



**CHARACTERIZING EMISSIONS FROM OPEN BURNING OF MILITARY
FOOD WASTE AND PACKAGING FROM FORWARD OPERATING BASES**

THESIS

Thomas Dominguez, Captain, USAF

AFIT-ENV-MS-16-M-145

**DEPARTMENT OF THE AIR FORCE
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WASTE AND PACKAGING FROM FORWARD OPERATING BASES

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Captain, USAF

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Abstract

Emissions from open burning of Meals, Ready-To-Eat (MRE) and MRE packaging were characterized in response to inhalation concerns at military forward operating bases. Emissions from four compositions of MREs and four types of packaging were determined and compared to assess contributions of individual components. Measurements of particulate matter (PM_{2.5}, PM₁₀) and metals showed that MREs account for 70 to 90 percent of PM emissions when burned in unison with the current fiberboard container and liner, PM_{2.5} constitutes a vast majority of the particulates emitted from both MREs and their fiberboard packaging, and that metal emission factors were similar regardless of material tested. Measurements of volatile organic compounds (VOCs) and polycyclic aromatic hydrocarbons (PAHs) indicate that the targeted replacement of MRE constituents may be more effective in reducing PAH and VOC emissions than variation of fiberboard packaging designs, while polychlorinated dibenzodioxin and -furan (PCDD/PCDF) emissions are believed to be more closely related to other components of a military waste stream. MRE compositions and fiberboard types each respectively produced essentially equivalent PM, PAH, VOC, and PCDD/F emission factors. This study provides the first representative characterization of open burning emissions associated with military rations separate from comprehensive military waste streams.

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Thomas Dominguez

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List of Acronyms

ACGIH	American Conference of Governmental Industrial Hygienists
AFIT	Air Force Institute of Technology
ANOVA	Analysis of Variance
AOR	Area of Responsibility
ATSDR	Agency for Toxic Substances and Disease Registry
BTEX	Benzene, Toluene, Ethylbenzene, Xylene
CAA	Clean Air Act
CFNP	Corrugated Fiberboard Container & Liner, No Wet Strength Polymer
CFP	Corrugated Fiberboard Container & Liner with Wet Strength Polymer
CH ₄	Methane
CO	Carbon Monoxide
CO ₂	Carbon Dioxide
DOD	Department of Defense
EF	Emission Factor
EPA	Environmental Protection Agency
ES&T	<i>Environmental Science and Technology</i>
ESTCP	Environmental Security Technology Certification Program
FDA	Food and Drug Administration
FOB	Forward Operating Base
FRH	Flameless Ration Heater
HAP	Hazardous Air Pollutant

HRGC/HRMS	High Resolution Gas Chromatography/High Resolution Mass Spectrometry
IARC	International Agency for Research on Cancer
IOM	Institute of Medicine
IRIS	Integrated Risk Information System
K-W	Kruskal-Wallis
LDPE	Low Density Polyethylene
LRGC/LRMS	Low Resolution Gas Chromatography/Low Resolution Mass Spectrometry
MCE	Modified Combustion Efficiency
MRE	Meal, Ready-to-Eat
MRL	Minimum Risk Level
NIOSH	National Institute for Occupational Safety and Health
NSRDEC	Natick Soldier Research, Development and Engineering Center
OBTF	Open Burn Test Facility
OSHA	Occupational Safety and Health Administration
PAH	Polycyclic Aromatic Hydrocarbon
PCDD	Polychlorinated Dibenzo-p-Dioxin
PCDF	Polychlorinated Dibenzo-p-Furan
PEL	Permissible Exposure Limit
PM	Particulate Matter
PUF	Polyurethane Foam
REL	Recommended Exposure Limits

SFNP	Solid Fiberboard Container & Liner, No Wet Strength Polymer
SFP	Solid Fiberboard Container & Liner with Wet Strength Polymer
SVOC	Semi-Volatile Organic Compound
TEF	Toxic Equivalent Factor
TEQ	Toxic Equivalent
TLV	Threshold Limit Value
US	United States
USA	United States Army
USCENTCOM	United States Central Command
VOC	Volatile Organic Compound
WHO	World Health Organization
XRF	X-ray Fluorescence

CHARACTERIZING EMISSIONS FROM OPEN BURNING OF MILITARY FOOD WASTE AND PACKAGING FROM FORWARD OPERATING BASES

I. Introduction

General Issue

In the past, open burning in the form of “burn pits” has been utilized to reduce large volumes of waste that the United States (US) military generates daily, especially during Operation Enduring Freedom (October 2001 – December 2014) in Afghanistan and Operation Iraqi Freedom (March 2003 – August 2010) in Iraq (Department of Defense, 2010; Torreon, 2015). A report by the Government Accountability Office in 2010 estimated that there were 251 active burn pits in Afghanistan with another 22 active burn pits in Iraq (United States Government Accountability Office, 2010). These burn pits destroyed tons of uncharacterized waste in large pits that could be as wide as 90,000 square feet for 4,000 or more personnel (U.S. Army, 2013). Burn pits were originally meant to be one of the last options considered for the rapid destruction of solid waste, with host-nation contracts, US-operated landfills, incinerators, and waste-to-energy systems as preferred options (Barnes, Aurell, & Gullett, 2016; Department of Defense, 2010). However, burn pits had become the standard operating procedure for the destruction of large volumes of waste for deployed military operations.

The considerable volume of waste generated daily had forced US Central Command (USCENTCOM) commanders to take immediate action and rely on burn pits to dispose of solid waste while concurrently implementing recycling efforts and arranging for incinerator use upon award of the appropriate contracts (Department of Defense, 2010). Military waste generation estimates vary but a 2011 approximation indicated eight to ten pounds of waste were generated

per person per day, which led to many hundreds of tons of waste being openly burned daily in the USCENTCOM area of responsibility (AOR) (Institute of Medicine, 2011).

A sharp decrease in openly burned waste, from approximately 200 tons in 2007 to 10 tons in 2009, was due to a multitude of factors, with one of those factors being concerns that the emissions from the open burning of military waste were associated with detrimental health effects (Institute of Medicine, 2011; Jones et al., 2012; Szema, 2009; United States Government Accountability Office, 2010; Weese, 2010). Environmental laws in the US such as the Clean Air Act (CAA) do not generally apply overseas (United States Government Accountability Office, 2010), therefore, USCENTCOM issued guidance (USCENTCOM, 2009) in 2009 to reduce environmental and personnel health risks from burn pits. Based upon continuing concerns, the Department of Defense (DOD) also charged the Institute of Medicine (IOM) in 2011 to conduct a thorough investigation to determine the long-term health consequences of exposure to burn pits, specifically from burn pits found within Afghanistan and Iraq (Institute of Medicine, 2011). However, this IOM study found inconclusive results that could not link burn pits as the primary cause of adverse health effects seen within exposed military populations (Institute of Medicine, 2011).

Additional studies taking advantage of the Millennium Cohort Study epidemiological data (Jones et al., 2012; Powell et al., 2012; B. Smith et al., 2012) attempted to find adverse health effect relationships associated with exposure to burn pits. These studies too, however, could not directly attribute burn pit emissions to negative health-related outcomes.

A significant limiting factor in these studies was the lack of characterization of the military waste streams that was being openly burned. Food and paperboard products have been known to comprise significant portions of municipal solid waste (Marsh & Bugusu, 2007; Staley

& Barlaz, 2009) and it has been shown that fiberboard products and food can comprise a significant portion of military waste streams as well (Rock et al., 2000; Ruppert et al., 2004). In 2009, the US Army food program was charged with managing “over 400 active dining facilities worldwide with 19 [dining facilities], 97 forward operation bases (FOBs) or coalition outposts ... in Afghanistan alone, and another 71 [dining facilities] and 52 FOBs in Iraq” (Barrett & Cardello, 2012). With the approved US Army field feeding system standard requiring three quality meals per day (U.S. Army, 2012) and over 144 million MREs being purchased in 2004 alone for field and base camp activities, nearly 67,000 tons of ration-related waste were generated and required disposal (Barrett & Cardello, 2012).

Every ounce of a 1.5 pound (US Army NATICK, 2015) military ration has an associated cost, whether it is measured by the pounds of fuel or by the number of personnel required for transportation from its point of origin to its destination. Therefore, any reduction in the weight associated with a military ration can provide large, long-term savings. Research into alternatives for both MRE primary packaging (DoD Combat Feeding Directorate, 2014) and fiberboard secondary packaging (Ratto et al., 2012) has been ongoing (Barrett & Cardello, 2012). These alternative designs could potentially provide weight reductions, thereby reducing costs in production, storage, and transport.

Many hazards have been associated with open burning (Cançado et al., 2006) and it has been speculated that respiratory ailments in veterans can be attributed to burn pit emissions (Riess, 2012; United States Government Accountability Office, 2010). However, the specific emissions associated with the open burning of military rations remains to be seen within the literature and, as such, cannot assist in determining whether burn pits may have a role in adverse health effects seen in veterans. By quantifying these airborne emissions, potential health risks

could be elucidated and further studies could be accomplished with greater understanding of the ration-related waste contributions to the overall open burning of military waste.

Problem Statement

Open burning of military waste in the form of burn pits is a limited but continued practice within the DOD. The emissions associated with this open burning have yet to be thoroughly characterized but limited studies suggest that food and paperboard products comprise a significant portion of military waste. Despite this prevalence of food and paperboard products within military waste streams, as well as suspected health risks associated with military waste open burning, a thorough characterization of the emissions associated with these food and paperboard products remains to be seen in the literature.

Research Questions

The primary objectives of this research effort are to characterize and compare the emissions of key pollutants of interest associated with the open burning of food and paperboard waste in the form of military rations and their fiberboard packaging. Alternatives to the current designs of MREs and their fiberboard packaging have been proposed and require characterization alongside the current military rations. Following characterization, airborne pollutant concentrations modified to the form of emission factors require human and environmental health context in order to provide clarity to health risk assessors. Therefore, this research will address the subsequent questions:

1. What are the airborne emission factors for the current and alternative designs for MREs and their fiberboard packaging?
2. How do the emission factors for the current and alternative designs compare with one another and to open burning situations within the literature, such as military waste burns?

2.1. What are the airborne emission factor effects of varying the nanocomposite content within the alternative MRE design?

Scope and Approach

Two distinct sampling campaigns were conducted in support of this research effort at the US Environmental Protection Agency's (EPA) Open Burn Test Facility (OBTF) situated in Research Triangle Park, North Carolina. The first campaign consisted of the characterization of the current fiberboard secondary packaging, as well as three alternative designs, associated with MREs. The second campaign consisted of the characterization of the MREs themselves, similarly with three alternative designs. Pollutants sampled for included particulate matter 10 μm (PM10) and 2.5 μm (PM2.5) in diameter or less, polycyclic aromatic hydrocarbons (PAHs), volatile organic compounds (VOCs), and polychlorinated dibenzo-p-dioxins (PCDDs) and -furans (PCDFs). Following sample analyses, pollutant concentrations were converted to emission factors for comparison purposes and are briefly discussed within Chapter II but are explained in greater detail within Chapter III and Appendix D. Statistical analyses were performed on the emission factors to determine statistical significance between the materials and are outlined within Appendix C while health-related outcomes associated with the pollutants are described thoroughly within Chapter III. Lastly, Chapter IV provides conclusions drawn throughout the research effort, identifies research limitations, and presents future research opportunities to further protect the health of service members.

Significance

Though the practice of open burning of military waste has been significantly reduced, it is still an expedient method of disposal suited for small contingency units where no other means

of waste disposal are available or practical. By characterizing the emissions associated MREs and their fiberboard packaging, their specific contributions to US military waste open burning emissions can be isolated and quantified. Additionally, differences found in emissions between the current and alternative designs for MREs and fiberboards could potentially identify packaging materials that can reduce weight and waste, emit less emissions, and possibly reduce pollutant exposures to better protect the health and meet the various needs of US military personnel. Furthermore, quantification of uncharacterized pollutant emissions could potentially lead to the establishment of health risk threshold criteria in the future and can supplement USCENTCOM burn pit guidance. Emission factors generated from this research effort establish the first baseline of what MREs and their fiberboard packaging solely contribute to military waste burn emissions and can assist the US military in quantifying, establishing, and understanding the potential health risks associated with such a prevalent waste stream product.

Preview

This thesis is written in the Scholarly Article format and the culmination of this research effort is encompassed within Chapter II, which contains a draft manuscript for projected publication in the journal entitled *Environmental Science and Technology (ES&T)*. Chapter II is presented in a modified format appropriate for submission to *ES&T* as well as for documentation as an Air Force Institute of Technology (AFIT) thesis. The article comprises the bulk of this research endeavor, answers the research questions investigated, and has independent sections that mirror those presented within the thesis (such as a distinct Abstract, Introduction, etc.). Chapter III explores the health effects associated with the aforementioned pollutants as well as the significance associated with the emission factors determined through this research while Chapter IV presents a summary of findings and presents opportunities for future research.

Supplemental information for concurrent submission to *ES&T* is presented within the thesis as Appendix A: Scholarly Article Supplemental Information. Appendix B: Material Composition describes the MREs and fiberboard packaging in greater detail. Analytical methods utilizing inferential statistics, such as analysis of variance (ANOVA) and Tukey's honest significant difference tests, were used to determine distinctions amongst the various materials tested and are expanded upon within Appendix C: Elaborated Statistical Methodology. Lastly, Appendix D: details the emission factor calculations.

II. Scholarly Article

Written for consideration of submission to
Environmental Science and Technology
(<http://pubs.acs.org/journal/esthag>)

CHARACTERIZING EMISSIONS FROM OPEN BURNING OF MILITARY FOOD WASTE AND PACKAGING FROM FORWARD OPERATING BASES

Thomas Dominguez, Johanna Aurell, Brian Gullett, Robert Eninger, Dirk Yamamoto

Abstract

Emissions from open burning of Meals, Ready-To-Eat (MRE) and MRE packaging were characterized in response to inhalation concerns at military forward operating bases. Emissions from four compositions of MREs and four types of packaging were determined and compared to assess contributions of individual components. Measurements of particulate matter (PM_{2.5}, PM₁₀) and metals showed that MREs account for 70 to 90 percent of PM emissions when burned in unison with the current fiberboard container and liner, PM_{2.5} constitutes a vast majority of the particulates emitted from both MREs and their fiberboard packaging, and that metal emission factors were similar regardless of material tested. Measurements of volatile organic compounds (VOCs) and polycyclic aromatic hydrocarbons (PAHs) indicate that the targeted replacement of MRE constituents may be more effective in reducing PAH and VOC emissions than variation of fiberboard packaging designs, while polychlorinated dibenzo-dioxins and -furans (PCDDs/PCDFs) emissions are believed to be more closely related to other components of a military waste stream. MRE compositions and fiberboard types each respectively produced essentially equivalent PM, PAH, VOC, and PCDD/F emission factors. This study provides the first representative characterization of open burning emissions associated with military rations separate from comprehensive military waste streams.

Introduction

The United States Department of Defense (DOD) has traditionally depended upon field expedient methods such as open burning, air curtain burners, and incinerators as a practical means to reduce volume and generally dispose of United States military waste in contingency environments overseas, such as within Iraq and Afghanistan. Open burning, commonly referred to as “burn pits”, provides relatively small forward operating bases (FOBs) the flexibility to dispose of waste without placing additional service members in a direct line-of-fire from enemy forces while transporting the FOB waste (i.e., backhauling) to a larger base with incineration capabilities (U.S. Army, 2006). By destroying the waste, it avoids vermin problems and vector-borne disease, while also preventing enemy combatants from gaining access to potential resources and intelligence. However, service members may be simultaneously exposed to pollutants originating from open burning (Aurell, Gullett, & Yamamoto, 2012; Woodall, Yamamoto, Gullett, & Touati, 2012). Conclusions from investigations into the long-term health effects associated with open-pit burning in a deployed military setting have generally been inconclusive (Institute of Medicine, 2011; “Special Issue - Health Effects of Deployment to Afghanistan and Iraq,” 2012).

Military rations have been a staple of the US military since the American Revolution (Barrett & Cardello, 2012) and the current version, the Meal, Ready-to-Eat (MRE), is often relied upon for short-term land operations. A single fiberboard package and liner contains 12 MREs that provide a convenient manner for the warfighter to retain adequate caloric intake to continue necessary operations (Feagans, Jahann, & Barkin, 2010; Institute of Medicine, 2005). The utility of the MRE can be demonstrated in overall consumption; in 2004 alone, over 144 million MREs were purchased for United States Army (USA) camp feeding activities, leading to

approximately 67,000 tons of solid waste being generated (Barrett & Cardello, 2012). With the current MREs and their fiberboard packaging considered non-recyclable, there is increased emphasis to investigate alternative materials and designs for combat ration MRE/fiberboard packaging applications.

Recent research by the US Army Natick Soldier Research, Development and Engineering Center (NSRDEC) into alternative fiberboard packaging for MREs has focused on compostability, Food and Drug Administration (FDA)-compliance, and replacing the hazardous chemicals used for water resistance with coatings designed to be non-hazardous. This research effort also includes altering the current solid fiberboard packages by corrugating the fiberboard to save on weight and reduce solid waste for the military (Ratto et al., 2012). Additional research by NSRDEC is ongoing for the potential MRE replacement of aluminum-based pouches with those that are nanocomposite-based and can provide greater barrier and mechanical properties (DoD Combat Feeding Directorate, 2014). By improving the current designs for both the fiberboard packaging and the aluminum-based pouches for MREs, a reduction in solid waste associated with the consumption of military rations is anticipated.

Few studies have been conducted, however, to specifically characterize the emissions from the open burning of military waste. Other open burning situations encountered within the literature, such as biomass and household waste burns, have previously characterized pollutants such as metals, volatile organic compounds (VOCs), polycyclic aromatic hydrocarbons (PAHs), polychlorinated dibenzo-dioxins and -furans (PCDDs/PCDFs), and particles with an aerodynamic diameter of 10 μm (PM_{10}) and 2.5 μm ($\text{PM}_{2.5}$), respectively (Aurell et al., 2012; Woodall et al., 2012). Many of these VOCs are on the US Environmental Protection Agency's (EPA) list of hazardous air pollutants (HAP) (US EPA, 2008) due to human toxicity concerns of

varying degrees and are classified as possibly, probably, or a confirmed carcinogen to humans (Group “2B” or greater) by the World Health Organization’s (WHO) International Agency for Research on Cancer (IARC) (WHO - IARC, 2015). Other VOCs have the ability to form ground-level ozone, which can cause various respiratory ailments (Pappas et al., 2000). The EPA’s 16 PAH priority pollutants have been investigated for over three decades in various capacities, with some of these pollutants having mutagenic and carcinogenic properties (Keith, 2015; Pickering & Pickering, 1999). PCDDs/PCDFs are of concern due to their toxicity and carcinogenicity, and are recognized as bioaccumulative and persistent within the environment (US EPA, 2003). PM_{2.5} has been known to cause adverse health effects (Samoli et al., 2008) and is a criteria pollutant regulated by the US EPA.

The objectives of this study were to (1) characterize the emissions from current and newly-developed MRE material designs and variations of their fiberboard packaging material, and (2) compare derived emission factors for the various packaging materials against the literature. The two variables that were studied included (1) the variation by mass of MRE aluminum-based pouches with pouches that are nanocomposite-based and (2) the presence or absence of a polymer that provides strength in high moisture conditions (i.e., wet-strength) on both solid and corrugated fiberboard packaging designs for MREs. To our knowledge, this is the first study in the literature to characterize emissions from MREs and their packaging separately from a comprehensive military waste stream.

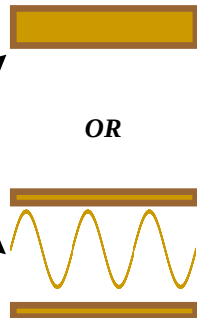
Methodology

Materials.

One of the variables of this experiment was to vary the aluminum/nanocomposite composition of an MRE by mass. This is expressed in the form of a percentage of aluminum

content replaced; for example, MRE-32% would indicate that 32% of aluminum-based components of an MRE were replaced by mass with components composed of nanocomposites for this study (DoD Combat Feeding Directorate, 2014). Therefore, four distinct MRE materials were created and tested for this experiment: MRE-0%, MRE-32%, MRE-66%, and MRE-100% and are shown in Table 1. Variations of the fiberboard packaging that 12 MREs are normally contained within were also studied. Two fiberboard structural designs were utilized within this experiment: the current MRE's solid design and a newly-developed corrugated design (Ratto et al., 2012). Additionally, these two fiberboard structural designs are characterized as having or not having a wet-strength polymer coating. Therefore, four distinct fiberboard packaging materials were tested and are also shown in Table 1.

Table 1. Variations of Test Materials

			Visual Depiction	
Test Material	Description			
Fiberboards	SFP	Solid Fiberboard Container & Liner with Wet Strength Polymer		
	SFNP	Solid Fiberboard Container & Liner, No Wet Strength Polymer		
	CFP	Corrugated Fiberboard Container & Liner with Wet Strength Polymer		
	CFNP	Corrugated Fiberboard Container & Liner, No Wet Strength Polymer		
MREs	MRE-0%	MRE (No Changes)		
	MRE-32%	MRE with 32% of Al-based contents replaced by Nanocomposite		
	MRE-66%	MRE with 66% of Al-based contents replaced by Nanocomposite		
	MRE-100%	MRE with 100% of Al-based contents replaced by Nanocomposite		

A: MRE breakdown depicting aluminum-based (OLD) versus nanocomposite-based (NEW) pouches

Test Setup.

Fiberboard packaging and MRE tests were conducted in November 2014 and July-August 2015, respectively. All tests were conducted at the US EPA's Air Pollution Prevention and Control Division Open Burning Testing Facility (OBTF) located at Research Triangle Park, North Carolina. The OBTF has been described and utilized previously to study open burning conditions (Aurell, Gullett, & Tabor, 2015; Grandesso, Gullett, Touati, & Tabor, 2011; Lemieux, Lutes, & Santoianni, 2004; Woodall et al., 2012) and was used in a similar experimental setting capacity (see Figure 10).

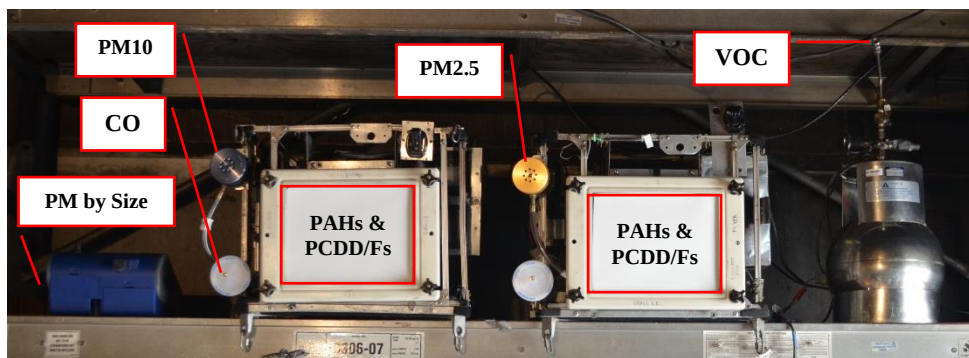


Figure 1. OBTF Sampling Package for PM, VOC, PAH, and PCDD/PCDF Emissions

The sampling package, Figure 1, consisted of PM_{2.5}, PM₁₀, VOC, and SVOC batch samplers as well as continuous measurements of carbon dioxide (CO₂), carbon monoxide (CO), PM by size, and temperature. The sampling package operated on battery (48V) and was remotely controlled for sampler operation while data were continuously logged to the computers. Further details on the sampling package, frequently referred to as “The Flyer,” can be found in the literature (Aurell, Gullett, Pressley, Tabor, & Gribble, 2011; Aurell et al., 2015; Aurell & Gullett, 2010, 2013; Grandesso et al., 2011) and is therefore only briefly described here. PM₁₀ and PM_{2.5} impact samplers, using 47 mm Teflon[®] filters with a pore size of 2 µm, were connected to Leland Legacy pumps (SKC Inc., USA) with Tygon[®] tubing utilizing a compensating pump with a set flow rate of 10 L/min. Filters for PM_{2.5} were also analyzed by X-ray Fluorescence (XRF) spectroscopy (US EPA, 1999a). VOCs were sampled with SUMMA[®] canisters, in accordance with US EPA Method TO-15 (US EPA, 1999c) with a 12-minute sampling time. The SUMMA[®] canister was also analyzed for CO, CO₂, and CH₄, following US EPA Method TO-25C (40 C.F.R. § 60, 2014). Quartz filters to collect PCDD/PCDFs were placed in-line with sampling media which consisted of polyurethane foam (PUF), followed by the high volume pump of a flow rate of 850 L/min (Aurell & Gullett, 2013). Quartz filters, PUF and a sorbent (XAD-2) were used to collect PAHs, with a high volume pump of a flow rate of 160 L/min (Aurell &

Gullett, 2010). PM impact samplers and the high volume blower quartz filters were replaced when pumps could no longer compensate and the high volume blower showed reduced flow rate, respectively. PCDDs/PCDFs and PAHs were analyzed utilizing High Resolution Gas Chromatography/High Resolution Mass Spectrometry (HRGC/HRMS) (US EPA, 1999d) and Low Resolution Gas Chromatography/Low Resolution Mass Spectrometry (LRGC/LRMS), respectively (Aurell et al., 2015; US EPA, 1999b). CO₂ and CO were continuously measured by the use of a LICOR 820 (Biosciences, USA), and a benchtop CO monitor (Analytical Instruments Model 200, USA) as well as an e2V EC4-500-CO sensor (SGX Sensortech, United Kingdom), respectively. Pre-calibration as well as drift check of the CO₂ and CO sensors were conducted daily in accordance with US EPA Method 3A (US EPA, 2014); calibration was conducted for all other equipment in accordance with manufacturer recommendations. A DustTrak DRX 8533 (TSI Inc., USA) was used for continuous measurements of PM₁, PM_{2.5}, PM₄, PM₁₀ and Total PM (up to 15 µm). A custom correction factor was applied, as per manufacturer's recommendation, to the DustTrak-derived PM concentrations by using simultaneously sampled PM_{2.5}, PM₁₀ concentrations (i.e., average continuous PM_{2.5} or PM₁₀ concentrations were divided by the PM_{2.5} or PM₁₀ filter impactor concentrations during the same time interval).

Test Samples.

Fiberboard test samples consisted of approximately 2.5 kg of material, which equated to two containers and liners for each fiberboard test material. Each MRE test sample consisted of six MREs, with all MREs consisting of the same entrée, plus half of a current fiberboard container and liner (see Table 13 for content). All MRE components were opened and randomly dispersed upon the OBTF test area, while 25% of all food content was included to simulate

uneaten food and to be consistent with US Central Command guidance of wet waste not exceeding a quarter of the waste input (USCENTCOM, 2009). All tests were conducted in a random order and a summary of the testing is shown in Table 2.

Table 2. Test Matrix

	Test Material ^a	# of Tests	Test Order
Fiberboards	NA: OBTF Blank	1	Pre-Test
	SFP	4	3,5,7,8
	SFNP	4	1,2,12,13
	CFP	4	6,9,10,11
	CFNP	4	4,14,15,16
	NA: OBTF Blank	1	Post-Test
MREs	NA: SFP (Baseline)	1	6
	MRE-0%	3	2,3,5
	MRE-32%	3	7,8,10
	MRE-66%	3	11,12,13
	MRE-100%	3	1,4,9
	NA: OBTF Blank	1	Post-Test

^aNA – Not Applicable; SFP – Solid Fiberboard Container & Liner with Wet Strength Polymer; SFNP – Solid Fiberboard Container & Liner, No Wet Strength Polymer; CFP – Corrugated Fiberboard Container & Liner with Wet Strength Polymer; CFNP – Corrugated Fiberboard Container & Liner, No Wet Strength Polymer

The MRE flameless ration heaters (FRHs) were excluded from the experiment design due to their volatility and potential for injury (Patel, 2002). The FRHs inclusion most likely would have resulted in the premature initiation of the FRH corrosion process due to the moisture content of the food. It is speculated that had the FRHs been included in the test samples, the corrosion process would have contributed a significant increase in heat and hydrogen available for combustion, as well as increasing the magnesium and iron particulates found within the PM_{2.5}

samples (Barrett & Cardello, 2012; Defense Logistics Agency, 2003). Whether or not the analytical results would have been discernibly different could be the focus of future research.

