EPA Public Access

Author manuscript

Waste Manag. Author manuscript; available in PMC 2020 July 13.

About author manuscripts

Submit a manuscript

Published in final edited form as:

Waste Manag. 2019 March 15; 87: 398–406. doi:10.1016/j.wasman.2019.02.031.

Methodology for characterizing emissions from small (0.5–2 MTD) batch-fed gasification systems using multiple waste compositions

J. Aurell^a, M. Barnes^b, B.K. Gullett^c, A. Holder^c, R. Eninger^b

^aUniversity of Dayton Research Institute, Energy Technology and Materials Division, 300 College Park, Dayton, OH 45469, USA

^bAir Force Institute of Technology, Graduate School of Engineering and Management, 2950 Hobson Way, Wright Patterson Air Force Base, OH 45433-7765, USA

^cU.S. Environmental Protection Agency, Office of Research and Development, National Risk Management Research Laboratory, Research Triangle Park, NC 27711, USA

Abstract

A compact, containerized gasification system was characterized for air emissions while burning four waste types. A methodology is presented for developing a standardized test waste composition and demonstrated using three military and one civilian waste types. Batch charges of waste were processed through a gasification chamber, afterburner, and wet scrubber. The 0.5–2 metric ton per day (MTD) system was designed for mobile deployment by the military in forward operations but would be applicable to small scale civilian applications. Emissions data from these types of small capacity, cyclically operated systems are lacking, limiting efforts to compare technologies and their environmental performance. Eight tests were conducted in a 7-day period at the Kilauea Military Camp (KMC) in Hawaii. The pollutants characterized were chosen based on their regulatory and health relevance: particulate matter (PM), mercury (Hg), elemental composition, volatile organic compounds (VOCs), polyaromatic hydrocarbons (PAHs), and polychlorinated dibenzo-*p*-dioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs). Averaged data from 4-hour runs, including startups and shutdowns, indicated that five of the nine EPA-regulated compounds (lead, cadmium, Hg, sulfur dioxide, and hydrogen chloride) were under the emission limits set for Other Solid Waste Incineration Units (OSWI) while four, PCDD/PCDF, PM, nitrogen oxides, and carbon monoxide, were higher. The procedures through which waste compositions were created and emissions were characterized provide a methodology by which differing waste to energy technologies can be compared on an equivalent basis. This system's emissions compare favorably with alternative disposal methods. PM and PCDD/PCDF emission factors were, respectively, over 39 and 9 times lower from this unit than from published data on burning simulated military waste in an air curtain incinerator and in open burn piles ("burn pits").

⁶Disclaimer

The views expressed in this article are those of the authors and do not necessarily represent the views or policies of the U.S. Environmental Protection Agency.

Keywords

Municipal waste; Gasification; Combustion; Emissions; Batch; Military waste

1. Introduction

Military waste management practices, specifically open air “burn pits”, have come under increased scrutiny due to concerns over potential health effects to service members from uncontrolled emissions (Baird Weese, 2010). Waste management practices in military-deployed environments vary depending on available resources, logistical constraints, and mission requirements; they include burn pits, burying, contracted disposal, or backhauling, to name a few. During Operation Iraqi Freedom and Operation Enduring Freedom open burn pits were frequently used in every size base camp. A wide variety of solid waste materials were burned for volume reduction and final disposal. Burning waste in open air burn pits is akin to uncontrolled burning of household waste in barrels and dumps, a common practice in rural areas and developing nations, and has been the subject of previous studies (Lemieux et al., 2003).

Service member concerns of exposure from uncontrolled burn pit emissions provoked investigations from the Institute of Medicine (IOM) upon request from the Veterans Administration (IM, 2011). The IOM was unable to definitively determine health risks due to limited, incomplete or inconclusive data, sampling, analysis, and epidemiological studies. Studies could not differentiate exposure from burn pit emissions, local conditions, generators, and other sources (IM, 2011, Baird Weese, 2010). Even with an irresolute link between burn pits and adverse health effects (Baird, 2011), burn pit use has been restricted by the Department of Defense (DoD), leaving field commanders to seek alternative options (DODI, 2011). To better quantify burn pit emissions, and in the absence of operating conditions that warrant heavy burn pit use (i.e., major conflicts), studies on emissions from simulated military waste have been conducted (Aurell et al., 2012, Woodall et al., 2012). Several waste stream characterizations in deployed environments have also been conducted to inform waste composition characteristics of deployed forces (Cospers et al., 2013). These composition studies are critical toward establishing a standardized waste in which to compare waste disposal technologies on a common basis.

Waste to energy (WTE) systems are an appealing and improving technology to help solve problems of disposal and energy needs (MCCDC, 2011). WTE systems process waste through various methods of thermochemical conversions, producing volume reduction and better control of emissions from that of burn pits (Bosmans et al., 2013). The principal combustion methods for WTE systems are mass burn, refuse-derived fuel-fired, fluidized bed, modular starved air, and modular excess air. The most appealing characteristic of these WTE systems are the various forms of usable energy which generally deal with a heat exchanger to produce hot water or steam to generate electricity (Tchobanoglous and Kreith, 2002). Generating energy aligns with DoD goals of reduced dependency on fossil fuels and sustainable resource management practices in contingency environments, which have been gaining momentum within the DoD (USDATL, 2012). Sustainability efforts have forced

service branches (Army, Navy, Marine Corps, Air Force, etc.) within the DoD to evaluate and improve resource management practices within base operations. Furthermore, designing small-scale WTE systems that process < 5 metric tons per day (MTD) for military use in austere conditions has broader relevance in waste management practices; for example, a system of this size could be used in rural communities or areas devastated by natural disasters where infrastructure cannot support larger and permanent waste management systems or practices. However, very little emission information is available to compare performance of these small, cyclically operating systems. This is particularly important for systems which incur frequent startups, as these transient conditions may be subject to non-optimal combustion conditions and higher emissions. According to regulations, this unit processing 0.5–2 MTD is classified as a “very small municipal waste combustion” (VSMWC) unit because it burns less than 35 tons (short) per day of municipal solid waste and would therefore be regulated as an Other Solid Waste Incinerator (OSWI) (EPA, 2005). Testing on these types of systems typically account only for steady state emission monitoring and do not capture emissions during startups and shutdowns.

