

Emissions from Washington State Compost Facilities: A Review of Volatile Organic Compound Data, and an Estimation of Greenhouse Gas Emissions

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Table of Contents

	<u>Page</u>
List of Figures and Tables.....	ii
Figures.....	ii
Tables.....	iii
List of Abbreviations	iii
Acknowledgements.....	iv
Abstract.....	v
Introduction.....	1
Review of existing emissions data from Washington State composting facilities	3
Washington State compost emissions data	4
Volatile organics emitted from Washington compost facilities.....	6
EPA green waste speciation profile	9
Duplicate samples	10
Composition differences from pile aging	10
Total VOC and compost emission factors	11
Environmental impacts of compost emissions.....	14
Criteria for Title V air operating permits and impact on Washington composting facilities	16
Assessment of VOC emission potential from compost facilities.....	16
Impact of Title V permitting.....	21
Comparison of greenhouse gas emission factors from landfilling and composting	21
Comparison using LandGEM	23
Comparison using WARM model	24
Conclusions.....	27
References.....	28
Appendix.....	31
Appendix 1. Compost Facility Emission Samples Collected by Ecology	31

List of Figures and Tables

Figures

Figure 1: Air quality agency jurisdictions in Washington state (Ecology, 2019a).....	1
Figure 2: Schematic showing surface flux isolation chamber sampling methodology on top of a static pile	5
Figure 3: Fractional composition of VOC emissions from the highest flux density sample collected at compost facility #2 and facility #4. In both samples, ethanol dominated overall emissions. Monoterpenes were not reported in the facility #2 sample analysis	7
Figure 4: Fractional composition of VOC emissions from the highest flux density sample collected at composting facilities #3 and #6.	8
Figure 5: Fractional composition of VOC emissions from the highest flux density sample collected at composting facilities #4 and #5. Monoterpenes were not reported in the facility #1 sample analysis, and alcohols were not observed in the facility #5 sample analysis	9
Figure 6: Green waste compost emissions profile from EPA speciate data base, profile number 8933 (EPA, 2016a).....	10
Figure 7: Comparison of total VOC flux density from windows at different ages at facility #5. 11	
Figure 8: Emissions profile of total daily VOC emitted from a green waste/food waste windrow at the Modesto, CA compost facility (CARB, 2007).....	13
Figure 9: Locations of composting facilities in Washington State (Ecology, 2019b).....	16
Figure 10: Methane emissions from a municipal landfill with a 50-year operating life as modelled by LandGem. Lower green curve shows emissions assuming 75% of emitted methane is captured on site.....	23

Tables

Table 1: Comparison of VOC emission flux densities ($\mu\text{g m}^{-2} \text{min}^{-1}$) for samples with the highest emission rates.	6
Table 2: Rate coefficients for the dominant volatiles emitted from composting and their maximum incremental reactivity values (MIR) from Atkinson, 2000; Atkinson et al., 2007; and Carter 2010.....	15
Table 3: Washington State’s largest composting facilities and 2017 annual throughputs (Ecology, 2019b)	17
Table 4: California Air Resources Board recommended emission factors for green waste composting (CARB, 2015).	18
Table 5: Estimated annual total VOC emission rates assuming California Air Resource Board green waste emission factors and 2017 annual throughputs. Uncontrolled emissions assumes no emissions control technology, while controlled emissions assume that emissions	20
Table 6: List of landfills with no LFG collection system in the state of Washington	25
Table 7: List of landfills with LFG collection system in the state of Washington	26
Table 8: Organic waste weight by material type used in WARM model for two landfill types: landfilling with and without LFG collection system.	26
Table 9: Net CO ₂ equivalent emissions in mega tonnes (MT CO ₂ e) from diversion of organic waste from landfills to composting.....	26

List of Abbreviations

ASP	Aerated static pile
ASTM	American Society for Testing and Materials
CARB	California Air Resources Board
C.F.R.	Code of Federal Regulations
CH ₄	methane
DNPH	2,4-dinitrophenylhydrazine

DMS	dimethyl sulfide
EPA	United States Environmental Protection Agency
FID	flame ionization detector
GC-MS	gas chromatography mass spectrometry
HAP	hazardous air pollutants
He	helium
kg	kilograms
L	liters
lbs	pounds
MT CO _{2e}	metric tons carbon dioxide equivalent
NH ₃	ammonia
N ₂ O	nitrous oxide
PM _{2.5}	particulate matter less than 2.5 microns in diameter
IPCC	Intergovernmental Panel on Climate Change
tpy	tons per year
µg	micrograms
U.S.C	United States Code
VOC	volatile organic compound

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Abstract

Volatile organic compound (VOC) emissions data collected from six Washington State compost facilities from a survey performed by the Washington State Department of Ecology (Ecology) between 2010 and 2013 was examined. The purpose was to determine what compounds are emitted and to determine total VOC emission rate factors. These factors are required to determine if annual emissions of volatile organic compounds from commercial compost facilities are large enough to warrant EPA Title V air permitting. The data reveal that total emissions are dominated by a few compounds, notably light alcohols such as ethanol and monoterpenes such as α -pinene and limonene. However, these compounds were often not reported as target compounds by the contracting analytical labs performing the analysis. As a result, this data set did not provide a complete picture of emissions. When both types of compounds were reported, as occurred for two facilities, the emissions of five compounds (ethanol, α -pinene, limonene, acetone, 2-butanone) accounted for 90% of all reported VOC emissions. While the survey examined several emitting sources within a facility, there was no emissions data from a windrow over its life-cycle from which an emission factor (lbs VOC / wet ton compost) could be derived. The Washington compost data differed significantly from the green waste composting VOC emissions profile listed in the EPA Speciate data base. This profile is based on a California compost study that reported i-propanol dominated the emissions (42%) while in Washington compost emissions it was seldom detected.

Estimates of annual VOC emissions from Washington compost facilities were determined using total VOC emission factors derived from California studies of green waste composting. Applying these factors to the annual material throughputs reported by Washington composting facilities indicates that perhaps seven facilities in the state may emit more than 100 tons VOC per year, the threshold for Title V permitting.

A simple assessment comparing methane and nitrous oxide emissions from composting and landfilling was performed using the EPA WARM model. Diversion of food waste from landfilling to composting significantly reduces emissions of methane, making composting of organic waste a preferred organic waste management practice to reduce greenhouse gas emissions from solid waste disposal.

PM_{2.5}, PM₁₀, lead), and ozone depleting substances (40 C.F.R. § 70.2). As stated in the Department of Ecology web site on Title V permitting, sources emitting less than 100 tpy threshold may still be subject to permitting under federal rules of the Clean Air Act related to Title IV Acid Rain, New Source Performance Standards, and National Emission Standards for Hazardous Air Pollutants (40 C.F.R. § 70.3).

Lower total VOC and NO_x thresholds apply for regions that are listed as ozone non-attainment areas. As of April 30, 2019, there were no areas in Washington that were listed as ozone non-attainment by the US EPA (EPA, 2019a). However, ozone pollution in the Tri-Cities area of Central Washington is a recent concern. Ozone is now being regularly monitored by Washington State Department of Ecology in the Tri-Cities because summertime concentrations of ozone sometimes exceed National Ambient Air Quality Standards (Jobson and VanderSchelden, 2017). There are many ozone non-attainment areas in California that vary in their classification from “Extreme” to “Marginal.” Classification impacts emission thresholds of VOC and NO_x from major sources. For ozone non-attainment areas classified as “Serious,” the total VOC or NO_x emission threshold is reduced to 50 tpy; for “Severe” classifications the threshold it is 25 tpy, and for “Extreme” classification it is 10 tpy (40 C.F.R. § 70.2).

Volatile organic compounds are defined as any compound containing carbon that can participate in photochemical reactions in the atmosphere (40 C.F.R. § 51.100) with the notable exceptions including methane, ethane, acetone, carbon monoxide, carbon dioxide, which have negligible photochemical reactivity in the troposphere. Many of the compounds emitted from composting facilities would be regarded as regulated VOC compounds. Photochemical reactions of VOC compounds are known to produce ozone and PM_{2.5} but these impacts are typically limited to summer when photochemical reactivity is greatest, and to air sheds with large emissions of both VOCs and NO_x.

Many VOCs are further classified by the EPA as hazardous air pollutants (HAPs) because exposure leads to elevated risk of cancer or other serious health risks. A list of these HAP compounds (42 U.S.C. Chap 85, § 7412) includes several compounds that have been identified in compost emissions, notably methanol, acetaldehyde, and formaldehyde. Actual cancer risk depends on air concentrations of these pollutants and this varies from compound to compound.

This review project had three principle tasks. The first task was to review existing emission data collected at commercial composting facilities in Washington by Ecology’s Solid Waste Management Program. The purpose of this task is to establish emission rates of VOCs in consultation with the Solid Waste Management and Air Quality Programs within the Department of Ecology, and to evaluate the types of compounds emitted and their potential impact as air toxics and as precursors for photochemical ozone and PM_{2.5}. The task will evaluate whether total VOC emission rates can be used as reliable estimates of VOC and air toxic emission rates from facility to facility.

The second task was to review the criteria established for Title V permitting and evaluate the potential impact of such regulation on commercial composting facilities. Title V air operating permits are used for facilities that are major emission sources of pollutants such as VOCs.

The third task was to compare greenhouse gas emissions rates from composting to organic debris disposal in landfill.

Review of existing emissions data from Washington State composting facilities

Ecology sampled emissions from six Washington State composting facilities from 2010 to 2013. These are among the largest compost facilities in the state as measured by material throughput. Samples were collected to determine surface emission rates of VOCs, sulfur compounds, and ammonia (NH₃) using a flux isolation chamber method prescribed by the EPA (EPA,1986). Typically, nine samples were collected from each facility from several types of emission sources, including aerated static piles (ASP), finished and fresh piles, covered piles, and leachate ponds. Air samples from the flux isolation chamber were collected into SUMMA canisters for VOC analysis, into Tedlar bags for sulfur compounds analysis, and onto DNPH impregnated absorbent tubes for aldehyde analysis. Samples were analyzed by three different contract laboratories, using standard EPA methods for analysis: TO-15 for VOCs, TO-11a for aldehydes, and method 3C for helium. Sulfur compounds were determined by ASTM D 5504-08. Lab#1 used ASTM D14965 for helium analysis and a modified TO-15 method utilizing a flame photometric detector for sulfur compound analysis. The data examined in this report consist of the original lab reports (pdf files) from the contracting laboratory and data from these reports compiled into Excel spreadsheets by Ecology.

