# Advancing Organics Management in Washington State: The Waste to Fuels Technology Partnership 2019-2021 Biennium

**Solid Waste Management Program**

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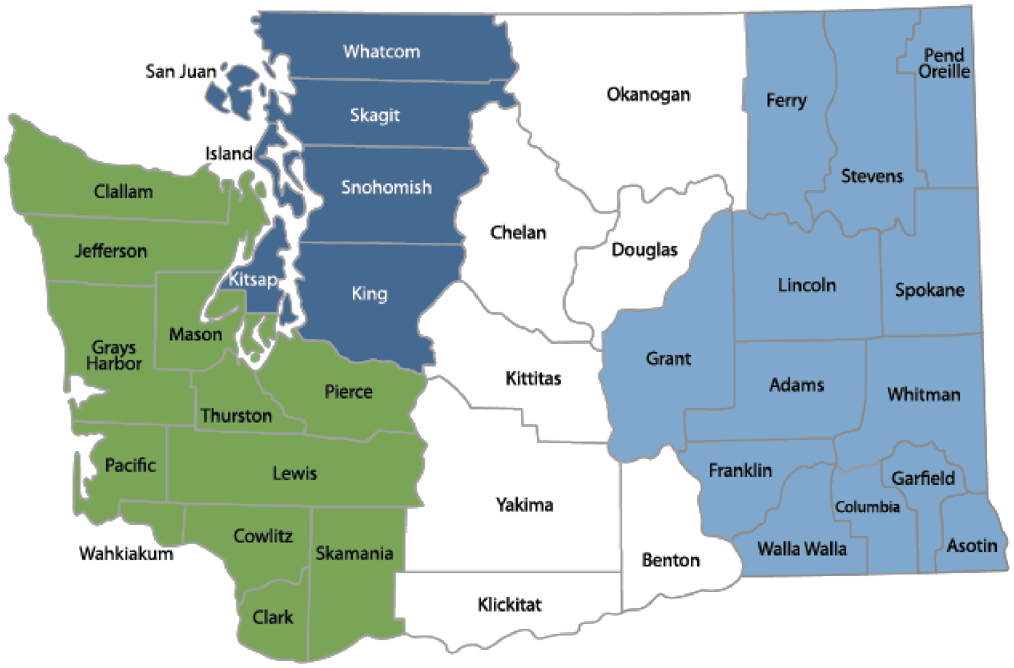

| Southwest Region 360-407-6300 | Northwest Region 425-649-7000 | Central Region 509-575-2490 | Eastern Region 509-329-3400 |
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| Region | Counties served | Mailing Address | Phone |
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# Advancing Organics Management in Washington State

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## The Waste to Fuels Technology Partnership 2019-2021 Biennium

*By*

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DEPARTMENT OF
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State of Washington

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# List of Abbreviations

| | |
|---|---|
| AD | anaerobic digestion |
| ASP | aerated static pile |
| BGRAM | Biochar Global Response Assessment Model |
| C | carbon |
| Ceq | carbon equivalent |
| $CH_4$ | methane |
| C:N | carbon to nitrogen ratio |
| $CO_2$ | carbon dioxide |
| $CO_2eq$ | carbon dioxide equivalent |
| CSANR | Center for Sustaining Agriculture and Natural Resources |
| CSI | Center for Sustainable Infrastructure |
| DF | Douglas fir forestry residuals |
| DMS | dimethyl sulfide |
| DMDS | dimethyl disulfide |
| EC | electrical conductivity |
| eGFP | enhanced green fluorescent protein |
| FTIR | Fourier Transform Infrared Spectroscopy |
| GC-MS | gas chromatography-mass spectrometer |
| ha | hectare |
| $H_2S$ | hydrogen sulfide |
| Mg | magnesium |
| MS | malate synthase |
| MSW | municipal solid waste |
| Mt | million metric tons; megatonne |
| N | nitrogen |
| N-doped | nitrogen-doped |
| N-Mg doped | nitrogen and magnesium doped |
| $NH_3$ | ammonia |
| $NH_4$-N | ammonium nitrogen |
| $NO_3$-N | nitrate nitrogen |