Burns.

Tests were initiated by weighing each test sample, arbitrarily dumping the test sample on the OBTF burn pan, and igniting the sample with a propane torch (see Figure 10). In order to simulate open burning conditions, the oxygen concentration was maintained at atmospheric levels by ventilating with approximately 40 air changes per hour through an exhaust duct. The number of tests per material was dictated by the amount of sample mass required to exceed detection limits of the most trace target compounds, typically PCDDs/PCDFs. This amount was estimated by the mass of carbon collected, as CO₂, as a surrogate for the amount of combustion products sampled. Data collection began once the test material was ignited. SUMMA[®] canisters were started and stopped at a sufficiently high CO₂ concentration of 1200 ppm in order to avoid diluting samples with non-detectable levels of the target compounds. When the burns appeared to be smoldering and CO₂ levels approximated ambient levels of 450±50 ppm, all sampling equipment was remotely stopped. Sampling times were approximately 15 minutes for each burn.

Calculations.

Modified combustion efficiencies (MCE), an indicator of the effectiveness of a combustion process (Hardy, Conard, Regelbrugge, & Teesdale, 1996; Reid, Koppmann, Eck, & Eleuterio, 2005; Ward & Hardy, 1991), were calculated for each sample collected; 1) CO₂/(CO₂+CO+CH₄) for VOCs and 2) CO₂/(CO₂+CO) for all other pollutants (Urbanski, 2013). Additionally, PCDD/PCDF and PAH toxicity equivalent (TEQ) values were calculated using the World Health Organization 2005 toxic equivalent factors (TEFs) (Van den Berg et al., 2006) and

TEFs relative to benzo-[a]-pyrene TEQ (J.C. Larsen, P.B. Larsen, Chemical carcinogens, in: Harrison & Hester, 1998), respectively.

Emission factors (EF) were calculated utilizing the carbon balance method (Laursen, Ferek, Hobbs, & Rasmussen, 1992), which assumes that all carbon in the fiberboard or MRE are emitted as CO₂, CO, CH₄, and total hydrocarbons and are thoroughly mixed with the pollutants in the smoke plume (Urbanski, 2013). The emission factor for a pollutant, $EF_{pollutant}$, could therefore be determined by:

$$EF_{pollutant} = \frac{m_f * m_{pollutant}}{m_{Total Carbon}} \quad (1)$$

Where:

m_f = mass fraction of carbon in the respective fiberboard or MRE

$m_{pollutant}$ = above-ambient mass concentration of the pollutant of interest (typically mg pollutant m⁻³)

$m_{Total Carbon}$ = above-ambient mass concentration of carbon (typically mg carbon m⁻³) determined by the measurement of major carbon-containing combustion products (Laursen et al., 1992)

Since CO₂, CO, and specifically CH₄ could be accounted for by SUMMA[®] canister for VOCs, all three were utilized in calculating $m_{Total Carbon}$ for VOC emission factors while CO₂ and CO were used for all other pollutants (Laursen et al., 1992). The emission factors can therefore be expressed as mass of pollutant per mass of the fiberboard or MRE burned, as appropriate, hereby referred to collectively as “waste” (e.g., mg pollutant per kg waste). The m_f for the fiberboards were obtained from ultimate proximate analyses and were determined to be 0.46, 0.47, 0.45, and 0.46 for SFP, SFNP, CFP and CFNP, respectively (see Table 12 for the full ultimate proximate analysis). Mass fractions of the carbon for the various MRE test materials were derived by the use of literature values (Liu & Lipták, 1999), the list of contents on the food pouches, and the fraction of each component within the MREs and were determined to be 0.42, 0.44, 0.43, and

0.46 for MRE-0%, MRE-32%, MRE-66%, and MRE-100%, respectively (see Table 13 and Table 14).

Statistical Analyses.

With emission factors calculated, statistical analyses were accomplished with the data utilizing the statistical software JMP[®] 12 (SAS Inc., USA). A confidence level of 95% (overall $\alpha=0.05$) was determined to be an acceptable threshold for statistical significance prior to conducting the experiments. Pollutants that did not have greater than 75% of their samples available, had emission factor values less than three times the limits of detection (as a conservative approach to identify emissions above analytical noise) (Armbruster & Pry, 2008), or did not have a corresponding pollutant to compare to (e.g., metals were analyzed for MREs but not for fiberboards) were excluded from statistical testing. Numerous multiway Analysis of Variance (ANOVA) tests were conducted to screen for significant differences in emissions factors by material type and MCE. If the ANOVA test indicated a possible statistical distinction, verification of the assumptions required for an ANOVA were conducted to include: (1) histograms of the studentized residuals to identify outliers; (2) overlay plots of Cook's distances to identify highly influential data points; (3) Durbin-Watson tests for independence; (4) Shapiro-Wilk tests for normality; and, (5) Breusch-Pagan tests for constant variance. Data points deemed highly influential based upon Cook's distances were excluded and ANOVA tests were re-conducted without those data points to validate statistical significance. If the assumptions of the ANOVA test held true, then a Tukey honest significant difference test was conducted to specifically identify which, if any, of the test materials were significantly different from one another.

In order to compare between the MREs and the fiberboards as a whole, a non-parametric Kruskal-Wallis (K-W) rank sums test was performed on the aggregate data for each pollutant in order to account for unequal sample sizes. Additionally, a time-series plot of the emissions factors in sequential burn order was utilized to verify the absence of autocorrelation, which is required for the Kruskal-Wallis test.

Results and Discussion

Two hundred and fourteen samples were collected in total for PM_{2.5}, PM₁₀, PAHs, VOCs, and PCDD/PCDF. Results are presented by pollutant class, followed by the comparison of MRE/fiberboard emission factors and modified combustion efficiencies to each other and the literature.

PM and Metals.

After removing highly influential data points, Tukey tests amongst the fiberboards indicated a statistical difference between SFNP versus CFNP and CFP, as well as between SFP and CFP (p-value=0.0001) for PM₁₀. The same can be said for PM_{2.5}, as SFNP was found to be statistically different from CFP (p-value=0.0109) when highly influential points were excluded. As Figure 2 indicates, however, when including all emission factors regardless of influence, the variability amongst the fiberboards prevents a distinction amongst the fiberboards from occurring. This would indicate that, from a practical perspective, the fiberboard types led to similar particulate matter emissions regardless of composition.

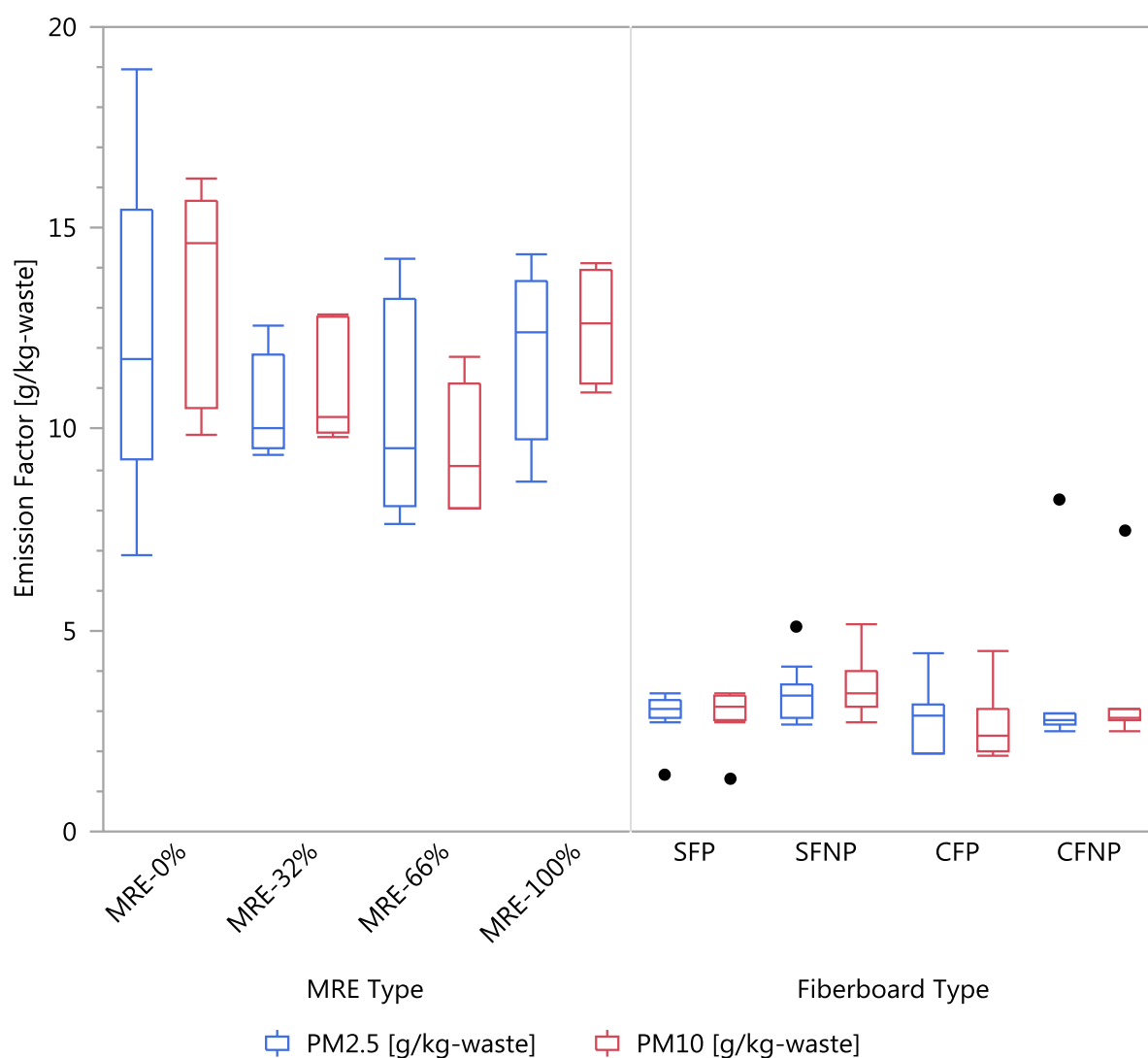


Figure 2. PM_{2.5} and PM₁₀ emission factors from open burning of four MRE test materials (MRE-0% = all current, aluminum-based pouches and 100% = all nanocomposite-based pouches) and four different fiberboard types

PM_{2.5} emissions factors for the various MRE types were not determined to be statistically different (p-value=0.4073) while a statistical distinction was found for MRE-0% and MRE-100% versus MRE-66% (p-value=0.0051) for PM₁₀. However, this statistical distinction is believed to be due to the low variability of MRE-100% emissions, the relatively low PM₁₀ emissions from

MRE-66%, as well as the high variability of MRE-0% emissions. Therefore, as Figure 2 illustrates and not unlike the fiberboards, particulate matter emissions for the MREs were essentially equivalent regardless of composition. Plots for PAHs, VOCs, and PCDD/PCDF drew similar conclusions and were, therefore, excluded.

When comparing the PM emissions for the MREs against the fiberboards, there is a significant difference: the MREs emitted up to five times greater PM emissions than the fiberboards (K-W p-value=0.0001) (see Figure 2 and Table 3). Additionally, PM₁₀ and PM_{2.5} emission factors were also closely correlated with one another across all material types, indicating that a majority of particulate emissions were 2.5 µm in size or smaller, regardless of material.

MRE PM_{2.5} emission factors (12±3 g/kg waste) were comparable to previous studies burning simulated military waste: small-scale open burns (20±6.4 g/kg waste), burn piles (39±24 g/kg waste), and an air curtain burner (12±12 g/kg waste) specifically, thereby suggesting that the combustion technique is more important than the difference in military waste composition on the emissions (Aurell et al., 2012; Woodall et al., 2012). A recent small scale gasification study for waste-to-energy purposes that burned military waste, including MREs (Barnes et al., 2016), showed significantly lower PM emission factors (0.39±0.22 g/kg waste) than found in this study, further suggesting that the combustion technique may be more important than different material types.

Despite varying the presence of aluminum-based pouches for the MRE types, metal emissions were found to be relatively equivalent amongst all MREs for all metals with values greater than the limits of detection. Evidence suggests that the heat of combustion did not reach levels great enough to volatilize significant quantities of metal for a long enough period of time

to distinguish between the MRE types. Further comparison and discussion of the emission factors for MRE metals can be found within the Appendix A: Scholarly Article Supplemental Information.

Table 3. Select Average Emission Factors ± 1 Standard Deviation for PM_{2.5}, PM₁₀, VOCs, and PCDD/PCDF^a

Pollutant	Unit	SFP	SFNP	CFP	CFNP	MRE-0%	MRE-32%	MRE-66%	MRE-100%
PM									
PM _{2.5}	g/kg Waste	2.9 $\pm$ 0.63	3.4 $\pm$ 2.0	2.8 $\pm$ 0.86	3.4 $\pm$ 0.72	12.3 $\pm$ 3.3	10.5 $\pm$ 0.7	10.4 $\pm$ 1.8	11.9 $\pm$ 0.7
PM ₁₀	g/kg Waste	2.9 $\pm$ 0.70	3.4 $\pm$ 1.7	2.7 $\pm$ 0.89	3.6 $\pm$ 0.72	13.6 $\pm$ 1.8	11.0 $\pm$ 0.9	9.5 $\pm$ 1.3	12.6 $\pm$ 0.1
VOCs									
Benzene ^{(1),b}	mg/kg Waste	79 $\pm$ 75	111 $\pm$ 16	153 $\pm$ 102	91 $\pm$ 49	211 $\pm$ 89	266 $\pm$ 10	182 $\pm$ 12 ^c	225 $\pm$ 36
1,3-Butadiene ^{(1),b}	mg/kg Waste	19 $\pm$ 22	24 $\pm$ 5	44 $\pm$ 43	34 $\pm$ 23	103 $\pm$ 38	126 $\pm$ 12	82 $\pm$ 2.3 ^c	94 $\pm$ 55
Styrene ^{(2B),b}	mg/kg Waste	5.6 $\pm$ 2.5	13 $\pm$ 1.6	60 $\pm$ 64	13 $\pm$ 7.5	206 $\pm$ 192	283 $\pm$ 198	210 $\pm$ 22 ^c	120 $\pm$ 79
Ethylbenzene ^{(2B),b}	mg/kg Waste	2.4 $\pm$ 2.2	4.1 $\pm$ 0.60	15 $\pm$ 13	4.6 $\pm$ 2.9	16 $\pm$ 12	24 $\pm$ 14	18 $\pm$ 0.33 ^c	15 $\pm$ 6.0
Toluene ^{(3),b}	mg/kg Waste	19 $\pm$ 19	26 $\pm$ 4.6	57 $\pm$ 44	28 $\pm$ 16	63 $\pm$ 36	79 $\pm$ 37	52 $\pm$ 11 ^c	58 $\pm$ 18
Acrolein ^{(3),b}	mg/kg Waste	9.4 $\pm$ 4.8	36 $\pm$ 5.9	81 $\pm$ 64	76 $\pm$ 60	125 $\pm$ 40	145 $\pm$ 16	126 $\pm$ 12 ^c	126 $\pm$ 27
PCDD/PCDF									
2,3,7,8-TCDD ^{(1),b}	ng TEQ/ kg Waste	0.003 $\pm$ 0.001 ^c	0.006 $\pm$ 0.002 ^c	0.028 $\pm$ 0.018	0.013 $\pm$ 0.002 ^c	0.293 $\pm$ 0.236	0.064 $\pm$ 0.035 ^c	0.033 $\pm$ 0.021 ^c	0.065 $\pm$ 0.042
2,3,4,7,8-PeCDF ^{(1),b}	ng TEQ/ kg Waste	0.009 $\pm$ 0.003	0.020 $\pm$ 0.013	0.026 $\pm$ 0.003	0.010 $\pm$ 0.001 ^c	0.479 $\pm$ 0.293	0.265 $\pm$ 0.074	0.075 $\pm$ 0.047	0.250 $\pm$ 0.168
Σ PCDD/ PCDF TEQ Total	ng TEQ/ kg Waste	0.046 $\pm$ 0.016	0.075 $\pm$ 0.041	0.28 $\pm$ 0.14	0.10 $\pm$ 0.055	1.802 $\pm$ 1.040	0.904 $\pm$ 0.291	0.301 $\pm$ 0.145	0.877 $\pm$ 0.527

^aFull lists can be found within SI. ^bHazardous Air Pollutants in accordance with the Clean Air Act (US EPA, 2008), 2,3,4,7,8-PeCDF specifically as a dibenzofuran. ^cRelative difference. ^(#)International Agency for Research on Cancer classification (WHO - IARC, 2015).

Considerable MCE ranges for the MREs (0.912-0.963) and fiberboards (0.866-0.944) are believed to be attributed to material composition and the orientation of those materials when burned (Gullett, Lemieux, Lutes, Winterrowd, & Winters, 2001). These MCE ranges were found to be comparable to Aurell et al. (Aurell et al., 2012) for combustion via open burning and air curtain burners, but were much lower than the Barnes et al. (Barnes et al., 2016) MCEs for gasification via a prototype waste-to-energy unit (0.999).

PAHs.

No statistical differences were observed in fiberboard emissions factors for the sum 16-EPA PAHs among the various types (p-value=0.0854) with SFNP showing a higher mean

emission factor. MRE emissions were equivalent amongst the various material types as well when highly influential points were excluded, with the exception of MRE-0% being statistically different from MRE-32% and MRE-66% ($p\text{-value}=0.0258$) with high variability preventing further distinction. Collectively, emission factors for the PAHs were essentially equivalent amongst the MREs, as well as for the fiberboards, respectively.

For both the MREs and fiberboard types, naphthalene was the greatest pollutant of the PAHs with at least twice the emission factor than the second greatest constituent, which was phenanthrene for MREs and acenaphthylene for the fiberboards (see Table 4). Both SFP and CFP naphthalene emissions showed the greatest distinction from the MREs, with means of 7 ± 3 mg/kg-waste and 15 ± 7 mg/kg-waste, respectively. When naphthalene emission factors are compared for the greatest MRE emitter, MRE-100% (65 ± 39 mg/kg-waste), against the smallest fiberboard emitter, SFP (7 ± 3 mg/kg-waste), the nanocomposite MRE emission factor surpasses the solid fiberboard with the wet strength polymer emission factor by at least two and a half times.

Table 4. SUM 16-EPA PAH Emission Factors for All Material Types in mg/kg-waste

Compound	SFP	SFNP	CFP	CFNP	MRE-0%	MRE-32%	MRE-66%	MRE-100%
Naphthalene ^{(2B)*}	7.1±2.5	28±21	15±6.7	21±9.9	46±15	40±8.7	31±3.3	65±39
Acenaphthylene	3.3±1.4	13±9.3	4.8±2.2	6.9±2.7	15±5.7	12±2.1	11±1.3	19±9.8
Acenaphthene ⁽³⁾	0.17±0.090	0.68±0.45	0.35±0.20	0.43±0.19	0.69±0.25	0.58±0.14	0.57±0.011	0.87±0.57
Fluorene ⁽³⁾	0.74±0.32	3.0±1.6	1.8±0.79	1.4±0.56	4.5±1.9	3.2±0.47	3.5±0.68	4.8±2.6
Phenanthrene ⁽³⁾	3.0±0.90	10±4.7	6.8±3.0	4.4±1.3	16±7.0	11±2.0	14±3.3	16±7.3
Anthracene ⁽³⁾	0.61±0.21	2.4±1.3	1.7±0.87	0.99±0.33	2.5±1.0	2.0±0.38	2.3±0.36	2.9±1.5
Fluoranthene ⁽³⁾	1.6±0.57	4.9±2.7	2.7±1.2	1.8±0.29	3.2±1.4	2.4±0.37	2.9±0.029	3.7±1.4
Pyrene ⁽³⁾	1.6±0.58	4.5±2.5	2.4±1.1	1.7±0.24	2.2±1.0	1.9±0.39	2.1±0.18	2.9±1.3
Benzo(a)anthracene ^(2B)	0.32±0.13	1.2±0.65	0.75±0.35	0.44±0.13	1.3±0.65	0.82±0.20	1.1±0.023	1.4±0.69
Chrysene ^(2B)	0.31±0.11	1.1±0.57	0.72±0.34	0.41±0.11	1.5±0.79	0.99±0.22	1.4±0.021	1.5±0.75
Benzo(b)fluoranthene ^(2B)	0.18±0.068	0.62±0.32	0.33±0.16	0.23±0.068	0.91±0.45	0.57±0.14	0.77±0.016	0.85±0.45
Benzo(k)fluoranthene ^(2B)	0.23±0.085	0.91±0.45	0.40±0.17	0.27±0.055	0.83±0.47	0.60±0.11	0.72±0.10	0.89±0.36
Benzo(a)pyrene ⁽¹⁾	0.25±0.095	1.0±0.55	0.46±0.22	0.31±0.067	0.79±0.43	0.56±0.14	0.65±0.079	0.86±0.44
Indeno(1,2,3-cd)pyrene ^(2B)	0.16±0.051	0.66±0.33	0.28±0.13	0.20±0.035	0.57±0.30	0.36±0.088	0.43±0.072	0.59±0.27
Dibenz(a,h)anthracene ^(2A)	0.029±0.012	0.11±0.067	0.067±0.038	0.037±0.011	0.21±0.11	0.12±0.031	0.14±0.0048	0.19±0.10
Benzo(ghi)perylene ⁽³⁾	0.19±0.067	0.73±0.37	0.27±0.11	0.21±0.035	0.50±0.25	0.37±0.10	0.37±0.11	0.58±0.27
SUM 16-EPA PAHs	20±7.2	74±40	38±16	41±16	96±37	77±15	73±1.8	122±67
SUM 16-EPA PAHs TEQ^a	0.54±0.31	1.6±0.82	0.77±0.38	0.51±0.12	1.4±0.76	0.98±0.22	1.2±0.093	1.5±0.75

^(#)International Agency for Research on Cancer classification (WHO - IARC, 2015). ^{*}Hazardous Air Pollutant in accordance with the Clean Air Act (US EPA, 2008). ^amg B[a]P TEQ/kg-waste

When comparing the emission factors for each PAH pollutant between the MREs and the fiberboards as two distinct groups, all MRE group means were statistically greater than the fiberboard group means with the exception of fluoranthene (K-W p-value=0.0710), which were equivalent, and pyrene (K-W p-value=0.3291) where the fiberboard group mean was greater than the MRE group mean.

A comparison of the sum 16-EPA PAH emission factors between this study and the Aurell et al. (Aurell et al., 2012) air curtain burner (43 ± 50 mg/kg-waste) and burn pile (129 ± 50 mg/kg-waste) study reveals that the mean MRE PAH emission factors (96-122 mg/kg-waste) were similar, while the fiberboards, specifically SFP (20 ± 7 mg/kg-waste), were distinctly lower (Aurell et al., 2012). When compared to the Barnes et al. (Barnes et al., 2016) gasification study, the fiberboards emitted comparable PAH emissions while the MRE emissions were nearly three to six times greater. It is speculated that this was most likely due to combustion conditions from the open burns with lower MCEs ranging from 0.941 to 0.986 when compared to the more complete combustion occurring during the gasification process (MCE=0.999). The Woodall et al. (Woodall et al., 2012) simulated military waste study yielded significantly higher Sum 16-EPA PAH emissions (376 ± 108 mg/kg-waste) when compared to Aurell et al. (Aurell et al., 2012), Barnes et al. (Barnes et al., 2016), and this study. MRE-0% emissions (96 ± 37 mg/kg-waste) were half that of the Woodall et al. (Woodall et al., 2012) study and CFP emissions (38 ± 16 mg/kg-waste) nearly an order of magnitude smaller. It is believed that this difference can be attributed to the percent of food content remaining within their samples (50%), as well as the smoldering conditions that the high food content percentage may have caused (Woodall et al., 2012).

PCDD/PCDFs.

PCDD/PCDF TEQ total emission factors for CFP (0.283 ± 0.136 ng-TEQ/kg-waste) were three times higher than CFNP (0.102 ± 0.055 ng-TEQ/kg-waste) and six times higher than SFP (0.046 ± 0.016 ng-TEQ/kg-waste). Although the difference is slight, it is speculated that the polymer that gives wet-strength found upon CFP has a larger chlorine content and is potentially increasing PCDD/PCDF emissions. Additionally, as indicated by the MCEs for MREs (0.920-0.957) and fiberboards (0.969-0.989), all fiberboard types burned with greater combustion efficiency than all of the MRE types. It is believed that the moisture content of the food present within the MRE samples hindered the combustion process and allowed greater PCDD/PCDF emissions to be emitted from the MRE waste versus the fiberboard types. The mean PCDD/PCDF TEQ total emission factor for MRE-66% (0.301 ± 0.145 ng-TEQ/kg-waste) was determined to be three times less than MRE-32% (0.904 ± 0.291 ng-TEQ/kg-waste) and MRE-100% (0.877 ± 0.527 ng-TEQ/kg-waste), and six times less than MRE-0% (1.802 ± 1.040 ng-TEQ/kg-waste). However, no statistical distinction was found for PCDD/PCDF TEQ total emissions amongst the various MRE and fiberboard types.

When comparing the MREs and fiberboards with one another as two distinct groups, a majority of the PCDD/PCDF congeners were statistically greater for the MREs versus the fiberboards with the following exceptions: 1,2,3,6,7,8-HxCDD (K-W p-value=0.3001); 1,2,3,7,8,9-HxCDD (K-W p-value=0.1143); 1,2,3,4,6,7,8-HpCDD (K-W p-value=0.4576); and 1,2,3,4,6,7,8,9-OCDD (K-W p-value=0.5462). The PCDD/PCDF TEQ total emission factor for both the MREs and the fiberboards were exceptionally small, especially when compared to significantly higher PCDD/PCDF emissions seen by the Barnes et al. (Barnes et al., 2016) gasification system (3.6 ± 2.4 ng-TEQ/kg-waste), the Aurell et al. (Aurell et al., 2012) air curtain

burner (35 ± 24 ng-TEQ/kg-waste) and burn pile ($1,765 \pm 1,474$ ng-TEQ/kg-waste), as well as the Woodall et al. (Woodall et al., 2012) OBTF burn pile (127 ± 136 ng-TEQ/kg-waste). Although samples were certainly different, these large differences in PCDD/PCDF TEQ total emission factors when compared to other simulated military waste burns would suggest that dioxin and furan formation is not primarily associated with MREs or their fiberboard packaging and may be more closely related to the combustion conditions, as well as other components, of military waste.

VOCs.

MRE VOC emissions were generally one to seven times greater in magnitude than fiberboard VOC emissions, with the greatest difference seen with propene. With the exception of 2-butanone (K-W p-value=0.0608), group mean VOC emission factors for MREs were statistically greater than group fiberboard emissions for pollutants that met the statistical criteria outlined within the methodology. Emission factors for 1,3-butadiene amongst the fiberboards were not statistically different from one another (p-value=0.5752) while benzene was determined to be statistically different for the corrugated fiberboard with the wet strength polymer (CFP) and the solid fiberboard without the wet strength polymer (SFNP) versus the corrugated fiberboard without the wet strength polymer (CFNP) (p-value=0.0001).