To field a safe and viable waste management alternative to burn pits while simultaneously tackling sustainability initiatives, extensive and uniform system testing will be required. Uniform testing which allows comparative analysis between systems requires reproducible and easily procurable standardized waste composition recipes. Therefore, a critical focus of this research was to establish procedures to build and test a standardized waste composition in a manner that can be replicated for future tests on similarly sized WTE systems. Uniform emission analyses, including operation-representative sampling, will also be a critical component to comparative testing. Emissions analysis must consider important health and environmentally toxic constituents, as these systems will be implemented in proximity to living spaces.

To our knowledge this study is the first to process a standardized waste composition through a small scale (0.5–2 MTD), batch-operated WTE system and to characterize a comprehensive array of emissions: nitrogen oxides (NO_x), oxygen (O₂), carbon monoxide (CO), carbon dioxide (CO₂), sulfur dioxide (SO₂), methane (CH₄), hydrogen chloride (HCl), total particulate matter (PM), metals including mercury (Hg), volatile organic compounds (VOCs), polycyclic aromatic hydrocarbons (PAHs), and polychlorinated dibenzo-p-dioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs). While regulatory limits are based on stack gas concentrations, useful units for comparing waste technology systems and evaluating overall burden to the environment are expressed through emission factors, or the amount of pollutant per amount of waste material processed. This study compared the emission factor results from testing this unit to previous burn box and burn pit emission studies. Four waste recipes were processed in a military owned, solid waste gasification system over the course of ten days at the Kilauea Military Camp (KMC), Hawaii on the National Park Service grounds of Volcano National Park.

2. Materials and methods

2.1. Gasification technology

A previously-operated, military-owned, prototype small batch gasification system was utilized (Supplemental Information (SI) Figure S1). Other similarly scaled gasification technologies are also in prototype stages (Margolin et al., 2015). The unit consists of two waste processing drums, or gasifiers, mounted inside a 6.1 m (20 ft) CONEX container and operating from a side and rear opening. Each waste processing drum is constructed with a thermally insulated heat exchange section that allows for indirect heating of the waste by the exhaust gases from the combustion chamber. Waste is loaded into the primary reactor and heated to about 760 °C. A controlled amount of pre-heated air is fed into the drum and brought in contact with the waste. The O₂ in the air reacts with the waste to convert organic molecules to a synthesis gas, or syngas, composed primarily of CO and hydrogen (H₂). The syngas then passes into the combustion chamber where it is ignited to provide energy for downstream processes.

The combustion chamber is a thermally insulated reactor, maintained at about 1,100 °C through the combustion of syngas or supplemental diesel. A two-stage burner system allows for heating of the combustion chamber during start-up, as well as the ignition of the syngas. The hot exhaust gases from the combustion chamber serve as the heat source for the primary reactor. Oxygen concentration is monitored at the stack output and the air intake to the combustion chamber is regulated to maintain the desired O₂ concentration. The gas flowrate through the system varies as a function of the production rate of combustible gas from the gasifier unit. The flow rate varied from approximately 18–23 standard m³/min (7% O₂ with the standard reference conditions 1 atm and 20 °C). Exhaust gases leaving the heat exchange section are quenched with water to a temperature of approximately 82 °C and then are cleaned by a caustic scrubber to remove particles, acid gases, and moisture prior to discharge. A 0.15 m diameter flexible duct was attached to the stack to bring the exhaust gas down to a 0.15 m diameter straight pipe exhaust manifold oriented parallel to the ground, providing multiple port locations for probe access for emissions testing. The process comes to completion when the organic waste is fully gasified and syngas production stops. The ash residue is comprised of inorganic carbon (carbonate minerals) and any metal and glass present in the original waste.

2.2. Simulated waste composition

The U.S. Army Research Laboratory (ARL) established a standardized waste recipe composition by waste category and corresponding percentage of respective material (Table 1) based on past DoD waste characterizations (Margolin et al., 2015). Waste recipes for emissions analysis were developed from materials representative of waste stream compositions for deployed forces at small and extra small base camps based on the ARL study. The recipes were designed to facilitate replication by using readily-available items; for example, dog food is used as a food replacement of comparable caloric and moisture content albeit much greater homogeneity. A primary focus of this study was to compare emissions from the previously developed “standard recipe” to “challenge” waste formulations with high plastic and military field ration waste (Table 1). A fourth challenge

waste source consisted of dumpster waste from KMC. Incorporation of these waste variations allows better comparisons of the methodology and emissions characterization to other waste and system scenarios. The four recipes tested and their notation used here were a standard waste (standard waste, SW), a plastic challenge (high plastic, HP), waste collected from KMC dumpsters, and a First Strike Rations (FSR) challenge. FSRs are “ready to eat” military rations manufactured with sufficient calories for one FSR to sustain one troop for one day. Food materials in FSR waste were used, rather than kibble dog food, cooking oil, and water (the SW recipe food surrogate). The SW, HP, and FSR wastes were constructed as outlined in Table 1; the plastic components are further specified in Table 2.

A 4-h run was deemed necessary to conduct a representative waste processing test and emissions characterization. Based on the manufacturer’s recommendations for throughput rate and the volumetric density of the waste, 16 waste-filled bags weighing 8.2 kg (18 lbs) each were constructed for a total of 131 kg (288 lb) per test. Each bag represents the waste generated from about four persons per day based on DoD waste characterizations, 2.0 kg/person/day (4.5 lb/person/day).

The KMC waste was constructed in a different manner than the other three wastes. The maintenance and cleaning staff at KMC filled a small dumpster with trash bags from various waste collection points such as rooms, kitchens, and recreation areas. Twenty bags were randomly selected from the dumpster, opened, and characterized according to waste categories identical to those used for the other three wastes. Characterization results are shown in Tables 1 and 2. The KMC waste was then re-bagged in the 20-bag average composition found in the dumpster and staged for gasification.

The FSR composition consisted of variation from the SW recipe, each bag including the remnants of four FSRs, or four service members’ daily food allotment. FSR food remnants, 2.6 kg (5.8 lb), as well as a proportional amount of packaging cardboard from the FSR cases, were included. Cardboard from the SW recipe was adjusted to account for the additional cardboard from the FSR cases. No plastics were removed to adjust for the additional material from the FSR food packaging (the additional weight per FSR was 0.20 kg (0.43 lb)). The metal cans were also removed from the FSR tests due to concerns of them jamming the system. All other waste categories for the FSR recipe were held constant to the standard recipe.