|  |  |
|---|---|
| npSOC | non-pyrogenic soil organic C |
| $N_2O$ | nitrous oxide |
| $PM_{2.5}$ | particulate matter less than 2.5 microns in diameter |
| ppbv | parts per billion by volume |
| ppm | parts per million |
| ppmv | parts per million by volume |
| PSD | prevention of significant deterioration |
| PTE | potential to emit |
| PTR-MS | proton-transfer reaction mass spectrometry |
| TCA | tricarboxylic acid |
| US EPA | United States Environmental Protection Agency |
| VFA | volatile fatty acid |
| VOC | volatile organic compound |
| XPS | X-ray photoelectron spectroscopy |
| VSC | volatile sulfur compound |
| WS | wheat straw |
| WSU | Washington State University |
| WTFT | Waste to Fuel Technology |

# Acknowledgements

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- Chapter 2: Rick Finch, WSU Composting Facility; Mark Fuchs, Ecology Solid Waste Division; Jeff Gage, Green Mountain Technology; and Tim O'Neill, Engineered Compost Systems offered feedback on survey questions. We would also like to thank the representatives of the compost facilities who shared information with us for this report.
- Chapter 4: Thida Tea and Brian Maupin of Washington State University for their help in conducting and collecting data for the potato field trial.
- Chapter 5: Brad Pugh, Thida Tea, and Brian Maupin of Washington State University for their help in conducting and collecting data for the potato and strawberry field trials; Dr. Ruifeng He and Garrett Gang for help with sample collection and tissue grinding.
- Chapter 7: The Pacific Northwest National Laboratory is operated for the U.S. Department of Energy by Battelle Memorial Institute under contract DE AC06 76RL01830.

The authors would like to offer sincere thanks to Mark Fuchs for being a visionary and steadfast partner since the inception of the Waste to Fuels Technology partnership.

# Executive Summary

Organic wastes – including materials such as lumber scraps, yard clippings, and food waste – represent an untapped resource hiding in plain sight. If recovered and used, these resources have exciting potential to help Washingtonians mitigate climate change, generate renewable energy and other sustainable products, and improve soil health, all while adding to our economy. Washingtonians generate an estimated 1,200 to 1,350 pounds of organic waste per person per year, nearly half of which is landfilled. Instead of disposal, organic wastes need to be diverted from conventional landfilling and solid waste practices in order to realize their benefits. In many cases, they also need to be processed so that the valuable energy, nutrients, carbon, and other materials can be recaptured in useable form.

Achieving these benefits will take progress on many fronts, including ongoing policy progress, citizen action, and technology development. Over the last fifteen years, a partnership between the Washington State Department of Ecology's Solid Waste Management Program and Washington State University (WSU) has supported research and extension on the technologies and economics that could guide the next generation of organic waste processing. Technologies including composting, anaerobic digestion, and pyrolysis are being further developed to maximize the environmental benefits provided, understand and address barriers to broader implementation, and explore opportunities to reduce technology costs and create higher value products to improve overall economics.

The WSU-Ecology partnership thus supports the Solid Waste Management Program's implementation of the Solid and Hazardous Waste Plan, which is currently being updated. Among the projects that the partnership tackled this biennium, several contribute to our understanding of new energy and material recovery technologies (Solid Waste Management Goal 15, 2016-2021 plan) and promote the development of new processing pathways that could add diversity to organics processing infrastructure (SWM Goal 22) and end use markets for recycled organic products (SWM Goal 24).[1] There are also projects that explore the use of soil amendments derived from recycled organics and the impact on soils, plant growth, and soil carbon sequestration (SWM Goals 18, 19, and 21).[2] Meanwhile, other projects explore air emissions and new potential strategies to manage odors at commercial composting facilities (SWM Goal 23).[3]

The WSU-Ecology partnership is working towards the development and appropriate implementation of municipal *biorefineries* – meaning facilities that sustainably convert biomass

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[1] SWM 15: State and local governments will have a better understanding of solid waste energy and material recovery technologies; SWM 22: More diversified organics processing infrastructure will exist in the state; SWM 24: Diversified end-use markets will be in place for recycled organic products.