In the data files obtained from Ecology, the flux density values ($\mu\text{g m}^{-2} \text{min}^{-1}$) for each compound had been calculated from the reported concentrations. These flux density values were used in this report's analysis. Determining the flux density is the first step in determining an emission inventory for a facility. The flux density must be determined over the course of the windrow life cycle (~30 days) in order to determine a total VOC or NH₃ mass emission per initial unit mass wet compost. The sampling conducted by Ecology was only performed once at each facility, so there is not enough data to determine an emission profile over the windrow life cycle from which to determine an emissions factor (lbs VOC / wet ton compost). The Ecology data were examined with the following objectives:

- 1) to determine what compounds are emitted from Washington State composting facilities,
- 2) to determine if emissions are similar from facility to facility,
- 3) to compare these emission profiles to those established in California,

- 4) to evaluate how the emission profile impacts calculation of total VOC emissions by EPA method 25 or South Coast Air Quality Management District (SCAQMD) method 25.3 used to determine VOC emission factors for compost in California, and
- 5) to determine potential environmental impacts of compost emissions.

Washington State compost emissions data

The samples collected by Ecology from Washington composting facilities are listed in Table A.1 through Table A.6 in Appendix 1. The sampling protocol called for a field blank sample and a duplicate sample to be collected at each facility for quality control. Field blanks are noted in the table as “Blanks” and duplicate samples are identified as “Duplicates.” Duplicate samples are two samples (i.e., canisters) collected from the flux isolation chamber at the same time. A schematic of the flux isolation chamber sampling is shown in Figure 2. In this method the chamber is placed on the windrow surface and clean air at a known flow (5 L min^{-1}) flows through the chamber and vents to ambient air via an opening in the top. Air sampling lines are passed into the chamber through this opening to collect air samples. Further dilution of the chamber is possible from positive pressure aerated piles (air flow from the pile into the chamber) and from wind forcing air into the chamber vent. For this reason, helium is added to the dry supply air to act as a tracer of extra dilution. In these studies, the helium concentration in the air flow supplied to the chamber was 10% by volume. The canisters were analyzed for helium to determine concentrations in the chamber. The measured helium concentration was used to correct the air flow into the chamber to account for extra dilution. The surface flux density of a compound can then be calculated, assuming steady state conditions, from the measured chamber concentration ($\mu\text{g m}^{-3}$), the supplied dry air flow rate into the chamber ($\text{m}^3 \text{ min}^{-1}$), the fractional helium recovery, and the surface area sampled by the chamber (0.13 m^2) (EPA, 1986):

$$\text{flux density} = \text{concentration} \times \text{air flow rate} \times \left(\frac{10\% \text{ He}}{\% \text{ He}_{\text{sample}}} \right) / \text{area} \quad (\text{eq. 1})$$

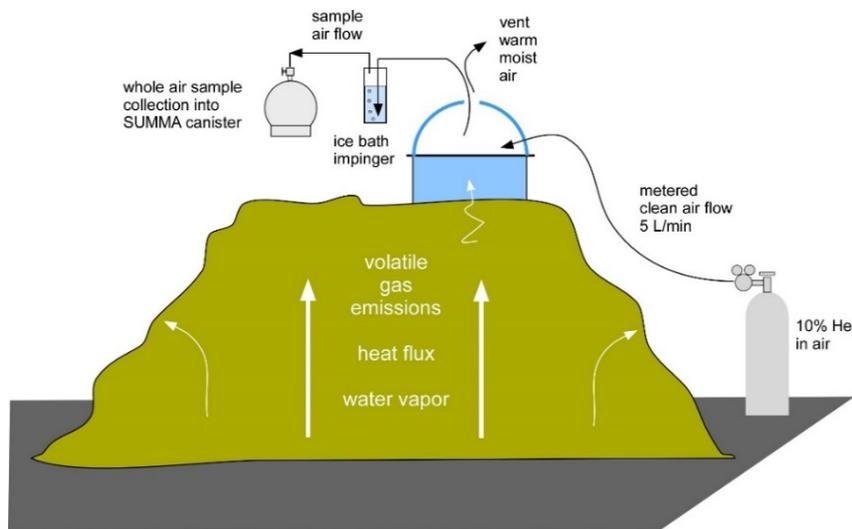


Figure 2: Schematic showing surface flux isolation chamber sampling methodology on top of a static pile

We noted in the contractor reports and indicated in the Table A.1 through Table A.6 that the He recovery was often quite low (sometimes < 0.1% instead of the expected 10%) indicating larger than expected dilutions of the flux chamber and substantial flow correction factors to calculate flux densities from equation (1). For the Cedar Grove Maple Valley sampling, where positive flow was indicated in the sample description for the aerated static pile sampling, the concentration was 0.21% and 0.30% for those samples, compared to 4.89% and 1.03% for negative aeration samples. This difference in helium concentration suggests the positive flow was actively diluting the flux isolation chamber with air coming up through the pile. For the positive aeration samples, the flow correction was thus a factor of 48 and 33 respectively. We note this difference because while positive flow might cause dilution of the helium in the chamber this would not cause dilution of the VOCs which in fact might be higher in concentration due to the air flowing up through the pile carrying emissions. In this case correcting for He might be a significant error. It is difficult to assess from the data how much the low helium recoveries were due to dilution by wind or dilution from aeration flows. Sampling from aerated static piles with flux isolation chambers could be problematic for this methodology.

Another potential sampling issue is that sampling of high humidity sources like compost piles typically requires water vapor removal to prevent water condensing in sampling lines and collection vessels. This is important for VOC sampling as many alcohols, acids, and aldehydes, are water soluble and would be lost to condensed water on sampling surfaces. Likely water vapor removal techniques are more important for certain times of year, especially when there are large temperature differences between hot compost piles with high water vapor flux and cold ambient temperatures, causing condensation of emitted water vapor on ambient temperature sampling lines and surfaces. Typically, an impinger containing a small volume of water held at ice temperatures is used to remove excess water before sampling into canisters, Tedlar bags, or adsorbent tubes. The impinger water is then also analyzed for dissolved organics and included in the surface flux calculations. An impinger apparently wasn't used in the sampling conducted at these facilities and so sampling losses of alcohols and aldehydes may have occurred.

Volatile organics emitted from Washington compost facilities

The data acquired by Ecology involved sampling from a variety of potential sources at the facilities. As a general observation, most of the VOCs reported in the TO-15 analysis were below method reporting limits. Thus, only a dozen or so compounds significantly contribute to the VOC flux density. To compare results between facilities we compared samples that had the highest flux densities, and summarized emission results for the dominant VOCs in Table 1.

Table 1: Comparison of VOC emission flux densities ($\mu\text{g m}^{-2} \text{min}^{-1}$) for samples with the highest emission rates.

Facility	#1	#2	#3	#4	#5	#6
Sampling Date	June 15, 2010	June 13, 2011	Nov 11, 2012	April 16-17, 2013	June 26-27, 2013	June 5-6, 2013
Analysis Lab	Lab #1	Lab #1	Lab #2	Lab #2	Lab #3	Lab #3
Sample	Fresh Pile	Chop Mix	Pile #3, 3 weeks	Fresh Built	Fresh ASP	Pre turn 1 week
methanol	498	1,720	NR	NR	NR	NR**
ethanol	593	10,273	10,516	82,359	ND	56,721
i-propanol	ND	ND	80	ND	ND	ND***
acetone	75	359	673	ND	ND	12,012
2-butanone	795	366	904	4,532	ND	4,338
α-pinene	NR	NR	989	23,071	57,841	28,361
limonene	NR	NR	610	21,835	43,702	22,021
DMS	ND	< MRL****	190	4,944	2,057	468
formaldehyde	ND	3.9	3	107	ND	177
acetaldehyde	61	237	610	317	27	137
Total VOC	NR	31,181	NR	90,634*	167,097*	166,827*

* reported as toluene equivalent

** NR = Not reported (no analysis for the compound)

*** ND = Not detected

**** < MRL = detected but below method reporting limit

The flux densities were dominated by three types of compounds. These were light the alcohols (ethanol, methanol), ketones (acetone, 2-butanone), and monoterpenes (α -pinene and limonene). In some of the analytical results light alcohols were not reported (methanol) and often not detected (i-propanol) and in other monoterpenes were not reported (facility #1 and facility #2, sample analysis by lab #1). This is likely an analytical omission rather than an absence of emission. In addition, there was a significant flux density of dimethyl sulfide (DMS), an odor-causing compound, at some facilities.

Smaller emissions of air toxic compounds formaldehyde and acetaldehyde were also observed but, in general, emissions of air toxic compounds in the samples were low and comprised a small fraction of total emissions. Also included in Table 1 is the alcohol isopropanol (i-propanol; aka 2-propanol) that was found to be the dominate emission (42% of total VOC flux) in a study of California green waste composting (Kumar et al. 2010). Results from that study are the only listing for a green waste composting VOC emission profile (profile #8933) in EPA’s Speciate Database (ver 4.4; EPA, 2016a). This database provides emissions profiles (relative mass emission rates) of volatile organic compounds from hundreds of pollution sources. Speciated emission profiles are used in air quality photochemical transport modeling. Isopropanol was above reporting limits for only one Washington composting facility (#3) but at very low emission rates compared to ethanol. The absence of i-propanol emissions and larger monoterpene emissions are notable differences from the California study and suggests that Washington compost facilities have significantly different VOC emission profiles than California green waste composting facilities.

Figure 3 compares the fractional amount of the VOC emission by compound class between the highest emitting samples from facility #2 (chop mix sample) and facility #4 (fresh built windrow sample) as noted in Table 1. While these facilities could be expected to have similar emission profiles, monoterpene compounds were not reported for the facility #2 samples, so its emissions are dominated by alcohols. Ethanol alone makes up 77% of the speciated emissions. In contrast, the facility #4 sample is dominated by ethanol (59%) and monoterpenes (32%) emissions. The facility #2 sample had reported concentrations for 13 compounds, while 19 compounds were reported for the facility #4 sample. Thus, relatively few compounds were speciated and a large fraction of the emissions are from only a few compounds.

