction and the main mechanism of chlorides removal is via leaching into water since chlorides is highly soluble. On the other hand, L/S ratio does not appear to have a significant effect on chlorides removal. For Sample A, fines removal does not contribute to total removal efficiency whereas for Sample D, a significant amount of chlorides were removed through fines removal due to the abundance of fines in the combined ash than the BA.

Figure 4-4 presents the effect of washing duration and L/S ratios on sulfate removal. Similar to chlorides removal, contact time had a significant influence on sulfate removal. In the initial washing stage, sulfate may be partly immobilized by the formation of ettringite. However, dissolution of ettringite is favored with the increase of washing time and thus leads to an increase of sulfate leaching (Alam et al., 2017). Though L/S ratio does not affect the removal efficiency overall, increase of L/S ratio gradually increased the fraction of materials dissolved into liquid phase. For example, for Sample A, at the washing time of 2 hours, 5% of sulfate leached into water at L/S of 1 while 26% of sulfate leached into water at L/S of 20. This indicates the leaching of sulfate is a solubility-controlled process and higher L/S ratio increased the solubility of sulfate.



Sample A



Sample D

Figure 4-3. Chlorides removal efficiency under the influence of different L/S ratios and contact time for two samples. Removal contributed by dissolution into wash water as well as removal of fines were presented respectively.



Sample A



Sample D

Figure 4-4. Sulfate removal efficiency under the influence of different L/S ratios and contact time for two facilities. Removal contributed by dissolution into wash water as well as removal of fines were presented respectively.

4.4.3 Removal of Metals by Washing

Extracting behavior of metals (Sb, Mo, Na, K, Ca, Al, selected for this study) for the two ash samples are displayed in Figure 4-5 and Figure 4-6, respectively. For WTE BA, based on the mechanism of removal efficiency, Na and K was mainly removed by dissolving into water whereas Sb, Mo, Ca and Al were predominately removed through fines removal. However, for

WTE combined ash, all metals were mainly removed through fines removal, the difference might due to sample variation since the combined ash derived sample contains more fines than the BA derived sample. Furthermore, removal of all metals examined exhibited the same trend in accordance with fines, chlorides and sulfate removal: the washing time plays a dominant role in extraction than L/S ratios. However, the release of Sb, Mo, Ca and Al into the wash water were clearly enhanced at higher L/S, which suggests that the leaching behavior of those metals is controlled by their solubility. Figure 4-7 displays the pH value of solution as a function of L/S ratio and contact time. The pH ranges from 8 to 11 for Sample A and 7 to 9 for the Sample D. For the BA, pH values of leachate gradually increased with longer washing time in general, indicating the dissolution of more alkaline compounds over time. For the combined ash, pH values of leachate remained mostly the same but gradually increased with the increase of L/S ratios, indicating the increase of the L/S ratios could facilitate the dissolution of alkaline phase in the ash sample. Again, the difference between two ash samples might resulted from the sample variance or the longer aging period of ash from Sample D compared to Sample A.

The observation that fines removal from washing contributes more to removal efficiency than dissolution into the wash water itself suggests that most contaminants are concentrated on the fines that coat the surface of coarser ash particles. In order to test this hypothesis, ash was tumbled on its own for 2 hours to abrade off the fines coated on the surface of the coarser ash particles. These fines were collected and analyzed for total concentration of heavy metals, chlorides, and sulfate; Table 4-1 shows these results.



Figure 4-5. Metal removal efficiencies under the influence of different L/S ratios and contact time for Sample A by dissolution into wash water (red) and by fines removal (blue).



Figure 4-6. Metals removal efficiencies under the influence of different L/S ratios and contact time for Sample D by dissolution into wash water (red) and by fines removal (blue).



Figure 4-7. The pH values of leachate as a function of L/S ratios and washing time for the two samples.

Element	Fines (mg/kg) for Sample A	Unwashed coarse (mg/kg) for Sample A	Fines (mg/kg) for Sample D	Unwashed coarse (mg/kg) for Sample D
Soluble Chlorides	7,310	3,130	811	324
Soluble Sulfate	9,430	3,760	10,800	4,300
Al	24,500	15,600	43,900	23,000
As	14.4	0.600	53.8	21.3
Мо	14.2	11.1	9.59	7.38
Pb	919	553	1,460	500
Sb	33.5	20.5	131	49.7
Cd	20.8	7.75	74.8	31.3
Са	77,500	54,200	99,500	58,400
Na	7,400	8,350	3,790	6,270
К	2,400	1,740	1,370	1,430

Table 4-1. Concentrations of total soluble chlorides, sulfate and total metals in fines as well as unwashed coarse ash.

4.4.4 Optimization of Washing Procedure

Based on the results for all the contaminants covered previously, removal efficiency exhibited the same trend with contact time being the most important factor over L/S ratio. Soluble contaminants like chlorides were mainly removed via leaching while insoluble or slightly soluble contaminants like Al were mainly removed via fines removal. Since most contaminants were concentrated in fines that coat the ash surface their removal plays a dominant role in pollutants extraction, especially for insoluble or slightly soluble pollutants. Since L/S ratio is not the dominant factor in removal efficiency, lower L/S ratios should be used from an environmental and economic standpoint. An L/S ratio of 3 was observed to provide the most optimum washing condition and is a feasible and economical choice. As for contact time, longer washing times showed higher removal efficiencies probably due to higher fines removal; thus, a L/S ratio of 3 and washing time of 2 hours was selected as the optimum washing process for large-scale washing. However, in a fullscale industrial operation, the washing time might be minimized by designing the washing system to speed up the process of fines removal by adding abrasive media that can be easily separated from the bulk ash afterwards (e.g., sand, grit).

4.4.5 Results from Large-Scale Washing

Large-scale washing ash was performed by placing approximately 25 kg of ash from each sample in a large rotator and rotating it at a L/S of 3 for 2 hours. The wash mixture was

then poured over a No. 4 sieve (4.76 mm) to remove the liberated fines and wash water from the washed coarse ash-aggregate. To test whether the fine ash produced from this process may be considered a hazardous waste, the TCLP was performed and six of the eight RCRA elements were analyzed. Table 4-2 shows the results from the TCLP. The TCLP determined that the fines collected for both facilities are not hazardous waste; however, Cd concentration of fines for Sample D is approaching the toxicity characteristic (TC) limit and has the potential to become hazardous waste.

Table 4-2. Leached concentrations of select RCRA toxicity characteristic elements for fines collected after washing the ash samples. This test was conducted in triplicate with the average reported.

Element	Sample A	Sample D	Toxicity Characteristic Limit (mg/L)
Arsenic	0.00847	0.00853	5.0
Barium	0.385	0.393	100.0
Cadmium	0.402	0.989	1.0
Chromium	0.0457	0.0132	5.0
Lead	1.30	1.15	5.0
Selenium	0.0281	0.00677	1.0

The results of the SPLP tests performed on the two ash samples are shown below in Table 4-3. Leachate concentrations of each element are reported as an average of three triplicates. Elemental concentrations which exceed the corresponding EPA RSLs are highlighted for identification.

Table 4-3. Average SPLP leachate concentrations for the two samples before and after washing treatment. Elemental concentrations which exceed the corresponding EPA RSLs are highlighted for identification.

	Unwashed Sample A	Washed Sample A	Unwashed Sample D	Washed Sample D	EPA Regional Screening Level (Residential Tap Water)
Element	(mg/L)	(mg/L)	(mg/L)	(mg/L)	(mg/L)
рН	10.72	10.53	8.99	8.71	-
Al	20.2	12.3	3.93	2.05	20
As	0.00403	< 0.004	< 0.004	< 0.004	0.01
В	0.226	0.132	0.255	0.122	4.0
Ba	0.0846	0.0268	0.0383	0.0176	2.0
Be	< 0.001	< 0.001	< 0.001	< 0.001	0.004
Ca	82.2	42.3	67.1	28.2	-
Cd	< 0.001	< 0.001	< 0.001	< 0.001	0.005
Со	< 0.006	< 0.006	< 0.006	< 0.006	0.006
Cr (total)	0.00383	0.00283	0.00267	0.00187	0.1
Cu	0.0507	0.0109	0.0188	0.00830	1.3
Fe	0.0258	< 0.002	< 0.002	< 0.002	14
K	17.4	6.08	3.81	1.26	-
Mg	0.110	0.138	2.17	0.537	-
Mn	0.00327	0.00163	0.00500	< 0.001	0.43
Мо	0.0136	0.00623	0.0360	0.015	0.1
Na	46.0	16.8	11.4	7.54	-
Ni	< 0.001	< 0.001	0.00143	< 0.001	0.1
Pb	0.0347	0.00603	< 0.004	< 0.004	0.015
Sb	0.105	0.0311	0.0719	0.0269	0.006
Se	< 0.002	< 0.002	< 0.002	< 0.002	0.05
Sn	0.00300	0.00287	< 0.002	< 0.002	12
Sr	0.219	0.0949	0.142	0.0641	12
V	0.0107	0.0065	< 0.001	< 0.001	0.086
Zn	0.0276	0.0118	0.0127	0.00627	6

The results of the SPLP test demonstrate that washing decreased the leaching for all elements. Washing successfully brought the Al and Pb leached concentrations for Sample A from exceeding the corresponding drinking water threshold (EPA RSL to lower than the threshold. Though Sb still exceeds its RSL after washing for the two ash samples, the leached concentration was significantly lower after washing treatment.

The results of leached heavy metal concertation in wash water was also analyzed and presented in Table 4-4. Elemental concentrations which exceed the corresponding EPA RSL are highlighted for identification. From the results, we can conclude that Sb concentration

in the wash water exceeded the RSL for both samples and thus needed to be treated in a wastewater treatment plant before release.

	Wash Water Sample A	Wash Water Sample D	EPA Regional Screening Level (Residential Tap Water)
Element	(mg/L)	(mg/L)	(mg/L)
рН	10.91	7.16	-
Al	17.2	0.393	20
As	< 0.004	< 0.004	0.01
В	0.518	0.656	4.0
Ba	0.153	0.0401	2.0
Be	< 0.001	< 0.001	0.004
Ca	183	247	-
Cd	< 0.001	0.00160	0.005
Со	<0.006	<0.006	0.006
Cr (total)	0.0113	0.0056	0.1
Cu	0.123	0.0570	1.3
Fe	< 0.002	0.00690	14
K	111.7	15.26	-
Mg	0.0212	10.7	-
Mn	0.00150	0.0312	0.43
Мо	0.0650	0.0317	0.1
Na	260	21.0	-
Ni	0.00150	0.00240	0.1
Pb	0.0101	< 0.004	0.015
Sb	0.0315	0.121	0.006
Se	<0.002	<0.002	0.05
Sn	0.00220	0.00310	12
Sr	0.955	0.423	12
V	0.00920	< 0.001	0.086
Zn	0.0453	0.0523	6

Table 4-4. Leachate concentrations in wash water for two samples. Elemental concentrations which exceed the corresponding EPA RSL are highlighted for identification.

4.4.6 Mixture Properties of Washed Ash

Based on the results of washing from a chemical and environmental standpoint, Sample A and B were washed using a L/S of 3 at a contact time of 2 hours. Washing the WTE ash has affected important mixture properties for PCC and HMA such as gradation, specific gravity, absorption, and surface pH. Table 4-5 shows the differences for these below for each ash stream.

	_	% Passing	g (by Mass)		
Gradation	Sam	ple A	Sam	Sample D	
Gradation		After		After	
	Untreated	Washing	Untreated	Washing	
19 mm (3/4 in.)	100	100	100	100	
12.5 mm (1/2 in.)	59	59	89	94	
9.5 mm (3/8 in.)	26	29	60	64	
4.76 mm (No. 4)	4	2	17	3	
2.38 mm (No. 8)	3	1	14	1	
1.19 mm (No. 16)	3	1	12	1	
0.6 mm (No. 30)	2	1	11	1	
0.3 mm (No. 50)	2	1	10	1	
0.15 mm (No. 100)	2	1	8	1	
0.075 mm (No. 200)	1.3	0.2	7.1	0.2	
	Sam	ple A	Sam	ple D	
Other Aggregate Properties		After	After		
	Untreated	Washing	Untreated	Washing	
Specific Gravity	2.311	2.392	2.090	2.353	
Absorption (%)	5.0	2.8	9.3	3.2	
Surface pH	10.61	10.26	8.66	8.22	

Table 4-5. Mixture properties of WTE ash before and after washing.