One of the greatest overall pollutants emitted was benzene, with similar emissions among all of the MRE types (see Table 3 and Table 5), with the exception of MRE-32% (266 ± 10 mg/kg-waste) and MRE-0% (211 ± 89 mg/kg-waste) showing statistically different emissions (p-value=0.0215) when a highly influential point was excluded from analysis. Overall, no practical difference in VOC emission factors was found amongst the MRE types and fiberboard types, respectively.

When compared to other studies with good combustion efficiency ($MCE > 0.95$), the Aurell et al. (Aurell et al., 2012) air curtain burner (243 ± 299 mg/kg-waste) and burn pile (260 ± 288 mg/kg-waste) benzene emissions were analogous to the MREs; however, in the Barnes et al. (Barnes et al., 2016) gasification study, the benzene emissions (43 ± 98 mg/kg-waste) were more similar to the fiberboard emissions. When comparing the MRE and fiberboard benzene emissions to poor combustion ($MCE < 0.90$) conditions, the Aurell et al. (Aurell et al., 2012) air curtain burner ($1,371 \pm 185$ mg/kg-waste) and burn pile ($2,421 \pm 1,265$ mg/kg-waste) emissions were greater by an order of magnitude, thereby reinforcing the importance of burn conditions over distinct material differences. Furthermore, the OBTF burn pile benzene emissions (940 ± 220 mg/kg-waste) tested by Woodall et al. (Woodall et al., 2012) were found to be nearly five times larger than MRE emissions and an order of magnitude larger than fiberboard emissions as well, most likely due to the larger amount of food waste present within their study, which was approximately 50% of total food waste available, compared to 25% in this study.

Reevaluated in 2012 and classified by the WHO IARC as a “carcinogenic to humans” substance (WHO - IARC, 2015), 1,3-butadiene mean emission levels for SFP (19 ± 22 mg/kg-waste) were found to be the lowest when compared to all other material types, to include the Aurell et al. (Aurell et al., 2012) air curtain burner (67 ± 95 mg/kg-waste) and burn pile (93 ± 150 mg/kg-waste) emissions and the Woodall et al. (Woodall et al., 2012) OBTF burn pile (254 ± 52 mg/kg-waste), under good burning conditions ($MCE > 0.95$). However, no statistical significance was determined between the fiberboards ($p\text{-value} = 0.5752$) or MREs ($p\text{-value} = 0.5881$) for 1,3-butadiene.

MCEs for the fiberboard types (0.966 to 0.987) and MRE variations (0.974 to 0.983) appeared to be linearly correlated to VOC emission factors and is a relationship previously

explored within the literature (Hardy et al., 1996; Urbanski, 2013; Ward & Hardy, 1991). It should be noted that this MCE relationship was much less robust for each of the other pollutant classes. Figure 3 depicts normalized emission factors as calculated percentages of the maximum emission factor for each of five IARC-classified (WHO - IARC, 2015), hazardous air pollutants (US EPA, 2008).

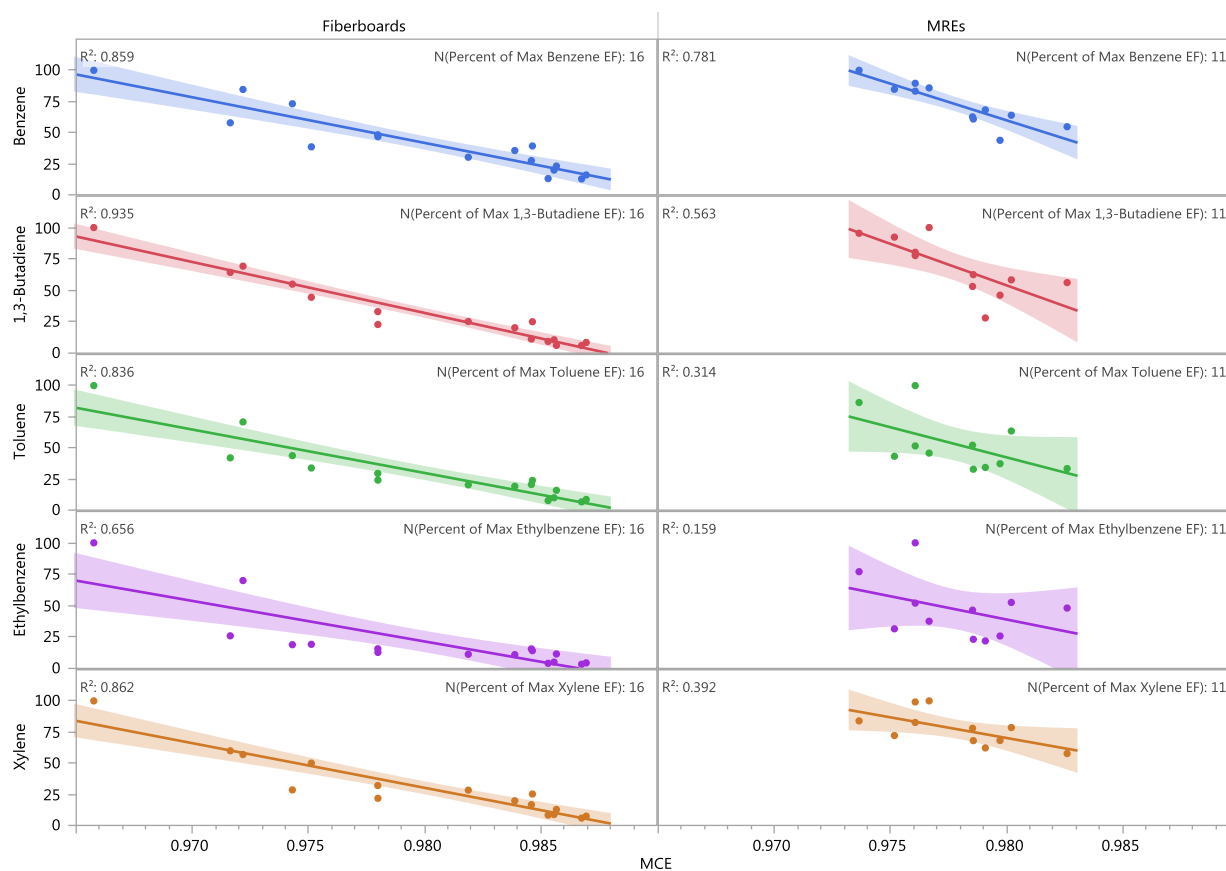


Figure 3. Percent of Maximum VOC Emission Factors vs MCE for Fiberboards and MREs with 95% Confidence Intervals

Coefficient of determination values for a linear correlation were generally greater for the fiberboards than for the MREs and is thought to be associated with the homogeneity of test material amongst the fiberboards versus the relatively heterogeneous nature of an MRE and its food contents. Interestingly, the fiberboards burned with slightly greater variance in the

combustion efficiency (~0.014 greater MCE range) when compared to the MREs and is believed to be attributed to minor deviation in the amount of carbon content collected.

Table 5 provides a synopsis of the simulated military waste studies discussed and provides perspective as to where select MRE and fiberboard emissions can be placed within the literature. The data suggest that MREs account for 70 to 90 percent of PM emissions when burned in unison with the current fiberboard container and liner, and that particulate matter equal to or less than 2.5 μm in size constitutes a majority of the particulates emitted from both MREs and their fiberboard packaging. The MREs emitted up to five times greater PM emissions than the fiberboards and average MRE emission factors were comparable to the Aurell et al air curtain burner (Aurell et al., 2012) and the Woodall et al. OBTF burn pile (Woodall et al., 2012) emission factors. PAH emission factors were similar to the Aurell et al. study (Aurell et al., 2012), greater than the Barnes et al. study (Barnes et al., 2016), and nearly four times less than the Woodall et al. study (Woodall et al., 2012). Due to the higher emission factors, the targeted replacement of MRE constituents may be more effective than the variation of fiberboard packaging designs to reduce PM, PAH, and VOC emissions. PCDD/PCDF emissions were minimal for both the MREs and the fiberboards; were by far the smallest emission factors compared to similar studies (Aurell et al., 2012; Barnes et al., 2016; Woodall et al., 2012), believed to be due to material composition and burn conditions; and appear to be more closely related to other components of military waste. Additionally, PCDD/PCDF formation is highly dependent upon chlorine availability, which is believed to be minimal for both MREs and their fiberboard packaging based upon the emission factors determined from this study. Overall, variations of the MRE and fiberboard types yielded minimal differences in PM, PAH, VOC, and PCDD/PCDF emissions generated.

Table 5. Select Emission Factor Comparisons for Simulated Military Waste Burns^a

Compound	Unit	Gasification System* (Barnes et al., 2016)	Air Curtain Burner (Aurell et al., 2012)	Burn Pile (Aurell et al., 2012)	OBTF Burn Pile ^f (Woodall et al., 2012)	MRE-0%	MRE-100%	SFP	CFP
ΣPCDD/PCDF TEQ	ng TEQ/kg waste	3.6±2.4	35±24	1,765±1,474	127±136	1.802±1.040	0.877±0.527	0.046±0.016	0.283±0.136
ΣPAH ^b	mg/kg waste	12±10	43±50	129±50	376±108	96±37	122±67	20±7	38±16
PM _{2.5}	g/kg waste	0.39±0.22 ^c	12±12	39±24	20±6.4	12±3	12±1	3±1	3±1
Iron	mg/kg waste	0.31±0.18	0.50±0.24	11±23	NA	0.29±0.05	0.30±0.03	NA	NA
Copper	mg/kg waste	1.2±0.33	0.18±0.11	0.89±0.92	NA	0.17±0.06	0.21±0.08	NA	NA
Cadmium	mg/kg waste	0.0058±0.0031	0.063±0.082	0.073±0.033	NA	0.089±0.038	0.11	NA	NA
Lead	mg/kg waste	0.66±0.35	0.55±0.42	0.37±0.22	NA	0.07±0.01	0.07±0.05	NA	NA
Benzene	mg/kg waste	43±98 ^d	243±299 ^d / 1,371±185 ^e	260±288 ^d / 2,421±1,265 ^e	940±220	211±89 ^d	225±36 ^d	79±75 ^d	153±102 ^d
1,3-Butadiene	mg/kg waste	ND	67±95 ^d / 375±31 ^e	93±150 ^d / 746±228 ^e	254±52	103±38 ^d	94±55 ^d	19±22 ^d	44±43 ^d
Styrene	mg/kg waste	0.019	256±382 ^d / 2203±634 ^e	284±463 ^d / 5099±3308 ^e	1880±230	206±192 ^d	120±79 ^d	6±3 ^d	60±64 ^d
Toluene	mg/kg waste	2.0±2.8 ^d	88±130 ^d / 652±111 ^e	109±170 ^d / 1,202±727 ^e	404±56	63±36 ^d	58±18 ^d	19±19 ^d	57±44 ^d
Acrolein	mg/kg waste	0.58±0.95 ^d	133±139 ^d / 463±33 ^e	98±108 ^d / 757±62 ^e	564±169	125±40 ^d	126±27 ^d	9±5 ^d	81±64 ^d
Vinyl chloride	mg/kg waste	0.25±0.32 ^d	3.7±2.5 ^d / 13 ^e	6.0±5.5 ^d / 26±3.3 ^e	7.1±3.5	0.55±0.27 ^d	0.52±0.01 ^d	0.22	ND
Vinyl acetate	mg/kg waste	0.57±0.74 ^d	79±97 ^d / 324±46 ^e	43±53 ^d / 688±195 ^e	705±268	101±41 ^d	109±27 ^d	33±13 ^d	98±70 ^d

^aRange of data denoted ±1 standard deviation; where no range is stated, only one sample with detectable levels. ND = not detected. NA – not analyzed. * MAGS unit. ^bSUM 16 EPA PAHs ^cTotal PM. ^dMCE > 0.95 ^eMCE < 0.90. ^fThese data were obtained by using Woodall et al. (Woodall et al., 2012) emission factors in µg/kg waste and multiplying by a correction factor (0.47) derived from their waste composition utilizing carbon fractions from each material obtained from Liu and Lipták (Liu & Lipták, 1999).

Emission factors have now been established for MREs and their fiberboard packaging and can directly contribute to future military, food, and packaging waste studies. Characterized emission factors can also be utilized for predicting and modeling potential future health risks associated with exposure to the materials and pollutants described within this study. This study sought to characterize emissions from four MRE variations and four fiberboard packaging types to establish emission factors, conduct a comparison of these emission factors to each other, and establish where these emissions stand within the literature data.

Acknowledgements

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Supporting Information

Supporting information can be found within Appendix A: Scholarly Article Supplemental Information.

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References

The references utilized throughout the scholarly article have been incorporated into the thesis document and format and can be found in the References section.

III. Human Health Perspectives

Chapter Overview

The purpose of this chapter is to outline the criteria utilized for the determination of targeted pollutants within this research effort, to provide an overview of the toxic characteristics of the targeted pollutant classes, and to provide a discussion of how emission factors fit into the context of human and environmental health risk determination.

Pollutant Determination

The pollutants that were characterized throughout this research effort consisted of PM, metals, PAHs, VOCs, and PCDD/PCDFs. These pollutants were targeted for characterization based upon open burning situations encountered within the literature (Aurell et al., 2012; Lemieux et al., 2004; Woodall et al., 2012), to enhance guidance issued by USCENCOM (USCENCOM, 2009), and due to the inherent toxicity associated with these pollutants (US EPA, 2008; WHO - IARC, 2015). The World Health Organization's International Agency for Research on Cancer (IARC) has classified pollutants based upon their carcinogenicity potential within humans and Table 6 outlines the various groups (WHO - IARC, 2015).

Table 6. International Agency for Research on Cancer Classifications and Descriptions

IARC Group	# of Agents in Group	Description
1	118	Carcinogenic to Humans
2A	75	Probably Carcinogenic to Humans
2B	288	Possibly Carcinogenic to Humans
3	503	Not Classifiable as to its Carcinogenicity to Humans
4	1	Probably Not Carcinogenic to Humans

The Clean Air Act identified and established 189 (now 187) Hazardous Air Pollutants (HAPs) that are regulated by the EPA (US EPA, 2008). Hazardous Air Pollutants were implemented into law as regulations that are geared towards industries to monitor and control the emissions of

suspected cancer-causing agents and to prevent potential adverse health effects. HAP limits are enforced within the United States when certain quantity thresholds are exceeded. Both the IARC classifications and the CAA Hazardous Air Pollutants were utilized as criteria for pollutants to be targeted for sampling within this research endeavor and are described in greater detail within this chapter. Figure 4 provides a summary of the health-based criteria that were considered for each pollutant.

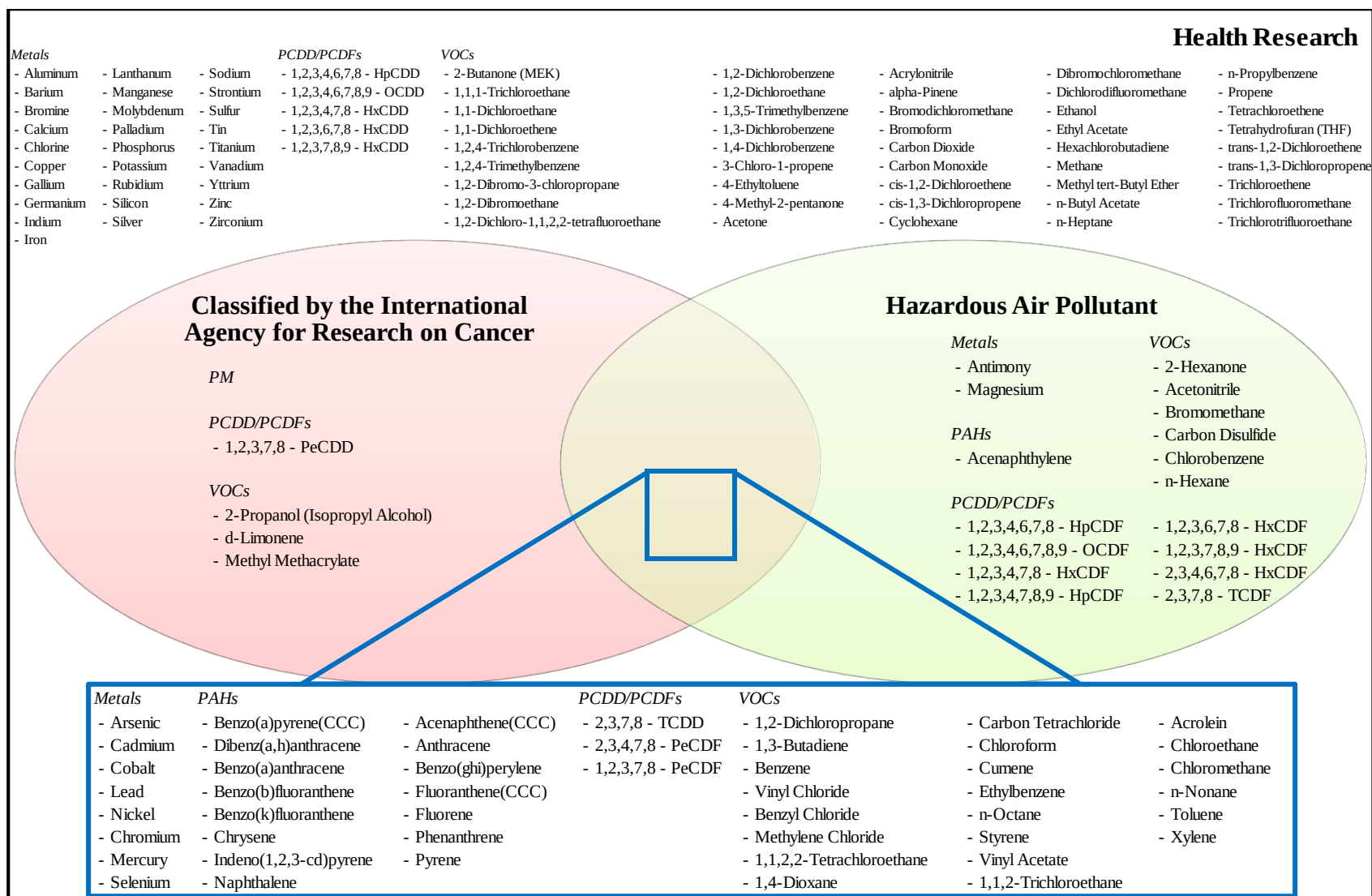


Figure 4. Health-based Criteria for Determination of Targeted Pollutants

Toxicity of Pollutants

The airborne emissions associated with these pollutants have reason to generate alarm. Particulate matter, in addition to carbon monoxide and lead, are considered criteria pollutants in accordance with the National Primary and Secondary Ambient Air Quality Standards. The Clean Air Act establishes the EPA as the publisher of these standards in accordance with 40 CFR, Subchapter C, Part 50, National Primary and Secondary Ambient Air Quality Standards, which defines primary standards as levels of air quality to protect the public health and secondary standards as levels with anticipated adverse effects from a pollutant, respectively (40 C.F.R. § 50, 2015). A comprehensive integrated science assessment into particulate matter by the EPA in 2009 synthesized toxicological and epidemiological evidence to suggest a causal relationship between long-term exposure to PM_{2.5} and cardiovascular effects, as well as a “likely to be causal” relationship for respiratory effects (US EPA, 2009). Additionally, particulate matter has been suggested to be associated with diseases such as lung cancer, reproductive and developmental effects, and mortality (Institute of Medicine, 2011; US EPA, 2009).

Polycyclic aromatic hydrocarbons are released into the atmosphere from both human-caused and natural sources and are commonly associated with the combustion of organic matter (Kim, Jahan, Kabir, & Brown, 2013). Incomplete combustion of waste, biomass, forest fires, and volcanic eruptions are all sources of PAH emissions and their long-term health effects are being studied extensively (Kim et al., 2013; Zhang & Tao, 2009). Table 7 provides a synopsis of the long-term health effects associated with PAHs from a 2011 Institute of Medicine study (Institute of Medicine, 2011); it should be noted

that a majority of these health effects were compiled from the EPA's Integrated Risk Information System (US EPA, 2015c), explained in further detail later within this Chapter.

Table 7. Summary of Long-term Health Effects Associated with PAHs, Reproduced from IOM Study (Institute of Medicine, 2011)

Chemical Name and CAS Number	Long-Term Health Effects ^a
Acenaphthene 83-32-9	Increased liver weight, increased cholesterol, vascular disorders, and degeneration in the internal organs and central nervous system
Acenaphthylene 208-96-8	NA
Anthracene 120-12-7	No observed effects at highest dose
Benz[a]anthracene 56-55-3	Probable carcinogen, ^b lung and liver cancers
Benzo[a]pyrene 50-32-8	Probable carcinogen, ^b stomach and respiratory tract tumors
Benzo[b]fluoranthene 205-99-2	Probable carcinogen, ^b lung and skin tumors, effects on liver
Benzo[e]pyrene 192-97-2	NA
Benzo[g,h,i]perylene 191-24-2	NA
Benzo[k]fluoranthene 207-08-9	Probable carcinogen, ^b lung and skin tumors
Chrysene 218-01-9	Probable carcinogen, ^b carcinomas and malignant lymphoma
Dibenz[a,h]anthracene 53-70-3	Probable carcinogen, ^b stomach and respiratory tract tumors
Fluoranthene 206-44-0	Nephropathy, increased liver weights, hematological alterations
Fluorene 86-73-7	Blood effects; increased liver, spleen, and kidney weights
Indeno[1,2,3-cd]pyrene 193-39-5	Probable carcinogen, ^b lung and skin tumors
Naphthalene 91-20-3	Possible carcinogen, ^b respiratory tumors; decreased body weights
Phenanthrene 85-01-8	NA
Pyrene 129-00-0	Nephropathy and decreased kidney weight

NA=Not Available

^aHealth effects by any route of exposure as described in EPA IRIS chemical profiles are presented unless otherwise noted in text (US EPA, 2015d); effects are based primarily on animal experiments.

^bCarcinogenicity determined by EPA IRIS as follows: Likely to be Carcinogenic to Humans : available tumor effects and other key data are adequate to demonstrate carcinogenic potential to humans, but does not reach the weight-of-evidence for the descriptor “carcinogenic to humans”; Suggestive Evidence of Carcinogenic Potential: evidence from human or animal data is suggestive of carcinogenicity, which raises a concern for carcinogenic effects but is judged not sufficient for a stronger conclusion; Inadequate Information to Assess Carcinogenic Potential: available data are judged inadequate to perform an assessment; Not Likely to be Carcinogenic to Humans: available data are considered robust for deciding that there is no basis for human hazard concern (US EPA, 2005a).

Metals such as zinc have been associated with lung inflammation and functional changes within the lungs (Chen & Lippmann, 2009; Institute of Medicine, 2011), especially when directly inhaling the emissions from zinc-containing items such as smoke bombs (Plum, Rink, & Hajo, 2010) . Lead exposure is known to be associated with loss in cognition for children (Schwartz, 1994) and is reasonably considered to be a human carcinogen (CDC, 2013; US EPA, 2015d). Antimony has been linked to heart, liver, and kidney damage in animals (CDC, 2013), as well as respiratory effects (Institute of Medicine, 2011).

As can be seen within Figure 4, the volatile organic compounds studied within this research effort occupy every niche of literature (Andersen & III, 1987; Astrand, Engstrom, & Ovrum, 1978; de Lacy Costello et al., 2014; Grosse et al., 2007; Kielhorn, Melber, Wahnschaffe, Aitio, & Mangelsdorf, 2000; Leibman, 1975; M. T. Smith, 2010; Stevens & Maier, 2008), as well as regulatory and health limits and recommendations (American Conference of Governmental Industrial Hygienists, 2015; CDC, 2013; US EPA, 2015d). A summary of the long-term health effects associated with some of the aforementioned VOCs can be found within Table 8.

Table 8. Long-term Health Effects Associated with Select Hazardous Air Pollutant VOCs, Adapted from IOM Study (Institute of Medicine, 2011; US EPA, 2015d)

IARC Class	Constituent	Long-term Health Effects ^a
1	1,3-Butadiene	Probable carcinogen, ^b liver, lung, ovary, mammary tumors; lymphohematopoietic cancers and leukemia; reproductive and developmental effects (e.g., ovarian and testicular atrophy, fetal skeletal variations, decreased fetal weight)
1	Benzene	Known carcinogen, ^b leukemia and hematological neoplasms; progressive deterioration of hematopoietic function with chronic exposure, suppression of circulating B-lymphocytes, menstrual disorders, limited evidence of reproductive toxicity and neurotoxicity
2A	Methylene Chloride	Probable carcinogen, ^b liver, mammary gland, salivary gland, lung tumors; liver toxicity (e.g., fatty changes)
2B	Ethylbenzene	Increased liver, kidney and spleen weights: developmental toxicity (e.g., skeletal variations)
2B	Styrene	Changes in red blood cells, reduced red blood cell counts and hemoglobin; increased liver weight, liver, kidney and stomach lesions; neurological effects (e.g., increased reaction time, decreased memory, concentration); possibly carcinogenic in humans (IARC Working Group on the Evaluation of Carcinogenic Risks to Humans, World Health Organization, & International Agency for Research on Cancer, 2002)
3	Acrolein	Respiratory and inflammatory responses, nasal lesions, increased heart and kidney weights, liver necrosis, decreased body weight gain
3	Chloromethane	Cerebellar lesions, central nervous system dysfunction
3	Toluene	Increased liver and kidney weight, nephropathy, neurological effects (e.g., vision impairment, increased performance time)
3	Xylene	Decreased body weight, increased mortality, eye and respiratory tract irritation, neurological effects (e.g., impaired learning and motor performance) (ATSDR, 2007; US EPA, 2015d)

^aHealth effects by any route of exposure as described in EPA IRIS chemical profiles are presented unless otherwise noted in text (US EPA, 2015d); effects are based primarily on animal experiments.

^bCarcinogenicity determined by EPA IRIS as follows: Likely to be Carcinogenic to Humans : available tumor effects and other key data are adequate to demonstrate carcinogenic potential to humans, but does not reach the weight-of-evidence for the descriptor “carcinogenic to humans”; Suggestive Evidence of Carcinogenic Potential: evidence from human or animal data is suggestive of carcinogenicity, which raises a concern for carcinogenic effects but is judged not sufficient for a stronger conclusion; Inadequate Information to Assess Carcinogenic Potential: available data are judged inadequate to perform an assessment; Not Likely to be Carcinogenic to Humans: available data are considered robust for deciding that there is no basis for human hazard concern (US EPA, 2005a).

Polychlorinated dibenzo-p-dioxins and -furans are byproducts of nearly all combustion processes (Fiedler, 2003), cannot be formed without chlorine (Zevenhoven & Kilpinen, 2004), and have been shown to be persistent and bioaccumulative (Fiedler,

2003), with the primary exposure route (90-95%) to humans through ingestion (Djien Liem, Furst, & Rappe, 2000; Fiedler, 2003; Schecter et al., 2001; US EPA, 2006; Van den Berg et al., 2006). Therefore, it is understood why Toxic Equivalency Factors are based primarily upon an ingestion route of exposure (Van den Berg et al., 2006). The Total Toxic Equivalent has best been described by the World Health Organization as the “sum of the products of the concentration of each compound multiplied by its TEF value” (Van den Berg et al., 2006). Furthermore, TEQs have been studied extensively for decades, especially by the EPA (US EPA, 2006). An emission inventory conducted by the EPA in 2006 (US EPA, 2006) found that the leading source of PCDD/PCDF emissions in the US in 1987 and 1995 was municipal waste combustion, while in 2000 it became the combustion of domestic refuse in backyard burn barrels (Lemieux et al., 2004). The PCDD known as 2,3,7,8-TCDD is considered to be the most toxic congener of all 17 congeners with a chlorine atom in the 2,3,7,8 positions, as well as the total 75 PCDD and 135 PCDF compounds (Fiedler, 2003). Exposure to high concentrations of PCDD/PCDFs are generally associated with skin lesions, weight loss, altered liver function, and depression of the immune system (Fiedler, 2003). Furthermore, 2,3,7,8-TCDD and (as of 2012) 2,3,4,7,8-PeCDF are both considered Class 1 carcinogens, while 1,2,3,7,8-PeCDD and 1,2,3,7,8-PeCDF are categorized as Class 3 carcinogens by the IARC (WHO - IARC, 2015).