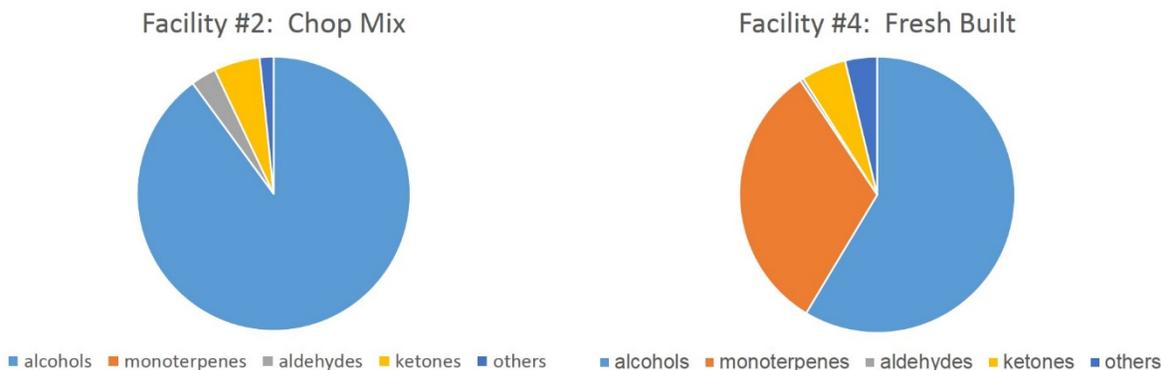


Figure 3: Fractional composition of VOC emissions from the highest flux density sample collected at compost facility #2 and facility #4. In both samples, ethanol dominated overall emissions. Monoterpenes were not reported in the facility #2 sample analysis

A total VOC emission was also reported as part of the analysis done at both these facilities. Total VOC amount was reported as toluene mixing ratio from the TO-15 analysis at facility #4, while total VOC was reported using EPA TO-14 method for the facility #2 samples. This metric would be a rough indicator of “unidentified” compounds in the sample analysis. Because these metrics determine total VOC in different ways, they are not directly comparable. The TO-14 method

would presumably use a flame ionization detector (though this was not specifically stated by the analytical lab) which acts as a carbon counter and is a standard detector for such purposes. By using a TO-15 method to determine total VOC, the reporting labs are summing the measured peak areas in the total ion chromatograms and using the toluene response factors to convert the total integrated peak area into an equivalent toluene mixing ratio. Because each compound has a unique GC-MS response factor, the sum of the speciated emission rates will not be equal to the total VOC as toluene metric. At best this metric can serve as a rough guide for unidentified peaks if this value is very large compared to the sum of the speciated flux densities. This appears to be the case for the facility #5 sample shown in Table 1 where alcohol concentrations were not reported.

Figure 4 compares the fractional amount of the VOC emission by compound class between the highest emitting sample from facility #3 (pile #3, 3 weeks old) and facility #6 (1 week old pile) and as noted in Table 1. These facilities are compared because the analytical reporting was similar. For both these samples ethanol was the dominant VOC emitted and comprised 70% and 45% of total emissions from the facility #3 windrow and facility #6 windrow, respectively. For the facility #6 sample, α -pinene made up 22% of total emissions and limonene 17%. In both samples emissions of acetone and 2-butanone (MEK) were also significant, making up 14% of facility #6 emissions and 10.5% of the facility #3 emissions. Again, only few compounds were reported, 22 compounds for facility #3 and 23 compounds for facility #6, and just a few compounds comprise the bulk of the emissions. The five principal compounds, ethanol, α -pinene, limonene, acetone and 2-butanone made up 97% of emissions from the facility #6 sample and 91% from the facility #3 sample. Interestingly acetaldehyde emissions, an air toxic, were as large as limonene emissions from the facility #3 sample, accounting for 4% of total emissions.

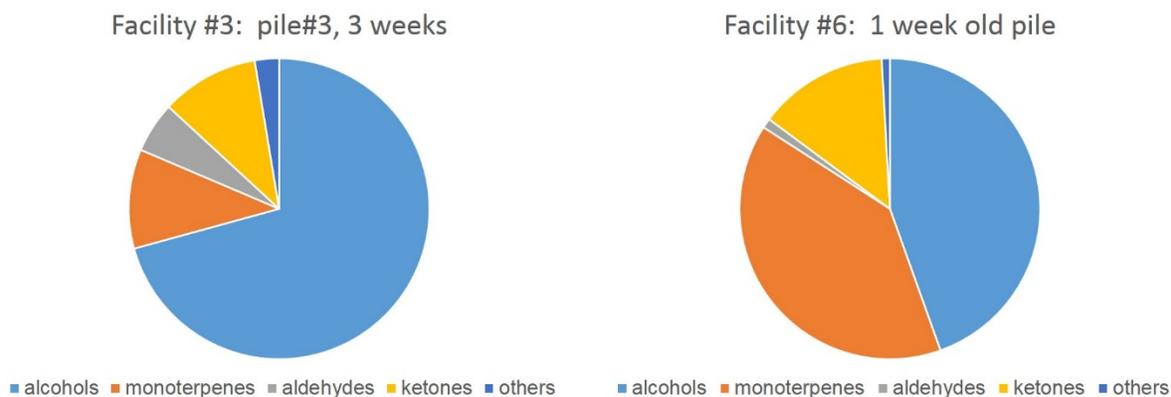


Figure 4: Fractional composition of VOC emissions from the highest flux density sample collected at composting facilities #3 and #6.

Figure 5 compares the fractional amount of the VOC emission by compound class between the highest emitting sample from facility #1 (fresh chopped sample) and facility #5 (fresh ASP sample) as noted in Table 1. These facilities were compared to contrast the very different conclusions that would be drawn based on the differences in the analytical reporting from the contract labs. The pie charts look very different because monoterpenes were not reported for the

facility #1 analysis and alcohols (ethanol and i-propanol) were not observed for the facility #5 analysis. Given the abundance of both alcohols and monoterpenes in the sample analyses from the other compost facilities, we conclude the emission data obtained from these two sites are likely not representative of the actual pile emissions and the measured flux densities are an underestimate of actual VOC emissions.

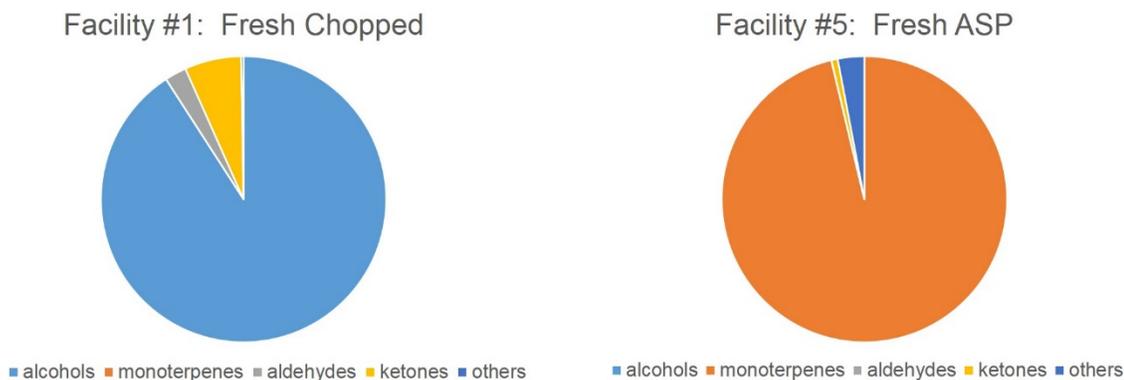


Figure 5: Fractional composition of VOC emissions from the highest flux density sample collected at composting facilities #4 and #5. Monoterpenes were not reported in the facility #1 sample analysis, and alcohols were not observed in the facility #5 sample analysis

EPA green waste speciation profile

The pie charts presented from these Washington facilities can be compared to the green waste composting profile listed in the EPA Speciate database (EPA, 2016a). This profile is shown in Figure 6 and the profile is based on the data obtained by Kumar et al. (2011). In these emission profiles monoterpene type compounds make-up only 6.6% of the total emissions by mass; the most abundant being: α -pinene (1.4%), limonene (2.3%), camphor (1.2%), and eucalyptol (0.6%). Light alcohols make up the majority of the emissions: i-propanol (42%), ethanol (18%), and methanol (13%), collectively accounting for 74% of total emissions. A large fraction of the emissions are acids (8.8%) which were not measured in the Washington compost analysis. The major acids in the EPA profile are acetic acid (6%) and butanoic acid (1.4%). Unidentified compounds made up 5.6% of the emissions and are included in the “others” category. There are significant differences between the EPA Speciate profile composition and those determined from the Washington compost facilities, notably the near absence of i-propanol in the Washington dataset while it was the dominant emission in California.

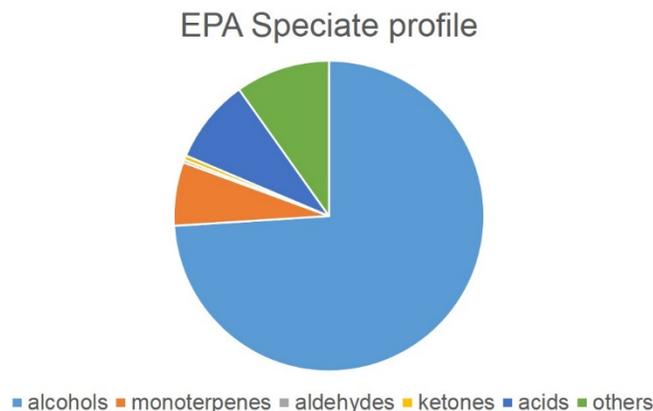


Figure 6: Green waste compost emissions profile from EPA speciate data base, profile number 8933 (EPA, 2016a).

Duplicate samples

The duplicate samples were examined to determine how well sampling was repeated with this methodology. The duplicate samples were collected from the flux isolation chamber at the same time as the other sample. This involved filling two canisters for VOC analysis by method TO-15, collection of air into two Teflon bags for analysis of mercaptans, and collection of two DNPH cartridges for analysis of aldehydes by method TO-11. In Table A.7 through Table A.9 in Appendix 1, the analyses from the duplicate samples are shown and a ratio, given by sample #1 divided by sample #2, of the results is shown as a metric of sample reproducibility. Shown are three duplicate samples collected from different facilities. These sample pairs were selected because a larger number of compounds were reported for these samples than others. The general conclusion is that sample replication is within a factor of two although, differences greater than a factor of 2 were often observed, even for compounds with high emission rates such as monoterpenes. Ratios varied within a duplicate sample set, suggesting potential analysis problems. For example, for the facility #4 duplicate in Table A.9, the acetonitrile ratio is 1.38 while α -pinene, a major emission, is 3.54; both these compounds would have come from the same TO-15 analysis of the canister sample. In many cases compounds were reported in one sample but not the other, most often occurring for compounds at low concentration and flux densities less than $5 \mu\text{g m}^{-2} \text{min}^{-1}$.

Composition differences from pile aging

The sampling at facility #5 provides an opportunity to examine how compost emissions can change with windrow age. At facility #5, flux samples were collected from a fresh pile (~1 day old), from a pile that was ~7 days old, and from a finished pile. The flux density data are tabulated in Table A.10 and total VOC as toluene flux densities shown in Figure 7. It is evident that emissions from the fresh pile are much higher than the day 7 pile or the finished compost pile. The total VOC as toluene value is 1000 times larger for the fresh pile than for the day 7 pile. Fresh pile emissions were dominated by monoterpenes, with a substantial flux of DMS, an odor compound. Monoterpene emissions from the 7-day old pile were a factor of 2500 lower than the

fresh pile, while DMS emissions and other sulfur compounds were a factor of 50 lower. Emissions from the finished pile were negligible in comparison to the fresh pile and the day-7 pile. Emissions from the compost material at facility #5 change by orders of magnitude over the pile life cycle. These data imply it will be important to determine flux densities for the first week of emissions in order to derive a compost emission factor. This is consistent with studies in California of green waste compost emissions that show 70% to 80% of the emission occur in the first two weeks (CARB, 2007).