[2] SWM 18: The use of soil amendments derived from recycled organics will increase, reducing the need for synthetic fertilizers, pesticides and herbicides; SWM 19: Agriculture, landscapes, and home gardens will need less water due to increased use of compost and other soil amendments derived from recycled organics; SWM 21: Soil organic sequestration using recycled organics will increase based on research recommendations.

[3] SWM 23: Composting facilities will produce clean end products (This goal includes activities to manage odors at compost facilities.)

to energy and other beneficial products. This idea is also closely related to the idea of industrial symbiosis, in which wastes or by-products from one industrial activity becomes the resource for another actor or process. In a biorefinery, the co-location and integration of various processes and technologies allows for the intake and conversion of organic wastes in order to generate higher value products, provide process improvements, or mitigate negative effects from emissions that cause odors, or climate or air quality impacts. For example, a composting facility could provide the core waste conversion technology (Figure 1), and this technology could be co-located with pyrolysis that treats excess woody materials and the compost "overs" (the portion of woody material that does not degrade). Pyrolysis generates biochar, a product that could be sold on its own or incorporated back into composting, with the potential to benefit the composting process and the resulting compost product.

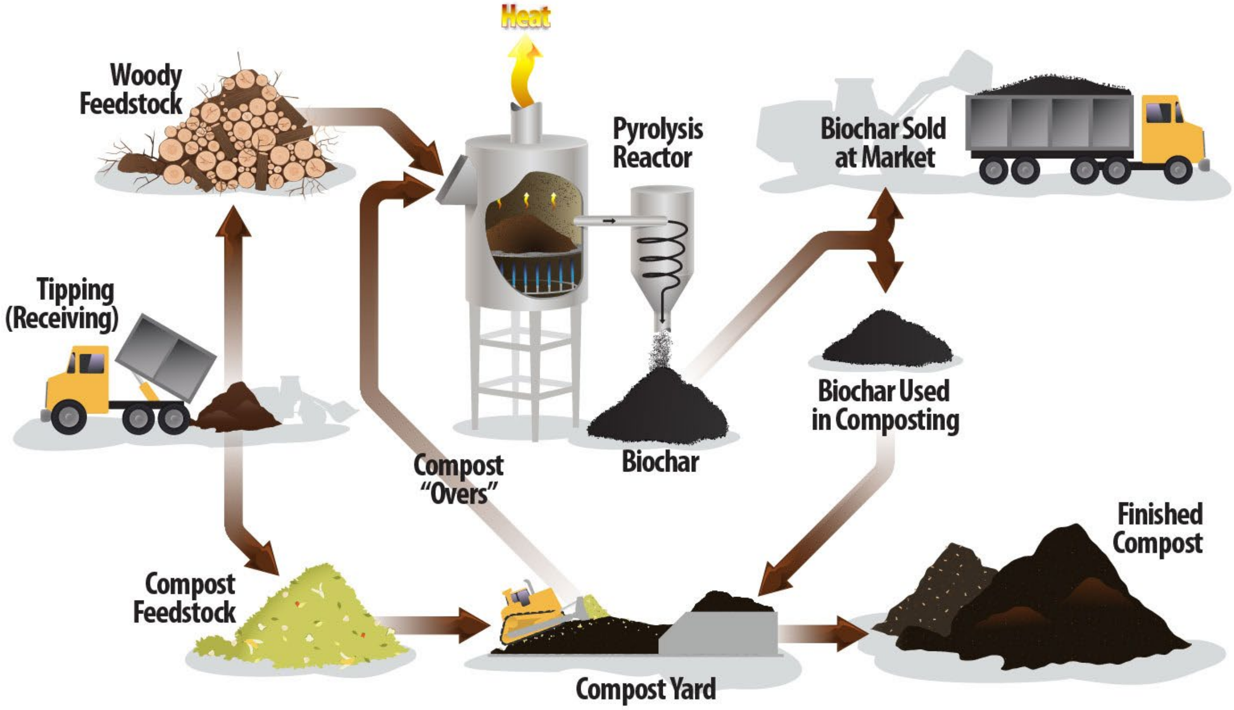

Figure ES.1: A schematic showing a possible integration of a pyrolysis reactor with a compost yard. (Figure credit: Karen Hills and Andrew Mack)

The biorefinery vision is modular rather than prescriptive, and the specific technologies that make sense will vary depending on the location and context. Furthermore, as biorefinery ideas continue to develop within and outside our region, they may include other technologies such as anaerobic digestion or other technologies not yet envisioned. Significant remaining barriers that have prevented wide adoption of a municipal biorefinery in the Pacific Northwest include the presence of inexpensive hydroelectric power in the Northwest (which impacts project economics), contamination of the organic waste stream, scale issues, and the need for additional technology development. The applied research and extension efforts carried out through the WSU-Ecology partnership aim to reduce these barriers and provide additional options for organics management throughout Washington.