Framing Emission Factors in a Health Context

Purpose of Quantifying Emission Factors.

Quantifying emission factors is an important first step in preventing deleterious health effects from affecting an individual exposed to a pollutant. An emission factor is “a representative value that attempts to relate the quantity of a pollutant released to the atmosphere with an activity associated with the release of that pollutant” (US EPA, 2015a). Emission factors come in various units but the most conventional form is simply a ratio of the mass of the pollutant emitted to a unit of activity that generated the emissions (e.g., milligrams of benzene emitted per kilogram of waste openly burned) (US EPA, 2013). Quantification of pollutant emissions for a particular process could be necessary for a variety of reasons. For example, the implementation of a new control into a process could require a reassessment of the airborne emission factors, verifying that pollutant emissions do not increase as a result of implementing said control (US EPA, 2013). However, for an uncharacterized process, the primary purpose of quantifying the emissions is to establish a standard emission factor for that particular process.

Purpose of Establishing Standard Emission Factors.

Once a standard emission factor is established for an uncharacterized process and is validated, it is placed within the EPA’s WebFire application (US EPA, 2015e) for online storage, retrieval, and development of emission factors, a repository for standardized emission factors of (currently) over 200 air pollution source categories, 420 pollutants, and thousands of processes (US EPA, 2013). The standard emission factor is also placed within AP 42, *Compilation of Air Pollutant Emission Factors*, a formerly published but now online-only book with chapters specific to sources and industries, but

that is also updated by chapter infrequently (US EPA, 2015a). The more a pollutant is studied and quantified for a particular process, the greater the confidence becomes in that pollutant's specific standard emission factor. With greater confidence comes greater reliability as to what a particular process will generate. When an emission factor is considered reliable, it can be used for many other purposes such as predicting and modeling spatial or temporal relationships through the EPA's Emission Modeling Clearinghouse (US EPA, 2015b). Another, more important, reason for establishing a standard emission factor for a process is to assist in establishing health and regulatory limits for the pollutant of interest.

Purposes of Health and Regulatory Limits.

The establishment of initial health and regulatory limits for a pollutant serve as a starting point for further research and analysis. For example, the Clean Air Act of 1990 established the initial list of 189 Hazardous Air Pollutants (US EPA, 2008). Since the establishment of the initial HAP list, the EPA has removed 2-Butanone, perhaps better known as Methyl Ethyl Ketone (MEK), since in-depth research had suggested that it was not anticipated to cause adverse human and environmental health effects (US EPA, 2005b). By initially establishing MEK as a HAP, additional research was conducted into the possible environmental and human health effects, thereby spurring its removal when it was anticipated to no longer be a significant hazard. Similar validation processes against health effects have been conducted on National Primary and Secondary Ambient Air Quality Standards, National Institute for Occupational Safety and Health (NIOSH) Recommended Exposure Limits (RELs), American Conference of Governmental Industrial Hygienists (ACGIH) Threshold Limit Values (TLVs), and Occupational Safety

and Health Administration (OSHA) Permissible Exposure Limits (PELs). If a health or regulatory limit should be more stringent or be maintained according to the research, this same validation process would occur, ensuring adverse human and environmental health effects are minimized to the greatest extent possible. Therefore, the primary purposes of the establishment of health and regulatory limits are to: (1) address and reduce environmental and human health issues, while attempting to at least maintain status quo; (2) to continue facilitating research into the health effects of pollutants; and, (3) to assist in the characterization, understanding, and prevention of adverse health effects associated with doses so that a formal revision of policy and regulatory limits can occur in the future.

The EPA's Integrated Risk Information System (IRIS) and the Agency for Toxic Substances and Disease Registry (ATSDR) are two systems in place that take advantage of this emission factor establishment process. The EPA's IRIS is a repository for toxicity information utilized globally. The IRIS identifies health hazards associated with exposure to either a specific chemical or a group of chemicals, conducts in-depth research into the potential health effects, and provides an assessment of the response associated with a range of doses from that chemical or group of chemicals. These "dose-response" assessments can assist health risk assessors in protecting people from hazardous exposures (US EPA, 2015c). The ATSDR provides similar capabilities as the IRIS but are additionally charged with protecting the general public from improperly-disposed hazardous waste, as well as from harmful chemical spills into the environment. The ATSDR was created and acts in accordance with the Comprehensive Environmental Response, Compensation, and Liability Act of 1980, commonly referred to as the

“Superfund” Act, to expand the understanding of health effects from exposure to hazardous chemicals (CDC, 2013). One of the non-cancerous, health-based research products that the ATSDR provides are Minimum Risk Levels (MRLs) that are intended to be used as screening levels, or health risk thresholds, for health risk assessors “to identify contaminants and potential health effects that may be of concern at hazardous waste sites” (CDC, 2001).

Summary

Ultimately, emission factors do not directly protect the health of humans and the environment but play a pivotal role in ensuring that the necessary health and regulatory limits are put in place to do so. This emission factor establishment process is vital in providing health risk assessors a human and environmental health risk threshold that can be used to communicate to stakeholders. Furthermore, establishment of emission factors for a particular process allows for the identification of completed exposure pathways between the source, the pathway it travels through, and a potentially exposed population and environment (or receivers).

IV. Conclusions

Chapter Overview

This study sought to characterize emissions from four MRE variations and four fiberboard packaging types to establish emission factors, conduct a comparison of these emission factors to each other, and establish where these emissions stand within the literature. This chapter serves to compile conclusions drawn throughout the thesis effort, succinctly answer the research questions posed within Chapter I, discuss limitations and significance of this study, and pose future action and research opportunities.

Review of Findings

A review of the findings is discussed below following a reproduction of the research questions posed within Chapter I, shown here:

1. What are the airborne emission factors for the current and alternative designs for MREs and their fiberboard packaging?
2. How do the emission factors for the current and alternative designs compare with one another and to open burning situations within the literature, such as military waste burns?
 - 2.1. What are the airborne emission factor effects of varying the nanocomposite content within the alternative MRE design?

Question 1: Emission Factors for Current and Alternative MREs and Fiberboards.

Airborne emission factors for all of the MREs and fiberboards can be found within the tables in Chapter II, as well as Appendix A. A comprehensive summary is

presented within Table 9 that directs the reader towards the appropriate tables to find the emission factors that are sought.

Table 9. Comprehensive Summary of Emission Factors for MREs and Fiberboards

	<i>PM</i>	<i>Metals</i>	<i>PAHs</i>	<i>VOCs</i>	<i>PCDD/PCDFs</i>
<i>MREs</i>	Figure 2 Table 3	Table 23	Table 4 Table 20	Table 22	Table 16 Table 18
<i>Fiberboards</i>	Figure 2 Table 3	-	Table 4 Table 19	Table 21	Table 15 Table 17

Question 2: Comparison of Emission Factors to Each Other/Open Burning

Literature.

A comparison between emission factors for MREs, fiberboards, and the literature can be found within Chapter II. Scholarly Article. Average PM values across all material types were found to be an order of magnitude greater than average VOC values, and two orders of magnitude greater than average PAH values. MRE emissions as a whole were determined to be between one to six times greater than fiberboard emissions for all emission types; however, this was expected due to the various components that comprise an MRE versus the relatively homogeneous composition of the fiberboards. Total VOC emissions for the MRE variations were generally one to seven times greater in magnitude than fiberboard type emissions with the greatest emitter being the current MRE-0% variation and the least emitter also being the current SFP type. As a whole, PAH emissions were similar amongst the MRE variations, as well as amongst the fiberboard types. When comparing the MREs and fiberboards to one another however, MRE PAH emissions are generally three to six times greater than the current SFP emissions and also two to three times greater than CFP emissions. Additionally, PM and

PCDD/PCDF emissions were found to be equivalent amongst the MRE variations, with emissions amongst fiberboard types also being comparable to one another. It was also observed that MRE variations generally emitted up to five times greater PM emissions and three times greater PCDD/PCDF emissions than the fiberboard types. Furthermore, the data suggests that MREs account for 70 to 90 percent of total emissions when burned in unison with the current fiberboard container and liner. Sampling variance ranged many orders of magnitude between material types and is believed to be attributed to burn qualities, as measured by MCE, and may have been affected by material composition of the waste stream, such as chlorine content. Table 5 provides a simple means to compare this research with the similar open burning situations within the literature, specifically simulated military waste burns. For additional context within the literature, Table 10 provides a comparison of MRE and fiberboard emission factors to other open burning situations encountered within the literature.

Table 10. MRE and Fiberboard Comparison of Select Pollutants to Additional Open Burning Literature, Extracted from (Lemieux et al., 2004)

Class	Pollutant	Units	Material Tested		Open Burning Literature	
			All MREs	All Fiberboards	Barrel Burning of Household Waste (Lemieux, 1998)	Extratropical Forest (Andreae & Merlet, 2001)
PM	PM2.5	g/kg-waste	11.271	3.192	-	13.0
	PM10	g/kg-waste	11.665	3.204	-	16.6
PAHs	Benzo(a)pyrene	mg/kg-waste	0.71	0.47	1.40	-
	Naphthalene	mg/kg-waste	45.23	17.06	11.36	-
VOCs	Benzene	mg/kg-waste	224.46	108.65	979.75	490
	1,3-Butadiene	mg/kg-waste	102.86	30.43	141.25	60
PCDD/PCDF	Total PCDD/PCDF	mg/kg-waste	0.07679	0.00875	0.0058 (Lemieux, Gullett, Lutes, Winterrowd, & Winters, 2003)	1.5E-4 to 6.7E-3 (Gullett & Touati, 2003)

As Table 10 depicts, MRE PM emission factors were nearly equivalent to the extratropical forest emission factors but slightly lower. Barrel burning of household waste had nearly double the benzo(a)pyrene emissions as MRE and nearly triple compared to the fiberboards; however, accounting for variance, these emission factors should be considered equivalent. Notably, the average MRE naphthalene emission factor was moderately higher than the household waste emission factor and is believed to be attributed to the material composition of the MREs (see Appendix B: Material Composition). However, VOC emission factors for the MREs and fiberboards were significantly lower than both of these studies. Lastly, MRE PCDD/PCDF emissions were approximately 10 times greater than both the open burning literature and the fiberboards but still relatively low. High chlorine availability (in the form of table salt) is hypothesized as the reason behind this difference in emissions.

Question 2.1: Emission Factor Effects from Nanocomposite Variation within MREs.

Chapter II provided details on the differences between the MREs but a brief summary follows. Little effect was seen from the variation of the nanocomposite content between the MRE types for all emission factors. The nanocomposite did not seem to affect particulate matter emission factors, as demonstrated within Figure 5.

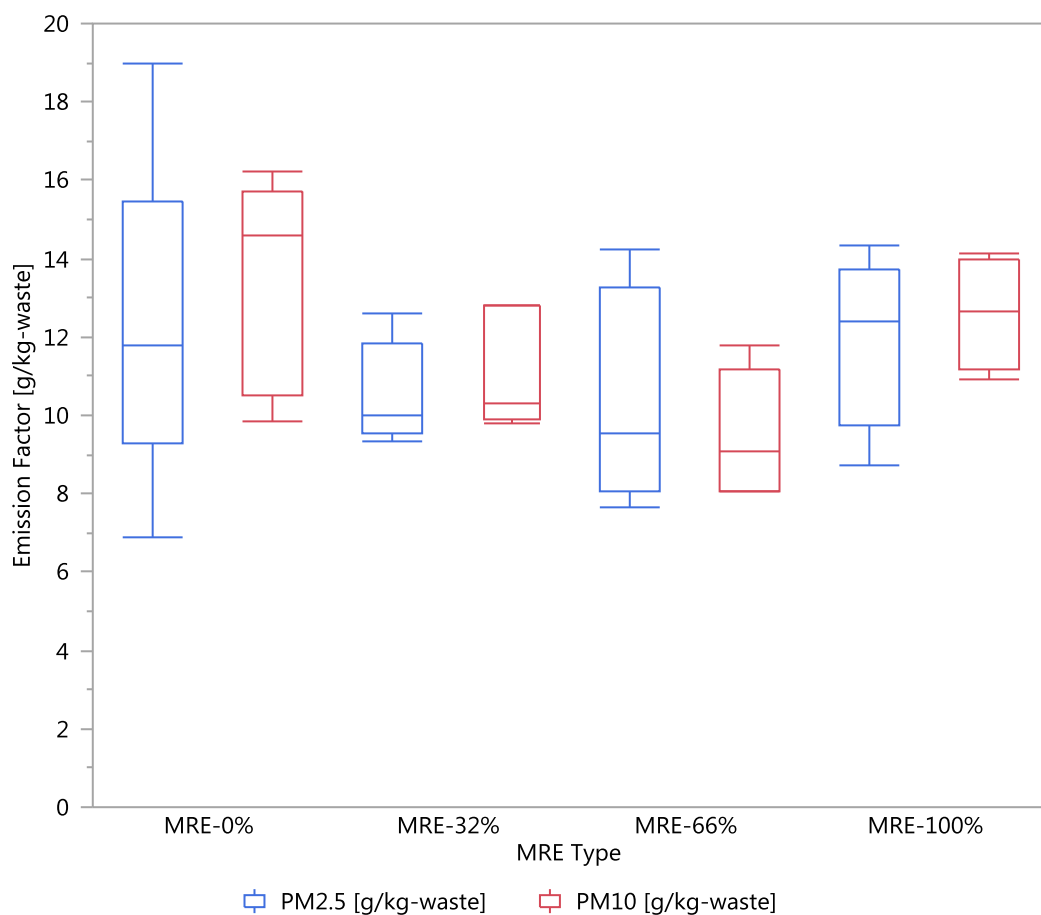


Figure 5. PM Box Plot Emission Factor Comparison Between MRE Types

A similar situation was seen with the top eight PAH emission factors, as depicted within Figure 6. Naphthalene had slightly higher emission factors for MRE-100% but the variance was too great for a definitive difference to be seen.

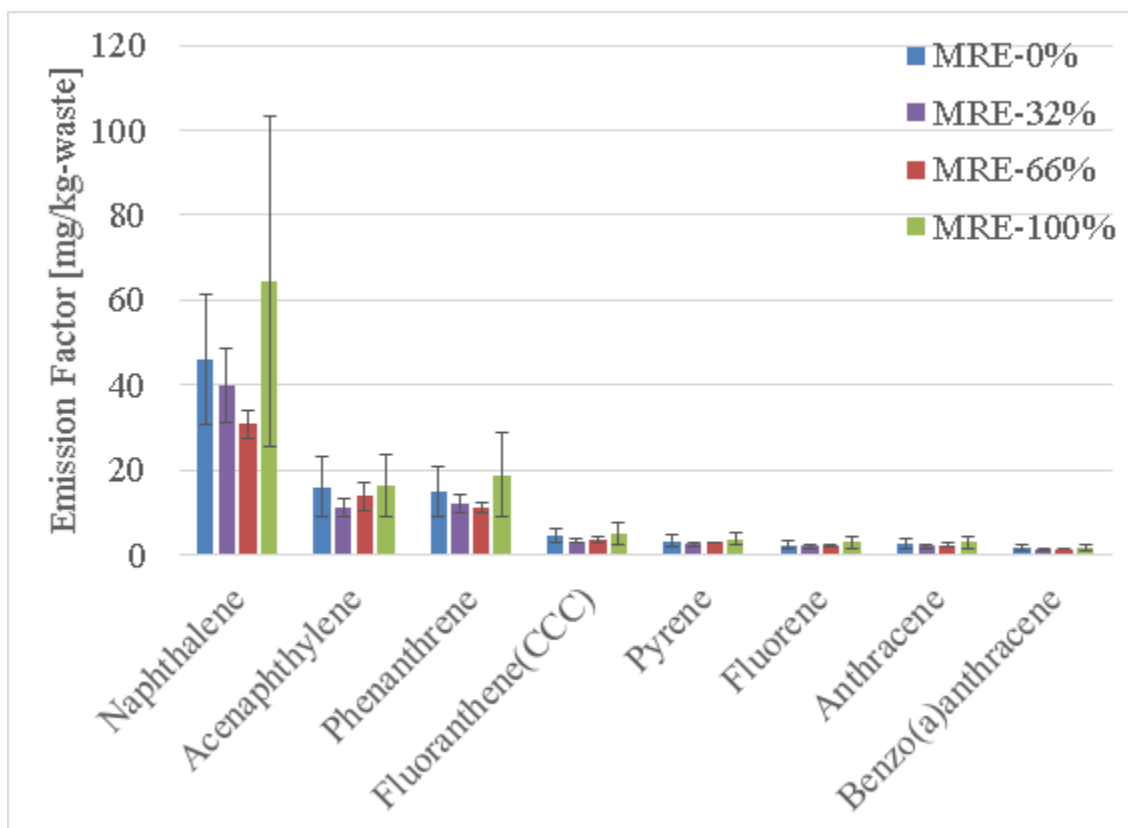


**Figure 6. PAH Emission Factor Comparison Between MRE Types with Error Bars
Depicting ± 1 Standard Deviation**

When the 10 greatest emission factors for VOCs are compared within Figure 7, a slight difference can be seen for 1,3-Butadiene, Benzene, and Propene between MRE-32% and MRE-66%. MRE-32% was consistently higher for these three VOCs and it is unclear as to why this was the case. Additional sampling would be recommended to determine whether these three VOCs were truly distinct and statistically different or if it was merely an instance where extreme values were observed.

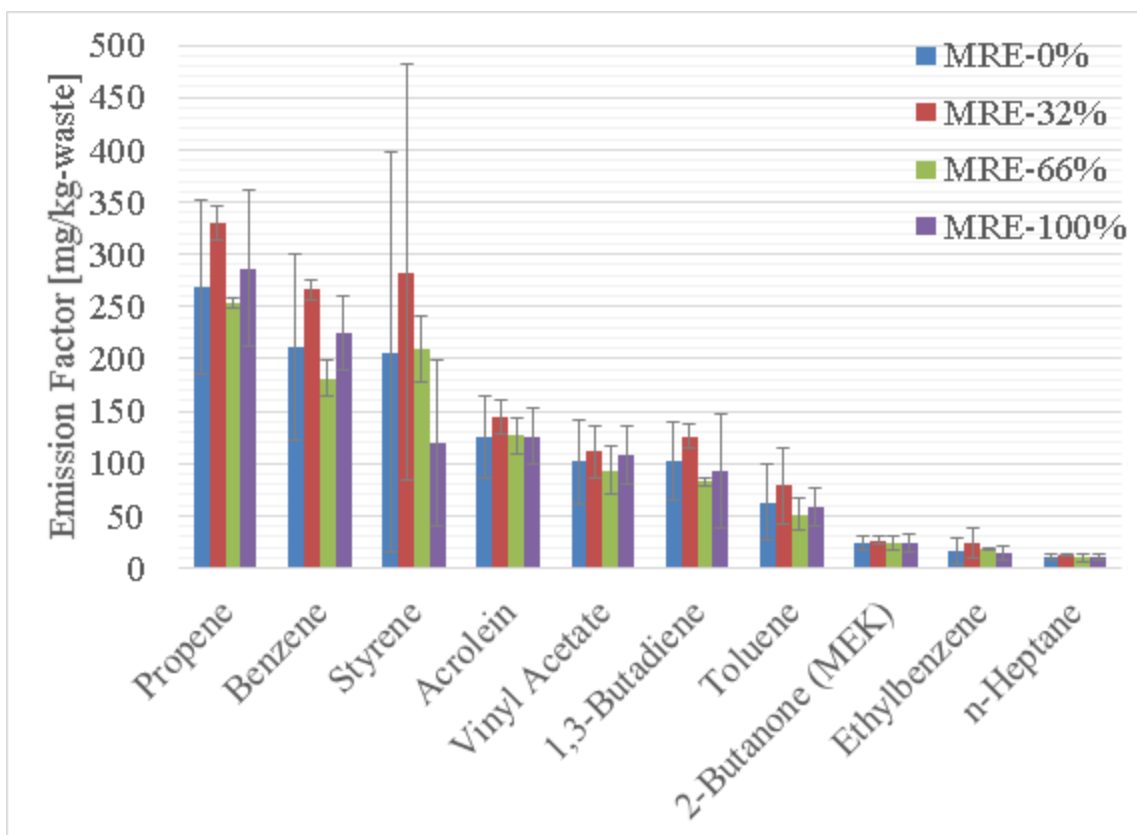


Figure 7. VOC Emission Factor Comparison Amongst MRE Types for 10 Greatest Pollutants with Error Bars Depicting ± 1 Standard Deviation

Figure 8 shows that MRE-32% is again slightly higher than MRE-66% for 2,3,7,8-TCDF and 2,3,4,7,8-PeCDF. However, the variation is within two standard deviations; therefore, it is likely that the MRE types emit similar emission factors. If they are truly different, it may be due to the unique ratios between the amount of aluminum-based components and nanocomposite-based components present. Upon further inspection, MRE-66% is slightly lower than MRE-100% in most pollutant categories and it may have to do with this proposed ratio. Further study would be warranted to determine whether these emission factors were isolated incidents or are revealing a hardly discernible difference.

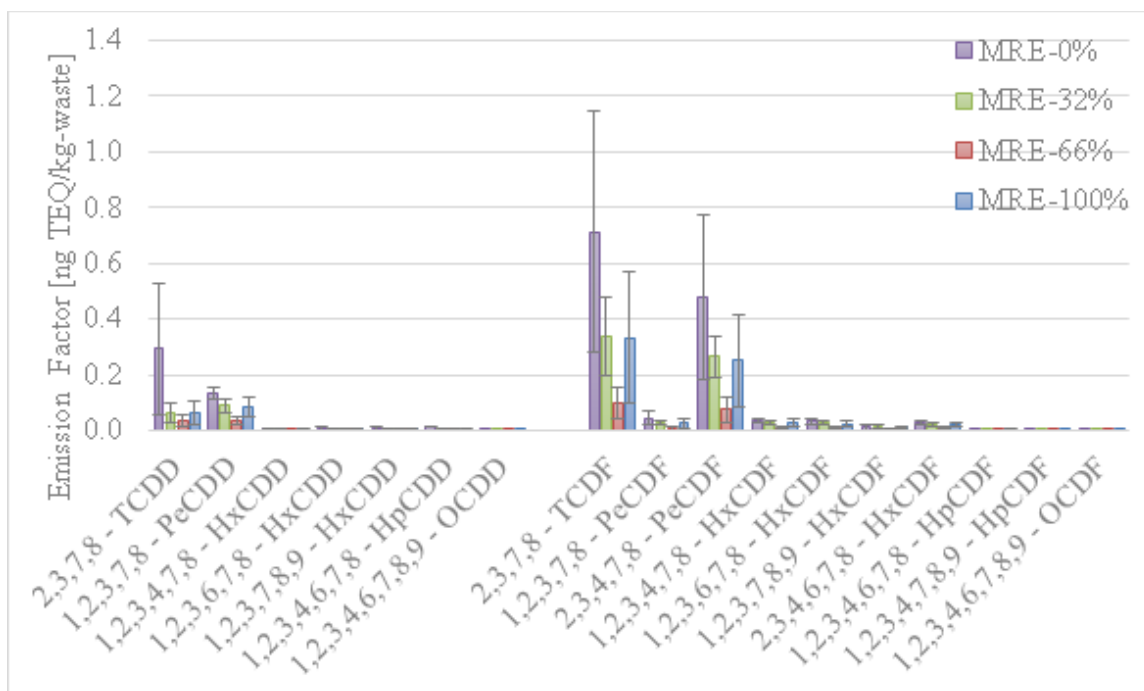


Figure 8. PCDD/PCDF Emission Factor Comparison Amongst MRE Types with Error Bars Depicting ± 1 Standard Deviation

Limitations

This research effort identified several limitations throughout the progression of these experiments. Although this study had more replicates than the studies compared within Chapter II (Aurell et al., 2012; Barnes et al., 2016; Woodall et al., 2012), the most significant limitation of this study was the limited sample size for each material. Although statistical analyses could be conducted, the exclusion of any outliers increased the variability of the material that the outlier belonged to, thus reducing the probability of finding a difference between the various material types. Additionally, this limited sample set size makes statistical significance limited in the ability to identify true differences between materials. If measurement bias was present, the population estimates for the

emissions may perhaps be skewed higher or lower than the true population mean (Reimann, Filzmoser, Garrett, & Dutter, 2011). Furthermore, any instances of possible autocorrelation could potentially be contributed to unequal time intervals between sample collections (Reimann, Filzmoser, Garrett, & Dutter, 2011), as it varied from test to test.

Actual burn pits typically utilize an accelerant, such as JP8, to initiate combustion which was not utilized within this study. It is speculated that utilizing an accelerant may produce higher initial burning temperatures, perhaps reducing overall emissions by increasing combustion efficiency. Additionally, minor amounts of starch-based glue would normally be present within the various fiberboard boxes but were not present for this study (Ratto et al., 2012). It is speculated that the glue's presence would not have influenced emission factors significantly but is worth noting.

Furthermore, pollutants that were not targeted within this study may still be of a health concern. A literature review of combustion products and known health hazards associated with the open burning of waste was utilized as a starting point in the characterization of emissions from MREs and their fiberboard packaging. However, other pollutants that were not sampled for that are also associated with combustion, such as criteria pollutants like ozone and sulfur dioxide (Institute of Medicine, 2011), may present an equivalent hazard and further study is warranted.

The FRH, depicted within Figure 9, is a “water activated exothermic-chemical heater” that is primarily a super-corroding alloy. This alloy consists of magnesium metal powder and food-grade electrolytic iron powder and function as anode and cathode, respectively, while the addition of water activates the electrolyte and promptly corrodes

the magnesium components. Products of the reaction include gaseous hydrogen, magnesium hydroxide, and heat (Defense Logistics Agency, 2003).

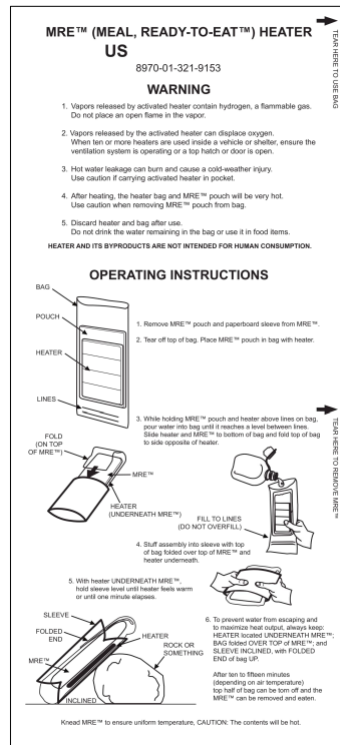


Figure 9. Depiction of Flameless Ration Heater (Defense Logistics Agency, 2013)

By taking advantage of the corrosion process, an FRH can raise the temperature of an MRE entrée 100°F within 12 minutes or less (Barrett & Cardello, 2012; Defense Logistics Agency, 2003); however, these unique capabilities posed credible safety concerns upon implementation of this experiment. MRE test samples were indiscriminately placed upon the industrial-grade aluminum foil sheets that materials were burned upon, to include food content. As mentioned within Chapter II, had the FRH been included in the burns, the moisture content of the food most likely would have initiated the corrosion process prematurely, with personnel in harm's way. It is speculated that, had the FRH been included in the test samples, the corrosion process

would have contributed a significant increase in heat and hydrogen available for combustion, as well as increasing the magnesium and iron particulates found within the PM2.5 samples. Whether or not the analytical results would have been discernably different remains to be seen and is worth further inquiry in a safe experimental manner.