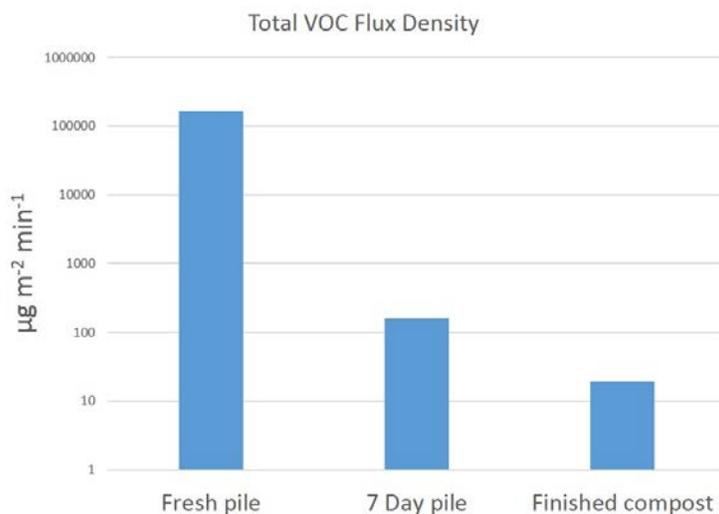


Figure 7: Comparison of total VOC flux density from windows at different ages at facility #5.

Total VOC and compost emission factors

Compost emission factor development in California has been accomplished using field measurements of total VOCs by EPA Method 25. This would be a more rigorous approach than reporting total VOC as toluene from method TO-15, a non-standard reporting metric. EPA Method 25 is the prescribed method for reporting total gaseous non-methane organic compound concentration. In EPA Method 25 (and California’s related SCAQMD Method 25.3) air samples collected from canisters are analyzed using a gas chromatograph system that first separates VOCs from carbon dioxide (CO₂), carbon monoxide (CO), and methane. VOCs are then oxidized to CO₂, and the CO₂ thus formed is reduced to methane. Methane is then quantified using a flame ionization detector. Thus, all the carbon from the volatile organic compound mixture is converted to methane and the total VOC is reported as a methane equivalent concentration. The method is an efficient carbon counter. To convert the measured methane into a mass of VOC requires knowledge of the VOC composition of the sample to account for the mass of oxygen and other atoms not detected by a flame ionization detector (FID); that is the average molecular weight per carbon of the sample must be known. SCAQMD Method 25.3 specifies how to calculate this from a known sample composition. The default composition is hexane if nothing is known about the sample composition. For example, if hexane (C₆H₁₄) is the

only VOC present in the sample, then Method 25 converts the 6 carbon atoms in the hexane molecule to 6 methane molecules as follows:



The reported concentration of methane measured by Method 25 can be converted to a concentration of hexane by the following equation to account for difference in molecular weights and the methane molar yield from hexane:

$$\text{VOC as hexane } (\mu\text{g m}^{-3}) = \text{methane } (\mu\text{g m}^{-3}) \times (1/6) \times (86.17/16.04) \quad (\text{eq. 3})$$

For compost, where most of the emission might be dominated by light alcohols such as methanol and ethanol, the mass emission rate would be significantly underestimated if hexane is used as the default. If compost emissions are all methanol, Method 25 converts methanol from the canister sample into methane and reports a methane concentration:



The methanol concentration is determined from the measured methane by:

$$\text{methanol } (\mu\text{g m}^{-3}) = \text{methane } (\mu\text{g m}^{-3}) \times (1/1) \times (32.04/16.04) \quad (\text{eq. 5})$$

If the data were reported as total VOC as hexane, the emission rate would be under reported by about half, given by the ratio of VOC as hexane to the actual methanol concentration:

$$\frac{\text{hexane } (\mu\text{g m}^{-3})}{\text{methanol } (\mu\text{g m}^{-3})} = \left[\frac{1/6}{1} \right] \times \left[\frac{86.17}{32.04} \right] = 0.448 \quad (\text{eq. 6})$$

For monoterpenes ($\text{C}_{10}\text{H}_{16}$), using hexane as the default VOC, Method 25 would slightly overestimate emissions:

$$\frac{\text{hexane } (\mu\text{g m}^{-3})}{\text{monoterpene } (\mu\text{g m}^{-3})} = \left[\frac{1/6}{1/10} \right] \times \left[\frac{86.17}{136.23} \right] = 1.05 \quad (\text{eq. 7})$$

For accurate estimates of mass emission rates using total VOC measurements by Method 25, the relative amounts of alcohols, ketones, monoterpenes, and sulfur compounds like DMS, must be known over the compost windrow lifecycle.

To determine a compost emission factor for total VOCs, as has been done in California, several steps are required (CARB, 2007). The first step is to determine the total VOC flux density using the flux isolation chamber. This must be done several times over the course of the windrow lifecycle to determine how emission change with time, for example, sampling on days 1, 3, 6, 18, 14, 30. The second step is to convert the measured flux densities into a daily pile emission rate by multiplying the measured flux density with the surface area of the pile.

$$\mu\text{g day}^{-1} = (\mu\text{g m}^{-2} \text{min}^{-1}) \times \text{pile surface area} \times 1,440 \text{ min day}^{-1} \quad (\text{eq. 8})$$

It is reported that most of the emission occur from the windrow ridge (the top of the pile) and these flux densities can be a factor of 16 larger than emissions from the windrow sides (CARB, 2007). One issue is that a pile may have hot spot emission locations where flux densities can be much larger (factors of 30) than normal ridge area emissions. Some accounting must be performed to estimate total area of so called “venting emissions” compared to normal emissions. The venting emission area can be on order of 10% of the total ridge area and have on average a factor of four higher flux densities (CARB, 2007).

From the daily emission rate an emissions profile over the life cycle of the windrow can be determined. An example of such a profile is given in Figure 8 taken from the Modesto compost emissions study (CARB, 2007). The area under the emission profile is the total VOC mass (lbs VOC) emitted over the windrow lifecycle. The total mass of VOC emitted is then divided by the initial pile mass to yield a VOC emission factor of lbs VOC per wet ton compost. The Modesto study report did not specify how the total VOC measured as methane by SCAQMD Method 25.3 was converted into a total VOC mass. It may be that default hexane values were used by the contracting analysis lab. If the emissions were dominated by light alcohols, like the EPA Speciate profile for green waste compost emissions, then the mass emission rates shown in Figure 8 are underestimated.

The data collected by Ecology’s Solid Waste Management Program from Washington compost facilities did not conduct flux density measurements over a windrow lifecycle from which an emissions factor can be calculated (lbs VOC / wet ton). There is still a need to conduct this kind of study in Washington.

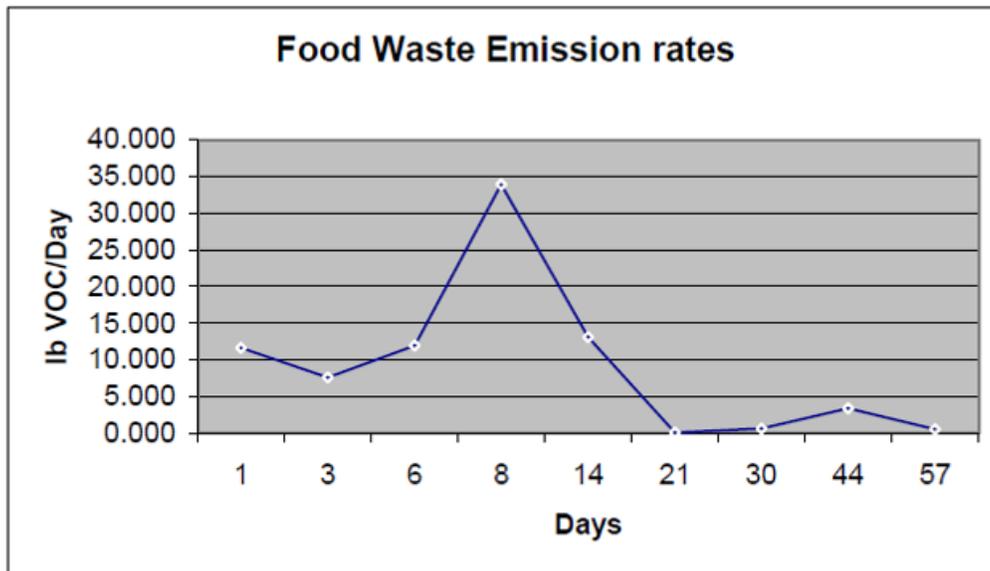


Figure 8: Emissions profile of total daily VOC emitted from a green waste/food waste windrow at the Modesto, CA compost facility (CARB, 2007).

Environmental impacts of compost emissions

The compounds that comprise the majority of the measured emissions in the Washington samples were light alcohols (ethanol, methanol, i-propanol), ketones (acetone, 2-butanone), monoterpenes (α -pinene and limonene), and to a lesser extent dimethyl sulfide. Their impact on photochemical ozone formation and PM_{2.5} would depend on the compound's emission rate and rate of oxidation in the air. From this perspective, monoterpenes will have the greatest impact on ozone and PM_{2.5} by virtue of their large emission rates and high reactivity with hydroxyl radical (HO), ozone, and nitrate radical (NO₃) compared to the light alcohols and ketones. Hydrocarbons that react rapidly with the HO radical can be precursors to photochemical ozone formation if sufficient nitrogen oxides (NO_x) are present. These reactions with HO are multi-step radical chain reactions and occur during the day when ultra-violet radiation from sunlight is sufficient to form HO radicals from various photolysis reactions. The rate of HO radical reactions with hydrocarbons depends on the chemical structure of the hydrocarbon. Alkene type compounds, such as monoterpenes, typically react rapidly with HO and thus can be efficient ozone precursors in urban areas. In contrast alcohols and ketones are less reactive. Acetone has such a low reactivity with the HO radical that the EPA does not classify acetone as a volatile organic compound. The HO radical reaction with hydrocarbons leads to organic photoproducts that may in turn react with the HO radical or ozone. The photoproduct typically has a lower vapor pressure than the precursor compound and if sufficiently low can partition to airborne particulate matter, enhancing PM_{2.5} mass concentrations. This process is called secondary organic aerosol formation and is thought to be an important fate for photoproducts from larger organic compounds (Odum et al., 1996). Ozone reactions with monoterpenes are relatively fast and are known to contribute to secondary organic aerosol formation (Yu et al., 1999; Lee et al., 2006; Xu et al., 2014). In contrast light alcohols and ketones are not thought to be significant sources of secondary organic aerosols. Table 2 lists reaction rate coefficients for these compounds with the principal oxidants, HO, ozone, and NO₃ (Atkinson, 2000; Atkinson et al., 2007). Alcohols and ketones are not reactive with ozone. The NO₃ radical can be an important oxidant at night if NO_x concentrations are low enough to allow for its formation. NO₃ is known to react rapidly with monoterpenes. Table 1 shows that monoterpenes are the most chemically reactive major constituent of compost emissions and, thus, would have the most impact on airshed ozone and PM_{2.5} formation. For example, α -pinene is 16.5 times more reactive, and limonene 53 times more reactive than ethanol with the HO radical.