Materials were purchased by category type to reproducibly build waste bags (see SI Figure S2) that were fed into the system; glass bottles, rubber mulch, plastic bottles, and cardboard are a few examples of purchased materials. The detailed materials list by category and waste recipe is included in SI Tables S1, S4.

2.3. Testing methodology

Eight tests using four different waste type compositions were conducted in a 7-day period. Triplicate runs were conducted for the standard waste (SW:1–3) composition, duplicate runs were performed for high plastic (HP:1–2) and FSR (FSR:1–2) waste compositions, and one run was performed for the KMC waste. The number of runs for each waste was limited by time and budget constraints. The standard waste was chosen for triplicate sampling to

establish a baseline to which duplicate tests could be compared. The tests were not performed in random order due to time limitations for waste production and waste characterization.

2.4. Instrumentation, sampling methods, and analytical methods

A range of pollutant emissions were measured using continuous and batch sampling (Table 3). The sampling period included the full cycle of waste processing, including daily system startups and daily system shutdowns, consistent with OSWI regulations.

The fourier transform infrared spectroscopy (FTIR) gas analyzer was multi-point calibrated daily for CO₂, CO, O₂, NO_x, SO₂ in accordance with U.S. EPA Methods 3A (EPA, 1989), 6C (EPA, 2014c), and 7E (EPA, 2014a), at the beginning of the test as well as a drift check at the end of each day. A precision gas divider Model 821S (Signal Instrument Co. Ltd., England) was used to dilute the high-level span gases for acquiring the mid-point concentrations for the calibration curves. The precision gas divider was evaluated in the field as specified in U.S. EPA Method 205 (EPA, 2014b).

Stack velocity was measured in accordance with U.S. EPA Method 2 (EPA, n.d.-d) using a Shortridge Instruments (Scottsdale, AZ) Airdata Multimeter with an Airfoil pitot head differential pressure device. The Airdata Multimeter measures temperature and barometric pressure which enables measurements in actual cubic meter per minute. A velocity head measurement and a temperature measurement were taken at each traverse point as determined in U.S. EPA Method 1 (EPA, n.d.-c). The Airdata Multimeter was also set up to continuously record stack velocities at a single center point.

Total PM, PAHs, and PCDD/PCDF were sampled using the same sampling train. The glass fiber filters underwent a 24-hour desiccation before the first tare and gross weighing (pre- and post-sampling). VOCs were sampled for the first 2 h of each test. The exhaust was passed through a Nafion dryer (Permapure, MD-50) before being sampled for VOCs to reduce the water vapor in the sample. For standard waste test number 3 (SW-3), four approximately 12–20 min samples were collected during hours 00:00–00:10, 00:11–00:30, 02:10–02:32 and 03:09–03:39 of the test run. Mercury sorbent tubes, as described in Method 30B (EPA, n.d.-b) cannot sample isokinetically as the tubes terminate in parallel to the probe.

Particle bound metal emissions were measured using a modified EPA Method 5 (EPA, n.d.-a). A heated sample (~170 °C) from the duct was pulled through a 47 mm Teflon filter at 2 L/min. Some compounds with higher vapor pressures that would be captured by EPA Method 29 (EPA, 2017) may not be captured on the heated filter, thus the elemental emissions may be underestimated. A subset of filters was analyzed by X-Ray Fluorescence (XRF) (EPA, 1999a) to determine the range of elements in the PM. All filters were subsequently analyzed by Inductively Coupled Plasma Atomic Emission (ICP) (EPA, 1999b) for quantitation of regulated compounds (Pb and Cd), (Fe and Cu), and major inorganic constituents (Na).

3. Calculations and comparisons

Measured pollutant quantities were evaluated as either a concentration or emission factor to compare with regulatory requirements and published data from related studies. Emission factors, which tally pollutant mass per waste input amount, allow for extrapolation to other waste scenarios and comparisons with other technologies where combustion emissions are not applicable (e.g., syngas production). In addition, comparison of concentrations in pollutant mass/volume units does not properly attribute the pollutant mass to the amount of waste processed. Pollutant concentrations were expressed in dry gas at 7% O₂ with the standard reference conditions 1 atm and 20 °C, referenced in accordance to Other Solid Waste Incineration Unit regulation limit equation (EPA, 2005) (SI Eq. (S1)). Emission factors in mass of pollutant per mass of waste were calculated using the carbon mass balance approach (Laursen et al., 1992). In this method, the pollutant was co-measured with the major carbon species in the form of CO, CO₂, and CH₄ resulting in mass of pollutant per mass of carbon emitted. The carbon fraction of the waste (reported in SI Table S5) was calculated from the waste composition (Table 1) and published carbon concentrations (Liu and Lipták, 1999) of materials. With the assumption that the waste is entirely combusted, the total amount of a pollutant per waste charge mass can be calculated by the product of the pollutant mass per carbon and the carbon fraction in the waste (SI Eq. (S2)). Emission factors were also calculated with a waste input method by determining the ratio of the pollutant mass (scaling the sampled gas volume with the duct flow rate) and the contemporaneous waste feed mass (SI Eq. (S3)). The data precision was checked by calculating the Relative Percent Difference (RPD) for any pair of duplicates, or the Coefficient of Variation (CV) if more than duplicate measurements were conducted.

Emissions were compared to regulations in the Code of Federal Regulations (CFR) and in the U.S. EPA's inventory of source emission strengths, AP-42 (EPA, n.d.-e). Emissions factors also allow for comparison to previous burn pit simulation studies, as well as future studies on waste technologies whose operating modes make comparisons of concentrations inapplicable.

4. Results and discussion

Complete waste feed schedules for each run are in SI Table S6. A representative continuous emissions record for CO₂ and CO (test SW-2) is plotted along with sampling collections periods for VOCs, PM, mercury, and PCDD/PCDF/PAHs in Fig. 1. Similar plots for each waste test are shown in SI Figs. S3–S8 except test SW-1 where an equipment failure precluded measurements. CO tends to spike upwards upon waste introduction to the gasifier chamber. CO₂ fluctuates multiple times between waste charging, up to a factor of two, likely due to door openings between the gasification and secondary combustion chambers.