Significance of Research

Emission factors for various pollutants have now been established for military rations to better protect the health and meet the needs of US military personnel. The establishment of standard emission factors for military rations is the first step in determining health-based risk threshold criteria to be utilized by health risk assessors in the future. By utilizing emission factors from this research effort, USCENTCOM burn pit guidance can be supplemented with additional emission factors specific to military rations and health impacts could potentially be further understood. Additionally, pollutants associated with the open burning of military rations can possibly be further understood as well. Furthermore, the quantification of the pollutants studied within this research effort have allowed for the isolation of military ration emission factor contributions to military waste streams. Though no significant practical emission differences were found to isolate one material type over another, with confirmation by additional studies, the least costly material type in terms of weight, volume, and practicality is recommended for further implementation when based upon an emission output criteria alone. If another MRE or fiberboard type saves even an ounce in weight with implementation, that ounce translates to fuel saved, payloads averted, and lives potentially saved by averting those sorties. Emission factors generated from this research

effort have established the first baseline of what MREs and their fiberboard packaging solely contribute to military waste burn emissions and can assist the US military in quantifying, establishing, and understanding the potential health risks associated with such a prevalent waste stream product.

Recommendations for Action

Combustion efficiency has been shown to play a role in how great emissions will be (Hardy et al., 1996; Reid et al., 2005; Urbanski, 2013; Ward & Hardy, 1991), where lower combustion efficiency (e.g., incomplete combustion and smoldering conditions) is associated with greater emissions. Therefore, for future open burning, incineration, or general situations regarding the disposal of military rations, it is recommended that liquid waste be separated from solid waste prior to destruction to increase combustion temperatures and, thus, reduce emissions from the smoldering of the waste (U.S. Army, 2013). Additionally, it is recommended that alternatives to open burning, such as targeted recycling efforts or waste-to-energy systems, be explored further for the destruction of military waste in order to reduce overall emissions (Barnes et al., 2016). Furthermore, from an emission factor standpoint based upon this limited research effort studying these targeted pollutants, it is recommended that the least costly material that still meets current performance specifications be selected amongst the various MRE and fiberboard material types for further study and implementation.

Recommendations for Future Research

There are many opportunities that remain unanswered in the literature. Establishing additional baselines for other components of a military waste stream is

recommended. Additionally, an in-depth risk assessment into the potential health consequences of being exposed to emission factors of this magnitude would prove useful. Furthermore, placing these emission factors into a larger-scaled context, such as being compared with industry emission factors on a ton rather than kilogram of waste basis, may be beneficial. A case study utilizing these emission factors for modeling may predict health risk better than current explanations.

Moreover, a long-term cost/benefit analysis for the various MRE and fiberboard types may potentially reduce significant amounts of cost. Saving pennies for each MRE or fiberboard would add up to millions of dollars within a short span of time. Additionally, a cost/benefit analysis for the potential implementation of an alternative waste disposal method(s) is recommended.

Replicating this experiment over a longer period of time or with the FRHs (in a safe manner) to simulate more realistic conditions may be informative. Additional pollutant and emission factor determinations from a future study of the open burning of MREs and their fiberboard packaging can further assist the military in quantifying and understanding the health risks. Investigating fully biodegradable MREs may also prove to be constructive.

It is the intent of this study that the included emission factors established for MREs and their fiberboard packaging can provide a basis for future health risk characterization, allow for potential modeling of burn pit emissions, and can help to protect the health of those exposed to similar pollutants.

Appendix A: Scholarly Article Supplemental Information

Table 11. Collected Samples for each Test Category

Test Category	VOCs SUMMA® Canisters	PM_{2.5}	PM₁₀	PAH	PCDD/PCDF
SFP	5	10	10	5	5
CFP	4	7	7	4	4
CFNP	4	8	8	4	4
SFNP	4	10	10	3	4
MRE-100%	3	6	6	3	3
MRE-66%	2	6	6	3	3
MRE-32%	3	6	6	3	3
MRE-0%	3	6	6	3	3
Background	2	2	2	2	3
Total	30	61	61	30	32

Table 12. Ultimate Proximate Analysis for Fiberboard Materials

	Current Fiberboard Solid Box	Current Fiberboard Liner	Corrugated Spektrakote Polymer Fiberboard Box	Corrugated Spektrakote Polymer Fiberboard Liner	No Spektrakote Box	No Spektrakote Liner	Current Fiberboard Box - No Wet Strength
Identification Code	CB	CL	SB	SL	NSB	NSL	NWS
Heat of combustion (BTU/lb)	8121	7955	7537	7166	7598	7979	8145
Loss on drying (%)	7.45	7.49	9.45	9.13	9.48	8.51	10.24
Carbon (%)	46.3	46.19	44.99	45.44	45.36	45.85	46.56
Chlorine (ppm)	188	185	297	251	97	57	139
Hydrogen (%)	6.08	6.16	6.13	6.02	6.29	6.20	6.03
Nitrogen (%)	<0.5	<0.5	<0.5	<0.5	<0.5	<0.5	<0.5
Sulfur (%)	0.234	0.243	0.162	0.155	0.149	0.140	0.236

Table 13. Content and Fraction of Material in each MRE Mixture

Material	MRE-0%	MRE-32%	MRE-66%	MRE-100%
Old material	1.1E-01	5.7E-02	3.6E-02	NA
New Material	NA	3.6E-02	7.0E-02	1.1E-01
Plastic	1.7E-01	1.8E-01	1.6E-01	1.7E-01
Cardboard	2.9E-01	3.0E-01	2.8E-01	2.9E-01
Paper	1.6E-02	1.6E-02	1.5E-02	1.6E-02
Al-bag	1.7E-02	1.7E-02	6.6E-02	1.7E-02
Matches	9.7E-03	9.9E-03	9.2E-03	9.7E-03
Adsorbent package	4.7E-03	4.8E-03	4.5E-03	4.7E-03
Food waste: Penne pasta	1.6E-01	1.7E-01	1.6E-01	1.6E-01
Beef stick	2.8E-02	2.8E-02	2.6E-02	2.8E-02
Toaster pastry	4.1E-02	4.2E-02	3.9E-02	4.1E-02
Crackers	2.8E-02	2.8E-02	2.6E-02	2.8E-02
Pretzels	2.6E-02	2.7E-02	2.5E-02	2.7E-02
Cappuccino	2.1E-02	2.1E-02	2.0E-02	2.1E-02
Salt	6.5E-03	6.6E-03	6.2E-03	6.5E-03
Seasoning	2.6E-03	2.7E-03	2.5E-03	2.7E-03
Ice tea	5.0E-02	5.1E-02	4.8E-02	5.0E-02
Chewing gum	8.2E-03	8.4E-03	7.8E-03	8.3E-03

**Table 14. Carbon Fraction in each Material and Total Carbon Fraction for each
MRE Mixture**

Material	Carbon fraction in material	MRE- 0%	MRE- 32%	MRE- 66%	MRE- 100%
Carbon fraction from each waste category^b					
Old material	0.37	4.2E-02	2.1E-02	1.3E-02	NA
New Material	0.74	NA	2.7E-02	5.2E-02	8.2E-02
Plastic	0.74 ^a	1.3E-01	1.3E-01	1.2E-01	1.3E-01
Cardboard	0.46	1.4E-01	1.4E-01	1.3E-01	1.4E-01
Paper	0.44 ^a	7.0E-03	7.1E-03	6.6E-03	7.0E-03
Al-bag	0.0076 ^a	1.3E-04	1.3E-04	5.0E-04	1.3E-04
Matches	0.50	4.9E-03	5.0E-03	4.7E-03	4.9E-03
Adsorbent package	0.0076 ^a	3.6E-05	3.7E-05	3.4E-05	3.6E-05
Food waste: Penne pasta	0.092	1.5E-02	1.5E-02	1.4E-02	1.5E-02
Beef stick	0.28	7.7E-03	7.8E-03	7.3E-03	7.7E-03
Toaster pastry	0.43	1.8E-02	1.8E-02	1.7E-02	1.8E-02
Crackers	0.51	1.4E-02	1.4E-02	1.3E-02	1.4E-02
Pretzels	0.39	1.0E-02	1.1E-02	9.9E-03	1.0E-02
Cappuccino	0.45	9.4E-03	9.6E-03	8.9E-03	9.4E-03
Salt	0.0076	4.9E-05	5.0E-05	4.7E-05	4.9E-05
Seasoning	0.49	1.3E-03	1.3E-03	1.2E-03	1.3E-03
Ice tea	0.49	2.4E-02	2.5E-02	2.3E-02	2.4E-02
Chewing gum	0.49	4.0E-03	4.1E-03	3.8E-03	4.0E-03
Total	NA	0.42	0.44	0.43	0.46

^aData from Liu and Lipták (Liu & Lipták, 1999). ^bCarbon Fraction in Material × Waste Fraction in Mixture

Table 15. PCDD/PCDF TEQ Emission Factors from Fiberboards

Isomer.	SFP		SFNP		CFP		CFNP	
	Average	STDV	Average	STDV	Average	STDV	Average	STDV
ng TEQ/kg fiberboard								
2,3,7,8 - TCDD	2.53E-03	8.17E-04	6.05E-03	2.11E-03	2.76E-02	1.78E-02	1.25E-02	2.17E-03
1,2,3,7,8 - PeCDD	1.15E-02	3.55E-03	1.63E-02	7.55E-03	1.18E-01	8.06E-02	5.17E-02	2.36E-02
1,2,3,4,7,8 - HxCDD	7.65E-04	3.68E-04	1.06E-03	4.28E-04	6.70E-03	4.21E-03	2.78E-03	9.28E-04
1,2,3,6,7,8 - HxCDD	1.64E-03	4.61E-04	1.72E-03	7.64E-04	2.07E-02	1.25E-02	9.23E-03	3.54E-03
1,2,3,7,8,9 - HxCDD	1.30E-03	4.50E-04	1.52E-03	4.28E-04	1.54E-02	9.77E-03	6.74E-03	2.60E-03
1,2,3,4,6,7,8 - HpCDD	1.59E-03	3.83E-04	1.65E-03	4.17E-04	2.52E-02	1.19E-02	1.20E-02	3.91E-03
1,2,3,4,6,7,8,9 - OCDD	1.29E-04	1.95E-05	1.26E-04	4.01E-05	2.05E-03	7.88E-04	1.01E-03	2.52E-04
2,3,7,8 - TCDF	6.51E-03	1.07E-03	8.71E-03	4.93E-03	2.38E-02	4.96E-03	7.09E-03	1.13E-03
1,2,3,7,8 - PeCDF	7.49E-04	8.50E-05	1.38E-03	1.16E-03	2.03E-03	2.47E-04	5.70E-04	3.88E-04
2,3,4,7,8 - PeCDF	9.06E-03	3.29E-03	1.97E-02	1.30E-02	2.56E-02	2.47E-03	7.40E-03	5.07E-03
1,2,3,4,7,8 - HxCDF	2.65E-03	1.49E-03	5.66E-03	3.38E-03	4.52E-03	8.73E-04	2.63E-03	5.45E-04
1,2,3,6,7,8 - HxCDF	2.31E-03	1.42E-03	5.02E-03	3.58E-03	3.51E-03	2.94E-04	2.05E-03	6.52E-04
1,2,3,7,8,9 - HxCDF	1.16E-03	9.56E-04	2.26E-03	1.21E-03	1.39E-03	1.68E-04	7.22E-04	7.79E-05
2,3,4,6,7,8 - HxCDF	3.49E-03	3.15E-03	5.79E-03	2.59E-03	4.58E-03	6.47E-04	2.18E-03	4.74E-04
1,2,3,4,6,7,8 - HpCDF	8.95E-04	9.25E-04	1.23E-03	5.72E-04	1.23E-03	2.30E-04	6.68E-04	1.17E-04
1,2,3,4,7,8,9 - HpCDF	1.53E-04	1.74E-04	2.33E-04	1.66E-04	2.23E-04	7.03E-05	1.31E-04	3.58E-05
1,2,3,4,6,7,8,9 - OCDF	2.09E-05	1.29E-05	2.27E-05	1.23E-05	4.86E-05	9.76E-06	2.48E-05	5.44E-06
PCDD TEQ Total	1.88E-02	5.03E-03	2.54E-02	1.11E-02	2.16E-01	1.37E-01	7.92E-02	4.94E-02
PCDF TEQ Total	2.70E-02	1.19E-02	5.00E-02	2.95E-02	6.70E-02	6.04E-03	2.27E-02	5.70E-03
PCDD/PCDF TEQ Total	4.57E-02	1.63E-02	7.54E-02	4.05E-02	2.83E-01	1.36E-01	1.02E-01	5.50E-02

Table 16. PCDD/PCDF TEQ Emission Factors from MREs

Isomer.	MRE-100%		MRE-66%		MRE-32%		MRE-0%	
	Average	STDV	Average	STDV	Average	STDV	Average	STDV
ng TEQ/kg MRE								
2,3,7,8 - TCDD	6.55E-02	4.22E-02	3.34E-02	2.05E-02	6.41E-02	3.47E-02	2.93E-01	2.36E-01
1,2,3,7,8 - PeCDD	8.63E-02	3.38E-02	3.65E-02	1.61E-02	8.81E-02	2.31E-02	1.31E-01	2.21E-02
1,2,3,4,7,8 - HxCDD	3.32E-03	1.32E-03	2.18E-03	4.11E-04	3.26E-03	1.10E-03	5.36E-03	2.09E-03
1,2,3,6,7,8 - HxCDD	8.05E-03	1.19E-03	6.44E-03	1.11E-03	7.76E-03	1.16E-03	7.20E-03	6.29E-03
1,2,3,7,8,9 - HxCDD	6.32E-03	5.38E-04	5.88E-03	8.58E-04	7.21E-03	2.03E-03	6.02E-03	5.28E-03
1,2,3,4,6,7,8 - HpCDD	9.81E-03	7.16E-04	8.95E-03	4.34E-04	8.84E-03	4.77E-04	1.17E-02	5.15E-04
1,2,3,4,6,7,8,9 - OCDD	8.71E-04	1.01E-04	8.14E-04	3.00E-05	7.45E-04	3.61E-05	9.92E-04	9.13E-05
2,3,7,8 - TCDF	3.32E-01	2.37E-01	9.67E-02	5.78E-02	3.37E-01	1.44E-01	7.12E-01	4.32E-01
1,2,3,7,8 - PeCDF	2.62E-02	1.72E-02	7.35E-03	4.66E-03	2.60E-02	7.85E-03	4.54E-02	2.42E-02
2,3,4,7,8 - PeCDF	2.50E-01	1.68E-01	7.46E-02	4.70E-02	2.65E-01	7.44E-02	4.79E-01	2.93E-01
1,2,3,4,7,8 - HxCDF	2.78E-02	1.19E-02	9.11E-03	2.96E-03	2.67E-02	8.44E-03	3.57E-02	9.18E-03
1,2,3,6,7,8 - HxCDF	2.44E-02	1.01E-02	8.62E-03	2.96E-03	2.66E-02	5.43E-03	3.19E-02	1.08E-02
1,2,3,7,8,9 - HxCDF	1.12E-02	4.23E-03	4.20E-03	1.85E-03	1.34E-02	5.48E-03	1.08E-02	1.04E-02
2,3,4,6,7,8 - HxCDF	2.10E-02	8.81E-03	8.61E-03	3.83E-03	2.38E-02	6.29E-03	2.74E-02	8.94E-03
1,2,3,4,6,7,8 - HpCDF	2.81E-03	8.62E-04	1.45E-03	2.50E-04	2.57E-03	6.20E-04	3.19E-03	1.35E-04
1,2,3,4,7,8,9 - HpCDF	6.39E-04	2.16E-04	2.53E-04	7.33E-05	7.42E-04	2.74E-04	3.89E-04	3.76E-04
1,2,3,4,6,7,8,9 - OCDF	5.99E-05	2.29E-05	4.80E-05	9.05E-06	6.31E-05	1.37E-05	7.07E-05	7.21E-06
PCDD TEQ Total	1.80E-01	7.54E-02	9.08E-02	2.86E-02	1.82E-01	5.02E-02	4.56E-01	2.63E-01
PCDF TEQ Total	6.97E-01	4.57E-01	2.11E-01	1.19E-01	7.22E-01	2.40E-01	1.35E+00	7.86E-01
PCDD/PCDF TEQ Total	8.77E-01	5.27E-01	3.01E-01	1.45E-01	9.04E-01	2.91E-01	1.80E+00	1.04E+00

Table 17. PCDD/PCDF Total Emission Factors from Fiberboards

Homologue	SFP		SFNP		CFP		CFNP	
	Average	STDV	Average	STDV	Average	STDV	Average	STDV
	ng/kg fiberboard							
TeCDD	0.32	0.061	0.39	0.12	0.84	0.40	0.39	0.13
PeCDD	0.19	0.045	0.20	0.033	0.80	0.47	0.31	0.17
HxCDD	0.17	0.030	0.19	0.016	1.7	1.0	0.67	0.30
HpCDD	0.28	0.066	0.29	0.071	4.0	1.9	1.9	0.65
OCDD	0.43	0.065	0.42	0.13	6.8	2.6	3.4	0.84
TeCDF	0.97	0.186	1.55	1.09	2.6	0.36	0.93	0.15
PeCDF	0.38	0.094	0.67	0.52	0.84	0.11	0.30	0.14
HxCDF	0.23	0.166	0.47	0.28	0.36	0.024	0.19	0.05
HpCDF	0.17	0.169	0.24	0.12	0.34	0.074	0.17	0.03
OCDF	0.069	0.043	0.076	0.041	0.16	0.033	0.086	0.023
PCDD Total	1.4	0.22	1.5	0.34	14	6.3	6.7	2.0
PCDF Total	1.8	0.60	3.0	2.0	4.3	0.48	1.7	0.30
PCDD/PCDF Total	3.2	0.81	4.5	2.2	19	6.6	8.3	2.2

Table 18. PCDD/PCDF Total Emission Factors from MREs

Homologue	MRE-100%		MRE-66%		MRE-32%		MRE-0%	
	Average	STDV	Average	STDV	Average	STDV	Average	STDV
	ng/kg MRE							
TeCDD	1.75	0.78	0.70	0.14	0.95	0.30	2.14	0.75
PeCDD	0.92	0.48	0.37	0.12	0.64	0.18	0.96	0.13
HxCDD	0.95	0.23	0.64	0.10	0.83	0.11	1.13	0.16
HpCDD	1.68	0.12	1.49	0.09	1.52	0.08	2.08	0.09
OCDD	2.90	0.34	2.71	0.10	2.48	0.12	3.31	0.30
TeCDF	59.03	25.14	25.75	8.04	56.98	15.02	92.11	48.13
PeCDF	8.59	5.32	2.43	1.21	9.05	2.20	13.66	6.51
HxCDF	1.86	0.75	0.66	0.25	1.96	0.43	2.37	0.85
HpCDF	0.44	0.07	0.25	0.02	0.50	0.14	0.49	0.16
OCDF	0.22	0.08	0.16	0.03	0.22	0.03	0.25	0.04
PCDD Total	8.21	1.61	5.91	0.52	6.42	0.51	9.61	0.96
PCDF Total	70.14	31.29	29.25	9.35	68.71	17.22	108.88	55.61
PCDD/PCDF Total	78.36	31.67	35.16	9.33	75.13	17.64	118.49	55.98

Table 19. PAH TEQ Emission Factors from Fiberboards in mg B[a]P TEQ/kg fiberboard

Compound	SFP		SFNP		CFP		CFNP	
	Average	STDV	Average	STDV	Average	STDV	Average	STDV
	mg B[a]P TEQ/kg fiberboard							
Naphthalene	NA	NA	NA	NA	NA	NA	NA	NA
Acenaphthylene	NA	NA	NA	NA	NA	NA	NA	NA
Acenaphthene	NA	NA	NA	NA	NA	NA	NA	NA
Fluorene	NA	NA	NA	NA	NA	NA	NA	NA
Phenanthrene	2.1E-03	1.5E-03	5.1E-03	2.4E-03	3.4E-03	1.5E-03	2.2E-03	6.4E-04
Anthracene	4.4E-04	3.2E-04	1.2E-03	6.3E-04	8.4E-04	4.3E-04	5.0E-04	1.6E-04
Fluoranthene	9.8E-02	4.6E-02	2.5E-01	1.3E-01	1.3E-01	5.9E-02	8.8E-02	1.4E-02
Pyrene	1.8E-03	8.0E-04	4.5E-03	2.5E-03	2.4E-03	1.1E-03	1.7E-03	2.4E-04
Benzo(a)anthracene	2.2E-03	1.4E-03	5.9E-03	3.3E-03	3.7E-03	1.7E-03	2.2E-03	6.3E-04
Chrysene	1.3E-02	8.4E-03	3.2E-02	1.7E-02	2.2E-02	1.0E-02	1.2E-02	3.3E-03
Benzo(b)fluoranthene	2.4E-02	1.4E-02	6.2E-02	3.2E-02	3.3E-02	1.6E-02	2.3E-02	6.8E-03
Benzo(k)fluoranthene	1.5E-02	9.2E-03	4.5E-02	2.3E-02	2.0E-02	8.3E-03	1.3E-02	2.7E-03
Benzo(a)pyrene	3.4E-01	2.2E-01	1.0E+00	5.5E-01	4.6E-01	2.2E-01	3.1E-01	6.7E-02
Indeno(1,2,3-cd)pyrene	2.2E-02	1.3E-02	6.6E-02	3.3E-02	2.8E-02	1.3E-02	2.0E-02	3.5E-03
Dibenz(a,h)anthracene	2.5E-02	1.8E-02	6.2E-02	7.3E-02	6.4E-02	5.5E-02	3.4E-02	2.4E-02
Benzo(ghi)perylene	5.3E-03	3.5E-03	1.5E-02	7.3E-03	5.4E-03	2.3E-03	4.2E-03	7.0E-04
SUM 16-EPA PAH	5.4E-01	3.1E-01	1.6E+00	8.2E-01	7.7E-01	3.8E-01	5.1E-01	1.2E-01

Table 20. PAH TEQ Emission Factors from MRE Packaging in mg B[a]P TEQ/kg MRE

Compound	MRE-0%		MRE-32%		MRE-66%		MRE-100%	
	Average	STDV	Average	STDV	Average	STDV	Average	STDV
	mg B[a]P TEQ/kg MRE							
Naphthalene	NA	NA	NA	NA	NA	NA	NA	NA
Acenaphthylene	NA	NA	NA	NA	NA	NA	NA	NA
Acenaphthene	NA	NA	NA	NA	NA	NA	NA	NA
Fluorene	NA	NA	NA	NA	NA	NA	NA	NA
Phenanthrene	7.9E-03	3.5E-03	5.6E-03	9.8E-04	6.9E-03	1.6E-03	8.2E-03	3.7E-03
Anthracene	1.3E-03	5.1E-04	9.9E-04	1.9E-04	1.2E-03	1.8E-04	1.5E-03	7.6E-04
Fluoranthene	1.6E-01	7.2E-02	1.2E-01	1.8E-02	1.5E-01	1.4E-03	1.8E-01	7.1E-02
Pyrene	2.2E-03	9.7E-04	1.9E-03	3.9E-04	2.1E-03	1.8E-04	2.9E-03	1.3E-03
Benzo(a)anthracene	6.3E-03	3.2E-03	4.1E-03	1.0E-03	5.5E-03	1.2E-04	6.8E-03	3.4E-03
Chrysene	4.5E-02	2.4E-02	3.0E-02	6.5E-03	4.1E-02	6.2E-04	4.6E-02	2.3E-02
Benzo(b)fluoranthene	9.1E-02	4.5E-02	5.7E-02	1.4E-02	7.7E-02	1.6E-03	8.5E-02	4.5E-02
Benzo(k)fluoranthene	4.1E-02	2.4E-02	3.0E-02	5.3E-03	3.6E-02	5.2E-03	4.4E-02	1.8E-02
Benzo(a)pyrene	7.9E-01	4.3E-01	5.6E-01	1.4E-01	6.5E-01	7.9E-02	8.6E-01	4.4E-01
Indeno(1,2,3-cd)pyrene	5.7E-02	3.0E-02	3.6E-02	8.8E-03	4.3E-02	7.2E-03	5.9E-02	2.7E-02
Dibenz(a,h)anthracene	2.3E-01	1.2E-01	1.3E-01	3.4E-02	1.5E-01	5.2E-03	2.1E-01	1.1E-01
Benzo(ghi)perylene	1.0E-02	4.9E-03	7.5E-03	2.1E-03	7.5E-03	2.2E-03	1.2E-02	5.5E-03
SUM 16-EPA PAH	1.4E+00	7.6E-01	9.8E-01	2.2E-01	1.2E+00	9.3E-02	1.5E+00	7.5E-01

Table 21. VOC Emission Factors from Fiberboard in mg/kg fiberboard^a

	SFP	SFNP	CFP	CFNP		SFP	SFNP	CFP	CFNP
Modified combustion efficiency (MCE, fraction)	0.983 ±0.006	0.981 ±0.004	0.977 ±0.010	0.978 ±0.006	Modified combustion efficiency (MCE, fraction)	0.983 ±0.006	0.981 ±0.004	0.977 ±0.010	0.978 ±0.006
Propene	38±42	51±9.4	118±94	84±57	1,4-Dioxane	ND [0.12]	ND [0.19]	ND [0.29]	ND [0.19]
Dichlorodifluoromethane (CFC 12)	ND [0.13]	0.024	ND [0.31]	0.044±0.0093 ^c	Methyl Methacrylate ^b	ND [0.24]	ND [0.38]	17±13	0.50
Chloromethane ^b	2.5±1.5	0.63±0.11	5.6±3.7	1.8±0.70	n-Heptane	ND [0.13]	ND [0.20]	0.18	0.64
1,2-Dichloro-1,1,2,2- tetrafluoroethane (CFC 114)	ND [0.15]	ND [0.23]	ND [0.34]	ND [0.23]	cis-1,3-Dichloropropene	ND [0.11]	ND [0.17]	ND [0.26]	ND [0.17]
Vinyl Chloride ^b	0.22	ND [0.20]	ND [0.31]	ND [0.21]	4-Methyl-2-pentanone	ND [0.12]	ND [0.19]	ND [0.29]	0.34
1,3-Butadiene ^b	19±22	24±5.3	44±43	34±23	trans-1,3-Dichloropropene	ND [0.12]	ND [0.19]	ND [0.29]	ND [0.19]
Bromomethane ^b	ND [0.15]	ND [0.23]	ND [0.34]	ND [0.23]	1,1,2-Trichloroethane	ND [0.12]	ND [0.19]	ND [0.29]	ND [0.19]
Chloroethane ^b	ND [0.13]	ND [0.20]	ND [0.31]	ND [0.21]	Toluene ^b	19±19	26±4.6	57±44	28±16
Ethanol	2.1±2.6	2.8	3.3±2.9 ^c	5.0±3.6	2-Hexanone	0.20±0.16	0.27±0.040 ^c	0.51±0.29	0.36±0.21
Acetonitrile	1.2±1.1	0.62±0.10	3.0±2.0	1.6±0.84	Dibromochloromethane	ND [0.12]	ND [0.19]	ND [0.29]	ND [0.19]
Acrolein ^b	9.4±4.8	36±5.9	81±64	76±60	1,2-Dibromoethane	ND [0.12]	ND [0.19]	ND [0.29]	ND [0.19]
Acetone	35±42	41	ND [1.4]	128	n-Butyl Acetate	ND [0.12]	ND [0.19]	ND [0.29]	ND [0.19]
Trichlorofluoromethane	ND [0.13]	0.024	ND [0.31]	0.026	n-Octane	ND [0.14]	ND [0.22]	ND [0.33]	0.45
2-Propanol (Isopropyl Alcohol)	ND [0.32]	ND [0.50]	ND [0.77]	ND [0.52]	Tetrachloroethene	ND [0.11]	ND [0.17]	0.66	ND [0.17]
Acrylonitrile ^b	0.54±0.51	0.27±0.072	1.5±1.0	0.55±0.30	Chlorobenzene ^b	ND [0.12]	ND [0.19]	ND [0.29]	ND [0.19]
1,1-Dichloroethene	ND [0.13]	ND [0.20]	ND [0.31]	ND [0.21]	Ethylbenzene	2.4±2.2	4.1±0.60	15±13	4.6±2.9
Methylene Chloride ^b	5.5±5.3	6.7±5.3	2.2±2.1 ^c	3.1±3.4	m,p-Xylenes ^b	1.9±1.7	3.7±0.82	6.9±6.1	5.5±3.5
3-Chloro-1-propene (Allyl Chloride) ^b	ND [0.12]	ND [0.19]	ND [0.29]	ND [0.19]	Bromoform	ND [0.12]	ND [0.18]	ND [0.28]	ND [0.19]
Trichlorotrifluoroethane	0.010	ND [0.20]	ND [0.31]	ND [0.21]	Styrene ^b	5.6±2.5	13±1.6	60±64	13±7.5
Carbon Disulfide ^b	1.4±0.68	1.8±0.74	1.1±0.95	1.1±1.2	o-Xylene ^b	0.74±0.58	1.5±0.32	2.9±2.6	2.2±1.4
trans-1,2-Dichloroethene	ND [0.15]	ND [0.23]	ND [0.34]	ND [0.23]	n-Nonane	ND [0.12]	ND [0.18]	ND [0.28]	ND [0.19]