As previously observed in Huang et al. (2006) and Chen et al. (2008), the gradation becomes much coarser after washing as the finer particulates are effectively suspended in the wash water, particularly the dust (< No. 200, 0.075 mm) material. As also observed in these studies, specific gravity increases while absorption decreases. These effects are attributed to the loss of finer aggregate and dust which consist of high surface areas and lower densities relative to the coarser materials that comprise WTE ash. As for surface pH, washing only reduced the pH by approximately 0.4 units. While this is slightly less of a reduction than that observed in Huang et al. and Chen et al., this may be attributed to the fact that prior to washing these ash streams were allowed to age in stockpiles on the working face of landfills for at least four weeks for Sample A and one year for Sample D.

4.4.7 Conclusion

Washing treatment was applied on BA as well as combined ash from two mass-burn facilities. Results showed that washing time plays the dominant role in removal of fines, chlorides, sulfate and heavy metals compared to L/S ratios. For both ash samples, the fines that attached on the surface appear to be more contaminated than the internal material

and thus the goal of washing is to remove the agglomerated fines. Based on an economic standpoint, a L/S ratio of 3 and washing time of 2 hours were selected as the optimum washing parameters for large-scale washing. After the washing treatment, the liberated fines collected for Sample D were approaching the TC limit for Cd and has the potential to become hazardous waste. Wash water collected exceeded the RSL for Sb for both samples, indicating further treatment of wash water is needed. The two washed ash samples showed lower leached concentrations for all heavy metals using the SPLP and successfully brought the Al and Pb to lower than the RSL, indicating that washing can effectively decrease the environmental leaching risk of ash when beneficially reused.

5.0 Examining the Effects of Washed Ash as an Aggregate in HMA

5.1 Introduction

WTE ash has been used in HMA applications for decades. Early works, such as by Walter (1976) demonstrated that WTE ash could be utilized at replacements up to 50% while still maintaining acceptable physical properties. More recent works show that WTE ash may be utilized at replacements of 40%, by mass, and below while still maintaining acceptable physical and environmental properties (e.g., leaching of heavy metals). However, one trend is clear: the addition of WTE ash almost always requires an increase in asphalt binder content (Ogurno et al., 2004; Huang et al., 2006; Chen et al., 2008) despite the fraction of WTE ash used (e.g., coarse vs. fine aggregate).

This increase in asphalt binder demand is typically attributed to the finer particles or dust attached to the surface of the coarser WTE ash particles. While WTE ash can be dry screened for the desired particle size distribution, often this method fails to fully to remove finer particulates. The finer particulates tend to have a higher surface area leading to a higher absorptivity resulting in more asphalt binder being absorbed by this dust rather than adhering aggregate particles to one another.

The issue with an increase in asphalt binder content is that while asphalt binder typically only amounts to approximately 5% of the pavement, by mass, this component comprises most of the cost of an asphalt pavement roadway. Hence, even with potential cost savings associated with using WTE ash as an aggregate replacement, this increase in asphalt binder demand quickly surpasses any immediate financial benefits derived from ash reuse. Not only do the presence of fines add to the amount of binder required but they also may interfere with HMA physical performance. As discussed in Section 2.2.3, the presence of fines on the ash particles may prevent sufficient adhesion by acting as a barrier between the aggregate surface and the asphalt binder which can lead to poor physical properties for the overall HMA mixture. In addition to finer particles and dust, the alkalis present in WTE ash (i.e., Na, K) may also interfere with adhesion in HMA mixtures and result in subpar performance.

One treatment strategy that has been previously used for ash reuse in HMA is a washing pretreatment step (Huang et al., 2006; Chen et al., 2008). In these works, WTE ash was washed to produce a higher quality aggregate for reuse. However, these studies did not directly compare ash-amended mixtures for unwashed and washed to examine the changes in asphalt binder demand and physical properties of HMA. The work in this section examines HMA mixtures made using both unwashed and washed coarse (3/4"-No. 4, 19.0-4.76 mm) ash-derived aggregate and compares the physical properties of the aggregates themselves along with their physical influence on the HMA mixtures they are utilized in.

5.2 Materials and Methods

For this study, a total of five mix designs were developed. All mixes were developed as fine, SP-12.5 mixtures meaning that a nominal maximum aggregate size (NMAS) of 19.0 mm (3/4") was selected. The design traffic level for these mixtures was C corresponding to 3 to 10 million equivalent single axial loads (ESALs). The aggregates used in this study include reclaimed asphalt pavement (RAP), a #7 (S1A) stone, a #89 (S1B) stone, W-10 screenings, and natural sand. The S1A stone, S1B stone, and screenings are derived from granite. The gradations and specific gravities of these aggregates are shown in Table 5-1 below.

	Stone	Crushed	#7 (S1A)	#89	W-10	Sand
		R.A.P.	Stone	(S1B)	Screenings	
_				Stone		
	3/4"	100	100	100	100	100
%)	1/2"	100	95	100	100	100
ze (1g)	3/8"	93	63	94	100	100
Siz Ssii	No. 4	71	15	39	96	100
eve Pa:	No. 8	55	4	10	75	100
Sie	No. 16	45	3	4	48	100
	No. 30	37	2	3	29	98
	No. 50	29	2	2	17	80
	No. 100	16	2	2	10	14
	No. 200	9.4	1.5	1.5	7.0	0.5
Gsb		2.631	2.627	2.625	2.580	2.633

Table 5-1. Gradation and specific gravity values for conventional aggregates used for the mixtures in this study.

The WTE ash-derived aggregate used in this study came from Ash Sample A and Ash Sample D as described in Section 4 previously. Ash-derived aggregate was also produced from these stockpiles that underwent a washing treatment as described in Section 4. This process was designed to remove as much finer material below 4.76mm as possible that may contribute to higher asphalt binder demand and impeded physical performance properties. The gradation and specific gravities from these treated ash-derived stockpiles were again determined using FM 1-T011 and ASTM C127 (2015), respectively, and these values are shown in Table 5-2 as well for review.

	Ash Type	Sample A	Sample A	Sample D	Sample D
		(Ash A)	(Ash A)	(Ash D)	(Ash D)
		Unwashed	Washed	Unwashed	Washed
	3/4"	100	100	100	100
ng)	1⁄2"	59	59	89	94
ssii	3/8"	26	29	60	64
Pa	No. 4	4	2	17	3
%	No. 8	3	1	14	1
, e (No. 16	3	1	12	1
Siz	No. 30	2	1	11	1
ve	No. 50	2	1	10	1
Sie	No. 100	2	1	8	1
	No. 200	1.3	0.2	7.1	0.2
Gsb		2.311	2.392	2.090	2.353

Table 5-2. Ash-Derived Aggregates Gradations and Specific Gravities.

The binder selected for these mixtures was influenced by reclaimed asphalt pavement (RAP) content, which was used in each asphalt mixture at a replacement of 16-30% (by mass). Because of this distinction, a PG 58-22 asphalt binder was used as per Section 334 in the FDOT Standard Specifications for Road and Bridge Construction (FDOT, 2019). This binder contained 0.5% of antistrip.

The objective of utilizing WTE ash in these mixtures was to simulate the aggregate stockpile "as-is" and developing mixture designs based on gradations of all the aggregate stockpiles. Mix A was an FDOT approved mixture which met all gradation and volumetric Superpave control points. To confirm, this mixture was replicated in the laboratory and used as the control mixture which job mix formulas (JMF), or the total composite gradation of the aggregate mixture, were based on. Mixtures including unwashed and washed ashderived aggregate was implemented at a 15% replacement, by mass, while limiting JMF variations so that mixture gradation does not act as a variable when assessing volumetrics and physical performance properties. Excel Solver was used to develop mixtures with consistent JMFs using 15% by mass of WTE ash from their respective facilities and pretreatments (i.e., washed) along with maintaining a RAP replacement of between 16-30% by mass to use the same binder grade for each mixture. Table 5-3 below shows the relative replacements of each component (by mass) for each mixture, while Figure 5-1 below visually presents the JMF for each individual mixture. The JMFs must follow within specific gradation control points as required under Section 334 Superpave in the FDOT Specifications for Road and Bridge Construction (FDOT, 2018c). Figure 5-1 displays these control sieve points as red dots. Table 5-4 presents these IMFs in tabular form.

Mixture	R.A.P.	#7 (S1A) Stone	#89 (S1B) Stone	Screenings	Sand	Ash A (unwashed)	Ash A (washed)	Ash D (unwashed)	Ash D (washed)
А	30	23	6	30	11	-	-	-	-
В	28	0	15	31	11	15	-	-	-
С	28	0	15	31	12	-	15	-	-
D	23	7	11	33	11	-	-	15	-
E	23	10	4	36	12	-	-	-	15

Table 5-3. The aggregate components by mass (%) of each individual mix.



Figure 5-1. JMFs for experimental mixtures. The red dots represent the Superpave gradation control points that must be met for an acceptable mixture. The 0.45 Power Line (in black) displays the gradation resulting in the maximum packing of the aggregates which must be avoided to prevent performance issues (e.g., achieving required volumetrics).

	Mixture	А	В	С	D	Е
	³ /4"	100	100	100	100	100
1g)	1/2"	99	94	94	98	99
ssiı	3/8"	89	86	86	89	89
Pas	No. 4	67	67	67	67	66
%	No. 8	52	52	52	52	53
je (No. 16	40	40	40	40	40
Siz	No. 30	31	31	31	31	31
ve	No. 50	23	23	23	23	23
Sie	No. 100	10	10	10	10	9
	No. 200	5.5	5.0	5.0	6.0	5.0
Gsb		2.614	2.561	2.576	2.516	2.567

Table 5-4. Tabulated JMFs for each mixture.

The most significant departure from the IMF for the ash-amended mixtures was for Mixes B and C at the $\frac{1}{2}$ " and $\frac{3}{8}$ " gradations. These discrepancies were allowed since this study aimed to reflect utilizing real aggregate stockpiles as-produced and these ratios were computed using Excel Solver to minimize the gradation differences between the experimental mixtures (B-E) and the overall particle size distribution of the Control mix (A). For each mixture, several trial asphalt binder contents were tested to achieve the optimum asphalt content for that specific mix. As mentioned previously in Section 2, the optimum asphalt content is the amount of asphalt required, by mass percentage of the total mix, to achieve a central range of volumetric properties including an air void content of approximately 4.0%. Table 5-5 describes the volumetric parameters examined in this study. These volumetric ranges change depending on Traffic Level (i.e., VFA) and the nominal aggregate size used in the asphalt mixture (i.e., VMA). Asphalt volumetrics include several properties: design asphalt binder content, bulk specific gravity, maximum specific gravity, air voids, voids in mineral aggregate (VMA), voids filled with asphalt (VFA), effective asphalt binder content, and dust-to-binder ratio. Table 5-4 provides a description and meaning for each of these properties.

These volumetric properties can be determined by knowing the maximum specific gravity (Gmm) and bulk specific gravity (Gmb) for a given asphalt mixture at a specific asphalt binder content. The Gmm was determined using FM 1-T209 (FDOT, 2015), and the Gmb was determined using FM 1-T166 (FDOT, 2016b). Determining Gmb involved creating compacted specimens as per ASTM D6925 using a gyratory compactor as shown in Figure 5-5.