Average pollutant concentration and emission factor results for each waste-specific recipe are presented in Table 4 along with the eight-run composite average. Run-specific continuous emission monitor results are in SI Table S7. PCDD/PCDF concentration data are presented in SI Tables S8–S9 and emission factors by the carbon balance and waste input methods are included in SI Tables S10–S13, respectively. PAH concentrations and emission

factors by the carbon balance and waste input method are presented in SI Tables S14–S16. VOC data are presented in SI Tables S17 and S18 for 2-h concentrations and emission factors, respectively. Similar data for 12–20-min samples are shown in SI Tables S19 and S20. Elemental concentrations and emission factors of PM samples from each waste type are included in SI Tables S21–S23.

In general, variation between the waste-specific emission factors up to a factor of four was common with no one waste type consistently having higher emissions for all pollutants. Waste types SW and FSR combined had over 75% of the highest emission factors. The high plastic (HP) waste had less than 10% of the high emission factors contrary to commonly held concerns regarding thermal disposal of plastics. Concentration limits for OSWIs are presented in Table 4 as a comparator. For the composite concentrations, the average values for four of the nine regulated emissions (PM, PCDD/PCDF, NO_x, and CO) exceed the OSWI limits. Gasifying SW, HP, and KMC wastes resulted in similar PCDD/PCDF stack concentrations, 0.26–0.27 ng Toxic Equivalency Factor (TEQ)/m³ at 7% O₂, while FSR waste generated a notably higher stack concentration of 0.68 ng TEQ/m³ at 7% O₂. The PM emissions, similarly, were higher from gasification/combustion of the FSR waste composition, 60 mg/m³ at 7% O₂, than the other waste composition types, 18–41 mg/m³ at 7% O₂. The mercury concentration was lower when gasifying waste with the higher plastic content (HP), 0.31 ± 0.037 µg/m³ at 7% O₂, than the other waste types. Benzene, toluene, and propene were the most abundant VOCs for all waste types. Also included are moisture, ash, waste load, and stack flow for each of the recipes. The limited number of replicates suggests that these conclusions can only be considered tentative.

Emission factors for pollutants corresponding to Table 4 and calculated using the carbon balance method are shown in Table 5 for the individual waste types and the overall four-waste composite. These values are compared in the right-most columns to published emission factors for (EPA, n.d.-e) modular, starved air combustors. Since the work reported herein uses a starved air gasifier with a caustic scrubber, published data for units operating with uncontrolled (UnC) emissions and with electrostatic precipitator (ESP) emissions treatment are presented. The average values for three of the ten emissions (NO_x as NO₂, CO, and CO₂) for all four waste types exceed the published EFs. In 8 of 11 cases the emission factors for the gasifier unit were lower than either published values for uncontrolled or ESP-equipped modular starved air incinerators. Values for NO_x, CO, and CO₂ were higher.

Iron (Fe) was the metal with the highest concentration in the waste (1.5–3.6% waste by mass) but only a negligible percentage was emitted in the exhaust (5.1–50.1 × 10⁻⁷% or 0.051–0.505 mg/kg waste). Copper (Cu), tin (Sn), and lead (Pb) were all present in the PM at 2–3.5 times higher concentrations than Fe. In addition, zinc (Zn) was present at 22 times the concentration of Fe, suggesting that Zn was enriched in the PM emissions. Approximately 20–25% of the PM is composed of K and Cl, which are likely salts produced from the treatment of acid gases in the wet scrubber (SI Tables S21–23). Other elements were present in the PM at only trace levels (<1–2% of the PM mass), suggesting that incomplete combustion was only a minor contributor to PM emissions. There were no large differences in the PM composition among the different wastes, however, there were only limited numbers of samples for comparison. The major element species in the ash from all

four waste types were CI and AI (SI Table S24). The ash amounts from all four waste types were statistically indistinguishable at $12\% \pm 3.3\%$.

Benzene and toluene (Table 5) along with acetone and propene (SI Table S17) were the most abundant VOCs for all waste types. The high benzene concentration (2-h sample) noted in the SW tests ($>4,000 \mu\text{g}/\text{m}^3$ at 7% O_2 , SI Table S17) is likely due to suboptimal combustion conditions which most probably occurred during an unexpected system shut-down and before the sampler could be stopped. The higher levels of chlorinated VOCs from the FSR waste (SI Table S17), such as vinyl chloride and chloromethane, may be due to higher salt content in the FSR food. The higher concentrations observed are not limited to chlorinated VOCs, as compounds such as acrolein and naphthalene are higher than the other waste types as well perhaps due to the composition of the FSR packaging material.

Emission factors for PM, PCDD/PCDF, Hg, and PAHs are presented in Fig. 2 for each waste recipe using the waste input and carbon mass balance approaches. These two methods compare well. For example, the correlation coefficient, R^2 , between PCDD/PCDF emission factors for these two methods is 0.75. This suggests that with sufficient knowledge of waste composition or waste feed rate, valid emission factors can be determined from either method.

Table 6 presents the emission factors in comparison with other methods of waste disposal in military theater, including an air curtain incinerator, a burn pile, and an open burn laboratory test facility. It is important to note that the waste compositions are not equivalent between this study and the referenced studies and there is a significant difference in the waste processing rate (see Table 4).

5. Conclusion

A gasifier/combustor unit was tested using four waste compositions (SW, HP, KMC, and FSR) specifically constructed to simulate U.S. DoD military deployment waste and evaluate compositional variations. Seven days of testing (~ 10 h/day) processed a daily average of 30 kg/h of waste during which emissions were sampled. Testing of this batch-fed unit included sampling during typical daily startups and shutdowns to fully characterize the emissions of a cyclically-operated unit. The procedures for creating a waste feed recipe outlined in this study can be applied to tests to compare alternative waste disposal technologies. Similarly, the emissions results can be compared to alternative waste processing techniques appropriate for rural and austere environments, in both military and civilian applications, such as other gasifiers, burn pits, etc. Five of the nine EPA regulated compounds (Pb, Cd, Hg, SO_2 , and HCl) from this system were under the set emission limits for Other Solid Waste Incineration Units (OSWI). The average PCDD/PCDF, PM, NO_x , and CO stack emissions were all above the set emission limits. Some distinctions were noted in the emissions from the waste types. For example, the FSR waste appeared to have higher PCDD/PCDF emissions than the other three types, although any waste-specific conclusions should be tempered by the limited number of samples. As shown in Table 6, the PM emission factors, however, were 39 and 100 times lower from this gasification unit than from published data on burning simulated

military waste in an air curtain incinerator and in open burn piles, respectively, while the PCDD/PCDF emissions were 9 and 460 times lower.