Table 21 (continued). VOC Emission Factors from Fiberboard in mg/kg fiberboard^a

	SFP	SFNP	CFP	CFNP		SFP	SFNP	CFP	CFNP
Modified combustion efficiency (MCE, fraction)	0.983 ±0.006	0.981 ±0.004	0.977 ±0.010	0.978 ±0.006	Modified combustion efficiency (MCE, fraction)	0.983 ±0.006	0.981 ±0.004	0.977 ±0.010	0.978 ±0.006
1,1-Dichloroethane	ND [0.12]	ND [0.19]	ND [0.29]	ND [0.19]	1,1,2,2-Tetrachloroethane	ND [0.12]	ND [0.18]	ND [0.28]	ND [0.19]
Methyl tert-Butyl Ether	ND [0.13]	ND [0.20]	ND [0.31]	ND [0.21]	Cumene ^b	0.33	0.20±0.027	1.5±1.4	0.29±0.12
Vinyl Acetate ^b	33±13	48±9.7 ^c	98±70	50	alpha-Pinene	0.27±0.18	0.84±1.1	0.33±0.19	0.37±0.17
2-Butanone (MEK)	9.3±11	9.6±3.0	23±19	24±16	n-Propylbenzene	ND [0.12]	ND [0.19]	0.71±0.65	0.31±0.10
cis-1,2-Dichloroethene	ND [0.12]	ND [0.19]	ND [0.29]	ND [0.19]	4-Ethyltoluene	0.10±0.020	0.36±0.084	0.53±0.50	0.49±0.31
Ethyl Acetate	0.088	1.0	0.57	2.3	1,3,5-Trimethylbenzene	ND [0.12]	0.23	0.51±0.37 ^c	0.35±0.11
n-Hexane	0.11±0.11	0.35±0.21	0.25±0.10	0.68±0.88	1,2,4-Trimethylbenzene	0.14±0.050	0.53±0.15	0.85±0.95	0.90±0.57
Chloroform ^b	ND [0.13]	ND [0.20]	ND [0.31]	ND [0.21]	Benzyl Chloride ^b	ND [0.086]	ND [0.13]	ND [0.20]	ND [0.14]
Tetrahydrofuran (THF)	0.61±0.50 ^c	0.44±0.20	0.70±0.61	0.71±0.54	1,3-Dichlorobenzene	ND [0.12]	ND [0.18]	ND [0.28]	ND [0.19]
1,2-Dichloroethane ^b	ND [0.12]	ND [0.19]	ND [0.29]	ND [0.19]	1,4-Dichlorobenzene	ND [0.11]	ND [0.17]	ND [0.26]	ND [0.17]
1,1,1-Trichloroethane ^b	ND [0.13]	ND [0.20]	ND [0.31]	ND [0.21]	1,2-Dichlorobenzene	ND [0.12]	ND [0.18]	ND [0.28]	ND [0.19]
Benzene ^b	79±75	111±16	153±102	91±49	d-Limonene	0.14±0.043	0.79±0.90	0.42	0.30±0.15
Carbon Tetrachloride ^b	0.013±0.0045	0.015	ND [0.28]	ND [0.19]	1,2-Dibromo-3-chloropropane	ND [0.078]	ND [0.12]	ND [0.18]	ND [0.12]
Cyclohexane	ND [0.23]	ND [0.35]	ND [0.53]	ND [0.35]	1,2,4-Trichlorobenzene	ND [0.12]	ND [0.19]	ND [0.29]	ND [0.19]
1,2-Dichloropropane	ND [0.12]	ND [0.19]	ND [0.29]	ND [0.19]	Naphthalene	8.9±5.7	34±2.8	25±23	20
Bromodichloromethane	ND [0.12]	ND [0.18]	ND [0.28]	ND [0.19]	Hexachlorobutadiene	ND [0.11]	ND [0.17]	ND [0.26]	ND [0.17]
Trichloroethene	ND [0.11]	ND [0.17]	ND [0.26]	ND [0.17]					

^a Range of data = ±1 STDV

^b On EPA's list of hazardous air pollutants (US EPA, 2008).

^c Relative difference.

No range of data = only one sample with detectable levels

ND = not detectable. Detection limit within brackets.

Table 22. VOC Emission Factors from MRE Packaging in mg/kg MRE^a

	MRE- 100%	MRE- 66%	MRE- 32%	MRE- 0%		MRE- 100%	MRE- 66%	MRE- 32%	MRE- 0%
Modified combustion efficiency (MCE, fraction)	0.979 ±0.002	0.981 ±0.002 ^c	0.976 ±0.001	0.977 ±0.003	Modified combustion efficiency (MCE, fraction)	0.979 ±0.002	0.981 ±0.002 ^c	0.976 ±0.001	0.977 ±0.003
Propene	286±75	253±3.3 ^c	329±16	268±83	1,4-Dioxane	ND [0.23]	ND [0.22]	0.25	0.11
Dichlorodifluoromethane (CFC 12)	0.073	ND [0.24]	ND [0.11]	0.10	Methyl Methacrylate ^b	ND [0.43]	ND [0.42]	ND [0.20]	ND [0.41]
Chloromethane ^b	3.1±0.43	3.2±1.3 ^c	2.3±0.25	3.2±1.9	n-Heptane	10±3.2	10±2.5 ^c	12±0.98	10±3.1
1,2-Dichloro-1,1,2,2-tetrafluoroethane (CFC 114)	ND [0.27]	ND [0.26]	ND [0.12]	ND [0.25]	cis-1,3-Dichloropropene	ND [0.20]	ND [0.19]	ND [0.09]	ND [0.19]
Vinyl Chloride ^b	0.52±0.0046 ^c	0.42±0.048 ^c	0.59±0.048	0.55±0.19 ^c	4-Methyl-2-pentanone	1.3±0.55	0.67	1.3	0.79±0.26
1,3-Butadiene ^b	94±55	82±2.3 ^c	126±12	103±38	trans-1,3-Dichloropropene	ND [0.23]	ND [0.22]	ND [0.10]	ND [0.21]
Bromomethane ^b	0.38±0.095 ^c	0.22	0.33±0.11	0.45±0.25 ^c	1,1,2-Trichloroethane	ND [0.23]	ND [0.22]	0.13	0.092
Chloroethane ^b	0.28±0.022 ^c	0.23	0.25±0.038	0.16±0.072 ^c	Toluene ^b	58±18	52±11 ^c	79±37	63±36
Ethanol	5.2±0.78 ^c	5.5	1.6	3.1±1.3	2-Hexanone	5.3±1.0	5.0±1.3 ^c	5.6±0.82	4.9±1.1
Acetonitrile	13±1.9	8.8±2.0 ^c	12±1.3	7.6±2.5	Dibromochloromethane	ND [0.23]	ND [0.22]	ND [0.10]	ND [0.21]
Acrolein ^b	126±27	126±12 ^c	145±16	125±40	1,2-Dibromoethane	ND [0.23]	ND [0.22]	ND [0.10]	ND [0.21]
Acetone	115	88	103±1.9 ^c	126	n-Butyl Acetate	ND [0.23]	ND [0.22]	0.23±0.050 ^c	0.15
Trichlorofluoromethane	ND [0.24]	ND [0.24]	ND [0.11]	ND [0.22]	n-Octane	7.7±2.7	7.4±1.5 ^c	9.4	7.7±2.0
2-Propanol (Isopropyl Alcohol)	2.0	ND [0.58]	ND [0.27]	ND [0.56]	Tetrachloroethene	ND [0.20]	ND [0.19]	ND [0.09]	ND [0.19]
Acrylonitrile ^b	8.4±0.48 ^c	5.2±1.1 ^c	7.1±0.43	5.6±2.0	Chlorobenzene ^b	0.18±0.049 ^c	ND [0.22]	0.24±0.044 ^c	0.21±0.0066 ^c
1,1-Dichloroethene	ND [0.24]	ND [0.24]	ND [0.11]	ND [0.22]	Ethylbenzene	15±6.0	18±0.33 ^c	24±14	16±0.51 ^c
Methylene Chloride ^b	17±14 ^c	ND [0.24]	ND [0.11]	8.5	m,p-Xylenes ^b	5.7±1.4	4.8±0.71 ^c	6.0±0.93	5.4±0.10 ^c
3-Chloro-1-propene (Allyl Chloride) ^b	ND [0.23]	ND [0.22]	ND [0.10]	ND [0.21]	Bromoform	ND [0.22]	ND [0.21]	ND [0.10]	ND [0.21]
Trichlorotrifluoroethane	ND [0.24]	ND [0.24]	ND [0.11]	ND [0.22]	Styrene ^b	120±79	210±22 ^c	283±198	206±192

Table 22 (continued). VOC Emission Factors from MRE Packaging in mg/kg MRE^a

	MRE- 100%	MRE- 66%	MRE- 32%	MRE- 0%		MRE- 100%	MRE- 66%	MRE- 32%	MRE- 0%
Modified combustion efficiency (MCE, fraction)	0.979 ±0.002	0.981 ±0.002 ^c	0.976 ±0.001	0.977 ±0.003	Modified combustion efficiency (MCE, fraction)	0.979 ±0.002	0.981 ±0.002 ^c	0.976 ±0.001	0.977 ±0.003
Carbon Disulfide ^b	ND [0.22]	ND [0.21]	0.59	32	o-Xylene ^b	3.1±0.72	2.6±0.39 ^c	3.2±0.53	2.7±0.36
trans-1,2-Dichloroethene	0.20	ND [0.26]	0.26±0.030 ^c	0.19	n-Nonane	6.1±2.4	6.2±1.0 ^c	7.8±0.66	6.2±1.8
1,1-Dichloroethane	ND [0.23]	ND [0.22]	ND [0.10]	ND [0.21]	1,1,2,2-Tetrachloroethane	ND [0.22]	ND [0.21]	0.13	ND [0.21]
Methyl tert-Butyl Ether	ND [0.24]	ND [0.24]	ND [0.11]	ND [0.22]	Cumene ^b	0.82±0.33	1.2±0.061 ^c	1.5±0.89	0.95±0.69
Vinyl Acetate ^b	109±27	93±16 ^c	111±24	101±41	alpha-Pinene	0.71±0.16 ^c	0.87±0.14 ^c	0.81±0.10 ^c	0.85±0.30 ^c
2-Butanone (MEK)	24±7.8	25±4.8 ^c	26±3.8	23±6.9	n-Propylbenzene	1.49±0.40	1.5±0.38 ^c	1.8±0.26 ^c	1.2±0.29
cis-1,2-Dichloroethene	0.31	ND [0.22]	ND [0.10]	ND [0.21]	4-Ethyltoluene	0.70±0.19	0.56±0.077 ^c	0.77±0.030 ^c	0.63±0.10
Ethyl Acetate	9.5	ND [0.47]	ND [0.23]	ND [0.46]	1,3,5-Trimethylbenzene	0.50±0.17 ^c	0.25	0.46±0.0075 ^c	0.41±0.014 ^c
n-Hexane	11±4.6	9.0±8.0 ^c	12±1.2	10±3.0	1,2,4-Trimethylbenzene	1.1±0.63	0.86±0.16 ^c	1.2±0.055 ^c	0.79±0.076
Chloroform ^b	0.29	0.29	0.25	ND [0.22]	Benzyl Chloride ^b	0.18	0.17	0.14±0.019 ^c	0.19±0.049 ^c
Tetrahydrofuran (THF)	2.6±0.82	2.8±0.27 ^c	2.7±0.18	2.5±0.61	1,3-Dichlorobenzene	ND [0.22]	ND [0.21]	ND [0.10]	ND [0.21]
1,2-Dichloroethane ^b	ND [0.23]	ND [0.22]	ND [0.10]	ND [0.21]	1,4-Dichlorobenzene	ND [0.20]	ND [0.19]	ND [0.10]	ND [0.19]
1,1,1-Trichloroethane ^b	ND [0.24]	ND [0.24]	ND [0.11]	ND [0.22]	1,2-Dichlorobenzene	ND [0.20]	ND [0.21]	ND [0.10]	ND [0.21]
Benzene ^b	225±36	182±12 ^c	266±10	211±89	d-Limonene	0.94±0.17 ^c	0.96±0.30 ^c	0.98±0.25 ^c	1.0±0.074 ^c
Carbon Tetrachloride ^b	0.023±0.011 ^c	0.055	0.016±0.0074	0.024±0.0022 ^c	1,2-Dibromo-3-chloropropane	ND [0.14]	ND [0.14]	ND [0.06]	ND [0.13]
Cyclohexane	0.48	ND [0.39]	0.51	0.30	1,2,4-Trichlorobenzene	ND [0.23]	ND [0.22]	ND [0.10]	ND [0.21]
1,2-Dichloropropane	0.21	ND [0.22]	0.28	0.22	Naphthalene	38±5.3	35±11 ^c	35±3.2	35±7.3
Bromodichloromethane	ND [0.22]	ND [0.21]	ND [0.10]	ND [0.21]	Hexachlorobutadiene	ND [0.20]	ND [0.19]	ND [0.09]	ND [0.19]
Trichloroethene	ND [0.20]	ND [0.19]	ND [0.09]	ND [0.19]					

^a Range of data = ±1 STDV

^b On EPA's list of hazardous air pollutants (US EPA, 2008).

^c Relative difference.

No range of data = only one sample with detectable levels

ND = not detectable. Detection limit within brackets.

MRE-0% iron emissions (0.30 ± 0.03 mg/kg waste) were comparable to incinerator (0.31 ± 0.18 mg/kg waste) (Barnes et al., 2016) and air curtain burner (0.50 ± 0.24 mg/kg waste) emissions, but less than burn pile (11 ± 23 mg/kg waste) emissions (Aurell et al., 2012). MRE-100% copper emissions (0.21 ± 0.08 mg/kg waste) were equivalent to air curtain burner (0.18 ± 0.11 mg/kg waste) emissions, but were less than incinerator (1.2 ± 0.33 mg/kg waste) (Barnes et al., 2016) and burn pile (0.89 ± 0.92 mg/kg waste) military waste emissions (Aurell et al., 2012). Cadmium emissions were similar to Aurell et al. (Aurell et al., 2012) air curtain burner and burn pile emissions, but were at least an order of magnitude greater than Barnes et al. (Barnes et al., 2016) incinerator (0.0058 ± 0.0031 mg/kg waste) emissions. MRE-0% lead levels (0.07 ± 0.01 mg/kg waste) were significantly lower than all three other studies, with the burn pile (0.37 ± 0.22 mg/kg waste), air curtain burner (0.55 ± 0.42 mg/kg waste) (Aurell et al., 2012), and incinerator (0.66 ± 0.35 mg/kg waste) emissions (Barnes et al., 2016) all approximately equivalent.

Table 23. MRE Emission Factors for Metals in mg/kg-waste

Pollutant	MRE-0%	MRE-32%	MRE-66%	MRE-100%
Aluminum	2.7 ± 0.78	2.3 ± 0.30	2.8 ± 0.75	2.3 ± 0.43
Iron	0.29 ± 0.046	0.31 ± 0.11	0.51 ± 0.45	0.30 ± 0.030
Chromium	0.040 ± 0.025	0.027 ± 0.0058^a	0.15 ± 0.19	0.059 ± 0.027
Lead	0.071 ± 0.012	0.045 ± 0.016	0.086 ± 0.033	0.056 ± 0.044
Copper	0.17 ± 0.062	0.17 ± 0.057	0.23 ± 0.079	0.21 ± 0.075
Zinc	0.78 ± 0.24	0.74 ± 0.15	0.91 ± 0.31	0.97 ± 0.18
Chlorine	68 ± 18	59 ± 25	54 ± 18	60 ± 23

^a = Relative Percent Difference.

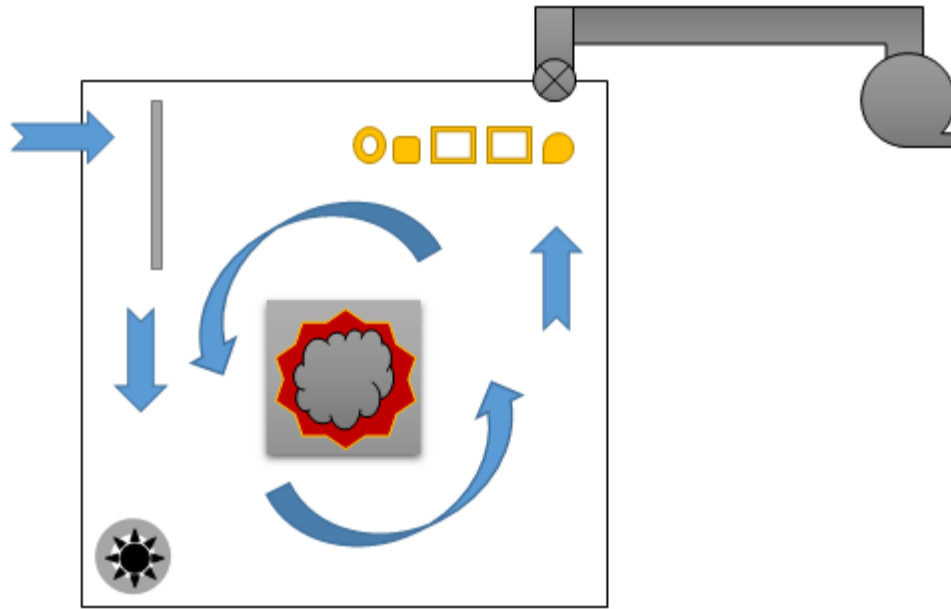


Figure 10. Plan View Schematic of the Open Burn Test Facility

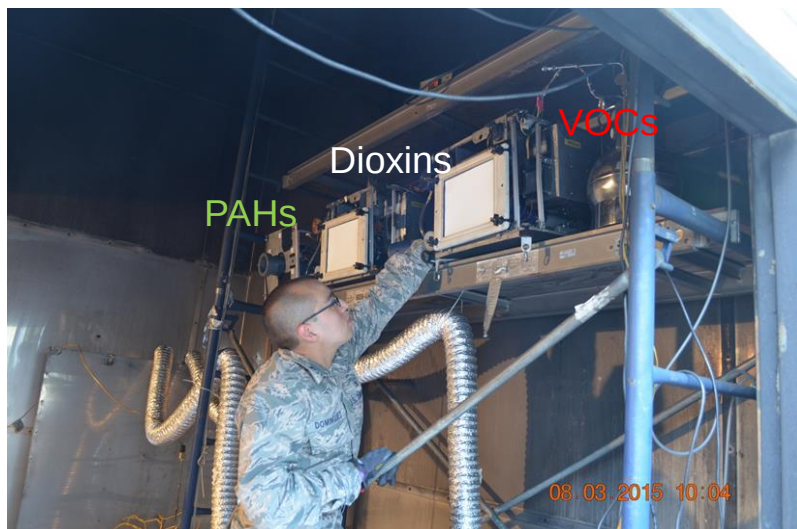


Figure 11. Flyer Sampling Instruments

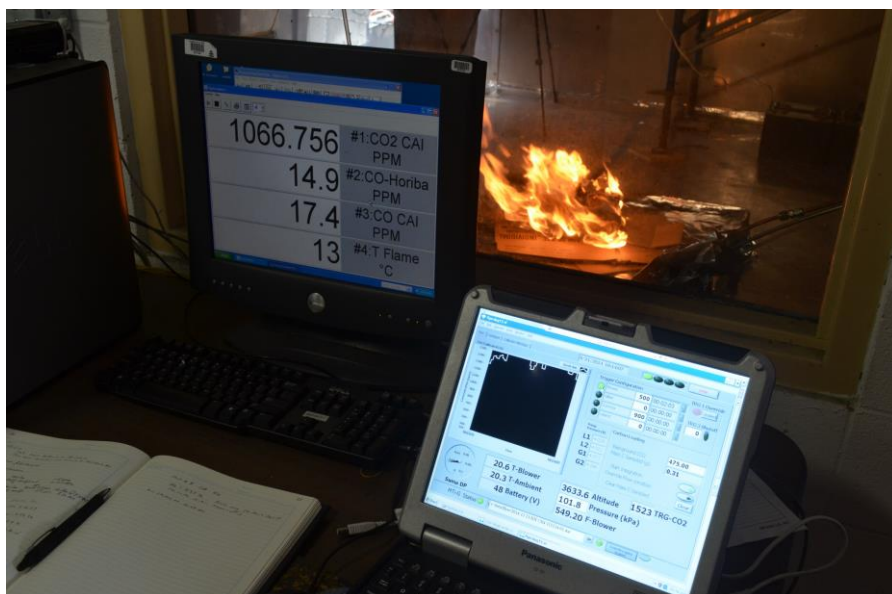


Figure 12. Monitoring of CEMs, SVOC Flow Rate, Temperature, and SUMMA[®] Canister Pressure