Parameters	Description
Design Asphalt Binder	Amount of asphalt binder added (by mass percent) to an
(%)	aggregate mixture to achieve approximately 4.0% air voids at
	the specified number of gyrations by design (N _{des}).
Bulk Specific Gravity	The specific gravity of an asphalt mixture when compared to
(Gmb)	N _{des} at approximately 4.0% air voids.
Maximum Specific	The maximum specific gravity of an uncompacted asphalt
Gravity (Gmm)	mixture.
Air Voids (%)	The percentage of air voids (by volume) within the asphalt
	mixture.
VMA (%)	The volume between each aggregate particle in a compacted
	asphalt mixture.
VFA (%)	The percentage of VMA that is filled with asphalt binder.
Effective asphalt	The asphalt binder that contributes to aggregate-aggregate
binder content (%)	adhesion or binder film thickness (minus the asphalt binder
	that is absorbed by the aggregate).
Dust-to-Binder Ratio	The ratio of dust (material passing No. 200) to asphalt binder
	content in an asphalt mixture.

Table 5-5. Asphalt volumetric parameters and their significance.



Figure 5-2. A PINE Gyratory Compactor.

5.3 HMA Performance Testing Results

The volumetrics for each mixture are shown below in Table 5-6. As mentioned previously, the volumetrics of an HMA mixture reflect the anticipated performance qualities of an inservice asphalt pavement. The Control mixture (Mix A) meets all Superpave requirements for a Traffic Level C SP-12.5 mixture at a design asphalt content of 5.3%. Interestingly, Mix B, which was made with unwashed ash from Sample A, also met these Superpave volumetric guidelines while its washed counterpart (Mix C) is slightly below VMA (based on an average of two specimens, one at 14.0 and one at 13.8). These differences are attributed to the variability of the WTE ash, as the gradation differences between the two mixtures are minimal between unwashed and washed (as discussed in Section 4). For this particular ash stream and gradation, the washing did not drastically alter mixture properties aside from reducing the binder content by 0.1%. This is speculated to be the result of the original unwashed ash stream already having a limited amount of minus No. 4 material present already (4%, by mass).

Volumetric Parameters	Superpave Traffic Level C. SP-12.5	Mix A	Mix B	Mix C	Mix D	Mix E
Design						
Asphalt Binder (%)	n/a	5.3	5.4	5.3	5.8	5.7
Bulk Specific						
Gravity (Gmb)	n/a	2.357	2.330	2.342	2.320	2.336
Maximum						
Specific	nla	2156	2/10	2 1 2 6	2/17	2 1 2 0
Gravity	II/a	2.450	2.419	2.430	2.417	2.430
(Gmm)						
Air Voids (%)	3.0-5.0	4.0	3.8	3.9	4.0	3.9
VMA (%)	≥14.0	14.6	14.1	13.9	13.1	14.2
VFA (%)	65-75	73	73	71	69	73
Effective						
asphalt	r / a	1.0	4 5		4.0	4 5
binder	n/a	4.0	4.5	4.4	4.0	4.5
content (%)						
Dust-to-	0.6-1.2	12	11	11	14	11
Binder Ratio	0.0-1.2	1.4	1.1	1.1	1.7	1.1

Table 5-6. Superpave volumetric ranges with Control and 15% ash mixture results.

This same reduction in binder content was also noticed for Mixes D and E despite the washed ash having a significant reduction in minus No. 4 material (14%, by mass). One potential explanation for this phenomenon is that the drastic increase in VMA from Mix D (13.1%), which used unwashed ash, and Mix E, which used the washed ash product

(14.2%). This increase means that there is more space between each individual aggregate particles which would also result in more asphalt binder being required to fill in that space. Simultaneously, an increase in VFA and P_{ba} was observed between Mix D and E suggesting that not only does utilizing the washed ash result in a slightly lower asphalt binder demand but also a more optimal use of the existing asphalt binder in supporting particle-to-particle adhesion. This is significant as the binder is now used to contribute to mixture stability rather than becoming "lost" to absorption. The washing process also reduces the absorption of the ash-aggregate to approximately 3% which mirrors that of typical limestone also used for HMA aggregate. Discussions with FDOT engineers reveals that a mixture with similar gradation but using granite vs. limestone aggregate could anticipate an approximate 1% increase in design asphalt content based on the differences in absorption of limestone aggregate alone.

Another potential explanation for this phenomenon is related to the distribution of WTE ash by size fraction throughout the HMA specimens for each mix. Both mixtures use 15%, by mass, of WTE ash. However, the gradation of the washed ash of Sample D is significantly coarser than the gradation of unwashed ash of Sample D. Since the batching step for preparing HMA specimens is based on the total gradation of each aggregate, this process actually resulted in more coarse aggregate, especially between 3/8" (9.5 mm) to No. 4 (4.76 mm) existing in the Mix E. During compaction, WTE ash is prone to breakdown causing these coarser particles to breakdown and create more fines which contribute to a higher asphalt binder demand (Huang et al., 2006). This may potentially explain the still relatively high asphalt binder demand despite removing the WTE ash fines.

As for physical performance such as strength and durability, washing ash for use in HMA pavement may have some beneficial effects. As mentioned previously, moisture susceptibility testing examines an HMA mixture's ability to resist moisture damage. Moisture damage occurs when the interaction between the aggregate and water is preferred over the aggregate and binder. This causes a loss of cohesion between aggregate particles which can lead to durability issues. This damage is indicated by a percent loss of indirect tensile strength (ITS) for unconditioned mixture specimens ("dry") compared to "wet" specimens subjected to conditioning involving water filtration followed by a freeze-thaw cycle. ITS testing is shown below in Figure 5-3 with a closeup view of the tensile cracking that occurs for an asphalt specimen during the test.



Figure 5-3. The moisture susceptibility indirect tensile strength testing apparatus.

Moisture pills were created for each mixture as per FM 1-T283 (FDOT, 2018a) using the design asphalt binder content derived from volumetric testing (as previously shown in Figure 2-16). These pills were gyrated such that they had an air voids content of $7 \pm 0.5\%$ to reflect compacted HMA pavements. Six specimens for each mixture were selected for testing. From each set of six, three specimens were selected as the "dry", or unconditioned specimens, and the other three were selected as "wet", or conditioned specimens. Dry specimens were set aside in an environmental chamber for approximately two days at a constant temperature of 77 ± 1°F. Meanwhile, the wet specimens were vacuum saturated with water such that they were 70-75% saturated with water in their pore structure. These specimens were then completely sealed with plastic film so that no water could escape and placed in a freezer for at least 16 hours at a temperature of 0 ± 5°F. Following this, these samples were immediately unwrapped and placed in a hot water bath for 24 ± 1 hr at $140 \pm$ $2^{\circ}F$. Lastly, the wet specimens were placed in a room temperature water bath (77 ± $1^{\circ}F$) for 2 hours ± 10 minutes then immediately removed and tested for tensile strength. The dry specimens were also tested at this point for their indirect tensile strength (ITS). Figure 5-4 shows the ITS for each specimen. All specimens met the minimum ITS requirement of 100 psi (FDOT, 2018c). Overall, the dry ITS for each ash stream increased after washing which may be contributed to a greater bonding strength between aggregate and asphalt. Interestingly, the washed HW ash had a higher dry ITS than the control mixture. Increases in ITS with increasing ash replacement has been observed in the literature as well and may be attributed to the high angularity and surface texture of the ash particles (An et al., 2014; Luo et al., 2017).



Figure 5-4. Indirect tensile strength (ITS) for each mixture with (wet) and without (dry) conditioning as per FM 1-T283.

Tensile strength of the conditioned (S_{conditioned}) and unconditioned (S_{unconditioned}) specimens for each mix are then divided and multiplied by 100 to develop a tensile strength ratio (TSR). This TSR value is used to qualify if an HMA mixture may be more prone to moisture damage. FDOT specifies that an HMA mixture should have a TSR of at least 80% in addition to meeting minimum tensile strength of 100 psi (FDOT, 2018c). The TSR calculation is shown below in Equation 5-1.

Tensile Strength Ratio (%) =
$$\left(\frac{"Dry" Strength}{"Wet" Strength}\right) * 100$$
 (5-1)

Where:

- S_{conditioned}: the average tensile strength of specimens subjected to the freeze/thaw cycle, in psi
- Sunconditioned: the average tensile strength of specimens as kept in the environmental chamber at 77 ± 1°F, in psi

Table 5-7 shows the TSR results for the ash-amended and control mixtures. Mix A had the highest TSR of 91.01%, while mixtures utilizing the unwashed ash (Mixes B and D) had the lowest TSRs of 79.13 and 73.18%, respectively. While these are outside the FDOT requirement as per FM 1-T283, it is not outside the limit of TSRs typically achieved by HMA mixes without ash (Pavement Interactive). Anecdotal information from FDOT/UF staff have also observed mix designs with TSRs in the high 50's to 60's range. It is also important to consider that these specimens were subjected to a freeze-thaw cycle, which represents a worst-case scenario in which water does manage to freeze in the HMA pore structure. It is also important to note that further discussions with FDOT/UF staff have revealed that even mixtures that "passed' the TSR test on occasion exhibit poor moisture resistance under field conditions. For the mixtures utilizing washed WTE ash (Mixes C and E), moisture

susceptibility only increased by approximately 1% between unwashed (Mix B) and washed (Mix C), suggesting that washing did not significantly improve the moisture susceptibility for this stockpile. Visual examination of the broken specimens reveals that for specimens tested in the "wet" condition that stripping is a major form of failure due to the glass and ceramics that comprise the ash, while for "dry" condition specimens these ceramics and glass fracture causing failure. These visual examinations are shown in Figure 5-5 below.

Table 5-7. Moisture susceptibility results presented in tensile strength ratio (TSR %) as per FM 1-T283.

Moisture Susceptibility (TSR %)					
Mix A (Control)	91.01				
Mix B	79.13				
Mix C	80.37				
Mix D	73.18				
Mix E	86.13				
Typical Range of Mixtures (without ash)	70-90*				
FDOT Requirement (Section 334)	80%				

*Pavement Interactive.





Figure 5-5. Visual observation of stripping in conditioned HMA specimens.

As mentioned previously, the rutting susceptibility of an HMA mix measures how well it can handle deformation due to repeated unloading/loading from anticipated traffic. The Asphalt Pavement Analyzer (APA) was used to assessing rutting susceptibility for all mixtures in accordance with AASHTO TP 63 (2009). This test involves rolling a loaded wheel back and forth across a pressurized hose over an HMA specimen. This is done for 8,000 cycles using a 100 lb load and 100 psi hose pressure at a temperature of approximately 140°F. After 8,000 cycles, the deformation, or groove depth, made by the loaded wheel is measured. This measurement is then correlated to expected in-field performance. APA rutting results are shown below in Table 5-8 for all mixtures (A-E) along with values typically found for Florida HMA (which do not include WTE ash as an aggregate) including the prescribed FDOT limit of deformation of 4.5 mm. These results show that there is only a slight increase in rutting susceptibility for the ash mixtures compared to the Control mixture (Mix A), except for Mix B, which actually decreased by nearly 0.2 mm. However, Mix B's washed counterpart (Mix C) did exceed Mix A's rutting deformation by approximately 0.15 mm. When comparing the unwashed and washed ash mixtures, there was a significant increase in rutting susceptibility for Mix E (1.878 mm increase). This is believed to be attributed to the higher proportion of 3/8" to No. 4 sized aggregate replaced by ash-derived aggregate when compared to Mix D.
Table 5-8. APA rutting test results.

Mixture	Deformation (mm)		
Mix A	1.878		
Mix B	1.709		
Mix C	2.040		
Mix D	2.252		
Mix E	3.850		
Typical Florida HMA	1.5 - 2.3		
FDOT Limit	4.5		

Although washing did not have much effect on increasing (or decreasing) moisture susceptibility for Ash Sample A, it does point out that the presence of large particles of glass and ceramic play a key role in moisture susceptibility. Hence, a growing focus should be on limiting the amount of glass and ceramic materials present in the WTE ash if decided for use in aggregate applications. This may be done by increasing local recycling programs for glass or out-right having a ban on glass. Such programs have been shown to reduce the amount of glass present in WTE ash over time (del Valle-Zermeño et al., 2017).