A diverse set of projects was carried out via the partnership in the 2019-2021 biennium. *Chapter 1, Extension, Engagement, and Technology Transfer*, describes the team's extension efforts. Through this work, the team sought to educate stakeholders broadly about issues relating to sustainable organics management and next generation processing, and share the results of prior and ongoing partnership work with a diversity of stakeholders throughout the region, including those who work in the organics management industry, purchasers and users of organic residuals, others working on sustainable organics management at non-profits, county and local governments, and private companies, students, and Washington's residents. Opportunities for education and engagement were provided through live in-person and virtual presentations and discussions (both group and one-on-one), and through a variety of publications, recordings, and other online resources. In the 2019-2021 biennium, live interaction opportunities were provided to 1,448 individuals (in person and virtual). Resources developed as part of the partnership (in some cases with complementary funding) including publications, webpages, and recordings, were viewed at least 16,887 times. A special focus in this biennium was related to coordinating a group of roughly forty biochar producers, practitioners, scientists, and engineers to chart a roadmap for future development of the biochar industry in the Pacific Northwest and beyond. The group's recommendations are captured in a report *Biomass to Biochar: Maximizing the Carbon Benefit*.

While the impacts of this extension work will occur over the long term, there are several indications that these efforts are having the intended impact. First, work carried out under the partnership led to the team successfully competing for additional funds to continue work relevant to the priorities described in the Solid and Hazardous Waste Plan. A total of $2,689,778 was awarded to WSU partnership members during this biennium, and just over $3 million since tracking of this began in 2017. Meanwhile, while the team is not always able to capture the direct impacts of interactions with individuals across the state, existing survey data suggest that team members are having an influence. As one example, 84% of participants in the *Biomass to Biochar* workshop said that they anticipated that new knowledge or connections that they made as a result of the workshop would impact their work relating to biochar over the next year.

The work described in *Chapter 2, Comparing Methods to Measure Air Emissions from Commercial Compost Facilities,* was motivated by the need to better understand volatile organic compound (VOC) emission factors to inform questions relating to air emissions permitting of commercial composting facilities in Washington State. This has become a more pressing issue as policies encouraging diversion of urban food and green waste from landfills to composting facilities are pursued, with the simultaneous risk that these same facilities may be subject to more costly and complicated air permitting applications as they expand their facilities to meet demand.