A method to quantify relative potential impacts on ozone formation of hydrocarbons in urban air mixtures is to use the Maximum Incremental Reactivity (MIR) metric developed by Carter (Carter et al., 1994) and used by the California Air Resources Board to classify organic compounds for their potential to form ozone (Carter, 2010). The MIR provides a scale that takes into account detailed atmospheric chemistry reactions and is determined from photochemical modeling. The scale quantifies the extra amount of ozone produced per unit of extra mass emission of the compound into an urban air mixture. The MIR values, in units of grams ozone per grams VOC, are shown in Table 2. The monoterpenes are efficient at making ozone

compared to the light alcohols and ketones. Thus, monoterpenes can contribute to both ozone formation and secondary organic aerosol formation.

Emissions of compounds that were observed in the compost samples and are classified as air toxics by the EPA (42 U.S.C. Chap 85, § 7412) include formaldehyde, acetaldehyde, acetonitrile, carbon disulfide, carbonyl sulfide, cumene (isopropylbenzene), hexane, methanol, naphthalene, propanal, and vinyl acetate. Flux densities for most of these compounds were typically not above reporting limits but were occasionally observed. When reported, flux densities were typically $< 100 \mu\text{g m}^{-2} \text{min}^{-1}$. The largest flux densities were observed for methanol at $6,575 \mu\text{g m}^{-2} \text{min}^{-1}$ (facility #1, unchopped tipping sample). Acetaldehyde was always identified in at least some of the samples from each of the facilities and had the second largest mass emission rates. The acetaldehyde flux density ranged from $3,561 \mu\text{g m}^{-2} \text{min}^{-1}$ from facility #4 (tipping biofilter sample, representing 8% of total speciated emissions) to $250 \mu\text{g m}^{-2} \text{min}^{-1}$ as the highest value observed in the facility #6 samples (fresh built pile, pre-turn sample). Carbonyl sulfide (OCS) was also identified in a least one of the samples from the facilities, with flux densities ranging from a high value of $293 \mu\text{g m}^{-2} \text{min}^{-1}$ from facility #4 (tipping biofilter) to $12 \mu\text{g m}^{-2} \text{min}^{-1}$ as the highest value observed in the facility #3 samples (pile #3 sample). The percent emissions of most air toxic compounds as a fraction of the total flux density were typically low, with the exceptions of acetaldehyde and methanol.

Table 2: Rate coefficients for the dominant volatiles emitted from composting and their maximum incremental reactivity values (MIR) from Atkinson, 2000; Atkinson et al., 2007; and Carter 2010.

Compound	Bimolecular rate coefficients ($\text{cm}^3 \text{molecule}^{-1} \text{sec}^{-1}$)			MIR
	K_{HO}	K_{O_3}	K_{NO_3}	
methanol	9.3×10^{-13}	NA	1.3×10^{-16}	0.65
ethanol	3.2×10^{-12}	NA	$< 2 \times 10^{-15}$	1.45
i-propanol	5.1×10^{-12}	NA	1.4×10^{-15}	0.59
acetone	1.7×10^{-13}	NA	$< 3 \times 10^{-19}$	0.35
2-butanone	1.2×10^{-12}	NA		1.43
α -pinene	5.3×10^{-11}	8.5×10^{-17}	6.2×10^{-12}	4.38
limonene	1.7×10^{-10}	6.4×10^{-16}	1.3×10^{-11}	4.40
DMS	6.5×10^{-12}	NA	1.1×10^{-12}	NA

NA = not available (ozone reactions with some compounds are negligible)

Criteria for Title V air operating permits and impact on Washington composting facilities

Assessment of VOC emission potential from compost facilities

A simple assessment was made of potential VOCs emission from Washington compost facilities to assess if Title V permitting might be required for a facility. There are 63 compost facilities in Washington State. Figure 9 shows a map of their locations and Table 3 lists the 25 largest facilities by reported 2017 annual throughputs to Ecology. The two largest compost facilities in the state are operated by Cedar Grove, at their Maple Valley and Everett locations, with annual throughputs in 2017 of 232,000 tons and 140,000 tons respectively. The amount of material processed annually varies widely from the largest facility in Maple Valley to the smallest compost facility in Centralia (Centralia Composting) where only 8 tons of material was reported.

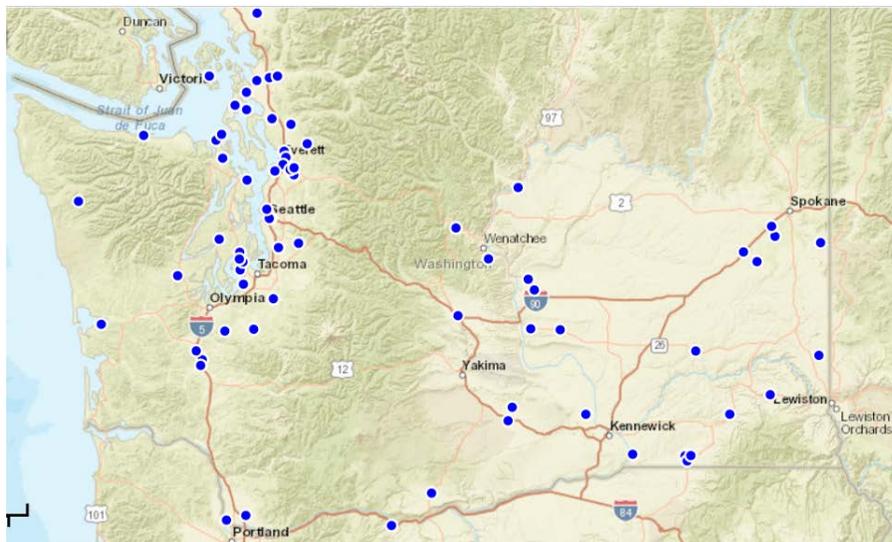


Figure 9: Locations of composting facilities in Washington State (Ecology, 2019b)

Most of the facilities, including those shown in Figure 9 accept a wide variety of material inputs that can be broadly categorized as green waste: yard debris, yard debris mixed with food scraps, wood debris, land clearing debris, and food waste. Some accept manure and agricultural organics. The Boise White Paper facility is perhaps somewhat different in that its principle inputs are wood debris, industrial organics, and food waste. The Ovenell Farms, Sunnyside Dairy, and Thomas Farms facilities are principally agricultural waste and manure composters.

Table 3: Washington State’s largest composting facilities and 2017 annual throughputs (Ecology, 2019b)

Rank	Facility	Location	Annual Throughput (tons)
1	Cedar Grove Composting Co. Maple Valley	Maple Valley	231,639
2	Cedar Grove Composting, Inc.	Everett	139,825
3	Barr-Tech Composting Facility	Sprague	85,703
4	LRI Compost Factory	Puyallup	73,940
5	Lenz Enterprises Inc	Stanwood	73,359
6	Silver Springs Organics Composting LLC	Rainier	64,038
7	Pacific Topsoils - Maltby	Woodinville	54,051
8	Boise White Paper LLC	Walla Walla	50,404
9	Natural Selection Farms Composting Facility	Sunnyside	49,686
10	Pierce County (Purdy) Composting Facility	Gig Harbor	42,845
11	Dirt Hugger LLC	Dallesport	36,548
12	La Conner WWTP Skagit Co SD #1	La Conner	32,475
13	Sunnyside Dairy	Sunnyside	30,000
14	Green Earth Technology (Compost)	Lynden	26,665
15	Royal Organic Products	Royal City	23,256
16	Ovenell Farms Composting Facility	Quincy	21,292
17	Thomas Farm Agricultural Composting	Snohomish	20,000
18	Bailand Farms Yardwaste (Bailey) Compost	Snohomish	17,000
19	Stemilt World Famous Compost Facility	Wenatchee	16,582
20	North Mason Fiber Co.	Belfair	15,799
21	Olympic Organics LLC	Kingston	15,009
22	Cheney WWTP & Compost Facility	Cheney	12,460
23	WSU Compost Facility	Pullman	11,392
24	Skagit Soils Inc	Mount Vernon	10,240
25	Lawrence Farms LLC Compost Facility	Royal City	6,810

An estimate of the annual total VOC emissions for green waste composting facilities can be calculated from the reported annual material throughputs following the emissions inventory procedure developed by the California Air Resources Board in 2015 (CARB, 2015). California has done the most work on estimating compost emission from green waste composting and has documented emission factors from a number of studies in California that can be usefully employed to estimate compost facility emissions. In the California emissions inventory procedure, the annual facility VOC emissions can be calculated from

$$\text{Annual Emissions (lbs)} = [CEF \times (1 - CE) \times TP] + [SEF \times SD \times TP] \quad (\text{eq. 9})$$

where *CEF* is the compost emissions factor (lbs/wet-ton), *CE* is the control technology efficiency (such as a biofilter or windrow cover), *TP* is the annual material throughput (wet tons/yr), *SEF* is the stockpile emission factor (lbs/wet ton day), and *SD* is the average number of days material is in stockpiles (days). The recommend VOC and NH₃ emissions factors used in California for green waste/food waste composting are given in Table 4 (CARB, 2015). These values are applicable if the amount of food waste, biosolids, and manure are less than 15% by weight. These recommended values are averages from nine compost emission studies conducted in California. It is noted in the methodology document that VOC emission factors ranged from a low of 0.85 lbs VOC/wet ton to a high of 10.03 lbs VOC/wet ton. It is not known if the emission factors determined from California facilities are applicable to Washington green waste compost facilities. The factor of 10 range of total VOC emission factors implies that these emission factors could be expected to vary depending on facility material inputs and facility operation.

Table 4: California Air Resources Board recommended emission factors for green waste composting (CARB, 2015).

Pollutant	Compost Emission Factor (lbs/wet ton)	Stockpile Emission Factor (lbs/wet ton day)
VOC	3.58	0.20
NH₃	0.78	NA

Various emission control technologies have been noted to be effective and include using finished compost as a topping layer on windrows to act as an absorbent filter, using membrane covers (such as the Gore cover), or venting emission for aerated static piles through a biofilter. In the simplest emission control approach, covering the windrow with a cap of finished compost is estimated to reduce VOC emission by 75% (CARB, 2007).

If it can be assumed that stockpile holding time is one day, a good estimate for large facilities handling food waste (John Cleary, pers. com.), the annual potential VOC emission rates can be readily calculated from equation (9). The potential emissions are used for air permitting calculations and thus do not include the effectiveness of control technologies. For example, the estimated annual VOC emission rate for the Cedar Grove Maple Valley facility would be:

$$\frac{lbs}{yr} = \left[3.58 \frac{lbs}{wet\ ton} \times (1 - 0) \times 231,639\ tons \right] \\ + \left[0.2 \frac{lbs}{wet\ ton\ day} \times 1\ day \times 231,639\ tons \right]$$

$$VOC \frac{lbs}{yr} = [8.293 \times 10^5\ lbs] + [4.632 \times 10^4\ lbs] = 8.756 \times 10^5\ lbs = 438\ tons$$

Annual VOC emission rates were calculated in a similar manner for the other facilities, assuming green waste composting emission factors, and are listed in Table 5. Shown are potential emissions (no emission control technology) and emissions with control technology, assuming a 75% efficiency. Seven facilities in Washington have potential annual VOC emissions that exceed the 100 tpy threshold for air permitting.