5.5 Conclusions

Overall, washing WTE ash does reduce the asphalt binder demand needed to attain optimum volumetrics (i.e., 4.0% air voids), albeit slightly (~0.1%, by mass) when compared to the control mixture (Mix A). This is speculated to be due to the fact that absorptivity for the washed ash (~2.5-3.5%) are similar to that of limestones. It has been observed in the literature that mixtures using limestones instead of granites, with all other parameters such as gradation held constant, typically result in a binder increase of approximately 1%. However, VMA did significantly increase for the Mix E washed mixture from its unwashed counterpart (Mix D), which is attributed to the significantly higher specific gravity attained from the washed Sample D ash-aggregate. This may also partly explain why the DAC remains relatively high as while there is less absorptive finer material to absorb asphalt, there is also now more void space that asphalt must fill.

Washing WTE ash has also shown a slight decrease in the moisture susceptibility when tested using FM 1-T283. Mix C's TSR slightly increased after washing. This was especially apparent for Mix E as between its unwashed and washed counterpart an increase of 12.95% of observed. Under FDOT Section 334, both washed mixtures passed TSR, although this is marginally so for Mix C. The significant increase in TSR for Mix E is believed to be due to the removal of deleterious fines while being mostly comprised of material between 3/8" to No. 4 which, while still considered a coarse aggregate, still provides a small area for stripping to occur.

Ash Mixes B and D are slightly outside the acceptable range based on FM 1-T283 but are still within the typical range (70-90%) found in HMA mixes without ash (Pavement Interactive). Furthermore, unconditioned and conditioned specimens all met the minimum

100 psi requirement specified in Section 334 (FDOT, 2018c). It is also important to consider that FM 1-T283 involves a freeze-thaw cycle in the method which reflects absolute worst-case conditions where water in the pavement layer is able to freeze, which is particularly rare throughout most of Florida.

Rutting susceptibility was only slightly higher in Mix C when compared to its unwashed counterpart (Mix B). All mixtures were below the FDOT limit of 4.5 mm deformation and mostly within the range of typical Florida HMA (1.5 - 2.3 mm deformation). However, this was a significant increase in rutting susceptibility for Mix E. This is believed to be attributed to the higher proportion of 3/8" to No. 4 sized aggregate replaced by ashderived aggregate when compared to Mix D.

While it has been shown here that washing the ash can make HMA cheaper to construct compared to unwashed ash from an asphalt demand standpoint, the economic costs associated with washing ash are unknown. In the next section, the economic implications of glass pozzolan creation and washing WTE ash are explored.

6.0 Economic Feasibility of GP Recycling and WTE Ash Reuse

6.1 Introduction

Previous sections have shown that utilizing GP and processed (i.e., washed) WTE ash can be feasible from a physical performance and environmental standpoint. However, little has been discussed about the economic feasibility and infrastructure necessary for full-scale GP and WTE ash processing for market. Here, several factors must be considered:

- 1. What are the materials that these waste-derived products need to compete with and what's their cost? For this Report, market values in Florida were chosen for comparison.
- 2. What is the infrastructure (i.e., facility and operational costs)) necessary to produce these materials at full-scale?
- 3. What are effects of supply (i.e., how much waste is generated) and transportation (i.e., how far away is "too far")?

The following sections explore these factors with respect to GP and WTE ash.

6.2 Economic Feasibility of GP Recycling

Glass powder has been shown in previous literature (see Section 2) as an effective pozzolan in PCC. Furthermore, it has been shown that waste glass can also be utilized as a GP pozzolan (Jani et al., 2014; Jiang et al., 2014; Patel et al., 2019). In the municipal waste recycling industry, recovered glass containers from residential and commercial recycling operations typically produce items with low market value when compared to other materials (Dhir and Lymbachiya, 2001). This effect is so drastic that some municipalities have decided to stop collecting glass in their residential recycling programs altogether, or even pay for its disposal (Ng, 2015). An example of disposed waste glass is shown below in Figure 6-1.



Figure 6-1. Waste glass disposed at a landfill.

The most common recycling market for recovered glass from a material recovery facility (MRF) is cullet, or recycled crushed glass, to facilities that will recycle it in new glass containers. Other less common applications include use as raw materials in fiberglass manufacturing, paint fillers, sandblasting media, and aggregate materials (Chen et al., 2002). However, there is an opportunity for glass to be beneficially reused as a valuable component in PCC manufacturing. To produce glass powder (GP) from waste glass for use in PCC, several factors must be considered. The first consideration is processing waste glass to achieve the necessary particle size for GP. Glass for use as GP must be processed to an average particle size below 20 μ m, as that is when the glass becomes reactive for the pozzolanic reactions that occur within PCC (Ferraro et al., 2017). Another factor is contamination, as glass collected from recyclables typically has some non-glass constituents in it such as aluminum, steel, paper, and plastic products as shown in below in Figure 6-2.



Figure 6-2. Close-up view of waste glass and the contaminants present in it.

Another factor that must be considered is if this type of processing is feasible at large production scales. This subsection explores the considerations needed for processing waste glass and if this process may be economically feasible in comparison to the two most common systems of current cullet management: (1) traditional recycling into new containers and (2) disposal to a landfill.

6.2.1 System Boundaries and Definitions

An economic comparison of each system was conducted through evaluation of known costs associated with current management strategies (traditional recycling and landfilling) and application of a cost analysis estimated of potential GP pozzolan processing costs based on annual cullet throughputs in USD per metric tonne (t). Household recyclables in the USA are typically collected by recycling trucks through either a single-stream or dual-stream recycling system. In a single-stream system, all recyclable materials (e.g., paper, plastic, glass, metal) are commingled in a single container. In a dual-stream system, fiber materials such as newspaper and cardboard are separated from recycled containers including plastic, glass, and metal products to create two separate material streams. These two material

streams are transported to MRFs for further materials separation and preparation for enduser product manufacturers. Glass in this process is crushed and reduced to down to a material known as cullet which can be recovered from MRFs and recycled in the production of new glass containers (GPI, 2016a). However, recycling cullet can be challenge due to color contamination (e.g., mixing of green glass with opaque glass) and often this material is disposed of in landfills (Dhir and Lymbachiya, 2001). The aforementioned two most common management systems for recovered MRF cullet were compared to a third potential system where cullet is size-reduced on-site at an MRF to a powder form for reuse as a pozzolanic material in concrete production. For purposes related to this study, all three systems were evaluated starting from the point where recycled glass is recovered from an MRF processing line as shown in Figure 6-3.

The steps involved in this research's economic analysis are designated by a system boundary. Of the 230 million t of MSW generated in the United States in 2013, 80 million t were recovered through either recycling (75% of recovered waste) or composting (25% of recovered waste) (USEPA, 2014). Glass accounted for approximately 3.6%, or approximately 3 million t, of the waste recovered (for this exercise, glass was assumed to be recovered solely through recycling and not composting), implying that 3.0 million t of glass were sent to MRFs nationwide in 2013. In 2012, approximately 736 MSW MRFs were reported to be operating in the USA, with roughly 52% operating as dual-stream and 33% as single-stream (Gershman and Bratton, 2015; Kessler Consulting, 2009). The landfilling rates for single-stream and dual-stream recycled glass were 40% and 10%, respectively (CRI, 2009). Higher glass breakage and cross-contamination rates occur during singlestream processing, generating larger quantities of unmarketable lower-quality cullet which accounts for its significantly higher landfilling rate (GPI, 2014). Applying 2012 figures, it is estimated that approximately 160,000 tons of dual-stream and 400,000 t of single-stream recycled glass, or at least 560,000 total t of recycled glass, were landfilled in the United States in 2013, accounting for roughly 20% of the total recycled glass recovered from the MSW stream. This landfilling rate implies that approximately 2.5 million t of recycled glass containers were actually recovered for recycling into new glass containers. According to the Glass Packaging Institute, 80% of all recycled glass containers recovered by MRFs are used in the manufacturing of new glass containers (GPI, 2016b). Therefore, 2.0 million t of recycled glass containers were estimated as used in manufacturing new marketable glass containers nationally in 2013.

6.2.2 System #1: Traditional Container Recycling

Traditional glass recycling involves recovered MRF cullet being sent to a glass processing facility for further contamination removal (e.g., paper by air jets, metals by magnets or eddy current separators), crushing for size uniformity (maximum 19 mm), and color sorting (e.g., clear, green, and amber) by means of optical sorters (GPI, 2016b). Processed cullet is then sent to a glass container manufacturing facility where it is first combined with other raw materials such as sand, soda ash and limestone, then melted in a furnace and eventually molded into new glass containers ready for market distribution. Recycled glass can substitute up to 95% of raw materials used in the manufacturing of glass containers (GPI, 2016a).



Figure 6-3. Step-by-step process for each system of MRF cullet management: (1) Traditional container recycling, (2) Landfilling and (3) Use as a pozzolan in concrete manufacturing. Adapted from Tucker et al. (2018).

Recovered MRF cullet is typically contaminated and not color sorted making it of low quality, and therefore, of low value. Mixed color cullet consisting of commingled flint (clear), green, and amber (brown) glass is referred to as '3-mix.' Low-grade 3-mix cullet currently has a negative market value of approximately \$-20/tonne and considered unprofitable (Recycling Markets Limited, 2016). Alternatively, the reported current average commodity values for processed and separated flint, amber, and green cullet are \$33/tonne, \$30/tonne and \$14/tonne, respectively. MRFs must pay glass processing companies to process '3-mix' due to its negative market value. For example, twenty years ago, Strategic Materials Inc. (SMI) - the largest private glass processor in North America paid MRFs for their recovered recycled glass as incoming truckloads contained 98% glass with only 2% contamination (Ng, 2015). However, due to increasing contamination rates over the years as a result of many MRFs shifting from dual-stream to single-stream recycling, SMI's incoming material contamination rate is currently closer to 50% (Ng, 2015). This higher contamination rate has caused SMI to invest in more expensive sorting equipment to separate out non-glass materials, and in turn, has increased its residue disposal cost. To offset the costs of new equipment and increased disposal rates, SMI now charges MRFs \$11 to \$44 per tonne for their contaminated cullet (Ng. 2015).

6.2.3 System #2: Landfill Disposal

An MRF may dispose of its recovered glass as residue in a landfill if it is economically more favorable than participating in a traditional glass container recycling market. When an MRF landfills its cullet as residue, there is an associated transportation cost, additional labor and administration costs, and a possible landfill tipping fee in addition to the MRF's standard materials processing and equipment costs. In 2013, the average national MSW landfill tipping fee in the USA was approximately \$55 per tonne of material disposed (USEPA, 2014). An MRF typically incurs both a transportation cost and a landfilling fee when under different ownership than the landfill receiving its residue. Some MRFs send their lowquality cullet to operating landfills for use as a material to fill depressions, supplement landfill cover, and for drainage; this may require approval and permitting by the appropriate state regulatory agency. At closed landfill sites glass cullet may fulfill the reoccurring need for fill material as a result of waste degradation and settlement. Although beneficial use at landfills does involve transportation, the landfill operator may be able to use the glass to offset costs associated with standard fill materials such as sand and gravel (Dhir and Lymbachiya, 2001; Disfani et al., 2011). Natural material costs are highly location-specific; in 2015, the average price of sand and gravel in the USA was \$8 per tonne (USGS, 2016a). Depending on local conditions, the landfill operator may offer a reduced tipping fee when the glass is utilized for a beneficial purpose and may possibly waive tipping fees altogether.