Figure 13. Fiberboard and MRE Test Material on 3.2 × 3.2 foot Steel Plate

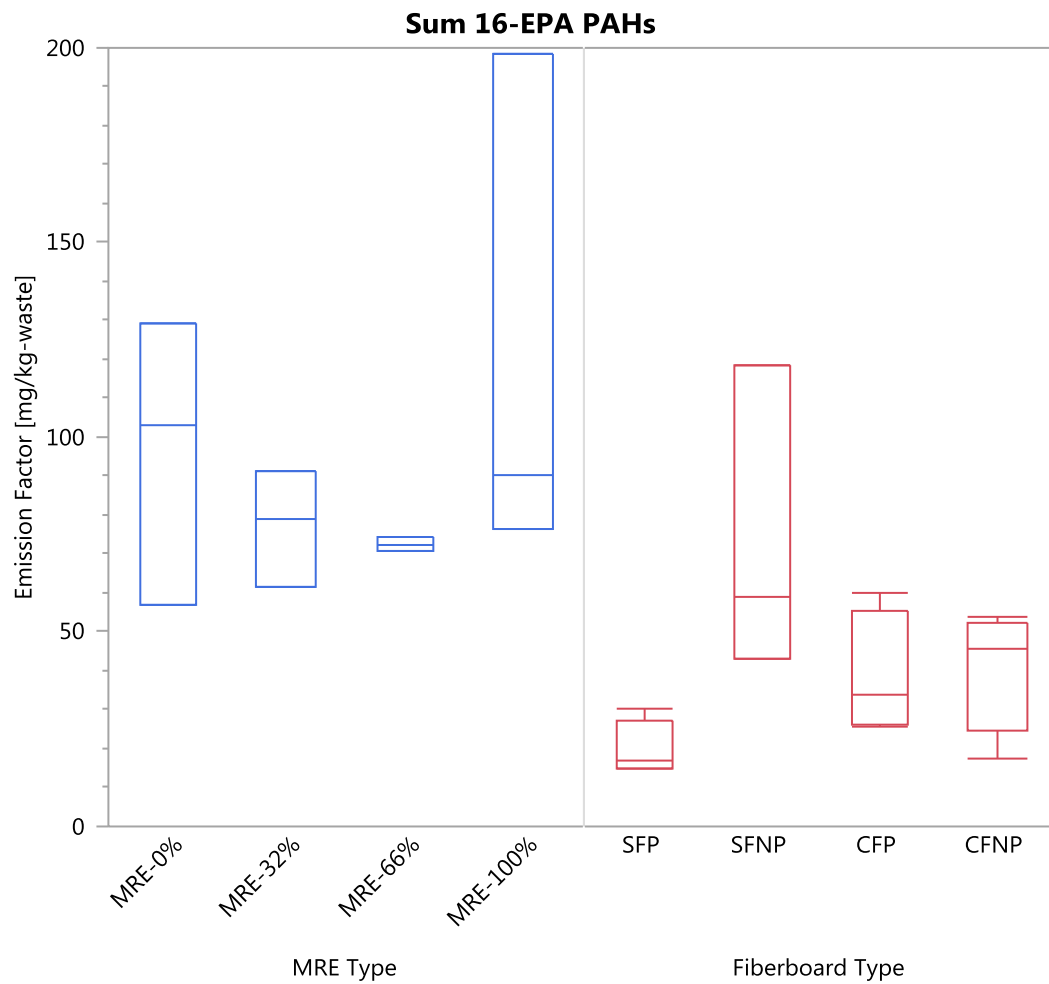


Figure 14. SUM 16-EPA PAHs by Material Type in mg/kg-waste

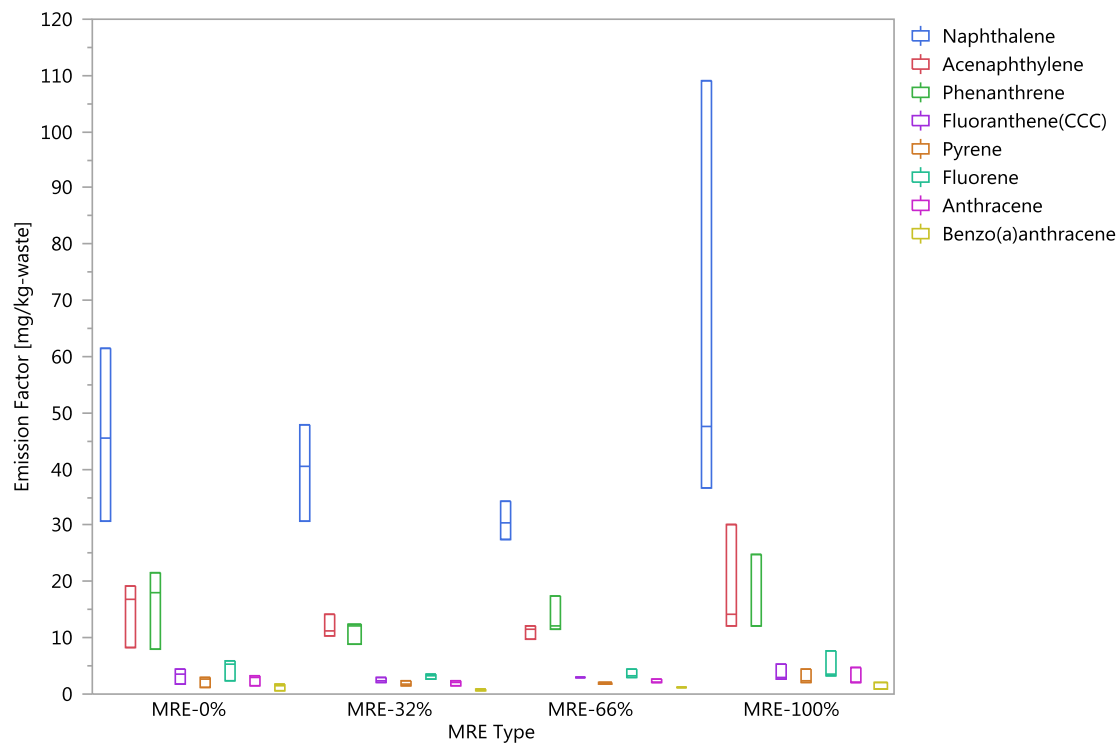


Figure 15. PAH Box Plot Emission Factor Comparison Between MRE Types

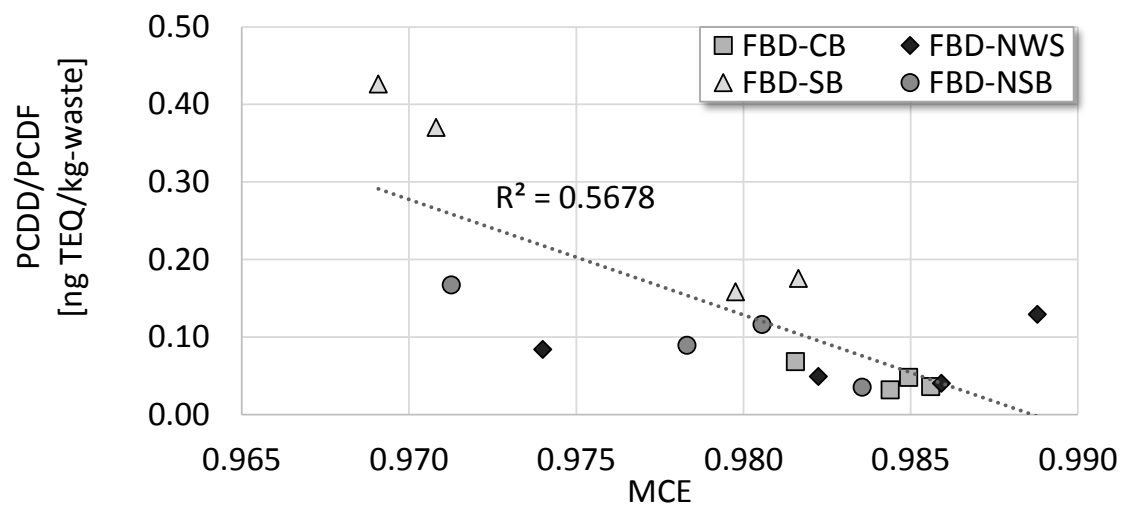


Figure 16. PCDD/PCDF TEQ Total Fiberboard Emission Factors in ng-TEQ/kg-waste

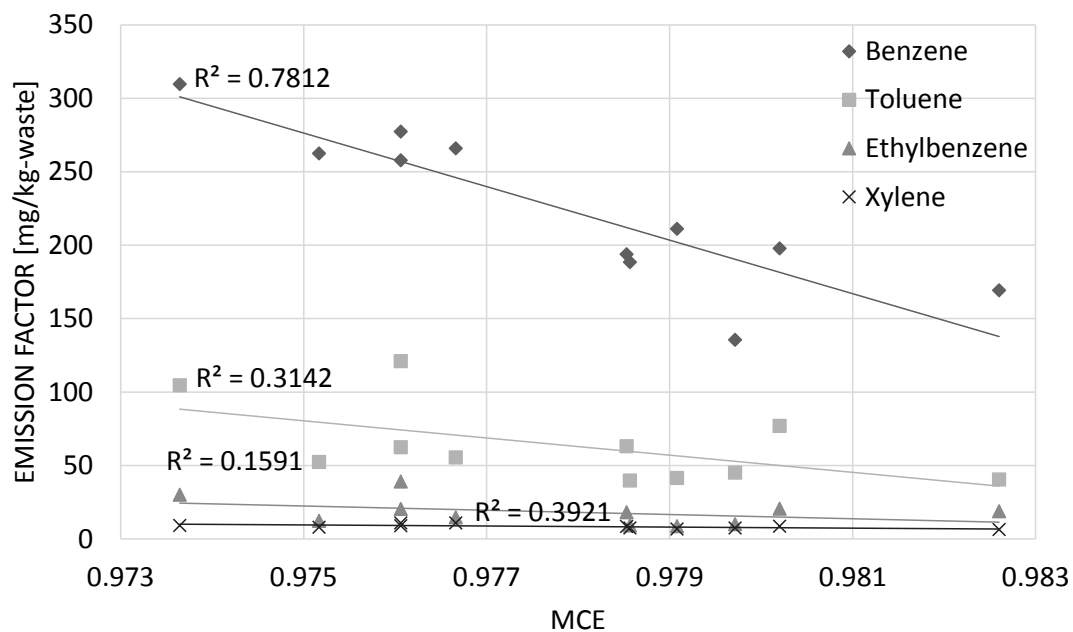


Figure 17. BTEX Emission Factors Versus MCE for All MRE Variations with Associated Coefficients of Determination

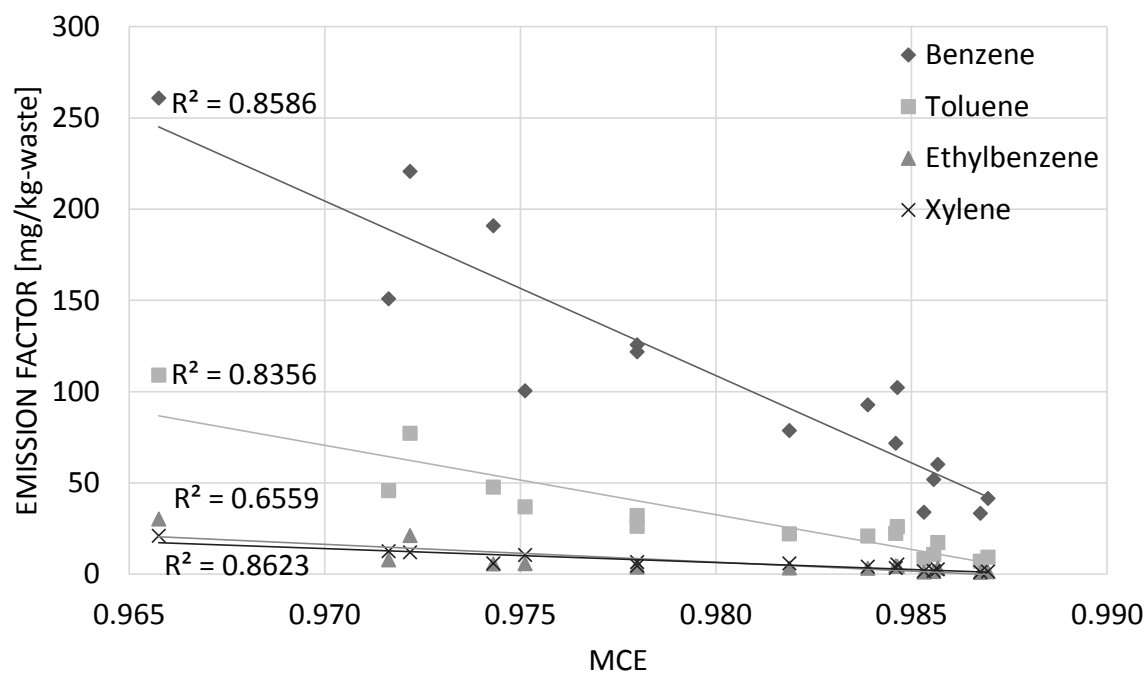


Figure 18. BTEX Emission Factors Versus MCE for All Fiberboard Variations with Associated Coefficients of Determination

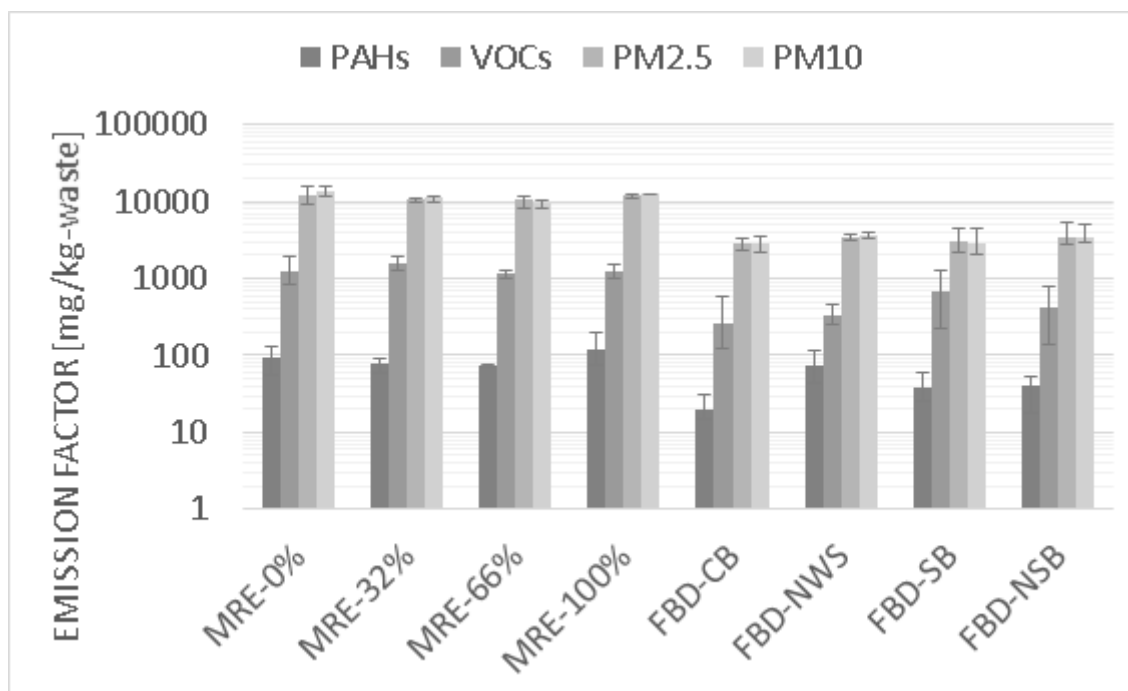


Figure 19. Log-normal comparison of total mean emission factors for PAHs, VOCs, PM2.5, and PM10 from open burning of four MRE material mixes (MRE-0% = all old Al-laminated material, 100% = all new plastic nanocomposite material) and four different fiberboard types. Error bars depict maximum and minimum average values.

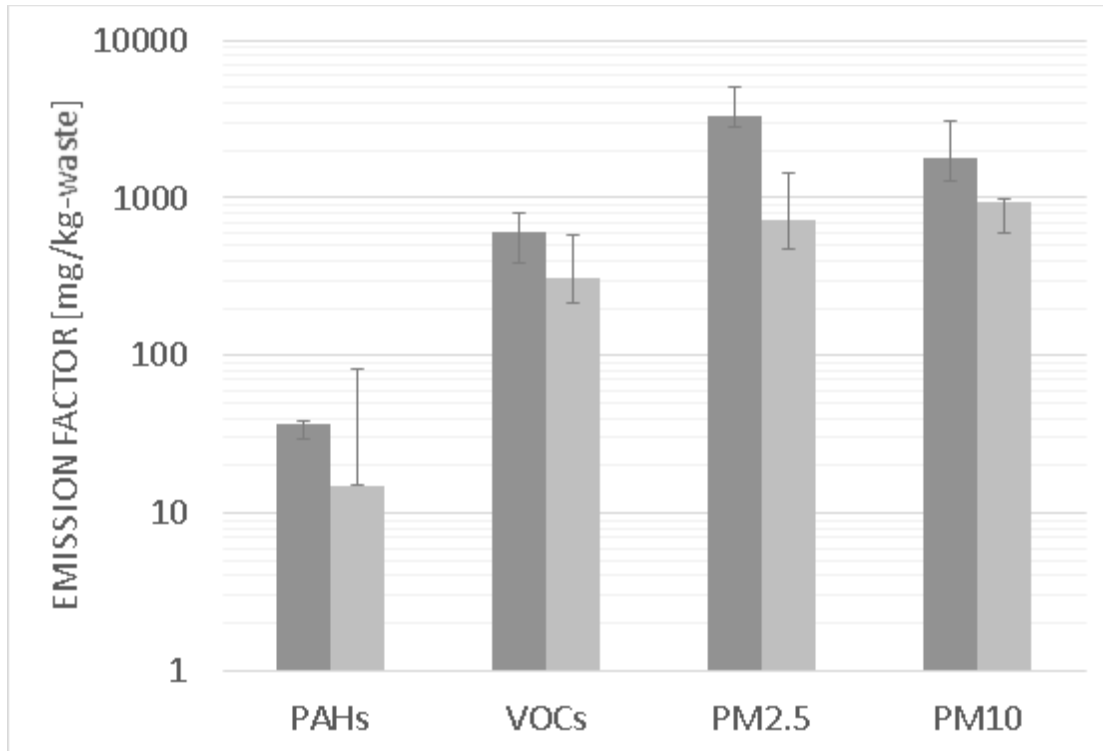


Figure 20. Log-normal comparison of combined total mean emission factors for PAHs, VOCs, PM2.5, and PM10 from open burning of four MRE material mixes (MRE-0% = all old Al-laminated material, 100% = all new plastic nanocomposite material) and four different fiberboard types. Error bars depict maximum and minimum average values across all burns.

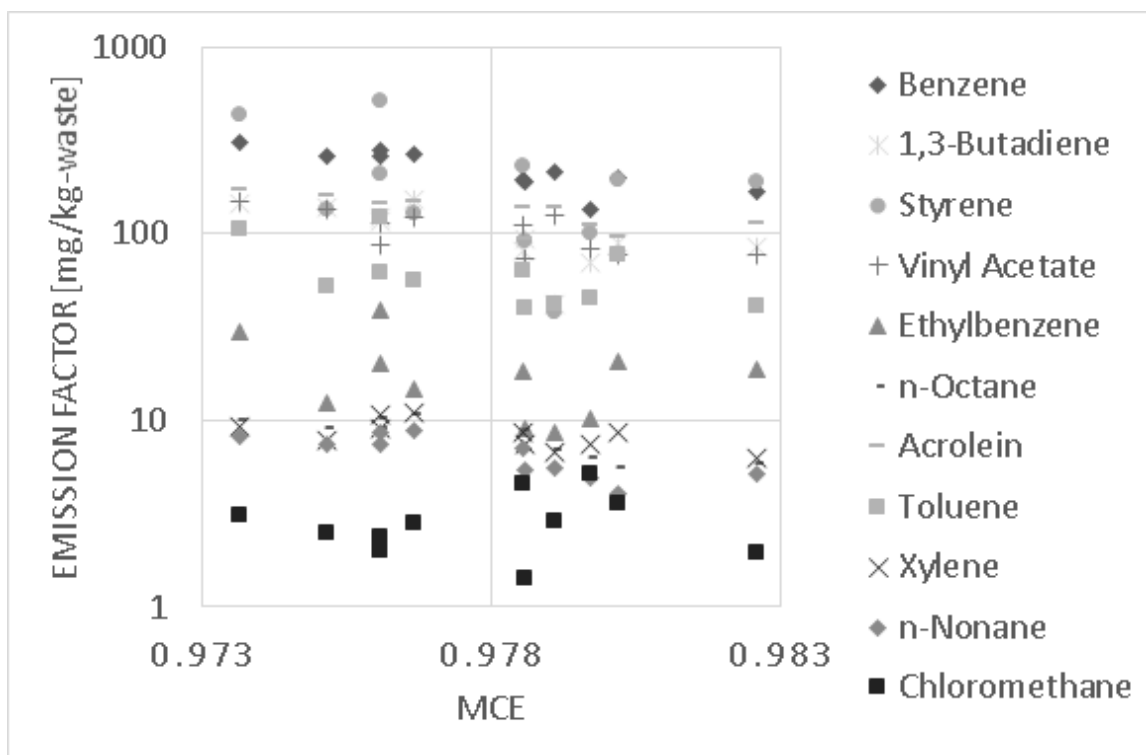


**Figure 21. MRE Emission Factors versus MCE for
Dual IARC/HAP-Classified VOCs in mg/kg-waste**

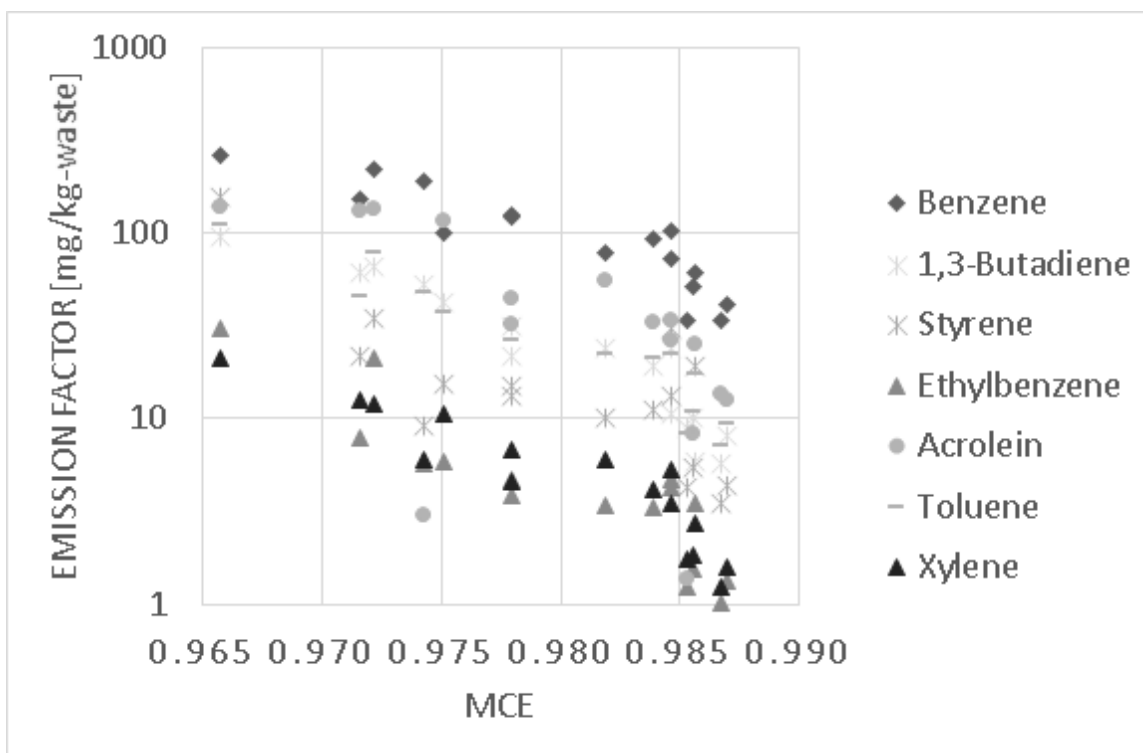


**Figure 22. Fiberboard Emission Factors versus MCE for
Dual IARC/HAP-Classified VOCs in mg/kg-waste**

When the greatest VOC emission for the fiberboards within Figure 3, the fourth and final burn for CFP, was excluded from the figure, a comparison could be made to determine the influence of that single point on the relationship between the emission factors and MCE. As can be demonstrated within Figure 23, the coefficient of determination decreased slightly for the fiberboards but still held a linear relationship. Additional sampling would further elucidate this relationship. Furthermore, as can be seen in both Figure 3 and Figure 23, the MRE MCEs were clustered between 0.973 and 0.983 while the fiberboard MCEs had a greater range from 0.965 to 0.987. These differences in MCE range between the fiberboards and the MREs are believed to be

minute (less than 1% difference) with the exclusion of the aforementioned greatest VOC emission.

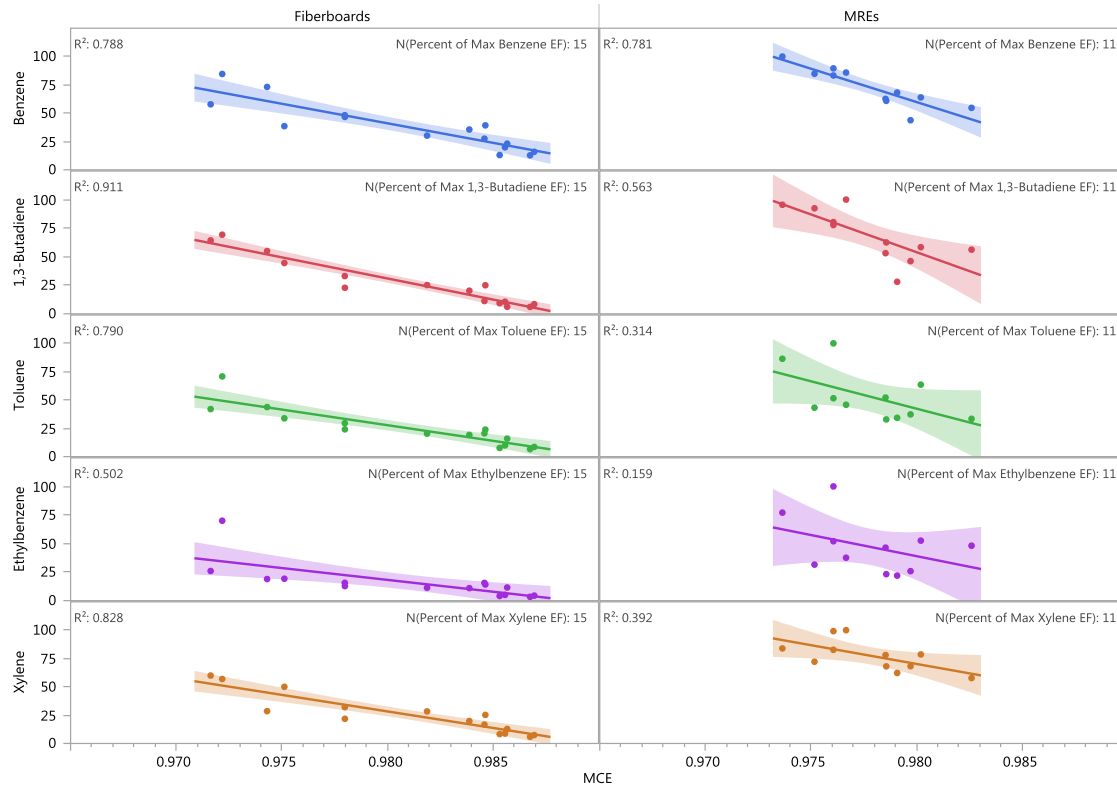


Figure 23. Percent of Maximum VOC Emission Factors vs MCE for Fiberboards and MREs with 95% Confidence Intervals, Excluding Greatest Emission for Fiberboards

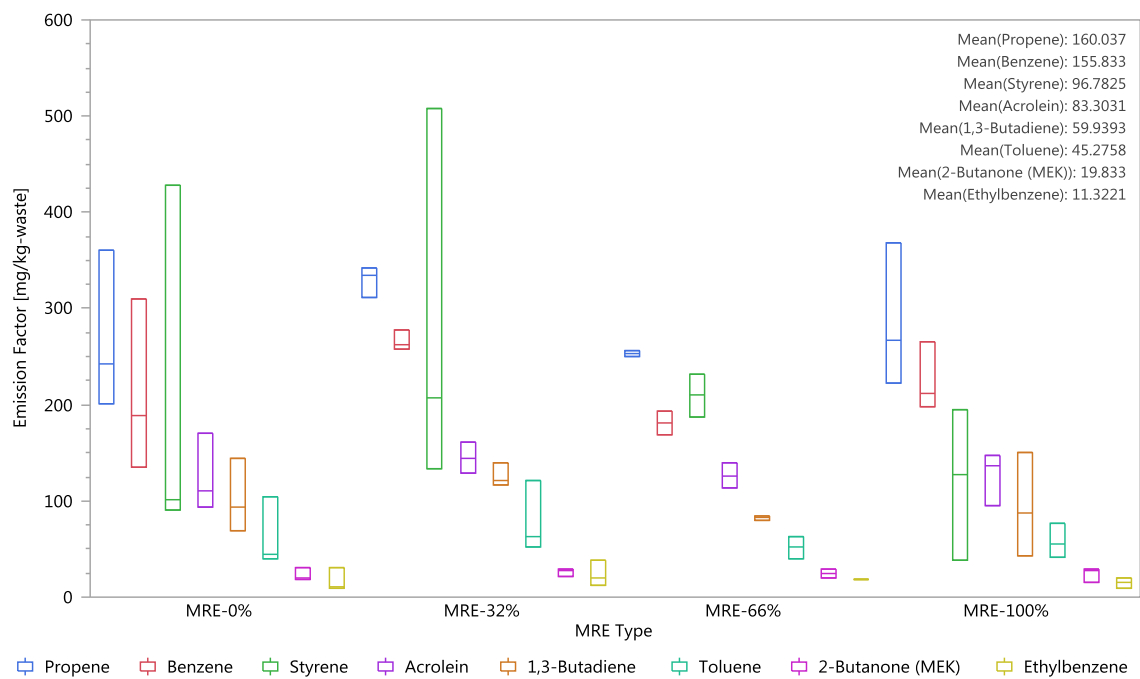


Figure 24. VOC Box Plot Emission Factor Comparison Amongst MRE Types

Appendix B: Material Composition

Overview

This appendix provides a brief background description of MREs and their fiberboard packaging. All test materials for MREs and their fiberboard packaging were procured from the US Army NSRDEC to conduct emissions testing.

Levels of Packaging

The components that make up an MRE are considered primary packaging since they come in direct contact with the food items, provide the initial barrier to the cultivation of harmful microorganisms, and maintain the oxygen and freshness quality (Barrett & Cardello, 2012). Primary packaging is the MRE, while 12 MREs are contained within a solid secondary package (i.e., the fiberboard packaging being varied within this experiment). Tertiary “stretch wrap” packaging envelops 48 secondary packaging fiberboard containers, 24 of which are menu items 1-12 and the other 24 being menu items 13-24, and were not investigated within this study (Barrett & Cardello, 2012; Defense Logistics Agency, 2015). Figure 25 demonstrates the levels of packaging.

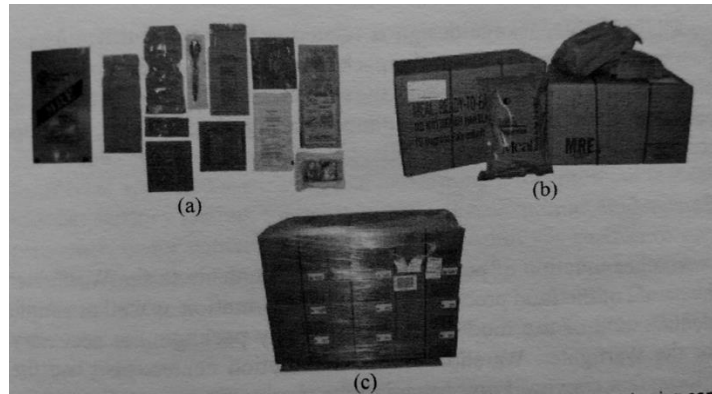


Figure 25. Primary(a), Secondary(b), and Tertiary(c) Levels of Packaging for the MRE (Barrett & Cardello, 2012)

MREs

MREs have been the DOD's military ration since 1975, are strategically prepositioned across the globe as a Class I war reserve item and are required to have an extended shelf life, with a minimum of three years at 80°F or six months at 100°F (Barrett & Cardello, 2012). They must also provide 1300 kilocalories, for a total consumption of 3900 kilocalories if all three meals are eaten in their entirety (Barrett & Cardello, 2012). Twelve MREs are contained within a single fiberboard secondary package [case], 48 cases are packed within a tertiary package [pallet]; therefore, 576 MREs are contained within one pallet, which feeds 192 personnel for one day (Barrett & Cardello, 2012; Defense Logistics Agency, 2015).