Supplementary Material

Refer to Web version on PubMed Central for supplementary material.

Acknowledgments

This project was supported by the Navy Expeditionary Combat Command, Headquarters U.S. Pacific Command (Ms. Joelle Simonpietri, lead), and the U.S. Environmental Protection Agency's Office of Research and Development. The authors appreciate the critical field and laboratory assistance provided by Mr. Dale Greenwell (EPA), Mr. Dennis Tabor (EPA), Mr. Peter Kariher (ARCADIS US, Inc.), and Mr. Kawakahi Amina (Cubic Applications, Inc.). Review expertise was provided by Mr. Steffan Johnson, Ms. Gerri Garwood, Dr. Nabanita Modak, and Ms. Charlene Spells of EPA's Office of Air Quality Planning and Standards.

References

- Aurell et al, 2012. Aurell J, Gullett BK, Yamamoto D, Emissions from open burning of simulated military waste from forward operating bases Environ. Sci. Technol, 46 (2012), pp. 11004–11012 [PubMed: 22992062]
- Baird, 2011. Baird CP, Is deployment associated with an increased risk of respiratory outcomes? What do we know? What do we think? The Army Med. Department J, Jul-Sep (2011), pp. 80–86
- Baird Weese, 2010. Baird Weese C, Issues related to burn pits in deployed settings The Army Med. Department J, Apr-Jun (2010), pp. 22–28
- Bosmans et al, 2013. Bosmans A, Vanderreydt I, Geysen D, Helsen L, The crucial role of waste-to-energy technologies in enhanced landfill mining: a technology review J. Clean Prod, 55 (2013), pp. 10–23
- Cosper et al, 2013. Cosper SD, Anderson HG, Kinnevan K, Kim BJ, 2018. Contingency base camp solid waste generation. ERDC/CERL TR-13–17. US Army Corps of Engineers, Engineer Research And Development Center, Construction Engineering Research Laboratory, 9, 2013 <http://www.dtic.mil/dtic/tr/fulltext/u2/a613823.pdf> Accessed August 10.
- Dominguez et al, 2018. Dominguez T, Aurell J, Gullett B, Eninger R, Yamamoto D, Characterizing emissions from open burning of military food waste and ration packaging compositions J. Mater. Cycles Waste Manage, 20 (2) (2018), pp. 902–913
- DODI, Department, 2011. DODI, Department of Defense Instruction, 2011 Use of open-air burn pits in contingency operations. Number 4715.19. <http://www.esd.whs.mil/Portals/54/Documents/DD/issuances/dodi/471519p.pdf?ver=2017-10-06-071825-977> Accessed August 6, 2018.
- Epa, 2014a. EPA, U.S. EPA Method 7E, 2014 Determination of Nitrogen Oxides Emissions from Stationary Sources (Instrumental Analyzer Procedure). <http://www.epa.gov/ttnemc01/promgate/method7E.pdf> Accessed June 17, 2015.
- Epa, 2017. EPA, U.S. EPA Method 29, 2017 Determination of metals emissions from stationary sources. https://www.epa.gov/sites/production/files/2017-08/documents/method_29.pdf Accessed February 7, 2019.
- Epa, 2015. EPA, U.S. EPA Method 5. Determination of Particulate Matter Emissions from Stationary Sources. Title 40, Chapter 1, Subchapter C, Part 60, Appendix A-3 to Part 60 - Test Methods 4 through 5I, n.d.-a. <http://www3.epa.gov/ttnemc01/promgate/m-05.pdf> Accessed November 10, 2015.
- Epa, 2015. EPA, U.S. EPA Method 30B. Determination of Mercury from Coal-Fired Combustion Sources Using Carbon Sorbent Traps. Title 40, Chapter 1, Subchapter C, Part 60, Appendix A-8 to Part 60 - Test Methods 26 through 30B, n.d.-b. <http://www3.epa.gov/ttn/emc/promgate/Meth30B.pdf> Accessed November 10, 2015.