6.2.4 System #3: GP Pozzolan

As discussed in Section 2, when glass is finely ground it possesses characteristics that allow it to behave as a pozzolanic material and, in turn, act as a SCM to partially replace portland cement in concrete production (Zheng, 2016). Because waste glass is rich in silica and amorphous in structure, it is chemically able to enter pozzolanic reactions when finely pulverized (Rajabipour et al., 2010; Shayan and Xu, 2004). Waste glass is required to be finely ground into a GP to suppress the alkali reactivity of coarser glass particles, further mitigating alkali-silica reaction (ASR) occurring between the silica in the glass and the alkali present in the pore solution of concrete (Rajabipour et al., 2010; Shayan and Xu, 2004). The standard specification for the use of coal fly ash and raw or calcined natural pozzolan in concrete has a material fineness (particle size) requirement of 34% maximum retained when wet-sieved on a 45- μ m (No.325 mesh) sieve (ASTM Method C618, 2015). For the purposes of this study, glass powder classified as a pozzolan is required to have a particle size of at most 45 μ m. However, multiple studies have found that the pozzolanic activity of GP increases with increasing particle fineness, further reducing concrete expansion (Idir et al., 2001; Shao et al., 2000; Shi et al., 2005; Ferraro et al., 2017). Similarly, Shayan and Xu (2004) found that GP sized to less than 10 μ m can replace 30% of cement in concrete production without causing any long-term detrimental effects, including ASR-induced expansion.

To produce GP with an average particle size of 10 µm from recovered MRF cullet, a suggested size-reduction process was selected after analysis of published scientific literature and discussions with MRF personnel and relevant equipment manufacturers: The following steps describe the process envisioned in the analysis presented here. First, recovered MRF cullet would be introduced to a drum feeder by rolling stock (e.g., front-end loader) and then fed onto an initial conveyor belt (Pressley et al., 2014). Hand sorters would manually remove any bulky non-glass materials from the conveyor belt for disposal (two hand sorters assumed in this analysis). The material would then pass under a selfcleaning cross belt magnet to remove ferrous materials (e.g., steel bottle caps). Remaining nonferrous materials (e.g., aluminum cans) would be removed by a high-frequency eddy current separator. After removal of metal contaminants, the material would be conveyed into a horizontal impact crusher. The crusher would break the glass into smaller pieces, with the more pliable and durable materials (e.g., paper, plastic bottle caps, straws and corks) flattened and remaining intact. The crusher would uniformly size reduce the glass to approximately 25 mm; a horizontal impact crusher was selected as they are reported to be more resistant to abrasive materials such as glass compared to jaw crushers (Cassar and Camilleri, 2012). Once crushed, all material would be fed into a trommel screen where any material less 25 mm would be screened out and further processed with some type of air classification equipment to separate out materials with lighter densities; bottle glass has a much higher density (2.4 g/cm^3) than paper (0.8 g/cm^3) , plastic straws (0.11 g/cm^3) , plastic bottle caps (0.96 g/cm³) and wooden corks (0.26 g/cm³). The remaining 25-mm minus glass material would then be processed in a ball mill to achieve a mean output powder size of 10 µm.

6.2.5 Economic Analysis

To determine if MRF investment in a GP pozzolan market is economically feasible, GP processing costs (in USD per tonne) were estimated based on the suggested cullet size-reduction process to 10-µm and various annual throughput capacities of recovered MRF cullet. It was assumed that hourly throughputs would range from 1 tonne to 45 tonne cullet. Based on standard MRF operations, it was assumed that equipment processing

would occur for 16 h per day, 6 days per week, for 47 weeks per year, or approximately 4,500 h per year. Therefore, the yearly throughputs of MRF cullet for GP processing ranged from 4,500 tonne (t) per year (tpy) to 202,500 tpy. Although a MRF typically processes at maximum 100,000 t of incoming material every year (with glass expected to account for only 20%), this range was selected to account for the possibility of a MRF collecting additional cullet from other MRFs located in surrounding areas (Cimpan et al., 2016; Pressley et al., 2014). Annual fixed capital, operating, maintenance and labor costs were calculated to determine overall GP processing costs (USD per tonne) for the varying annual throughput capacities (4,500 to 202,500 tpy).

Seventeen pricing quotes from ten USA-headquartered equipment manufacturing companies were acquired for six necessary pieces of GP processing equipment: magnet, eddy current separator, crusher, trommel screen, air classifier and ball mill (see Table 6-1). Investment costs for a conveyor system, drum feeder, rolling stock and vacuum (for air classification) were obtained from a previous 2012 study and adjusted to 2017 costs using a 6.6% cumulative inflation rate (Combs, 2012; USBLS, 2017). An equipment installation rate of 43% of the total purchased equipment cost was assumed (KLM, 2014). Equipment purchasing and installation costs for each hourly throughput capacity (1–45 t) were depreciated over an assumed 10-year lifespan at a 10% annual rate. Purchasing costs were averaged for machines of the same type with the same maximum throughput capacities (e.g., 45 tonne per hour (tph) magnet) and machines of lower throughput capacities were assumed to be placed in parallel to estimate equipment costs for larger throughputs (e.g., the 45 tph crusher equipment cost consists of two 23 tph crushers in parallel).

The average size of a theoretical GP processing building to be constructed as an addition to an on-site MRF was assumed to be 930 m² through estimation of building sizes of ten nationally located glass processing facilities using Google Earth software. The construction cost of a building of such size was estimated using published unit-cost warehouse construction cost data (RSMeans, 2015). A project size modifier cost multiplier of 1.1 was applied as well as a cumulative inflation rate of 3.3% to adjust to 2017 costs (RSMeans, 2015; USBLS, 2017). After inclusion of an architectural/engineering cost estimated at 15% of building construction cost, the total construction cost of a 930 m² processing warehouse add-on was estimated to be \$950,000. An annual building capital cost of \$32,000 was determined using straight-line depreciation over an assumed 30-year lifespan with a no salvage value. Land capital cost was assumed to be 6.0% of the initial total purchased equipment cost (KLM, 2014). Since equipment costs varied by throughput capacity, the initial purchasing cost of land ranged from \$140,000 to \$200,000. It is important to consider that land costs are very site/location specific and that this serves as a general estimate. The estimated average land size for ten glass processing facilities was about 20,000 m², or $\frac{7}{m^2}$ to $\frac{10}{m^2}$. The land capital was allocated over an assumed 30-year lifespan to determine an annual land capital cost. The total annual fixed capital for each throughput capacity was inclusive of equipment purchasing and installation, building construction, land costs as well as assumed property insurance and tax rates of 1% and 1.5% of total fixed capital, respectively (KLM, 2014).

Equipment operating costs were calculated based on the average national industrial price of electricity (\$0.0674/kWh) reported for March 2017 and each equipment piece's required power, with an assumed operation time of 4500 h per year (see Table 6-1) (USDOE, 2017). An average diesel fuel estimate of \$0.66 per liter was utilized to calculate the rolling stock's (e.g., front-end loader) annual operating cost (based on a 4,700 L per month usage) (AAA, 2017; Combs, 2012). Annual building operating costs were determined using latest known national annual electricity (71.0 kWh/m² building) and fuel (5.91 m³ natural gas/ m^2 building) consumption data for commercial warehouses (USEIA, 2016). Applying the assumed 930 m² building size as well as March 2017 reported national commercial electricity and fuel consumption costs at \$0.1048/kWh and \$7.67/28.3 m³, respectively, total annual electricity usage was estimated at \$7,000 and fuel usage at \$1,500 (USEIA, 2017a; USEIA, 2017b). Therefore, the total annual estimated building operating cost was estimated to be \$8,500. Building maintenance was estimated at \$15,000 per year based on a \$3/h rate with a 4500 hpy operating time. Annual equipment maintenance was estimated by applying a 7% rate to annual purchased equipment costs (dependent on throughput capacities) (KLM, 2014). It was assumed that alumina ball mill grinding media (\$4.4/kg) would be used at a 25% mill loading rate, with replacement once annually. Alumina ball media was chosen as high-density zirconium oxide ball media would be economically unfeasible due its high estimated retail value (\$100/kg) and the mass of media required (> 29,000 kg). While steel media has an equivalent cost to alumina media, its usage would risk metal contamination in the GP pozzolan. The required alumina media masses at 25% loading were provided with the ball mill manufacturer quotes and interpolated to determine media costs for the entire hourly throughput range of 1–45 tph.

Equipment	Maximum Throughput (TPH)	Power (kW)	Purchasing Cost (\$)	Operating Cost (\$PY)	Lifespan (Y)
Rolling stock ^a	24	-	375,000	37,000	10
Conveyor ^a	30	6	50,000	2000	10
Drum feeder ^a	30	15	160,000	5000	10
Vacuum ^a	11	5	160,000	2000	10
Hand sorters (2)	-	_	-	135,000	-
Magnet	45	2	25,000	700	10
Eddy current separator	7	7 ^b	10,000	2000	10
	10	7 ^b	105,000	2000	10
	12	7 ^b	110,000	2000	10
	14	7 ^b	120,000	2000	10
	35	15 ^b	250,000	5000	10
	41	22 ^b	350,000	7000	10
	45	30 ^b	450,000	9000	10
Crusher	9	15	50,000	5000	10
	23	30	90,000	10,000	10
	27	30	95,000	10,000	10
	32	30	105,000	10,000	10
	45	60	180,000	20,000	10
Trommel screen	27	11	100,000	3000	10
	45	19	200,000	6000	10
Air classifier	36	2	75,000	700	10
	45	2	90,000	700	10
Ball mill	18	400	1,200,000	120,000	10
	45	950	1.700.000	290,000	10

Table 6-1. Equipment capital and annual operating costs based on manufacturer data and a 4,500 h per year operation time. Adapted from Tucker et al. (2018).

^a Combs, 2012.

^b Power assumed.

This study accounted for two hand sorters (\$15/h), six equipment operators (\$15/h), and one rolling stock operator (\$15/h) working during each hour of annual operation (4,500 hpy), totaling \$600,000 per year in operating labor. Annual supervision costs were assumed to be 15% of annual operating labor costs, totaling \$90,000 per year (KLM, 2014). Annual plant overhead costs (including fringe benefits) were assumed to be 60% of yearly total operating labor, supervision and equipment maintenance costs. Lastly, an annual general administration cost was assumed to be 25% of annual operating labor costs, totaling \$150,000 per year. Table 6-2 indicates total annual costs based on the four cost categories: fixed capital, operating, maintenance and labor. The fixed capital costs for purchased equipment and installation were depreciated over a 10-year lifespan at a 10% rate, while the building costs were depreciated over a 30-year lifespan. Land purchase costs were allocated over a 30-year period. The total annual costs inclusive of fixed capital, operating, maintenance and labor costs were then divided by 1-45-tonne annual throughput capacity to determine GP processing costs (in USD per tonne). The total GP processing cost (\$pt) versus annual throughput (tpy) with assigned parameters of a depreciated 10-year lifespan (excluding building and land capital costs with 30-year lifespans) and a 4,500 hpy operating time is depicted in Figure 6-4. The resulting trend line is provided in Figure 6-4. As designated in Figure 6-4, portland cement (used in concrete manufacturing) retails (at the time of this analysis) for approximately \$120/tonne (USGS, 2016b). Class F fly ash, a known effective pozzolan with the ability to replace 15–30% of PC, retails for approximately \$40/tonne (Roessler et al., 2016; Blissett and Rowson, 2012; FDOT, 2016); however, that market value is expected to increase over the next few years due to decreasing availability of fly ash throughout the USA as the power market continues to shift towards use of cheaper natural gas instead of coal (AASHTO, 2016).