Results of a survey of 10 compost facilities in Washington set the context for experimental work in the laboratory and field. Volatile organic compound emissions rates were successfully measured from the active phase of a negatively aerated 160-ton pile at the WSU Compost

Facility with feedstocks consisting of manure and livestock bedding (straw, etc.), which would differ from yard and food waste feedstocks. Emission rates through the negative aeration duct (that are subsequently treated through a biofilter) were 5.47 lbs per wet ton, while those from the surface averaged 0.084 lbs per wet ton, for a combined emission factor of 5.55 lbs VOCs per wet ton of compost. The small percentage (1.5%) of fugitive emissions from the surface suggests that negative aeration systems, such as the one at WSU, have a high potential to control VOC emissions if biofilters downstream of the duct can be operated efficiently. The very low emission rates at the end of active composting imply that fugitive VOC emissions from the stabilization and curing phases might also be very low (less than 1% of emissions during the active phase). SCAQMD Method 25.3 is a standard analysis method approved in California for emissions testing from combustion sources that has also been used to measure compost emissions. If emissions from this experiment are expressed in terms of mass carbon emitted, (the same units employed by SCAQMD Method 25.3), the results would be less than half the previously mentioned VOC mass emission rate (2.6 lbs carbon (C) per wet ton versus 5.4 lbs VOCs per wet ton). This is lower than the emissions factor of 5.71 lbs C per wet ton used by San Joaquin Valley Air Pollution Control District for windrow green waste composting, but larger than their value of 1.78 lbs C per wet ton for windrow manure composting.

Mass emission rates at the WSU Compost Facility through the duct were dominated by methanol (65%) followed by acetone (12%). Acetone is an US EPA exempt VOC and could therefore be discounted from the emission factor. If emission of methanol and other oxygenated compounds identified here are found to dominate emissions generally from Washington State compost facilities, then the Method 25.3 reporting of total VOC emissions as carbon mass would significantly underestimate VOC emission rates. Furthermore, the use of surface flux isolation chambers (domes over a small area of a compost pile) can cause problems for sampling water soluble organics since compost emissions contain a lot of moisture that will condense on sampling devices and water soluble organics will therefore be under-sampled. While surface flux isolation chamber sampling in combination with Method 25.3 has provided the initial baseline dataset for VOC emissions factors from windrows, this analysis methodology is less useful for sampling aerations duct exhaust and may be problematic for sampling from positively aerated piles. A partnership with stakeholders to develop more accurate, reliable, and cost-effective VOC sampling and analysis methods is recommended. To that end, the following recommendations were provided in this chapter with regards to sampling compost emissions: limit surface flux sampling, if possible; improve surface sampling representativeness; test semi-continuous in-situ sampling of negative aeration ducts; develop speciated VOC emission factors independent of facility testing; and use total temporary enclosures for testing emissions from biofilters and positively aerated piles.

*Chapter 3* investigated methods for creating higher value biochar for specialized uses, specifically, the *Development of Engineered Biochar Cocktails for Odor Removal in Composting Facilities*. Researchers developed twelve engineered biochar samples from locally available feedstocks, wheat straw and Douglas fir, pyrolyzed at either 400$^{0}$ C or 600$^{0}$ C in a furnace tube reactor. They then evaluated these biochars' capacity to adsorb particular gasses that can be emitted during the composting process: hydrogen sulfide (a flammable, toxic, odorous gas), ammonia (a gas that can react with other gases in the atmosphere to create air pollution) and carbon dioxide (if more carbon were captured, it could enhance the ability of the resulting compost to store carbon). Some biochars were also created under specialized conditions to enhance their adsorption properties, including nitrogen-doped biochar produced under the presence of ammonia, and magnesium- and nitrogen-doped biochar, obtained by impregnating the biomass with magnesium chloride hexahydrate ($MgCl_2•6H_2O$) prior to pyrolysis.