- Epa, 2015c. EPA, U.S. EPA Method 1. Sample and Velocity Traverses for Stationary Sources. Title 40, Chapter 1, Subchapter C, Part 60, Appendix A-1 to Part 60 - Test Methods 1 through 2F, n.d.-c <http://www3.epa.gov/ttnemc01/promgate/m-01.pdf> Accessed November 10, 2015.
- Epa, 2015d. EPA, U.S. EPA Method 2. Determination of Stack Gas Velocity and Volumetric Flow Rate (Type S Pitot Tube). Title 40, Chapter I, Subchapter C, Part 60, Appendix A-1 to Part 60 Test Methods 1 through 2F, n.d.-d <http://www3.epa.gov/ttnemc01/promgate/m-02.pdf> Accessed November 10, 2015.
- Epa, 2005. EPA, U.S. EPA. Environmental Protection Agency Standards of Performance for New Stationary Sources and Emission Guidelines for Existing Sources: Other Solid Waste Incineration Units (OSWI). Title 40, Part 60, 12 16 2005 pp. 74, 870–74, 924. <https://www.gpo.gov/fdsys/pkg/FR-2005-12-16/pdf/05-23716.pdf> Accessed August 6, 2018.
- Epa, 2014b. EPA, U.S. EPA Method 205, 2014 Verification of Gas Dilution Systems for Field Instrument Calibrations. <http://www.epa.gov/ttn/emc/promgate/m-205.pdf> Accessed June 17, 2015.
- Epa, 1989. EPA, U.S. EPA Method 3A, 1989 Determination of oxygen and carbon dioxide concentrations in emissions from stationary sources (instrumental analyzer procedure). <http://www.epa.gov/ttn/emc/promgate/m-03a.pdf> Accessed May 5, 2014.
- Epa, 2014c. EPA, 2014 U.S. EPA Method 6C. Determination of Sulfur Dioxide Emissions from Stationary Sources (Instrumental Analyzer procedure). <http://www.epa.gov/ttnemc01/promgate/method6C.pdf> Accessed June 17, 2015.
- Epa, 1999a. EPA, U.S. EPA Compendium Method IO-3.3, 1999 Determination of metals in ambient particulate matter using X-Ray Fluorescence (XRF) Spectroscopy. <http://www.epa.gov/ttnamti1/files/ambient/inorganic/mthd-3-3.pdf> Accessed May 5, 2014.
- Epa, 1999b. EPA, U.S. EPA Compendium Method IO-3.4, 1999 Determination of metals in ambient particulate matter using inductively coupled plasma (ICP) spectroscopy. <http://www.epa.gov/ttn/amtic/files/ambient/inorganic/mthd-3-4.pdf> Accessed May 5, 2014.
- Epa, 2018. EPA, U.S. EPA AP-42. The compilation of air pollutant emission factors, Volume I: Stationary point and area sources. n.d.-e. <https://www3.epa.gov/ttn/chief/ap42/ch02/final/c02s01.pdf> Accessed August 10, 2018.
- Epa, 1991. EPA, U.S. EPA Method 23, 1991 Determination of polychlorinated dibenzo-p-dioxins and polychlorinated dibenzofurans from stationary sources. 40 CFR Part 60, Appendix A. <http://www.epa.gov/ttn/emc/promgate/m-23.pdf> Accessed November 10, 2015.
- Epa, 1986. EPA, U.S. EPA SW-846 Test Method 0010, 1986 Modified method 5 sampling train. <https://www.epa.gov/sites/production/files/2015-12/documents/0010.pdf> Accessed June 3, 2016.
- Epa, 1999c. EPA, U.S. EPA Compendium Method TO-15, 1999 Determination of volatile organic compounds (VOCs) in air collected in specially-prepared canisters and analyzed by gas chromatography/mass spectrometry (GC/MS). <http://www.epa.gov/ttnamti1/files/ambient/airtox/to-15r.pdf> Accessed November 10, 2015.
- Epa, 2016. EPA, U.S. EPA Method 25C. Determination of nonmethane organic compounds (NMOC) in landfill gases. n.d.-f. <http://www.epa.gov/ttn/emc/promgate/m-25c.pdf> Accessed May 11, 2016.
- IM, 2011. IM, Institute of Medicine, 2011 Long-Term Health Consequences of Exposure to Burn Pits in Iraq and Afghanistan. The National Academies Press: Washington, D.C. <https://www.nap.edu/read/13209/chapter/1> Accessed August 6, 2018.
- MCCDC, 2011. MCCDC, Marine Corps Combat Development Command - Initial Capabilities Document, 2011 Initial capabilities document for United States Marine Corps - Expeditionary energy, water and waste. https://www.hqmc.marines.mil/Portals/160/Docs/MROCDM_46-2011_E2W2_ICD.pdf Accessed August 6, 2018.
- USDATL, 2012. USDATL, Under Secretary of Defense for Acquisition, Technology, and Logistics, 2012 Department of Defense Strategic Sustainability Performance Plan. <https://www.denix.osd.mil/sustainability/dod-sspp/unassigned/dod-sspp-fy-2012/> Accessed August 6, 2018.

- Laursen et al, 1992. Laursen KK, Ferek R, Hobbs P, Rasmussen RA, Emission factors for particles, elemental carbon, and trace gases from the Kuwait oil fires J. Geophys. Res, 97 (D13) (1992), pp. 14491–14497
- Lemieux et al, 2003. Lemieux PM, Gullett BK, Lutes CC, Winterrowd CK, Winters DL, Variables affecting emissions of PCDD/Fs from uncontrolled combustion of household waste in barrels J. Air Waste Manage, 53 (5) (2003), pp. 523–531
- Liu and Lipták, 1999. Liu DHF, Lipták BG, Hazardous Waste and Solid Waste Lewis Pub: CRC Press (1999), p. 288
- Margolin et al, 2015. Margolin JA, Marrone PA, Randel MA, Allmon WR, Mclean RB, Bozoian PM, 2015 Waste characterization analysis and standard test recipe development for contingency bases US Army Natick Soldier Research, Development and Engineering Center (NSRDEC) and US Army Research Laboratory (ARL): Unpublished Revision 1, January 26.
- Tchobanoglous and Kreith, 2002. Tchobanoglous G, Kreith F, Handbook of Solid Waste Management McGraw-Hill, New York (2002) Vol. Second Edition
- Woodall et al, 2012. Woodall BD, Yamamoto DP, Touati A, Gullett BK, Characterization of emissions from simulated deployed US military waste Environ. Sci. Technol, 46 (2012), pp. 10997–11003 [PubMed: 22947082]

Fig. 1.

Real time CO₂ and CO concentration versus time as well as the timing of waste loads and sampling collection for VOCs, PM, Metal/PM, mercury, and PCDD/PCDF/PAH during run number SW-2. Left axis corresponds to “CO₂ dry Vol-%” and the right axis corresponds to “CO conc.” values. Horizontal lines represent sampling duration for corresponding analyte. Left and right chamber loading is denoted by “x”.

Fig. 2.

PCDD/PCDF, mercury, PM, and PAH emissions factors. Bars denoted by “*” indicate range of data; all others are ± 1 std. dev.

Table 1.

Standard and challenge waste recipe by weight percent. standard and plastic waste recipes from ARL (Margolin et al., 2015).

Waste category	Standard	Challenge recipe		
	Recipe (SW)	Plastic (HP)	FSR	KMC ^c
Cardboard	15	10	12	16
Mixed paper	10	6.0	7.7	18
Food waste	32	21	43	44
Plastic	15 ^a	44 ^a	19 ^a	12
Wood	14	9.0	11	0.09
Metals	6.0 ^b	4.0 ^b	2.5 ^b	2.9
Glass	1.0	1.0	0.8	3.7
Rubber and neoprene	1.0	1.0	0.8	1.2
Textile	3.0	2.0	2.3	0.25
Miscellaneous waste/other	3.0	2.0	2.3	1.8
Total	100	100	100	100

^aPlastic breakdown in Table 2.

^b60% iron, 36% aluminum, and 4% other metals.

^c20-bag average composition sampled from dumpster.

Table 2.

Breakout of waste recipes by weight percent. Standard and plastic recipes from ARL (Margolin et al., 2015).

Plastic category	Standard	Challenge recipe		
	Recipe (SW)	Plastic (HP)	FSR	KMC
Plastic (Total)	15	44	19	12
#1 Polyethylene terephthalate (PET)	6.0	18	4.6	1.1
#2 High density polyethylene (HDPE)	2.7	7.8	2.1	2.5
#3 Polyvinyl chloride (PVC)	0.9	2.6	0.7	0
#4 Low density polyethylene (LDPE)	2.7	7.8	2.1	3.8
#5 Polypropylene (PP)	0.3	0.8	0.2	0.93
#6 Polystyrene (PS)	1.8	5.4	1.4	2.8
#7 Other (e.g., polycarbonate, acrylic, nylon, bioplastics, composites)	0.6	1.6	7.9	0.68

Table 3.