Table 5: Estimated annual total VOC emission rates assuming California Air Resource Board green waste emission factors and 2017 annual throughputs. Uncontrolled emissions assumes no emissions control technology, while controlled emissions assume that emissions are reduced by 75%, compared to uncontrolled emissions.

Rank	Facility	Location	Estimated VOC emissions (tons / yr)	
			uncontrolled	controlled
1	Cedar Grove Composting, Maple Valley	Maple Valley	437.8	126.8
2	Cedar Grove Composting, Inc.	Everett	264.3	76.6
3	Barr-Tech Composting Facility	Sprague	162.0	46.9
4	LRI Compost Factory	Puyallup	139.7	40.5
5	Lenz Enterprises Inc	Stanwood	138.6	40.2
6	Silver Springs Organics Composting LLC	Rainier	121.0	35.1
7	Pacific Topsoils - Maltby	Woodinville	102.2	29.6
8	Boise White Paper LLC	Wallula	95.3	27.6
9	Natural Selection Farms Composting	Sunnyside	93.9	27.2
10	Pierce County (Purdy) Composting	Gig Harbor	81.0	23.5
11	Dirt Hugger LLC	Dallesport	69.1	20.0
12	La Conner WWTP Skagit Co SD#1	La Conner	61.4	17.8
13	Sunnyside Dairy	Sunnyside	56.7	16.4
14	Green Earth Technology (Compost)	Lynden	50.4	14.6
15	Royal Organic Products	Royal City	44.0	12.7
16	WSU Compost Facility	Pullman	43.1	12.5
17	Ovenell Farms Composting Facility	Quincy	40.2	11.7
18	Thomas Farm Agricultural Composting	Snohomish	37.8	11.0
19	Bailand Farms Yardwaste (Bailey)	Snohomish	32.1	9.3
20	Stemilt World Famous Compost Facility	Wenatchee	31.3	9.1
21	North Mason Fiber Co.	Belfair	29.9	8.7
22	Olympic Organics LLC	Kingston	28.4	8.2
23	Cheney WWTP & Compost Facility	Cheney	23.6	6.8
24	Skagit Soils Inc	Mount Vernon	19.4	5.6
25	Lawrence Farms LLC Compost Facility	Royal City	12.9	3.7

Impact of Title V permitting

The Department of Ecology posts air operating permit annual fee schedules and air operating permit program reports that document costs of holding a permit. These fees can be used as an example to evaluate cost. Actual costs may vary between the different air agency jurisdictions. There are three costs: an emissions fee, a complexity fee, and a fixed cost fee. These costs may be annually adjusted to reflect costs associated with running the air operating permit program. From the 2018 air permit program report (the latest available) the emission fee was \$31.32 per ton, the flat fee was \$19,510, and the complexity fee varied from \$282 to \$177,726 (Ecology, 2018). The emission fee is charged to actual emissions. The 2019 fee schedule on the Ecology Air Quality Program website shows 31 facilities with air operating permits in their jurisdiction. The flat fee for 2019 was \$21,100.23 and the average complexity fee was also \$21,100. The complexity fee takes into account the work performed by Ecology in writing the permit and any inspections or compliance work performed.

The cost of having an air operating permit for the largest compost facility can be estimated from the documented fee schedule. Assuming a biofilter of some control technology was used on 100 ton/yr potential emission rate to yield a 30 tons/year actual VOC emission rate, the 2018 emission fee of \$31.32/ton yields an emissions cost of \$940. Together with the 2018 flat fee of \$19,510 and assumed complexity cost of \$21,100, the total annual cost for the permit would be \$41,550. The emissions fee is a small fraction of the permit cost.

Comparison of greenhouse gas emission factors from landfilling and composting

Landfilling and composting are two approaches for organic solid waste disposal. The objective of this section was to find information on greenhouse gas emissions from composting and landfilling used in greenhouse gas inventories and determine if composting would reduce greenhouse gas emissions compared to organic solid waste disposal in landfills. Landfills are a known source of methane. In landfilling organic waste is initially decomposed by aerobic bacteria, and when oxygen levels are depleted, anaerobic bacteria then decompose the waste, generating emissions of methane in the process over the course of many years. Some fraction of the landfilled organic waste is converted into CO₂ which is not treated as a greenhouse gas emission and some of the decomposed organic material becomes stable organic matter in the landfill. The generation of methane from landfills is treated as a greenhouse gas (GHG) emitting process. Methane emission models have been developed to calculate the emission rate (mass methane per year) from landfills. The EPA model is the Landfill Gas Emission Model (LandGem). The International Panel on Climate Change (IPCC) also has a landfill gas emission model (IPCC Waste Model) and has similar model inputs and parameters. The most recent national assessment of greenhouse gas emissions in the USA (EPA, 2016b), estimates that municipal solid landfills were the third largest methane source in the US, accounting for 20.2%

of total methane emissions in 2014, behind natural gas transmission systems and enteric fermentation.

In composting organic waste is aerobically decomposed to a stabilized solid product, known as compost. The composting process occurs over a shorter timescale than landfill decomposition. Some fraction of the organic material is decomposed to CO₂, estimated to be 0.44 kg CO₂/kg of dry material (RTI, 2010). The generation of nitrous oxide (N₂O) and methane (CH₄) can occur and emission factors have been reported on a wet material mass basis. A standard set of emission factors have been adopted by the IPCC from a review of the available literature and these values are used by the EPA (EPA, 2016b). For methane the emission factor is 0.004 kg CH₄/wet kg material and for nitrous oxide it is 0.0003 kg N₂O/wet kg material (IPCC, 2006). Emission of these compounds depend on composting conditions, with reported methane emissions ranging from < 1% to a few percent of the initial carbon content of the compost, and reported N₂O emissions have ranging from 0.5% to 5% of initial nitrogen in the material (IPCC, 2006). These IPCC emission factors were used by the EPA to determine emission rates from composting in the latest US national greenhouse gas emission inventory (EPA, 2016b). The EPA estimates that in 2014 composting was the 11th largest source of methane emissions in the US and the 8th largest source of nitrous oxide.

The emissions of CH₄ and N₂O can be converted into an equivalent CO₂ emission rate using the global warming potentials (GWP) of these gases. The GWP are recommended by the IPCC in their periodic assessment reports. GWP are dimensionless values referenced to the radiative forcing impact of CO₂. These values account for differences in atmospheric lifetime between CO₂ and the other greenhouse gas and differences in their infra-red absorption spectrum. For GHG reporting required by the EPA and internationally by the United Nations Framework Convention of Climate Change (UNFCCC), the GWP₁₀₀ values from the IPCC 4th Assessment Report (AR4) are used. These values are the 100-year time horizon values and are 25 for CH₄ and 298 for N₂O. The carbon dioxide equivalent (CO_{2e}) emission rates are thus determined by multiplying the emission rate by the corresponding GWP value. For example, for methane the CO_{2e} emission rate would be:

$$CO_{2e} = CH_4 \text{ emission rate} \times 25 \quad (\text{eq. 10})$$

The IPCC methane and nitrous emission factors can be converted into CO_{2e} emission factors using the GWP₁₀₀ values. For methane this is 0.10 kg CO_{2e} /wet kg material and for nitrous oxide it is 0.089 kg CO_{2e} /wet kg material, yielding a total emission factor of 0.189 kg CO_{2e} / kg wet material.

The EPA reports these emission factors in the form of metric tons CO_{2e} per short ton (MT CO_{2e} / ton), where a metric ton is 1000 kg. For methane, the IPCC emission factor in these units is 0.091 MT CO_{2e} / ton for CH₄, and for N₂O it is 0.081 MT CO_{2e} /ton. These values can be compared to a recent emissions factor review done by CARB of three California green waste composting studies and three other related green waste studies (CARB, 2017). They reported average emissions factors of 0.049 MT CO_{2e} / ton for CH₄ based on six studies and 0.021 MT

CO_{2e} / ton for N₂O based on four studies. These values are significantly lower than those used by the IPCC. Methane emission factors in the CARB review varied by a factor of 45, across the six studies while N₂O emission varied by a factor of 6. The large range suggests that a few studies with high emissions rates can significantly skew the data. More measurements of emission rates are required to better understand the emissions rate distribution from commercial facilities.

Comparison using LandGEM

A simple comparison of CO_{2e} emissions was performed using emissions of CH₄ from landfilling and emissions of CH₄ and N₂O from composting. For landfill methane emissions, the EPA model LandGEM ver 3.02 was used (EPA, 2019b). For purpose of comparison it was assumed that the landfill and the compost facility have constant annual throughputs of 100,000 tons/year. Conventional default values of LandGEM model variables were used: a methane generation rate k of 0.05 yr⁻¹, and a CH₄ potential generation capacity L_o of 100 m³/10⁶ g waste. The model was run with an assumed 50-year landfill operating lifetime. Model output of methane emission per year over the 50 year lifetime of the landfill and beyond are illustrated in Figure 10. It was assumed that methane would be captured onsite and burned, with a capture efficiency of 75% (EPA, 2008).

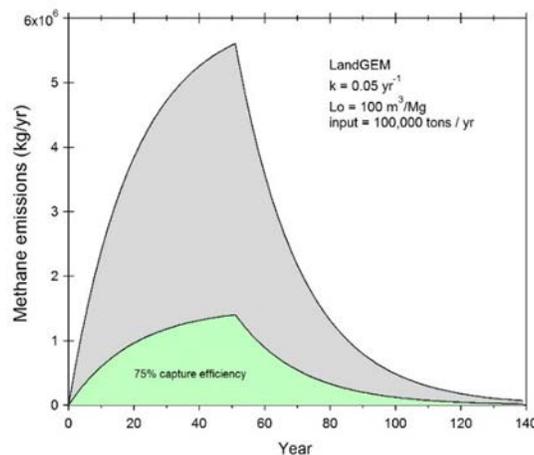


Figure 10: Methane emissions from a municipal landfill with a 50-year operating life as modelled by LandGem. Lower green curve shows emissions assuming 75% of emitted methane is captured on site.

The amount of methane generated over the 50-year lifetime of the landfill can be compared to the emissions of methane and N₂O from composting the same mass of material. While the solid material input types of landfills and compost facilities differ, the degradable organic carbon fraction of landfill waste was assumed to be 20% by wet weight to yield the L_o value used in the LandGEM model. This degradable organic carbon value is similar to that for materials sent for composting: food waste (15%), garden waste (20%), wood waste (40%), food processing industry waste (22%) (RTI, 2010). Thus, the amount of degradable carbon is assumed to be similar between the landfill waste types and compost waste type. The total amount of methane emitted over the 50-year lifetime of the landfill, assuming 75% capture efficiency, was 1.2 x 10⁹ kg CO_{2e}.