The resulting biochars were characterized, and adsorption tests measured their capacity to capture the target gases. Hydrogen sulfide and carbon dioxide adsorption experiments showed that the surface area, pH, and nitrogen content had a strong influence on the performance of biochar to remove these pollutants. For ammonia, the acidic functional groups on the adsorbent surface were the main factor determining adsorption and removal. Based on this work, additional studies are underway with methane and nitrous oxide, two other gases released during composting. From these preliminary results, plus additional results found in the literature, researchers explored whether a biochar blend could feasibly treat gaseous emissions of compost. They found that additional activation strategies need to be explored for the feedstocks available in the Pacific Northwest as the biochar blends that were studied did not have sufficiently high adsorption capacity to be economically used in composting facilities.

*Chapter 4, Impact of High Rates of Biochar on the Composting Process and Resulting Products,* explores biochar as a potential compost feedstock, a topic that has received much attention because of biochar's unique physical and chemical properties such as high carbon and porosity. To better understand biochar's impact on the composting process, researchers carried out a replicated composting trial. Two rates of biochar incorporation were evaluated, 20% and 40% by volume, in a chicken manure and wood chip compost, and were compared with an unamended control compost. To investigate the agronomic benefits of this strategy, the resulting compost products were utilized as a soil amendment in potato production. The retention of nitrogen did not result in significant differences in potato soil and plant biomass, except in comparisons with the unamended control (soil with no compost or compost and biochar), meaning that yield differences could only be attributed to the addition of compost. Biochar incorporation at 20% and 40% by volume only minimally impacted compost nutrient status, moisture content and temperature profiles The linear increase in biochar rate did not result in consistent and significant differences in the response variables measured. Potential reductions in nitrogen loss were observed in the 40% biochar amended composts, but this likely reflects the additional biochar nitrogen, not significant reductions in nitrogen loss. Those looking to incorporate biochar as a compost feedstock need to carefully consider the biochar feedstock and production type, and match these properties with appropriate expectations. As yield was not impacted, other benefits from the use of biochar such as carbon sequestration should be explored.

*Chapter 5, Integrating Compost and Biochar for Improved Air Quality, Crop Yield, and Soil Health*, explores the integration of composting with biochar. Biochar, compost, compost plus biochar (simply adding biochar to compost at the end of the composting process) and co-compost – the product of composting traditional feedstocks with biochar – have been identified as potential soil amendments that, after surface application and incorporation, can increase crop yield and improve soil health. Yet despite this promise, results need to be tested in regionally relevant crops before biochar use will be adopted. In previous work (during the 2017-

2019 biennium), the WSU-Ecology partnership explored the impacts on VOC emissions when biochar was added as a compost feedstock. Field and greenhouse trials with the resulting materials were initiated and this biennium's work continued and completed these investigations.

The greenhouse and field trials tested the effect of compost, biochar, co-compost, and compost plus biochar as soil amendments in a variety of different cropping systems and sites in Washington: sweet basil (field, Colbert, Washington), basil (greenhouse), strawberry (greenhouse), strawberry (field, Puyallup, Washington), and potato (field, Mount Vernon, Washington). Soil and yield data for strawberry, basil and potato field trials now include data and analysis from an additional growing season. This biennium's work expanded and replicated these trials and validated previous results, including finding that basil treated with different biochar-amended composts showed moderate increases in biomass production. Continued greenhouse-based experiments with strawberries indicated productivity increases were observed in some of the biochar-compost treatments but were only moderate overall.

Expanded results from additional growing seasons and analysis indicated in general, significant effects on crop yield, that varied by amendment type, crop, and soil type. In greenhouse experiments, a strawberry cultivar and two basil cultivars showed an increase in berry mass production or in biomass, respectively. In the field, a much larger increase (almost two-fold) was observed for basil produced under organic growth conditions. In field experiments, potatoes grown with co-compost also showed a significant (but smaller) increase in yield, but strawberries did not exhibit statistically significant differences. Amendments to the soil did not significantly affect the phytochemical composition of field- or greenhouse-grown sweet basil, an indication that product quality (flavor, smell) was not compromised by addition of biochar. Co-compost, compost, and the compost plus biochar amendments were typically observed to affect soil physical and chemical properties beneficially in Puyallup and Mount Vernon field trials, but it seems that this effect is somewhat dependent upon the native soil and the amendment's application rate and nutrient content. Depending on the plant species and growth conditions, treatments yielded minimal to dramatic increases in biomass.