Target pollutants and instrumentation/sampling methods.

Pollutant	Instrument/method(s)	Duration
Total PM	Modified U.S. EPA Method 5 (EPA, n.d.-a)	Integrated run, 0–4 h
PCDDs/PCDFs	U.S. EPA Method 23 (EPA, 1991)/HRGC-HRMS	Integrated run, 0–4 h
PAHs	U.S. EPA Method 0010 (EPA, 1986)/HRGC-LRMS	Integrated run, 0–4 h
VOCs	SUMMA Canister/U.S. EPA Method TO-15 (EPA, 1999c), including CO ₂ , CO, CH ₄ / U.S. EPA Method 25C (EPA, n.d.-f)	Integrated run, ~2 h
NO _x , O ₂ , CO, CO ₂ SO ₂ , CH ₄ , HCl	FTIR - Gasmet DX-4000 (Finland)	Real time
CO ₂	LI-COR 820 (Biosciences, USA)	Real time
Metal: Mercury	Sorbent trap/U.S. EPA Method 30B (EPA, n.d.-b)	Integrated run, 0–4 h
Metals: Cd, Pb, others	Teflon filters/Modified U.S. EPA Method 5 (EPA)/gravimetric and X-Ray Fluorescence (XRF) (EPA, 1999a) and/or Inductively coupled plasma (ICP) (EPA, 1999b)	Integrated run, 0–4 h

Table 4.

Emissions concentrations by waste type.

Pollutant	Units	SW (N = 3)			HP (N = 2)			KMC (N = 1)			FSR (N = 2)			Composite (N = 8)			OSWI ^d
		Mean	RPD (%)	CV (%)	Mean	RPD (%)	CV (%)	Mean	RPD (%)	Mean	Mean	RPD (%)	Mean	RPD (%)	Mean	CV (%)	
PM _{Total}	mg/m ³ at 7% O ₂	39		15	41	9.5		18	N/A	60			16	42	35	30	
PM _{Total} (Teflon)	mg/m ³ at 7% O ₂	38	7.7		34	19		16	N/A	13			15	27	48	...	
ΣPCDD/PCDF WHO-TEQ	ng TEQ/m ³ at 7% O ₂	0.27		22	0.27	7		0.26	N/A	0.68			3.4	0.37	52	...	
Total PCDD/PCDF	ng/m ³ at 7% O ₂	53		17	54	12		61	N/A	108			9.0	68	38	33	
PAH TEQ (16 EPA PAHs)	µg B[a]P/m ³ at 7% O ₂	33		127	6.8	3.4		6.1	N/A	6.9			26	17	159	...	
Total PAH (16 EPA PAHs)	µg/m ³ at 7% O ₂	2389		100	685	1.3		994	N/A	1101			58	1467	105	...	
NO _x	dry PPM	196	5.1		143	2.4		254	N/A	257			1.4	207	25	103	
SO ₂	dry PPM	0.02	100		0.011	27		0.16	N/A	0.44			30	0.16	137	3.1	
HCl	dry PPM	0.46	12		0.57	21		0.66	N/A	1.4			28	0.79	62	15	
Benzene	µg/m ³ at 7% O ₂	4377		170	666	42		1458	N/A	1119			88	2270	35	...	
Toluene	µg/m ³ at 7% O ₂	84		156	16	46		644	N/A	137			62	133	166	...	
Acrolein	µg/m ³ at 7% O ₂	26		14	25	94		13	N/A	129			14	50	145	...	
Vinyl Chloride	µg/m ³ at 7% O ₂	14	57		28	92		8	N/A	38			79	24	108	...	
Mercury	µg/m ³ at 7% O ₂	0.73		24	0.31	12 ^c		0.53	14	0.65			15.7 ^c	0.60	35	74	
Cd ^d	µg/m ³ at 7% O ₂	1.04		52	0.49	16		0.37	N/A	<MDL [0.089]			N/A	0.75	64	18	
Pb ^d	µg/m ³ at 7% O ₂	122		66	83	13		32	N/A	61			20	86	65	226	
Fe ^d	µg/m ³ at 7% O ₂	45		82	61	41		5.4	N/A	40			53	43	74	...	
Cu ^d	µg/m ³ at 7% O ₂	150		40	169	29		70	N/A	174			26	151	39	...	
Na ^d	µg/m ³ at 7% O ₂	3015		50	5829	25		2873	N/A	2420			22	3552	52	...	
CO	dry PPM	68	34		101	43		190	N/A	39			46	86	70	40	
CO ₂	dry Vol-%	9.9	0.08		9.4	2.1		8.3	N/A	9.8			0.60	9.5	6.0	...	
CH ₄	dry PPM	0.6	89		0.75	74		1.0	N/A	0.42			95	0.6	83	...	
O ₂	dry Vol-%	9.0		10	9.0	0.50		9.33	N/A	8.7			0.42	9.0	6.0	...	

Pollutant	Units	SW (N = 3)			HP (N = 2)		KMC (N = 1)		FSR (N = 2)		Composite (N = 8)		OSWI ^d
		Mean	RPD (%)	CV (%)	Mean	CV (%)	Mean	RPD (%)	Mean	RPD (%)	Mean	RPD (%)	CV (%)
H₂O	Vol-%	9.5	5.3	9.4	9.4	5.1	9.62	N/A	11	N/A	10	2.6	10
Moisture^b	Vol-%	10.7		6.4	10.5	9.9	10.70	N/A	13	N/A	11	2.2	1
Ash	%	12		34	15	N/A	11	N/A	9.0	N/A	12	N/A	28
Waste Load	kg/hr	28		17	18	25	30	N/A	26	N/A	25	0.65	25
Stack Flow	m ³ /min	4.2		4.4	4.5	0.66	4.2	N/A	3.9	N/A	4.2	8.8	7.3

^aInductively Coupled Plasma Spectroscopy (ICP).

^bDerived from Method 5.

^cValue is coefficient of variation (CV).

^dOther Solid Waste Incineration Unit regulation limits (EPA, 2005). [...] = No set regulatory standard. N/A = not applicable, single sample.

Table 5.

Emissions factors by waste type.