Concrete batch plant owners are unlikely to purchase recycled MRF GP unless it can compete with the retail values of Class F coal fly ash (\$40/tonne) or portland cement (\$120/tonne) (USGS, 2016a; FDOT, 2016a). Based on an assumed 4,500 hpy plant operating time and a 10-year payment plan for fixed capital of a standard GP processing system (inclusive of operation, maintenance, and labor costs), a yearly throughput of at least 50,000 tons of collected MRF cullet would be required before GP processing would cost less than the current retail value of Class F coal fly ash and become a profitable market (see Figure 6-4). If the retail value of fly ash increases to \$50/tonne, the required annual throughput decreases to 40,000 tons; 25,000 tpy for an increase to \$75/tonne FA; and 15,000 tpy for \$100/tonne FA. The throughput requirement is only 20,000 tpy of cullet when compared to the current market value of portland cement. It is estimated that the maximum annual throughput of glass material at an advanced MRF is approximately 20,000 ton (Cimpan et al., 2016; Pressley et al., 2014). Therefore, in order to compete with the current fly ash market value, additional cullet would have to be collected from MRFs located in the surrounding area to meet the annual throughput processing demand of 50,000 t of cullet. This outsourcing need coincidingly applies to MRFs of small and medium capacity; maximum <5,000 tpy glass and 5,000–15,000 tpy glass, respectively. The GP processing costs per tonne continually decrease as the annual throughput increases, further incentivizing the collection of additional MRF facility cullet for processing. Figure 6-5 shows how much glass was recycled in each Florida county in 2014 providing a

theoretical cullet supply capacity to support this potential demand. In regard to operational feasibility, it is important to note that GP processing is expected to produce significant amounts of dust, especially during the milling phase. This would likely result in the facility needing to acquire an air permit, the implementation of an air filtration system (e.g., installation of bag houses) and more frequent equipment cleaning to decrease risk to worker health. Social acceptance of new recycled materials which have different attributes may have barriers with respect to social acceptance. Faleschini et al. (2016) noted that social constraints resulting from noise and dust affected the beneficial use of recycled aggregates in the construction industry. The costs associated with dust management (including installation of an air filtration system) were not included in this study's economic analysis.

Categories	Annual Cost (\$PY)
Fixed Capital Purchased equipment + 43% Installation Building (930 m ²)	300,000 - 500,000 30,000
Land (20,000 m ²), 6% purchased equipment Property insurance, 1.0% total fixed capital Property taxes, 1.5% total fixed capital	5000 - 7000 4000 - 5000 5000 - 8000
Total Annual Fixed Capital Costs	345,000-550,000
<i>Operating</i> Equipment Building	175,000–350,000 9000
Total Annual Operating Costs	185,000-360,000
<i>Maintenance</i> Building, \$3/h (4500 hpy) Equipment, 7% purchased equipment Ball mill media (25% loading, 1 x annual replacement)	15,000 15,000–25,000 5000 – 275,000
Total Annual Maintenance Costs	35,000-315,000
Labor Operating Labor Supervision, 15% operating labor Plant Overhead, 60% operating labor + supervision + equipment maintenance General Administration, 25% operating labor	600,000 90,000 approx. 400,000 150,000
Total Annual Labor Costs	1,250,000
Total Annual Costs	1,815,000-2,475,000

Table 6-2. Annual costs associated with GP processing plant with an assumed depreciation of 10 years (excluding building and land fixed capital costs with 30-year depreciation/allocation). Adapted from Tucker et al. (2018).



Figure 6-4. Total cost to produce glass powder (GP) based on annual processing throughput capacity of MRF cullet (in t) and an assumed 10-year plant life (excluding building and land capital with 30-year lifespans). Current retail values of portland cement (PC) and Class F coal fly ash (FA) are represented. Adapted from Tucker et al. (2018).



Figure 6-5. Tons of glass recycled in 2014 by county in Florida. Adapted from Tucker et al. (2018).

6.3 Economic Feasibility of WTE Ash Recycling

6.3.1 Introduction

The beneficial reuse of WTE BA as an aggregate, in lieu of virgin aggregate, in construction material is currently an area of focus in research (Åberg et al., 2006; Hjelmar et al., 2007; Van Praagh et al., 2018; Clavier et al., 2019). Albeit the environmental implications of using WTE BA as an aggregate is important to consider, determining the economic feasibility of full-scale production is crucial as well. Currently, there is an absence of literature regarding the assessment of economic feasibility of using WTE BA in a full-scale processing plant as an aggregate in construction material. Nevertheless, the use of WTE BA as a replacement aggregate is currently being practiced in European countries.

The Confederation of European Waste-to-Energy Plants reports on several cases where WTE BA is being utilized throughout Europe. Legislation in Denmark has permitted the use of WTE BA aggregate for numerous years and as of 2012 has increased the scope of its utilization to high load roads; aiding in a recovery rate of approximately 99% of BA. In the Netherlands, all WTE facility operators signed the "Green Deal Bottom Ash" with the Dutch Government and currently use BA in road construction materials. The goals of this legislation were to recover 75% or more of non-ferrous metals from the BA and by 2020 to process BA of such high quality that 100% of it can be utilized for reuse (CEWEP, 2016). According to Tasneem (2014), France and Germany currently recycle WTE BA in construction materials at approximately 79% and 65%, respectively. In the United States (US), WTE BA has been extensively researched but currently the only active recycling operations are those that recover ferrous and non-ferrous materials (Tasneem, 2014). The principle causes for this contrast between the US and Europe is that the limited land availability in Europe causes the landfill fees and virgin aggregate prices to be higher than that of the US (CEWEP, 2017).

Given these differences, it is the goal of this report to assess the economic feasibility of implementing a full-scale WTE BA processing facility, for its use in road base as a virgin aggregate replacement, in the US. This is accomplished by comparing two different management strategies, screening of WTE BA only and screening of WTE BA with advanced metals recovery (AMR), with that of the current practice of landfilling. A typical claim against the notion that WTE BA, as a replacement aggregate, is not economically feasible is the additional costs associated with processing and transporting the material. While this is a valid argument, landfilling rates (EREF, 2018) and virgin aggregate prices (USGS, 2019) are steadily increasing; also, the revenue made from recovering ferrous and non-ferrous material may account for the additional costs.

6.3.2 Feasibility Analysis

To consider the economic feasibility of a full-scale WTE BA processing facility, for ashamended aggregate as a replacement for virgin aggregate, two systems were compared to the current management method of landfilling BA.

- 1. WTE BA processing without advanced metals recovery using conventional screening
- 2. WTE BA processing with screening and advanced metals recovery

Listed below are the starting assumptions considered in this economic analysis:

- 1. The production of WTE BA is in proximity to the processing plant and landfill thus rendering transportation costs to these facilities negligible
- 2. The addition of AMR machinery does not increase the costs associated with construction or result in additional land necessary for the processing facility
- 3. The cost per tonne saved from diverting WTE BA from being landfilled is approximately \$10/tonne. This is based upon the average capital costs of constructing a landfill (\$30,000,000), an average lifespan of 30 years and the average WTE BA generation for a WTE facility being 104,000 tonne per year (tpy)

There are two widely used combustion technologies implemented in WTE facilities that produce BA; mass burn facilities and refuse derived fuel systems (RDF). Mass burn facilities, the more common method in the US, combust sorted or unsorted MSW in a single combustion chamber under excess air conditions. RDFs utilize mechanical methods to shred MSW and remove non-combustibles to maximize the output. The amount of total ash (BA and fly ash) produced from the combustion of MSW is approximately 30% (by mass) of the original MSW. BA accounts for approximately 80-90% (by mass) of the total ash produced (USEPA, 2015a). In the US the BA is traditionally disposed of via landfill. For analysis purposes, all three methods were evaluated starting from the point when the BA arrived at the landfill or processing facility, refer to Figure 6-6.



Figure 6-6. System boundary is defined after the WTE BA has been transported to either the facility or the landfill.

Approximately 238 million tonne of MSW was generated in the US in 2015; of which, 30 million tonne was combusted in a WTE facility to produce energy (USEPA, 2015b). In 2018, there were 75 WTE facilities in 21 different States; the majority being located in Florida, New York and Minnesota with 11, 10 and 8 facilities, respectively (Michaels and Krishnan, 2018). Based on the daily MSW input reported by Michaels and Krishnan (2018), the national annual average BA ash production of a WTE facility is theoretically 104,000 tpy.

The traditional management of WTE BA in the US is disposal via landfilling. The economic factors related to this management method are the costs associated with tipping fees, a price per tonne (\$PT) basis, and in the case where the landfill is owned by the WTE facility, a cost of acreage used. The average capital costs per acre for the construction of a landfill is approximately \$1,000,000 and its average acreage spans 33 acres (Anderson et al., 2011). Accounting for a typical lifespan of 30 years and the national average WTE BA production for a WTE facility of 104,000 tpy, the cost savings is approximately \$10/tonne.

For System #1, the production of road base aggregate a screening process would be needed to separate the BA into different size fractions. Once divided into the appropriate size fractions, the WTE BA would then be loaded into an 18 tonne truck by a front-end loader to be transported to an aggregate stockyard for distribution (Addington et al., 2007). One major economic factor to be considered for this scenario is the distance travelled to the stockyard. The mileage considered is from 0 miles (i.e., reflecting processing and sale onsite) to 50 miles, roundtrip. Assumptions considered are that the truck miles per gallon (mpg) and maintenance cost per mile traveled are the same to and from (i.e., one way having full capacity and the other with no load). Also, it is assumed that residues from the screening process that are unsuited for road base applications such as size fractions larger than 2 inches (50.1 mm,) and unburned materials are disposed of via landfill.

Similar to System #1, for System #2 the recovered aggregate would be transported to an aggregate stockyard and examined at a range of miles, roundtrip. It is assumed that the screening and AMR would be executed in a single process. The recovered metals will be examined and evaluated based on a national average market value to assess its economic influence on the processing of WTE BA recycling.

6.2.3 Economic Analysis

For the determination of economic feasibility of processing WTE BA into an aggregate for road base construction, the processing costs (\$PT) were estimated based on the three potential management methods as discussed in the previous sections and at various annual throughput capacities of BA-derived aggregate. Based on current installations, it was assumed that hourly throughputs would range from 10 tonnes to 150 tonnes. Based on standard materials recovery processing operations, it was assumed that processing would occur for 16 hours per day, 6 days per week, for 47 weeks per year, or approximately 4,500 hours per year (hpy) (Tucker et al., 2017). Based on this information, the yearly throughputs for BA-derived aggregate range from 4,500 tpy to 675,000 tpy. Annual fixed capital, operating, maintenance, and labor costs were calculated to determine overall BA

processing costs (\$PT) for the varying annual throughput capacities. A summary of all associated costs for each system is provided in Table 6-3.

Regarding System #1, the fixed capital costs associated with equipment and building construction were considered. Equipment costs were determined by compiling and averaging cost information provided by a personal contact who is currently operating a BA processing facility and quotes from manufacturers. The equipment included screening machinery (screens), a front-end loader and an excavator which costs approximately \$211,250, \$250,000, and \$250,000 respectively. An installation rate of 43% of the total costs associated with processing equipment (screens and AMR system) was assumed (KLM, 2014). Therefore, the total equipment and installation costs for System #1 was \$1,013,338. Equipment purchasing and installation costs for each hourly throughput capacity (10 – 150 t) were depreciated over an assumed 10-year lifespan at 10% annual rate which equals \$101,334 annually.

The average size of a theoretical BA processing building to be constructed and overall land space needed on a preexisting landfill site was assumed to be 2,800 m² and 36,400 m² respectively, through estimation of several BA processing facilities using Google Earth software. The construction cost of a building of such size was estimated using published unit-cost warehouse construction data (RSMeans, 2015). A project size modifier cost multiplier of 1.1 was applied as well as a cumulative inflation rate of 7.3% to adjust to 2019 costs (RSMeans, 2015; USBLS, 2019). After inclusion of an architectural/engineering cost estimated at 15% of building construction cost, the total construction cost of a 30,000 ft² warehouse was estimated to be \$2,179,500. An annual building capital cost of \$72,650 was determined using straight-line depreciation over an assumed 30-year lifespan with no salvage value. The total annual fixed capital for each throughput capacity was inclusive of equipment purchasing and installation, building construction as well as assumed property insurance and tax rates of 1% and 1.5% of total fixed capital, respectively (KLM, 2014). Therefore, the total annual fixed capital costs for System#1 is \$209,083.

The factors for the fixed capital costs for System #2 are nearly equivalent to System #1 with the major difference being the inclusion of equipment costs for the AMR system (magnets and eddy current separators). Information provided from a currently operating BA metals recovery processing plant estimated that AMR equipment accounted for 65% of the capital costs, totaling \$1,321,000. In addition, another front-end loader is required to handle the additional materials flow (i.e., handling ferrous (FE) and non-ferrous metals (NFe)). Consequently, the total equipment costs plus installation and its annualized costs are \$4,473,368 and \$219,112, respectively. No additional building space was assumed to be needed for System #2 compared to System #1; therefore, annual building capital costs are assumed to be the same for System #2 as for System #1. Therefore, the total annual fixed capital costs, accounting for taxes and insurance, for System #2 is \$329,806.