Assuming that this mass of degradable organic material (5×10^6 tons) was composted instead of landfilled, the total emission of CH_4 and N_2O can be calculated from the IPCC emission factors noted above. In the compost scenario, CH_4 emission was calculated to be 453.5×10^6 kg CO_2e . The N_2O the emission was 405.4×10^6 kg CO_2e , giving a combined GHG emission of 0.86×10^9 kg CO_2e . This value is 25% less than the landfilling value. This simple calculation suggests that composting organic waste emits less GHG emissions than landfilling with landfill gas capture systems. However, it should be noted that the values for landfilling and composting are fairly similar, especially given the fact that emissions of methane and N_2O from composting are not well documented, and emission factors are estimated to have an uncertainty of $\pm 50\%$ (EPA, 2016b).

Comparison using WARM model

A further comparison was made using the EPA Waste Reduction Model (EPA 2019c) version 14. The WARM model applies a life-cycle analysis to determine overall greenhouse gas reduction benefits of waste management practices such as landfilling, incinerating, recycling, and composting. The model calculates changes in greenhouse gas emissions between a baseline waste management practice and an alternative practice. It includes all greenhouse gas emissions associated with the management practice such as fossil fuel emissions associated with waste transport and operation of on-site machinery. For composting it's assumed the compost is applied to agricultural fields and thus includes a benefit for increasing soil carbon storage.

For food waste composting the WARM model uses a combined methane and nitrous oxide emissions factor of 0.055 kg CO_2e / kg material, and for mixed organics (yard trimmings, grass, leaves) an emission factor of 0.077 kg CO_2e / kg is used. These values are significantly smaller than the 0.189 kg CO_2e / kg emission factors used by the IPCC and the EPA in greenhouse gas inventories. For diesel fuel consumption for transportation of waste, and diesel fuel consumption from equipment operation at the compost facility, an emission factor of 0.022 kg CO_2e / kg material is used. The soil carbon storage term used in WARM was determined by the EPA using a soil carbon storage model and accounts for the amount of carbon that is still in the soil at the end of ten years. The carbon storage benefit term for compost is 0.265 kg CO_2e / kg. Summing the emission factors and subtracting the soil storage term yields a total GHG emission factor from food waste composting of -0.188 kg CO_2e / kg material, or -0.171 CO_2e / ton. The EPA WARM documentation list this value as -0.18 CO_2e / ton. In the WARM model, composting removes greenhouse gases from the atmosphere by virtue of the soil storage term.

For landfilling, the WARM model assumes the same transportation emissions as in composting, and includes terms for methane generation and a benefit term for carbon storage. Methane emission factors are applied to different material types and the model assumes full yield of methane from these materials; that is, it does care that these methane emissions occur over many years. For example, for food waste, the methane yield is 42% of the carbon content of the waste. That is 42% of the carbon atoms in the food waste are converted into CH_4 . For branches the yield is 12%, for leaves 8%, and for grass it is 23%. Much of the branch material is stored as stable carbon material (i.e. lignin) in the landfill. The model allows the user to select if the generated

methane is captured on-site for power production and applies an additional benefit term for power production to account for avoided fossil fuel CO₂ emissions. For national average practices at landfills, the net emissions factor used in WARM for landfilling food waste is 0.54 kg CO₂e / ton, and for mixed organic waste it is 0.20 kg CO₂e / ton. Thus, landfilling is treated as a net source of greenhouse gases to the atmosphere. As an illustrative example of this accounting, the difference (Δ CO₂e) between landfilling and composting emissions for 1 ton of food waste is the difference in the net emissions factors:

$$\Delta \text{CO}_2\text{e} = 0.54 \text{ kg CO}_2\text{e} / \text{ton} - (-0.17 \text{ CO}_2\text{e} / \text{ton}) = 0.71 \text{ CO}_2\text{e} / \text{ton} \quad (\text{eq. 11})$$

Thus, if food waste is diverted from composting to landfilling there is a net gain of 0.71 CO₂e / ton of greenhouse gas emissions to the atmosphere.

As an exercise in greenhouse gas emission accounting for solid waste management, the WARM model was applied to Washington State landfill data to estimate the greenhouse gas emission reduction if the organic waste sent to landfills had instead been composted. Landfill data associated with Washington State landfills was downloaded from Landfill Methane Outreach Program (LMOP) database (EPA, 2019d). Landfills equipped with landfill gas (LFG) collection systems were analyzed separately from landfills without LFG recovery systems to account for power production benefits. In all landfills with LFG collection system in Washington except one, collected gases are used for energy production instead of being flared. The amount of material currently in the landfills (waste in place) in Washington are presented in Table 6 and Table 7. The total waste in place for landfills without gas collection was 8,541,689 tons, while for landfills with an LFG collection system the material mass is much larger at 62,729,995 tons. This mass of landfill material was classified into its constituent components to determine the mass of organics that could have been diverted to composting. The fraction of the landfill waste mass that is compostable organics was based on municipal solid waste stream analysis by the EPA (EPA, 2019e). From this analysis the food waste mass is 22%, yard trimmings mass is 8 %, and woody branch mass is 8%. These material mass fractions were used to calculate mass inputs for the WARM model. These masses are shown in Table 8.

Table 6: List of landfills with no LFG collection system in the state of Washington

Landfill Name	Waste in place (tons)
Rainbow Valley LF	300,000
Ryegrass LF	464,000
Sudbury Road LF	1,102,317
Terrace Heights LF	6,198,335
Vashon LF	477,037

Table 7: List of landfills with LFG collection system in the state of Washington

Landfill Name	Waste in place (tons)
Port Angeles LF	1,026,000
Purdy LF	473,243
Roosevelt Regional Municipal Solid Waste Landfill	53,506,278
Tacoma City Solid Waste Facility	5,588,721
Thurston County Waste and Recovery Center	2,135,753

Table 8: Organic waste weight by material type used in WARM model for two landfill types: landfilling with and without LFG collection system.

Material	weight in tons in landfills <i>without</i> LFG collection system	weight in tons in landfills <i>with</i> LFG collection system
Yard Trimmings	666,252	4,892,940
Branches	683,335	5,018,400
Food Waste	1,879,172	13,800,599

Output from the WARM model is shown in Table 9. The table shows the difference (eq. 11) in greenhouse gas emissions (net MT CO₂e) if the organic waste amounts listed in Table 8 had been composted instead of landfilled. Table 9 shows net emissions for each organic material type. Composting food waste instead of landfilling shows negative emissions which means that amount of greenhouse gas emissions has been avoided by composting. Composting both yard trimmings and branches yields a net emission of greenhouse gases compared to landfilling. According to the WARM model, composting food waste significantly reduces greenhouse gas emissions, while there is no benefit from composting yard trimmings and branches. Since food waste is the largest component of landfilling material (EPA, 2016b) and composting significantly decreases GHG emissions for this type of waste material, the WARM model indicates that composting organic waste is better than landfilling in reducing greenhouse gas emissions.

Table 9: Net CO₂ equivalent emissions in mega tonnes (MT CO₂e) from diversion of organic waste from landfills to composting.

Material	Net MT CO ₂ e	
	landfills <i>without</i> LFG collection system	landfills <i>with</i> LFG collection system
Yard Trimmings	20,020	147,028
Branches	148,745	1,169,390
Food Waste	-1,338,469	-9,829,690

Conclusions

Flux density emission data collected from six commercial compost facilities in Washington State by Ecology's Solid Waste Management Program from 2010 to 2013 was examined. The data show that volatile organic compound emissions from young windrows were dominated by two compound classes: light alcohols (methanol and ethanol) and monoterpenes (α -pinene and limonene). These compounds were not always reported by the different contract labs performing the VOC analysis by EPA method TO-15, making comparisons between facilities difficult. Methanol in particular was not typically reported as a target compound for analysis and information about this compound is lacking. Sulfur compound emissions (principally dimethyl sulfide) and emissions of ketones (acetone and 2-butanone) and some aldehydes (acetaldehyde) were also sometimes significant and important contributors to total VOC emissions. Largest flux densities were often observed from the youngest piles, as expected, and flux densities from some sampled older piles were very low. Unfortunately, there was no data collected from a single pile over its life cycle from which a VOC emissions factor could be derived. This should be done for one or more Washington compost facilities to determine a relevant VOC emissions factor for permitting purposes. The mixture of VOC emitted was significantly different compared to the emissions profile listed in the US EPA Speciate database. The US EPA Speciate profile for green waste compost (#8933; EPA, 2016a) uses data reported from a California study where emissions were dominated by i-propanol (42%), a compound rarely detected in the Washington data, and where monoterpenes comprised a small fraction of the total emissions (6%). The EPA Speciate profile is different enough from the Washington compost data that it does not appear useful for describing Washington compost emissions.

Future work on VOC compost emissions factors in Washington requires a VOC analysis method that can quantify both light alcohols (methanol, ethanol, i-propanol) and the various monoterpene compounds to get an accurate assessment of total VOC emissions. For the highest emitting samples collected at the Sudbury and Silver Springs facilities, just five compounds (ethanol, α -pinene, limonene, acetone, and 2-butanone) made up more than 90% of the emission rate. Applying a total VOC analysis method, such as EPA Method 25 that detects VOCs as methane, requires an accurate assessment of the emissions composition to convert measured methane back to a VOC mass. If emission rates are dominated by light alcohols, Method 25 would significantly underestimate VOC emission rates. The large difference in flux densities observed for piles of different ages at the Lenz facility also suggests the need for continuous VOC monitoring or more frequent grab sampling of VOC emissions during the first week to properly capture rapid changes in flux density with pile age.

In addition, the release of odorous compounds, such as dimethyl sulfide and ammonia, the principle environmental chemistry impact of compost VOC emissions would appear to be a result of monoterpene emissions, which are much more chemically reactive in the atmosphere than light alcohols or ketones. Atmospheric oxidation of monoterpenes occurs rapidly and could

contribute to photochemical ozone production in areas with elevated NO_x concentrations. Oxidation of monoterpenes can also contribute to PM_{2.5} formation.

An estimate of annual VOC emission rates for Washington composting facilities was made based on annual reported waste material through puts and VOC emission factors derived from California studies of green waste composting. For California composts an average emission factor of 3.58 lbs VOC / wet ton has been determined. Using these data, it was calculated that seven composting facilities in Washington have the potential to emit more than 100 tons of VOC per year and thus potentially be subject to EPA Title V permitting.