Currently, the MRE consists of primarily two kinds of multilayer, aluminum-foil-based pouches: (1) the retort pouch and, (2) the non-retort pouch. Retort simply refers to the process of utilizing high temperature and pressures to achieve sterilization; thermal retorting has been utilized since the 1940s to sterilize the main entrée within military

rations (Barrett & Cardello, 2012). Both pouches can be found depicted within Figure 26.

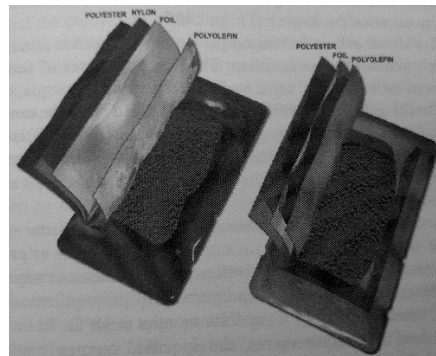


Figure 26. Retort(Left) and Non-retort(Right) MRE Pouches (Barrett & Cardello, 2012)

Both the retort and non-retort pouches consist of multiple layers that provide various capabilities. The retort pouch, which is commonly used for entrée and side items, consists of the following components from the outermost to innermost layer: (1) polyester; (2) polyamide (nylon); (3) aluminum foil; and, (4) polyolefin. The non-retort pouch is layered identically as the retort pouch, except with the exclusion of the nylon layer, and is utilized for food products that do not require sterilization, such as baked goods. The nylon is primarily introduced into the retort pouch in order to provide puncture resistance in extreme cold weather environments from potentially freezing liquids while the aluminum foil provides an excellent barrier to gas exchange and light penetration, thereby preventing spoilage for extended periods of time (Barrett & Cardello, 2012; NSRDEC, 2009). The outermost layer of an MRE consists of 11-mil monolayer low density polyethylene (LDPE) film that primarily provides a moisture barrier (but limited oxygen barrier) and typically depicts the DOD symbol and menu item number (Barrett & Cardello, 2012).

However, the aluminum foil within MRE pouches are susceptible to stress-induced fractures, such as during a military airdrop, can be difficult and time-consuming to manufacture as part of multilayer pouches, and can limit the type of food processing methods that can be applied to them that could potentially increase food acceptability, such as microwave sterilization (Barrett & Cardello, 2012). Additionally, since the aluminum foil is part of a multilayer product, it is considered non-recyclable and reduces the potential for waste reduction prior to disposal (Barrett & Cardello, 2012).

Alternatives to the current aluminum-based MRE primary packaging are expected to meet the same rigorous criteria for shelf-life, puncture resistance, moisture integrity, etc. The nanocomposite-based packaging utilized within this experiment meet these criteria while potentially reducing weight, offering recyclability potential, as well as enhancing mechanical and barrier properties of the primary packaging (Barrett & Cardello, 2012; DoD Combat Feeding Directorate, 2014).

All MREs within this experiment consisted of Menu #14, Penne Pasta. As depicted within Figure 27, MREs were varied by substituting aluminum-based packaging, by mass, with packaging that was nanocomposite-based.



Figure 27. MRE Aluminum-based versus Nanocomposite-based Primary Packaging

Fiberboards

The fiberboard packaging is a critical aspect to ensuring MREs arrive at their destinations to meet the needs of the warfighter. A package serves to “contain, protect, preserve, communicate, and perform” in a multitude of environments and conditions (Barrett & Cardello, 2012). The current solid fiberboard design for the secondary packaging associated with an MRE offers significant compressive strength with a paper weight of 90 pounds in addition to survivability against high moisture environments due to additives that provide wet strength (Ratto et al., 2012). These same additives however, prevent the current solid fiberboard from being repulpable and an alternative corrugated fiberboard design has been proposed that offers similar compressive strength with a paper weight of 69 pounds while offering repulpability and compostability (Ratto et al., 2012). Greater details regarding both the current and alternative designs for the fiberboard

secondary packaging are freely available from the US Army NSRDEC (Ratto et al., 2012).

Appendix C: Elaborated Statistical Methodology

Chapter II provides a general overview of the statistical methodology applied within this study, while this appendix provides examples of these analyses. Not all analyses are included; rather, example analyses are provided for each step and can be extrapolated for the other pollutants. As mentioned within Chapter II, following the calculation of emission factors, data was placed into JMP and verified upon entry. Interactions between main effects were considered but limited sample sets prevented statistical analyses from occurring, primarily due to the loss of degrees of freedom, and were therefore excluded.

Figure 28 illustrates a multiway ANOVA summary of fit for PM10 when testing the fiberboard types and MCE as main effects. Figure 29 displays the ANOVA p-value indicating a possible difference in the means for at least two fiberboard types.

Summary of Fit	
RSquare	0.671111
RSquare Adj	0.613913
Root Mean Square Error	0.329039
Mean of Response	3.011943
Observations (or Sum Wgts)	28

Figure 28. ANOVA Summary of Fit for PM10 for Fiberboards

Analysis of Variance				
Source	DF	Sum of Squares	Mean Square	F Ratio
Model	4	5.0812122	1.27030	11.7331
Error	23	2.4901308	0.10827	Prob > F
C. Total	27	7.5713430		<.0001*

Figure 29. ANOVA Report Displaying P-Value for PM10 for Fiberboards

Figure 30 specifies p-values associated with the particular main effects being tested, which in this case were fiberboard type and MCE. These p-values would indicate that fiberboard type most likely can explain some of the variance while the MCE may potentially explain some of the variance.

Effect Tests					
Source	Nparm	DF	Sum of Squares	F Ratio	Prob > F
Fiberboard Type	3	3	5.0517610	15.5535	<.0001*
MCE-PM10	1	1	0.5928923	5.4762	0.0283*

Figure 30. P-Values for Main Effects Tested for PM10 for Fiberboards

Residuals, studentized residuals, and Cook's distances, would then be saved as separate columns within JMP to test the ANOVA assumptions since the ANOVA p-value indicated a possible distinction between the fiberboard types for PM10. A histogram of the studentized residuals, depicted within Figure 31, would then be created and used to determine if there were particularly significant outliers beyond three standard deviations from the mean.

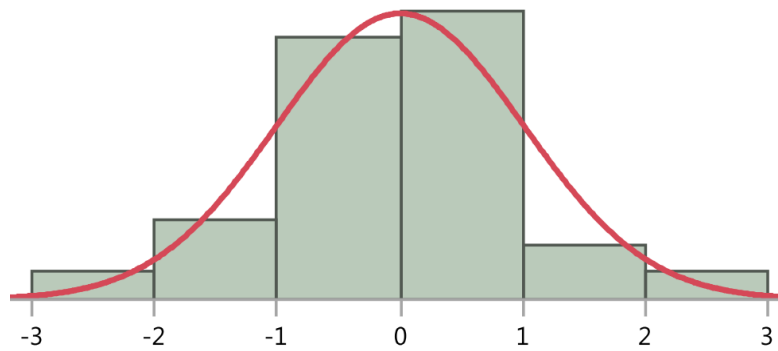


Figure 31. Histogram of Studentized Residuals for PM10 for Fiberboards

Following this verification, an overlay plot of Cook's distances would be created to identify highly influential data points. If a data point was found to exceed 0.5, it would be excluded and the ANOVA analysis would be conducted again. If the data points were below 0.5 but above 0.25, as depicted within Figure 32, they were inspected but ultimately remained within the ANOVA analysis.

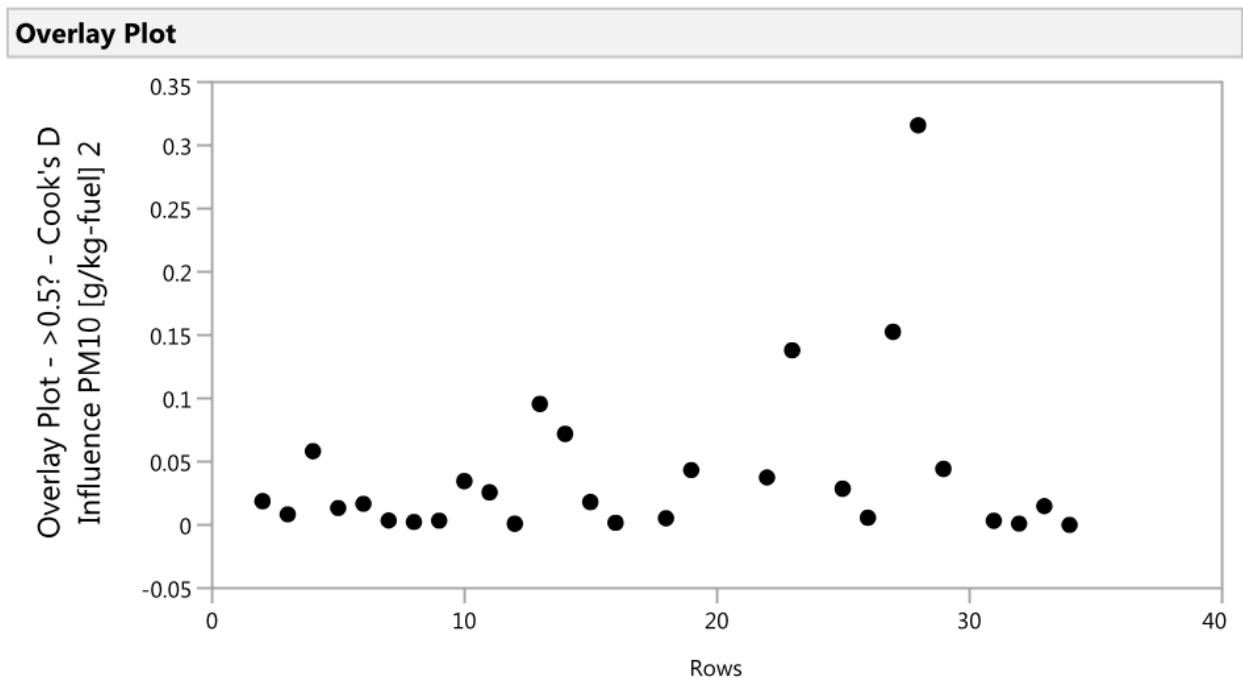


Figure 32. Overlay Plot of Cook's Distances for PM10 for Fiberboards

A Durbin-Watson test, illustrated within Figure 33 for PM10 for fiberboards, would then be conducted to verify that independence held true. A p-value greater than 0.05 would indicate that the residuals were independent with respect to time. Following the test for independence, a Shapiro-Wilk test for normality is conducted upon the residuals. A p-value greater than 0.05 would indicate that the residuals are normally distributed, such as within Figure 34.

Durbin-Watson			
Durbin-Watson	Number of Obs.	AutoCorrelation	Prob<DW
2.5706604	28	-0.2912	0.8533

Figure 33. Durbin-Watson Test for Independence for PM10 for Fiberboards

Goodness-of-Fit Test		
Shapiro-Wilk W Test		
W	Prob<W	
0.976649	0.7640	

Note: Ho = The data is from the Normal distribution. Small p-values reject Ho.

Figure 34. Shapiro-Wilk Test for Normality of the Residuals for PM10 for Fiberboards

The last verification of assumptions for the ANOVA test is that there is constant variance amongst the residuals. Constant variance can be determined through a Breusch-Pagan test but this test requires a separate calculation outside of the JMP program, specifically within a program such as Microsoft Excel®. Parameters required for the Breusch-Pagan test are the number of observations, the degrees of freedom associated with the ANOVA

model, the sum of squares term for the error, and the sum of squares term for the model for an ANOVA of the residuals squared utilizing the same main effects of the original ANOVA analysis. The first three parameters can all be found within tables such as Figure 28 and Figure 29. However, the last term requires another ANOVA analysis to be performed utilizing the squared residuals as the response term and the same main effects as the primary ANOVA analysis, which in this case would be fiberboard type and MCE. When an ANOVA analysis is conducted in this fashion, a sum of squares for the model term can be retrieved such as the one found within Figure 35.

Response ANOVA - Breusch-Pagan - Squared Residual PM10 [g/kg-fuel] - Fiberboards				
Whole Model				
Summary of Fit				
RSquare		0.189917		
RSquare Adj		0.049033		
Root Mean Square Error		0.150505		
Mean of Response		0.088933		
Observations (or Sum Wgts)		28		
Analysis of Variance				
Source	DF	Sum of Squares	Mean Square	F Ratio
Model	4	0.12214240	0.030536	1.3480
Error	23	0.52099207	0.022652	Prob > F
C. Total	27	0.64313447		0.2825

Figure 35. Retrieval of Breusch-Pagan Sum of Squares Term for PM10 for Fiberboards

With this term, a Breusch-Pagan test can be conducted within Microsoft Excel®. A test statistic must first be calculated following the formula:

$$Test\ Statistic = \frac{SSR/2}{\left(\frac{SSE}{N}\right)^2} \quad (2)$$

Where:

SSR = Sum of Squares for the Model term associated with Squared Residuals

SSE = Sum of Squares for the Error term associated with the Original ANOVA

N = Number of Observations

Once the test statistic is determined, it is utilized alongside the aforementioned degrees of freedom to calculate a p-value associated with the right-tailed probability of the chi-squared distribution. The Excel® formula, $CHIDIST(test\ statistic, degrees\ of\ freedom)$, returns a p-value such as the one found within Figure 36 which can then be interpreted. A p-value greater than 0.05 indicates constant variance.

Breusch-Pagan Calculations		FIBERBOARDS				
Source-->	ANOVA	ANOVA	ANOVA	ANOVA of Squared Residuals	Calculation	CHIDIST function
Pollutant	N	Degrees of Freedom (explained)	Sum of Squares-Error	Sum of Squares-Regression	Test Statistic	p-value
PM10	28	4	2.4901308	0.12214240	7.72161593	0.102

Figure 36. Breusch-Pagan Test for Constant Variance for PM10 for Fiberboards

With the assumptions for the ANOVA test verified and an original ANOVA p-value less than 0.05 indicating there may be a difference in the means of two or more fiberboard types, a Tukey honest significant difference test could be conducted to specifically identify which of the fiberboard types may be different. To interpret the Tukey test, the various fiberboard types are assigned letters identifying statistically similar and different groups, as depicted within Figure 37.

Level		Least Sq Mean
FBD-NWS	A	3.5003617
FBD-CB	A B	3.2409950
FBD-NSB	B C	2.8289448
FBD-SB	C	2.3069878

Levels not connected by same letter are significantly different.

Figure 37. Tukey Honest Significant Difference Test for PM10 for Fiberboards

In this instance, PM10 emission factors for SFNP would be considered statistically similar to SFP, while statistically different from CFNP and CFP. Also, SFP is statistically similar to CFNP but different from CFP. Lastly, CFNP is statistically similar to CFP. The fiberboard types are also placed in order by mean emission factor from greatest to least. Another way to arrive at similar conclusions is through the use of a least squares means plot, depicted within Figure 38. With conclusions drawn for the fiberboards for PM10 emission factors, similar analyses would be conducted for other pollutants and their emission factors.

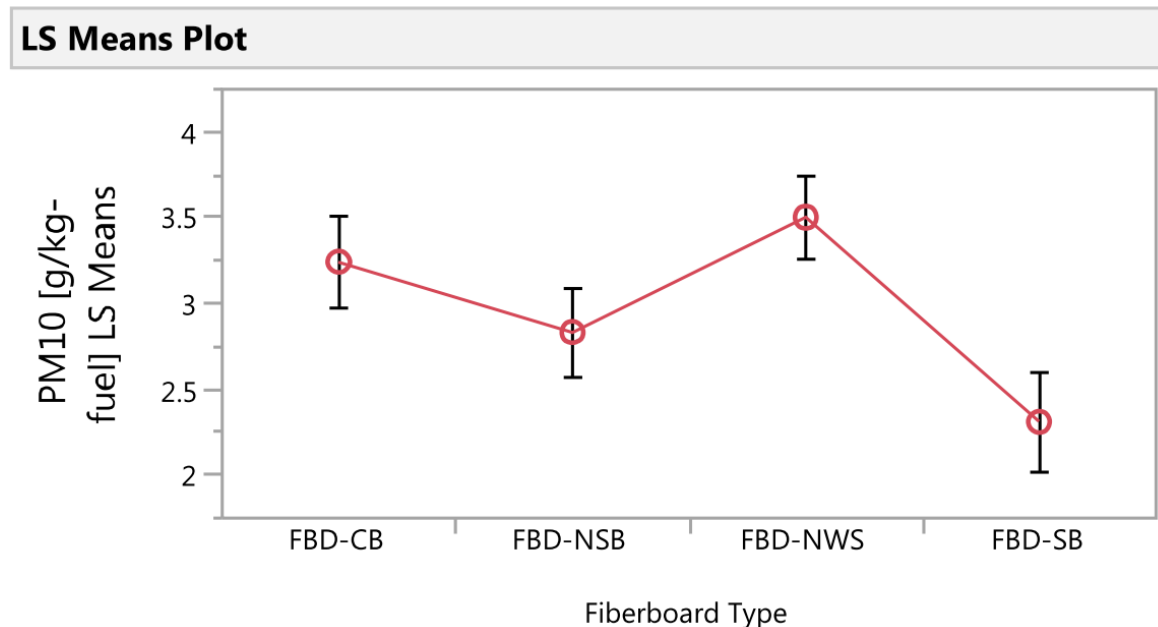
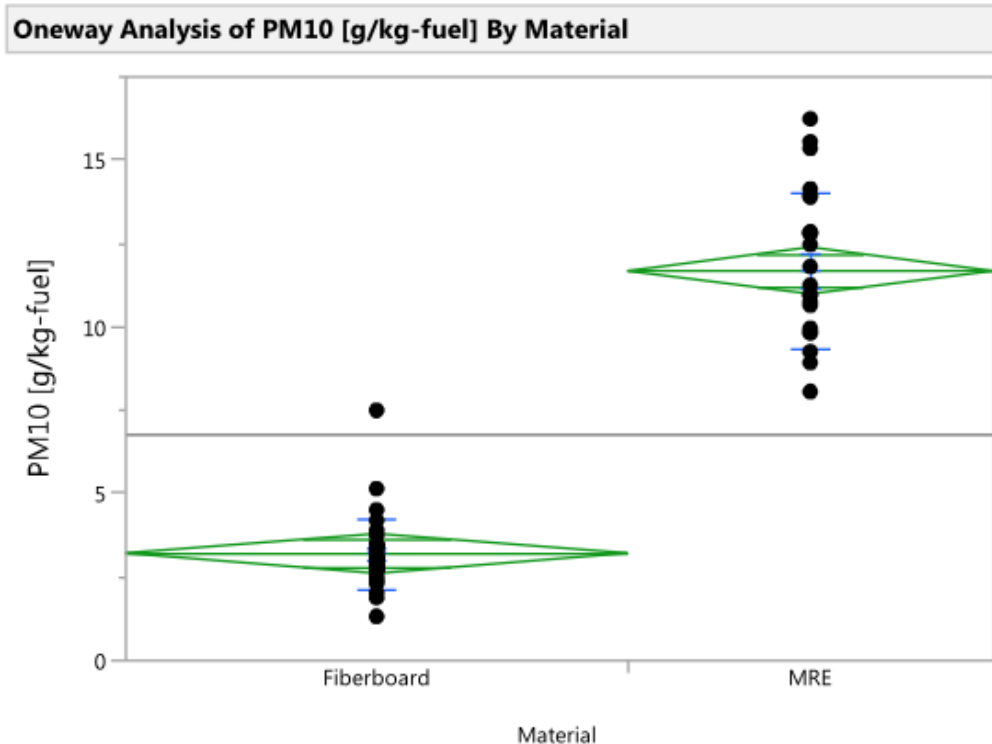


Figure 38. Least Squares Means Plot for PM10 for Fiberboards

With comparisons drawn within material types, a comparison between the fiberboards and MREs remains to be conducted. Since sample sizes differed greatly for the individual MRE (n=two to three) and fiberboard (n=three to four) types, an ANOVA could not be conducted. Therefore, a non-parametric Kruskal-Wallis rank sums test was utilized on the combined data for each pollutant. Figure 39 shows an example comparison between the PM10 emission factors for the fiberboards and MREs.



Missing Rows 29

Means and Std Deviations

Level	Number	Mean	Std Dev	Std Err Mean	Lower 95%	Upper 95%
Fiberboard	33	3.1954	1.06285	0.18502	2.819	3.572
MRE	24	11.6650	2.33276	0.47617	10.680	12.650

Wilcoxon / Kruskal-Wallis Tests (Rank Sums)

Level	Count	Score Sum	Expected Score	Score Mean	(Mean-Mean0)/Std0
Fiberboard	33	561.000	957.000	17.0000	-6.392
MRE	24	1092.00	696.000	45.5000	6.392

2-Sample Test, Normal Approximation

S	Z	Prob> Z
1092	6.39235	<.0001*

1-Way Test, ChiSquare Approximation

ChiSquare	DF	Prob>ChiSq
40.9655	1	<.0001*

Figure 39. Kruskal-Wallis Test for Differences Between MREs and Fiberboards for PM10

This Kruskal-Wallis test indicates that the MREs emitted statistically greater PM10 emissions than the fiberboards. Both a two-tail and one-tail p-value can be determined from this analysis as well. A time-series plot for both the MRE and fiberboard data was then conducted to verify the absence of autocorrelation, depicted within Figure 40.

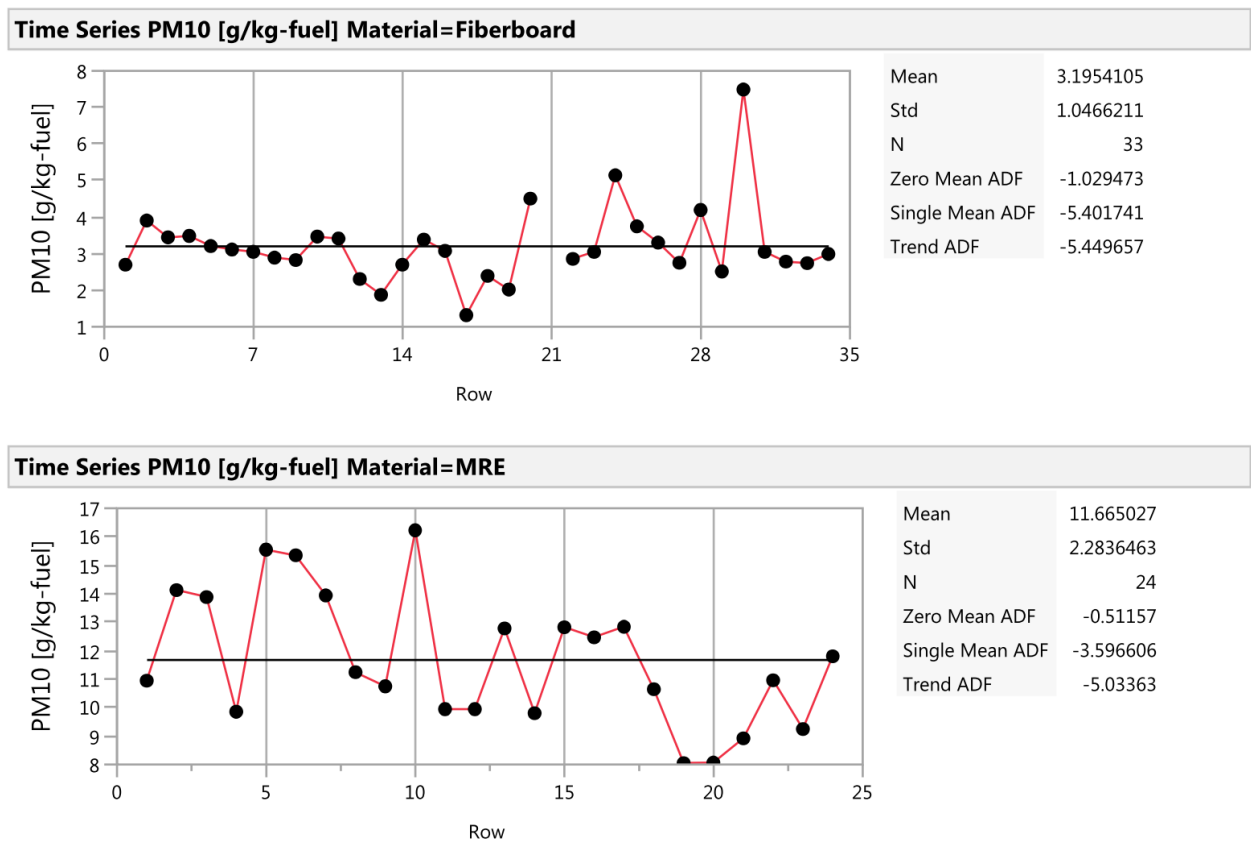


Figure 40. Time Series Plot of PM10 Emission Factors for Fiberboards and MREs

In this instance, based upon the limited samples gathered, a possible autocorrelation can be seen; however, without a greater sample size, a true correlation could not be determined.

Appendix D: Elaborated Emission Factor Calculation

Concentrations were background-corrected and converted to emissions factors utilizing the carbon balance method described within Chapter II. The elaborated equation follows:

$$EF \left(\frac{mg_{Pollutant}}{kg_{waste}} \right) = \left(\frac{10^6 mg_{waste}}{1 kg_{waste}} \right) * F_{Carbon} \left(\frac{mg_{Carbon}}{mg_{waste}} \right) * \frac{P_{Conc} (mg_{Pollutant}/m^3)}{\sum Carbon_{Conc} (mg_{Carbon}/m^3)} \quad (3)$$

Where:

EF = Emission Factor in mg-pollutant per kg-waste burned

F_{Carbon} = Carbon Fraction in mg-carbon per mg-waste burned

P_{Conc} = Pollutant Concentration in mg-pollutant per cubic meter

$\sum Carbon_{Conc}$ = Sum of Carbon Concentrations for CO and CO₂ (and CH₄ for VOCs) in mg-carbon per cubic meter

An example calculation for the fourth PM_{2.5} sample associated with SFP is provided below:

$$3103 \frac{mg_{PM2.5}}{kg_{waste}} = \frac{10^6 mg_{waste}}{1 kg_{waste}} * 0.46245 \frac{mg_{Carbon}}{mg_{waste}} * \frac{7.2 \frac{mg_{PM2.5}}{m^3}}{1067.9 \frac{mg_{Carbon}}{m^3}}$$

which is equivalent to 3.1 g-PM_{2.5} per kg-waste. Emission Factors can then be used for modeling, comparison purposes, or, as described in Chapter III, as a surrogate for health risk.

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14. ABSTRACT Emissions were characterized and compared from the open burning of four compositions of Meals, Ready-To-Eat (MRE) and four types of MRE fiberboard packaging in response to inhalation concerns at military forward operating bases. Measurements of particulate matter (PM2.5, PM10) showed that MREs account for 70-90% of PM emissions when burned in unison with the current fiberboard container and liner and that PM2.5 constitutes a vast majority of the particulates emitted. Targeted replacement of MRE constituents may be more effective in reducing volatile organic compound (VOC) and polycyclic aromatic hydrocarbon (PAH) emissions than the variation of fiberboard packaging designs, while polychlorinated dibenzo-dioxin and -furan (PCDD/PCDF) emissions are believed to be more closely related to other components of a military waste stream. MRE and fiberboard types each respectively produced equivalent PM, PAH, VOC, and PCDD/PCDF emission factors. This study provides the first representative characterization of open burning emissions associated with military rations separate from comprehensive military waste streams.						
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