For System #1, equipment operating costs were calculated based on the national average diesel fuel estimate of \$0.79 per liter with an assumed annual operation time of 4,500 hpy (USEIA, 2019a). This was utilized to calculate the screening equipment, front-end loader and excavator's annual operating cost based on a 3,563 L per month, 4,988 L per month

and 12,825 L per month, respectively (AAA, 2019; Combs, 2012). Therefore, the total annual equipment operating cost is estimated to be \$779,781. Annual building operating costs were determined using the latest known annual electricity (151 kWh/m²) and fuel (11.38 m³ natural gas/m²) consumption data for commercial warehouses (USEIA, 2016). Applying the assumed 30,000 ft² building size as well as April 2019 reported national commercial electricity and fuel consumption costs at \$0.1066/kWh and \$7.75/28.3 m³, respectively (USEIA, 2019b). Total annual electricity usage was estimated at \$45,158 and fuel usage at \$21,490. Therefore, the total annual estimated building operating cost was estimated to be \$53,754. Accounting for both equipment and building operating costs, the total annual operating costs for System #1 is estimated to be \$833,535.

For System #2, equipment operating costs were adjusted accounting for the energy consumption from the AMR system as well as the additional front-end loader (see Table 6-3). Otherwise, all other operating costs were assumed to be the same as System #1. The AMR equipment operating costs were calculated based on the average national industrial price of electricity (\$0.07/kWh) reported for 2019 and its required power, with an assumed operation time of 4500 hpy (USEIA, 2019c). Based on the additional equipment, the new total annual operating costs was estimated to be \$1,050,024.

Building maintenance was estimated at \$13,500 per year based on a \$3 per hour rate with a 4,500 hpy operating time (Tucker et al., 2017). Based on information provided by a personal contact who is operating a BA processing plant, the annual equipment maintenance costs for each system was calculated assuming \$1.37 per tonne processed.

Regarding System #1, for labor costs it was assumed that there would be one screen operator (\$15/h), one front-end loader operator (\$15/h) and one excavator operator (\$15/h) (Tucker et al., 2017). Assuming an annual operation time of 4500 hpy, the annual labor costs for System #1 is \$202.500. As for System #2, there would be two additional operators; one front-end loader and another screen operator for the AMR system. Assuming an annual operation time of 4,500 hpy, the annual labor costs for System #2 is \$337,500. Annual supervision costs were assumed to be 15% of annual operating labor costs, totaling an annual cost of \$30,375 and \$50,625 for System #1 and System #2, respectively (KLM, 2014). As per the KLM (2014) report the annual plant overhead costs (including fringe benefits) were assumed to be 60% of yearly total operating labor, supervision and equipment maintenance costs. Lastly, an annual general administration cost was assumed to be 25% of annual operating costs, totaling an annual cost of \$50,625 and \$84,375 for System #1 and System #2, respectively (KLM, 2014). Therefore, the total annual labor costs (excluding plant overhead due to it being influenced by maintenance costs) is \$283,500 and \$472,500 for System #1 and #2, respectively.

For System #1, the total annual costs for WTE ash processing are inclusive of fixed capital, operating, maintenance and labor costs were summed up and divided by the 4,500 – 675,000 tonne of annual throughput capacity to determine the BA processing costs (\$PT). Regarding System #2, the revenue from recovered Fe and NFe metals was accounted for in the analysis. Based upon data from a WTE BA processing facility that recovers metal, the percent recovery of Fe and NFe metals from WTE BA is approximately 0.4% Fe and 1.3%

NFe (Syc et al., 2018; Kahle et al., 2015). A value of approximately \$113/tonne and \$723/tonne was determined by a previous study conducted in 2015 and accounting for inflation for Fe and NFe, respectively (Kahle et al., 2015).

Table 6-3. An annualized cost breakdown of each system based upon fixed capital, operating, maintenance and labor costs.

Annualized Cost Breakdown for Each System			
Categories	System #1 Costs	System #2 Costs	
	(\$)	(\$)	
Fired Capital			
Fixed Capital	101 224	210 112	
	101,334	219,112	
$Building (30,000 ft^2)$	/2,650	72,650	
Land (9 acres), 6% purchased equipment	30,000	30,000	
Property Insurance, 1.0% total fixed capital	2,040	3,218	
Property Taxes, 1.5% total fixed capital	3,060	4,826	
Total Annual Fixed Capital Costs	209,083	329,806	
Onerating			
Equinment	779 781	996 270	
Building	53 754	53 754	
Dunung	55,751	55,751	
Total Annual Operating Costs	833,535	1,050,024	
Maintenance			
Building, \$3/h (4,500 hpy)	13,500	13,500	
Equipment, \$1.37/tonne processed	6,165 - 924,750	6,165 - 924,750	
Total Annual Maintenance Costs	19,665 - 938,250	19,665 - 938,250	
Labor			
Operatina Labor	202.500	337.500	
Supervision 15% operating labor	30 375	50 625	
Plant Overhead 60% operating labor + supervision +	143 424 - 694 575	234 574 - 787 725	
equinment maintenance	110,121 071,070	101,071 707,710	
General Administration. 25% operating labor	50.625	84.375	
		- ,	
Total Annual Labor Costs	426,924 - 978,075	707,074 -	
		1,260,225	
Total Annual Costs	1 480 207 -	2 106 569 -	
1.000111111111111111111111111111111111	2.958.943	3.578.305	

6.3.4 Feasibility

The total BA processing cost (\$PT) versus annual throughput (tpy) with assigned parameters of a depreciated 10-year lifespan (excluding building and land capital costs with 30-year lifespans) and a 4,500 hpy operating time is shown in Figure 6-7. BA-derived aggregates, when used in road base applications, are unlikely to be purchased unless it can compete with retail values of conventional aggregates (e.g., crushed stone). As shown in

Figure 6-7, the national average of conventional road base aggregates used in the US, such as crushed stone, as of 2018 is approximately \$11.90/tonne (USGS, 2019). Referring to Figure 6-7, for System #1, WTE BA becomes economically competitive with these aggregates at throughputs above 85,500 tpy. Meanwhile, at throughputs of 76,500 tpy, System #2 becomes economically competitive to conventional road base aggregates. System #2 provides an economically viable BA-derived aggregate at a lower throughput than System #1 (at 76,500 tpy compared to 85,500 tpy for System #1) but at the a much higher capital investment; \$1,013,338 for System #1 and \$4,473,368 for System #2. The varying distances to the aggregate stockyard in 10, 50, and 100-mile roundtrips is also depicted in Figure 6-7. Regarding System #1, if the stockyard is 5 miles away (10 miles roundtrip) then the tpy shifts from 85,500 tpy to 90,000 tpy, while 25 miles away (50 miles roundtrip) shifts the tpy to 108,000 tpy and at 50 miles (100 miles roundtrip) the tpy shifts to 139,500. Regarding System #2, 10-mile, 50 mile, and 100-mile roundtrips adjust the tpy from 76,500 tpy to 81,000 tpy, 90,000 tpy, and 103,500 tpy, respectively. It is estimated that the average maximum annual throughout of WTE ash produced at a WTE facility is approximately 104,000 tpy (Michaels and Krishnan, 2018). So, therefore the feasibility of implementing a processing facility is more likely for larger WTE facilities and it should incorporate AMR systems; unless the aggregate yard is approximately 5 miles from the processing facility. However, feasibility can also be influenced by the expected metals recovery rate with respect to System #2. A sensitivity analysis (see Table 6-4) shows that an increase of 20% in recovery of Fe and NFe metals shifts the tpy breakthrough point (before transportation to aggregate yard) from 76,500 tpy to 72,000 tpy; a decrease of 20% shifts the tpy breakthrough point (before transportation to aggregate yard) from 76,500 to 85,500. Furthermore, changes in Fe and NFe metals market values produce a change from 76,500 tpy to 72,000 tpy for an increase and to 90,000 tpy for a decrease. A summary of the sensitivity analysis including transportation to the aggregate yard can be found in Table 6-4. As shown, only when the aggregate yard is approximately 50 miles from the processing facility and a 20% decrease in either metals recovery or market value of metals does the economic feasibility become problematic for the average US WTE facility.

As shown in Figure 6-7, it appears that System #2 has the greatest impact of cost per tonne (\$PT). As illustrated, BA reuse feasibility is dependent on the throughput of WTE BA that can be generated by a facility and the intended market for the BA. The market is the same for Systems #1 and #2 (i.e., road base) but their breakthrough points differ greatly. While System #2's breakthrough point is lower than System #1 due to the AMR system, the initial capital costs for the AMR systems accounts for 65% of the budget; therefore, potentially increasing the payback period of the equipment.

Beyond concerns related to cost, there is the matter of evaluating potential environmental risks associated with misuse of the BA-derived aggregate. Currently in the US, there are no federal guidelines that exist to allow for WTE ash reuse, and this action is left to the state departments of environmental protection on a case-by-case basis. Some states have rules to address the reuse of WTE ash, but permission must be acquired from regulators and is typically a project specific determination. Often, approval is based on implementation of engineering controls, or conditions that must be met to ensure that there is no harm to human health and the environment during the project's lifespan. Such examples include

keeping the BA material covered at all times to prevent water contact, documentation of when the BA is acquired, stored, and used, where, and when, along with restrictions on where the BA-derived aggregate may be placed (e.g., not within 100 ft of a potable well used for human/livestock consumption). Further research should also consider the environmental effects associated with the reuse of WTE BA in lieu of conventional aggregate mining and transport.

There is also the question of long-term durability and effects of reusing WTE BA as a construction aggregate in road base applications. Long-term physical performance data is lacking with respect to road base applications. Existing data is gleaned from several studies by the Federal Highway Association (FHWA) during the 1970s on utilizing WTE ash in road construction and concrete manufacturing (Wiles and Shepherd, 1999). While most of the recorded observations were promising, some of these involved vague generalizations for road conditions such as "fair' or "good" (Wiles and Shepherd, 1999). These observations also only continued for a few years post-construction and thus long-term physical data is lacking.

There is also the aspect of social acceptance for reusing WTE BA in construction materials. Faleschini et al. (2016) mentions that the beneficial use of recycled aggregates in the construction industry may be affected be the less desirable qualities of the materials. Not to mention that there may be stigmas for reusing WTE BA due to the perceived notions that it is hazardous to human health and the environment even if approved for use by an environmental protection agency.



Figure 6-7. Total cost to process WTE BA based on processing throughout capacity (in tonnes) and an assumed 10-year plant life (excluding building and land capital with 30-year lifespans). The current price per tonne of sand and gravel and landfilling tipping fees are represented as well.

		% Change in Metals Yield		% Change in of M	Market Value etals
Distance to Aggregate Yard	Before %	20%	20%	20%	20%
(miles, roundtrip)	Change (tpy)	Increase	Decrease	Increase	Decrease
0	76,500	72,000	85,500	72,000	90,000
10	81,000	76,500	85,500	76,500	90,000
50	90,000	85,500	99,000	85,500	103,500
100	103,500	94,500	117,000	94,500	117,000

Table 6-4. Summary of sensitivity analysis results for System #2.