Compost emission factors for the greenhouse gases methane and nitrous oxide were also reviewed. These emission factors get used in official greenhouse gas inventories compiled by the EPA. A simple comparison was made comparing greenhouse gases emissions from composting or landfilling organic waste using the EPA's Waste Reduction Model (WARM). It was concluded that composting as an alternate waste management strategy will likely decrease greenhouse gas emissions from organic waste compared to landfilling, with most of the reduction attributed to two factors: removing food waste emissions of methane from landfills and the soil carbon storage benefit of applying compost to soils. The analysis contains uncertainties related to N₂O and CH₄ emission factors related to compost, which vary widely in the literature, and to quantifying soil carbon storage benefits. Establishing emission factors for N₂O and CH₄ from Washington state compost facilities is also required to more accurately quantify greenhouse gas reduction benefits of composting compared to landfilling.

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Appendix

Appendix 1. Compost Facility Emission Samples Collected by Ecology

Table A.1: Compost emission samples collected at Facility #1

Date	Source	Description	% He*
6/15/2010	Leachate in Clarifier	Quiescent sediment trap	9.31
6/15/2010	Process Biofilter	Northern Cells	0.100
6/15/2010	Finish Compost	Undisturbed pile	7.45
6/15/2010	Finish Compost	Duplicate Sample	5.03
6/15/2010	Unchopped Tipping	Vent on pile, one day onsite	1.87
6/15/2010	Fresh Chop Compost	Chopped and stored < one day	1.70
6/15/2010	Fresh Built Pile	Approx one day old in pile	0.923
6/15/2010	13 Day Covered	Cover removed and mixed	3.29
6/15/2010	ECS Cover Vent	Perforated vent on cover	2.90
6/15/2010	Blank	Sample blanks	11.5

Table A.2: Compost emission samples collected at Facility #2

Date	Source	Description	% He*
6/13/2011	Primary Effluent Lagoon	Effluent tested in wash tub	10.5
6/13/2011	Gore on Primary Compost	Day 3 Food Waste Compost	1.84
6/13/2011	Secondary ASP	Negative Aeration	4.89
6/13/2011	Secondary Biofilter	Postive Flow Biofilter	0.301
6/13/2011	Primary ASP	Negative Aeration	1.03
6/13/2011	Primary ASP	Duplicate sample	1.34
6/13/2011	Finish Compost Pile	Aged One Approx One Month	1.23
6/13/2011	Fresh Chopped Compost	Mix Ready for Primary ASP	4.89
6/13/2011	Fresh Chopped Compost	Duplicate sample	4.89
6/13/2011	Primary Biofilter	Postive Flow Biofilter	0.211
6/13/2011	Blank	Sample blanks	11.3

Table A.3: Compost emission samples collected at Facility #3

Date	Source	Description	% He*
11/15/2012	Finished Pile		6.1
11/15/2012	3 Week Old Pile 3		1.9
11/15/2012	3 Week Old Pile 2		3.2
11/15/2012	Leachate Pond	Effluent tested in wash tub	8.1
11/15/2012	3 Week Old Pile 1		3.3
11/15/2012	3 Week Old Pile 1	Duplicate sample	3.3
11/15/2012	Feedstock		3.2
11/15/2012	Blank	Sample blanks	3.2

Table A.4: Compost emission samples collected at Facility #4

Date	Source	Description	% He*
4/16/2013	Tipping Biofilter		0.046
4/16/2013	Fresh Built	Pile built 5-9 am, Sample Location at 7 am.	0.97
4/16/2013	Covered Pile Air On	14 days covered	2.1
4/16/2013	Covered Pile Air On	Duplicate	2.6
4/16/2013	Blank	Blank samples	3.2
4/17/2013	Crushing Biofilter	Crushing stopped during stabilization. Diesel exhaust from crusher.	0.096
4/17/2013	Leachate Tank	No gas tracer. Direct sample used.	NA
4/17/2013	Covered Pile Air Off		3.0
4/17/2013	Finished Covered		1.0
4/17/2013	Phase 3		1.4

Table A.5: Compost emission samples collected at Facility #5

Date	Source	Description	% He*
6/26/2013	ASP Biofilter	Phase I biofilter	0.18
6/26/2013	Tipping & ASP Biofilter	Phase I Biofilter and ASP	0.47
6/26/2013	Tipping & ASP Biofilter	Phase I Biofilter and ASP Duplicate	0.51
6/26/2013	Fresh ASP	Fresh Chopped, ~1 day old, turned once.	0.31
6/26/2013	7 Day ASP	ASP Bay 5, ~7 day old, turned once.	2.3
6/26/2013	Finished	Finished, unscreened	4.1
6/27/2013	Mass Bed NE Corner	8-10 days old	3.1
6/27/2013	Mass Bed NW Corner	15 days old	2.5
6/27/2013	Mass Bed Middle W	15 days old	3.9
6/27/2013	Mass Bed Middle S	10 days old	3.8
6/27/2013	Mass Bed Middle E	ASP Bay 5 into Mass Bed, ~7 day old, turned twice.	0.94
6/27/2013	Blank	Sample blanks	7.5

Table A.6: Compost emission samples collected at Facility #6

Date	Source	Description	% He*
6/5/2013	Biofilter Comp Dog	Phase II biofilter	0.14
6/5/2013	Mass Bed Biofilter	East side of facility	1.7
6/5/2013	Liquid Runoff Sump	Clarifier water, technically stormwater	4.1
6/5/2013	Product Pile	Coarse product, sampled 3/4 to top of pile	4.7
6/5/2013	Fresh Built Pre-turn	Chopped, ~1 day old, turned once.	1.2
6/5/2013	Fresh Pile Pre-turn	Chopped, ~1 day old, preturned	1.2
6/6/2013	Pre-turn 1 Wk	A lot of grass, <1 day old	1.2
6/6/2013	Post-turn	1/4-1/3 into mass bed, dark black material, recognizable feedstock in material.	3.3
6/6/2013	Post-turn	Duplicate	5.4
6/6/2013	2/3 Mass Bed	2/3 of total distance along mass bed. ~10 days processing. Fans on. Really hot!	3.4
6/6/2013	Blank	Sample blanks	5.6
6/6/2013	15 day Mass Bed	End of mass bed. Dark material. Dusty fungus growth on outside.	4.8

Table A.7: Facility #6 duplicate sample flux density results ($\mu\text{g m}^{-2} \text{min}^{-1}$)

Compound	Sample #1	Sample #2	Ratio
2-Hexanone	0.80	ND	
Valeraldehyde	2.43	1.3	1.93
Butyraldehyde	2.91	1.8	1.64
Propene	3.0	5.2	0.58
Acetonitrile	5.46	4	1.47
Dimethyl Sulfide	6.1	72.7	0.08
Acetaldehyde	6.43	3.6	1.81
Formaldehyde	7.40	4.2	1.75
Isovaleraldehyde	13.4	8.9	1.50
Ethanol	17.0	9	1.91
Naphthalene	24.3	17	1.42
alpha-Pinene	25.5	96.4	0.26
Vinyl acetate	27.9	15	1.88
d-limonene	31.5	48.2	0.65
2-Butanone	59.5	44	1.34
Acetone	316	230	1.37
Carbonyl Sulfide	ND	14.8	
Carbon Disulfide	ND	2.4	
Total VOC as toluene	1128	1038	1.09

ND = not detected

Table A.8: Facility #3 duplicate sample flux density results ($\mu\text{g m}^{-2} \text{min}^{-1}$)

Compound	Sample #1	Sample #2	Ratio
Formaldehyde	0.5	0.7	0.70
Butyraldehyde	0.6	ND	
Valeraldehyde	0.9	0.7	1.39
Methyl Mercaptan	1.2	ND	
Isovaleraldehyde	1.3	1.0	1.34
Hexane	1.6	ND	
m,p-Xylenes	1.6	ND	
Naphthalene	2.2	1.3	1.64
Nonane	2.5	ND	
n-Heptane	2.8	ND	
Carbon Disulfide	3.1	ND	
Benzene	3.4	0.88	3.84
Acetaldehyde	4.0	2.7	1.50
Styrene	4.6	2.7	1.73
Octane	4.8	ND	
4-Methyl-2-pentanone	8.4	4.5	1.86
Toluene	11.5	6.8	1.70
Chloromethane	14.5	1.5	10.0
Propene	41.2	3.5	11.7
Acetonitrile	54.5	35.1	1.55
2-Butanone	75.1	41.2	1.82
Acetone	95.7	40.0	2.39
Dimethyl Sulfide	194	77.5	2.50
d-limonene	339	291	1.17
alpha-Pinene	1574	618	2.55
Tetrahydrofuran	ND	8.2	
Ethyl acetate	ND	3.5	
TVOC	5207	3391	1.54

ND = not detected

Table A.9: Facility #4 duplicate sample flux density results ($\mu\text{g m}^{-2} \text{min}^{-1}$)

Compound	Sample #1	Sample #2	Ratio
Valeraldehyde	1	1.08	0.99
Formaldehyde	1.24	0.94	1.32
2,5-Dimethylbenzaldehyde	1.9	1.69	1.13
Chloromethane	2	0.88	2.39
Butyraldehyde	2	0.45	4.70
Naphthalene	2.3	1.3	1.81
1,3-Butadiene	3	1.69	1.80
2-Hexanone	3.2	1.84	1.75
Isovaleraldehyde	4	1.08	3.89
Toluene	4.6	1.31	3.50
Carbonyl Sulfide	5	3.2	1.47
Methyl Mercaptan	6	5.4	1.03
4-Methyl-2-pentanone	6.5	2.61	2.48
Styrene	8.8	3.1	2.85
Ethyl acetate	9	3.38	2.70
Acetaldehyde	10	4.61	2.23
Acetonitrile	13	9.7	1.38
Dimethyl Sulfide	15.2	8.8	1.74
Propene	15	8.91	1.73
2-propanol	21	12.30	1.70
Ethanol	57	23	2.48
alpha-Pinene	190	53.8	3.54
d-limonene	247	73.8	3.35
2-Butanone	266	169.07	1.58
Acetone	1389	922	1.51
Hexane	ND	0.25	
1,2,4-Trimethylbenzene	ND	0.25	
Tetrahydrofuran	ND	0.41	
Benzene	ND	0.46	
n-Heptane	ND	0.57	
Total VOC as toluene	2284	1153	1.98

Table A.10: Comparison of flux densities (ug m-2 min-1) for piles of different ages at facility #5.

Compound	Fresh ASP	7 Day ASP	Finished compost
Acetaldehyde	27.0	ND	ND
Carbon Disulfide	36.0	ND	ND
Carbonyl Sulfide	129	ND	ND
Propene	476	0.6	1
Toluene	476	1.5	ND
Ethyl acetate	746	22.5	ND
Dimethyl Sulfide	2,057	5.9	ND
d-limonene	43,702	27.7	0.09
alpha-Pinene	57,841	13.0	0.17
Acetone	ND	2.3	1.6
Acetonitrile	ND	4.2	0.7
n-Heptane	ND	0.24	ND
TVOC as Toluene	167097	161	19.4

ND = not detected