Pollutant	Units	SW (N = 3)			HP (N = 2)		KMC (N = 1)		FSR (N = 2)		Composite (N = 8)		AP-42 EF UnC ^d	EF ESP ^e
		Mean	RPD (%)	CV (%)	Mean	RPD (%)	Mean	RPD (%)	Mean	RPD (%)	Mean	CV (%)		
PM _{Total}	g/kg waste	0.23	15		0.39	10	0.166	N/A	0.62	4.7	0.38	50	1.72	0.174
PM _{Total} (Teflon)	g/kg waste	0.30	20		0.24	6.1	0.15	N/A	0.120	11	0.21	42	...	...
ΣPCDD/PCDF WHO-TEQ	ng TEQ/kg waste	1.7	20		2.5	6.1	2.4	N/A	7.1	7.5	3.6	68	...	...
Total PCDD/PCDF	ng/kg waste	317	18		507	11	573	N/A	1112	2.0	635	54	1470	1880
PAH TEQ (16 EPA PAHs)	µg B[a]P/kg waste	274	86		69	4.6	57	N/A	84	36	130	129	...	...
Total PAH (16 EPA PAHs)	mg/kg waste	17	83		6.5	0.68	9.3	N/A	12	65	12	90	...	...
NO _x (as NO ₂)	g/kg waste	2.9	4.98		2.95	4.6	5.47	N/A	5.25	1.9	4	33	1.58	*
SO ₂	g/kg waste	0.0003	N/A		0.0002	29	0.0025	N/A	0.006	30	0.003	122	1.61	*
HCl	g/kg waste	0.005	12.4		0.009	19	0.011	N/A	0.023	28	0.012	69	1.08	*
Benzene	mg/kg waste	95		172	7.1	42	16	N/A	13.6	92	43	228	...	...
Toluene	mg/kg waste	1.8		168	0.17	46	7.1	N/A	1.6	74	2.0	139	...	...
Acrolein	mg/kg waste	0.29		62	0.26	94	0.15	N/A	1.5	87	0.58	164	...	...
Vinyl Chloride	mg/kg waste	0.095	54		0.30	91	0.090	N/A	0.45	87	0.25	126	...	...
Mercury	µg/kg waste	6.48		51	2.86	18 ^c	4.86	17	6.3	28	5.16	47	2800	*
Cd ^a	mg/kg waste	0.0089	10		0.0038	39	0.0035	N/A	MDL	N/A	0.0058	54	1.2	0.23
Pb ^a	mg/kg waste	0.92	36		0.64	37	0.30	N/A	0.62	43	0.66	53	...	1.4
Fe ^a	mg/kg waste	0.31	61		0.41	18	0.051	N/A	0.34	32	0.31	58	...	...
Cu ^a	mg/kg waste	1.0	15		1.2	4.7	0.66	N/A	1.6	0.50	1.2	28	...	...
Na ^a	mg/kg waste	25	10		41	0.40	27	N/A	22	3.6	29	29	...	...
CO	g/kg waste	0.61	34		1.25	41	2.5	N/A	0.48	46	1.03	79	0.15	*
CO ₂	g/kg waste	1396	0.024		1842	0.04	1714	N/A	1912	0.019	1716	13.3	985	*
CH ₄	g/kg waste	0.003	98		0.005	73	0.008	N/A	0.0030	95	0	85	...	...

^aInductively Coupled Plasma Spectroscopy (ICP).

EPA Author Manuscript

EPA Author Manuscript

EPA Author Manuscript

^cValue is coefficient of variation (CV).

^dEmissions factors from AP-42 for uncontrolled units, UnC(EPA).

^eEmission factors from AP-42 for units with electrostatic precipitators, ESP (EPA). N/A = not applicable, single sample. [...] = No Emission Factor established by AP-42. [*] = Same as “uncontrolled” for these pollutants.

Table 6.

Gasification/combustion emission factors determined here compared to combustion emissions from burning of simulated forward operating base waste.^a

Compound	Unit	Gasification/ combustion	Burn box (Aurell, et al., 2012)	Burn pile (Aurell, et al., 2012)	MRE OBTF (Dominguez, et al., 2018)
Σ PCDD/PCDF	ng TEQ/kg waste	3.8 ± 2.8	35 ± 24	1,765 ± 1,474	1.802 ± 1.040
Σ PAH ^b	mg/kg waste	11 ± 9.8	43 ± 50	129 ± 50	96 ± 37
PM	g/kg waste	0.39 ± 0.22	12 ± 12 ^c	39 ± 24 ^c	12 ± 3 ^c
Iron	mg/kg waste	0.32 ± 0.18	0.50 ± 0.24	11 ± 23	0.29 ± 0.05
Copper	mg/kg waste	1.2 ± 0.40	0.18 ± 0.11	0.89 ± 0.92	0.17 ± 0.06
Cadmium	mg/kg waste	0.0058 ± 0.0031	0.063 ± 0.082	0.073 ± 0.033	0.089 ± 0.038
Lead	mg/kg waste	0.68 ± 0.36	0.55 ± 0.42	0.37 ± 0.22	0.07 ± 0.01
Benzene	mg/kg waste	43 ± 98 ^d	243 ± 299 ^d /1,371 ± 185 ^e	260 ± 288 ^d /2,421 ± 1,265 ^e	211 ± 89 ^d
Toluene	mg/kg waste	2.0 ± 2.8 ^d	88 ± 130 ^d /652 ± 111 ^e	109 ± 170 ^d /1,202 ± 727 ^e	63 ± 36 ^d
Acrolein	mg/kg waste	0.61 ± 1.1 ^d	133 ± 139 ^d /463 ± 33 ^e	98 ± 108 ^d /757 ± 62 ^e	125 ± 40 ^d
Vinyl chloride	mg/kg waste	0.27 ± 0.35 ^d	3.7 ± 2.5 ^d /13 ^e	6.0 ± 5.5 ^d /26 ± 3.3 ^e	0.55 ± 0.27 ^d
Vinyl acetate	mg/kg waste	0.62 ± 0.82 ^d	79 ± 97 ^d /324 ± 46 ^e	43 ± 53 ^d /688 ± 195 ^e	101 ± 41 ^d

^aRange of data denoted ± 1 STDV.

^b16 EPA PAHs (see Tables S14 and S15).

^cPM2.5.

^dMCE > 0.95.

^eMCE < 0.90.