Sensitivity Analysis for System #2 - Metals Recovery

6.4 Conclusions

When considering whether an MRF should landfill its recovered glass cullet, pay for it to be traditionally recycled into new glass containers, or further process it to a size suitable to act as a SCM in concrete production is highly dependent on the MRF's maximum annual throughput capacity. It is relatively inexpensive for an MRF to send its cullet to a closed landfill to be used as depression fill material if associated tipping fees are avoidable as a result of co-ownership or an applied fee waiver in exchange for free fill material. However, traditional recycling into new glass containers remains to be the most popular market, especially from an environmental standpoint as the amount of raw materials used as well as material landfilled is significantly reduced. Although MRFs typically pay private glass processing companies to transport their recovered cullet off-site, the associated costs would likely decrease if the processing technique was altered to produce cleaner cullet (e.g., through positive sorting, use of air classification, etc.). While it was found in Section 2 that using glass powder as a pozzolanic material to partially replace portland cement in concrete manufacturing certainly reduces environmental impacts (i.e., carbon emissions and energy usage) relative to landfilling, it provides similar (to slightly better) environmental results when compared to traditional glass recycling. The analysis conducted here found that the GP product resulting from cullet size-reduction processing can compete with the traditional glass recycling market as long as there is an annual processing throughput of at least 50,000 t of MRF cullet (based on a 10-year lifespan with a 4,500 h per year operating time). Technical specifications for the use of GP in concrete and the logistical issues associated with providing GP to concrete batch plants require examination and development. Further research should be conducted to predict the future demand for GP pozzolan based on expected decreases in coal fly ash supply, and in turn, increases in retail value.

The methods for recycling WTE BA into an aggregate for reuse into construction materials, such as road base, is affected by the desired market for the material. WTE BA, as currently generated in the US, is mostly suited for a road base application due to its generally well graded particle size distribution along with requiring limited screening to create a product. This market is the same for Systems #1 and #2 but System #2 requires a lower throughput value due to the revenue obtained from metals recovery. Also, accounting for the miles

travelled to an aggregate stockyard renders System #2 more feasible than System #1. It is important to reiterate that this model assumes that the generation, staging and processing of WTE BA would occur on premises. If transportation is necessary between these steps, or if wholesale of this material also occurs on premises, then estimated costs may increase or decrease accordingly. Technical and environmental specifications for the use of WTE BA in construction materials is an area that needs further research and development beyond cost analysis. For instance, if the use of BA-derived aggregates leads to the increased frequency of road maintenance then this will affect the economic feasibility; thus, the need for research on long-term durability. Further research should also consider the environmental effects associated with the reuse of WTE BA in lieu of conventional aggregate mining and transport.

7.0 Research Needs

Understanding the results gathered from the literature and UF's laboratory and modeling analysis, solutions and future work are proposed for increasing the beneficial use of waste glass as a pozzolan and WTE ash in Florida. These concepts are developed in the following sections.

7.1 Developing Specifications for Waste Glass Derived GP and WTE Ash

Results from the literature UF's laboratory analysis show promise in the use of GP and WTE ash in civil engineering construction applications such as PCC and HMA, while environmental and economic analysis shows that these reuse options can be beneficial and feasible. While the performance, environmental, and economic analyses show promise in successfully reusing waste glass and WTE ash in these products, there is a lack of material and construction specifications on how to discern whether the product should be reused. Since wastes such as glass and WTE ash are inherently variable in their sources, it is important to have standards developed knowing when a material should not be used based on composition or how to ensure quality control (QC) when used in construction purposes. Currently, there are no specifications for reusing waste glass in GP nor WTE ash in civil engineering applications. Some potential examples for each material are included in Table 7-1.

Material	Example Material	Example Construction	
	Specification	Specification	
	Shall not contain more than	Workability/slump.	
	5% of non-glass material.		
Wasto Class	Shall not contain residual		
Waste Glass	organics.	Shall meet requirements	
	Shall have an alkali content of	of ASTM C1567.	
	less than "x" Na		
	Shall have a loss on ignition	Shall have a limerock	
WTE Ash	(LOI) of less than "x"%	bearing ratio of at least	
		"X".	
	Shall have a particle size	Shall have a gradation	
	distribution of "X" to "Y".	meeting "x" specification	
		for road base.	

Table 7-1. Example specifications for waste glass as a pozzolan and WTE ash in PCC and HMA.

While the reuse of these materials may make sense from a physical, environmental, and economic standpoint, without a specification approved by a transportation authority (e.g., FDOT) many municipalities, counties, and industries are reluctant to "take a chance" on a material that has not be officially approved. This is also related to that there is no approved

methodology for using the material, which may cause confusion or neglect during usage which may result in subpar performance.

7.2 Evaluating Washing Systems for WTE Ash

While it was shown that removal of deleterious substances (i.e., alkalis, heavy metals, chlorides) can be effectively removed at L/S ratios currently used for aggregate processing, it appears that longer contact times also play a crucial role in removal efficiency. This is hypothesized to be linked to fines removal. The removal of fines is believed to be due to abrasion of the ash surface during washing through the collision of ash particles with one another which removes the fines and surface believed to contain most of the heavy metals, chlorides, sulfates, and alkalis. However, for washing to occur on an industry scale, the hydraulic residence time must be reduced to make it more economical and practical. It may be possible to include abrasive material into the washing system (e.g., grit) that could be easily filtered out of the coarser aggregate source to serve to speed up this process.

While utilizing ash as a coarse aggregate replacement (3/4"-No.4, 19.0-4.76 mm) has been the primary focus of research by UF for reuse in PCC and HMA to date, there is also potential to explore the potential of washed WTE ash as fine aggregate (e.g., No.4-No.50). Previously, this was limited by the quality of the finer ash-aggregate with considerations such as alkali content and dust which would impede the physical and chemical properties of PCC and HMA. However, based on washing coarse ash-aggregate, it may be possible that a washed fine ash-aggregate may perform well in PCC and HMA. This may be particularly advantageous for HMA applications as large pieces of glass and ceramic would not be present which has been shown to cause issues with moisture susceptibility. This observation is why utilizing glass in HMA (e.g., "glassphalt") typically limit the particle size of glass to that of a fine aggregate (< No.4, 4.76 mm).

7.3 Improving Recovery of Waste Glass from MSW Stream

Waste glass is common "negatively" sorted at MRFs, which means that is it treated more as a residual of recycling rather than a valuable commodity, if it's even collected for recycling at all. This approach not only reduces the profitability of waste glass cullet but introduces contamination into this stream, such as metal lids/caps, plastics, papers, and food and beverage residuals. Often, this results in glass not being recovered at all and ending up disposed in landfills. When considering that this waste glass may also be recycled or reused as a pozzolan, this material represents a "lost" resource similar to unrecovered ferrous/non-ferrous metals in WTE ash. The effects of glass recycling have also been shown to produce an ash product with less glass (del Valle-Zermeño, 2017).

7.4 Investigating Ferrous/Non-Ferrous Metals Quality in WTE Ash Stream

One factor that played into making ash-derived aggregates more economically feasible at lower throughputs was the recovery of ferrous and non-ferrous metals from the WTE ash stream. Increasing both the yield, or recovery, and the purity, or grade, of the metals, especially non-ferrous, may play a significant role in making this process more feasible. This should also involve understanding how metals are physically/chemically changed during combustion and subsequent aging. Previous works suggest that corrosion occurs during combustion which can limit metals recovery and that during aging over time (i.e., years) can erode metals resulting in lost material (Bunge, 2015). Understanding these factors may promote more efficient metals recovery systems that can make this process even more profitable and economical. It is also important to note that different nonferrous metals behave differently and that different systems can be used to separate out different nonferrous metals.

8.0 SUMMARY AND CONCLUSIONS

This project aimed to further the understanding of how waste glass and WTE ash can be beneficially reused within Florida. This was achieved by conducting an extensive literature review that sought to unveil how glass powder has been utilized in PCC and how pretreatments on WTE ash, such as washing, can improve its ability to be reused in PCC and HMA. In addition, the economic feasibility and development of infrastructure to support full-scale waste glass and BA recycling as GP and ash-aggregate, respectively, were also investigated to further support reuse activities in Florida.

Interest was focused on how glass powder affects the physical performance of PCC by acting as a SCM to convert weaker calcium hydroxide formed during curing into calcium-silica-hydrate (CSH) improving the strength of the final product. The environmental and economic impacts of reusing GP in PCC were also touched on by mentioning that the use of GP results in some significant GHG savings when compared with conventional portland cement as shown in work by Jiang et al. (2014). Yet, the economic impacts and necessities to utilize GP on a large-scale, particularly from waste glass, are nonexistent in the literature and demanded further examination as performed in Section 6.

Following the examination of GP, pretreatment methods for WTE ash were reviewed, which involved stabilization/solidification, thermal treatment, and separation processes such as screening, metals recovery, and washing. Out of these, the most commonly used method was washing. The effects of washing and what they may suggest for performance in PCC and HMA mixtures was also discussed.

Based on the literature review, it's understood that GP may be able to mitigate deleterious ASR reactions that occur in PCC. Since ASR has been detected in mixtures using BA, GP was added as a SCM in mixtures using BA to observe whether this addition may mitigate ASR. Overall, the addition of GP to WTE ash from one facility resulted in limited expansion; however, for the other two facilities, eventually deleterious expansion did occur, albeit at a slower rate than without GP.

Since it's speculated that washing WTE ash can remove alkalis, which facilitate ASR, a possible solution is washing. This strategy can also eliminate more absorptive fine aggregates and dust on the surfaces of coarser ash-aggregate that can cause increases in w/cm ratios and, for HMA, asphalt binder demand which has a significant performance and economic impact. Beyond physical performance, washing has been utilized in the literature to remove trace metals, especially for fly ash treatment. Washing was explored using various L/S and contact times to reflect practical washing procedures by industry. Testing revealed that removal of fines, heavy metals, alkalis, chlorides, and sulfates were not significantly affected by L/S ratio but rather by contact time with longer durations corresponding to higher removal efficiencies, particularly for fines. This resulted in a large-scale testing with larger quantities (approximately 25 kg) were washed to produce washed ash aggregate for use in PCC and HMA. Testing of this ash-derived aggregate revealed that there were slight reductions in heavy metal, chloride, and sulfate leaching from this ash

product. Physical properties revealed a significant reduction in absorptivity, an increase in specific gravity, and decrease for surface pH. These properties are significant for PCC/HMA mixtures as absorptivity decreases workability of PCC while also increasing binder demand for HMA. A higher density aggregate, in theory, produces a more durability material, while surface pH can play a role in adhesion in HMA applications.

To test the impact of washing on the physical properties of ash-amended HMA, specimens were prepared using WTE ash from two facilities using unwashed and washed ash-derived coarse aggregate (No. $4 - \frac{34}{7}$). These specimens were developed as Superpave SP-12.5 dense graded mixtures. Performance was evaluated based on asphalt volumetrics, moisture susceptibility, and rutting susceptibility. Ash-amended mixtures were compared to a control (non-ash) mix for comparison. For the two ash mixtures, an increase of 0.1% for Mix B and 0.5% for Mix D. Overall, the unwashed ash-amended mixtures had difficulty meeting VMA while the washed mixtures easily met VMA attributed to the higher specific gravity of the washed ash-aggregates compared to their unwashed counterparts. As for as rutting resistance is concerned, washing did not have much of an effect on improving or worsening resistance aside for Mix E; although, this may be attributed to changes in gradation primarily. As far as moisture resistance is concerned there was a minimal change between Mix B and C which may be attributed to the already large presence of glass and ceramics in that ash stream which allow for significant stripping of asphalt from these aggregates. While it was suspected that there would be a more significant reduction in asphalt binder content for Mix D, it was only 0.1% which suggests that the finer aggregate and dust is not the primary driver for asphalt binder demand but may actually be due to the highly angular ash-derived aggregate along with potentially higher surface area.

Even with adequate physical and environmental performance, there is still the means of producing GP and ash-derived aggregate on a full-scale basis. This involves developing the infrastructure such as processing facilities and transporting the material to the appropriate markets. Ultimately, the cost of processing waste glass to GP and BA relies heavily upon the throughput of material, with more throughput significantly decreasing the cost of both waste glass derived GP and BA-derived aggregates.

Lastly, from understanding the results gathered from the literature and UF's laboratory analysis, solutions and future work were proposed for increasing the beneficial use of waste glass and WTE ash in the Florida. These solutions involve further testing of waste glass derived GP and the development of specifications for its use (e.g., contaminant limits, fineness requirements). Specifications can also be developed for the reuse of WTE ash as well including parameters such as gradation and bearing strength (e.g., LBR).

9.0 References

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