In the work described in *Chapter 6,* researchers explored *Production of a Biochemical from Food Waste Through Integration of Anaerobic Digestion and Fermentation Processes.* Recovering and reusing organic wastes as a feedstock for anaerobic digestion (AD) can conserve resources, reduce greenhouse gas emissions, diminish odors, and stabilize waste. Currently, anaerobic digestion represents a well-developed technology that is viable for commercial-scale waste utilization, but economics can be challenging. While anaerobic digestion alone can produce methane, a renewable energy source, using the digester as a platform to create higher value products, such as bioplastic precursors, can make the economics for more favorable – especially in our region where renewable energy is plentiful and inexpensive. Likewise, the low or even negative cost of organic waste streams could overcome one of the primary hurdles to the production of bio-based products: high production costs. To this end, an innovative technical route was developed to make glycolic acid from organic wastes by integrating anaerobic digestion and aerobic cultivation. Glycolic acid is a valuable chemical that has broad industrial applications and already has a sizeable market. The first stages of the anaerobic digestion process were used to convert waste into simple carbon molecules called volatile fatty acids. The volatile fatty acids generated from anaerobic digestion were in turn were used to efficiently biosynthesize glycolic acid using a genetically engineered non-conventional yeast, *Yarrowia lipolytica*. This innovative biorefinery benefits both from the capacity of the anaerobic microbial consortia to handle complex waste, and from ability of the engineered cell factory to biosynthesize the target molecule. This technical pathway has potential to be further leveraged to generate other biochemical from organic wastes with high yield and at low cost.

*Chapter 7, Technical Potential for Carbon Dioxide ($CO_2$) Drawdown Using Biochar in Washington State,* explores the contribution that biochar could make towards helping Washington meet greenhouse gas reduction goals. It is the final report in a series that describes development and application of a high-resolution scalable method to estimate the technical potential for atmospheric carbon drawdown by biochar in Washington State. The method integrates spatially explicit information on soil productivity and crops at a 1-hectare resolution, assesses changes in soil organic carbon levels resulting from feedstock harvesting and biochar application, estimates increases in productivity stemming from biochar application to cropland, models the growth in biochar production capacity over time using an S-shaped technology-adoption curve, and tracks biochar production and soil storage capacities over time.

For each Washington county, 10 biomass-feedstock and biochar-process scenarios were developed, including one each for agricultural crop residues and for waste wood harvested from municipal solid waste (MSW) and processed at a central facility, and eight full scenarios for the combination of crop residues and MSW waste wood with forestry residues from four levels of timber harvest and two processing locations (central facility and in the field using a mobile unit). Individual results for each county were generated.

The combined results for Washington State from the eight full scenarios show that, over 100 years, 140-380 million metric tons (megatonnes; Mt) of biochar carbon could be produced. This effort would provide a total immediate climate offset of 170-430 Mt of carbon equivalents (Ceq), which equals 640-1600 Mt of carbon-dioxide-equivalents ($CO_2$eq) – or the equivalent of about 1.5 – 3.7 billion gallons of oil. We term this offset the "immediate" offset because it does not account for long-term responses of the earth's climate system. The ultimate "equilibrium" offset achieved after several centuries is smaller than the immediate offset by a factor of 2.17 to account for slow release of $CO_2$ from other labile reservoirs in the earth's climate system, primarily the oceans. Thus, after accounting for long-term climate-system responses, this effort would achieve an ultimate drawdown of 38-93 parts per billion by volume of atmospheric $CO_2$eq. At the maximum biomass-utilization rate, which is achieved after five decades, biochar production could offset between 8% and 19% of the greenhouse gas emissions in Washington State (taken at 2018 levels). If the same sustainably procured biomass were instead combusted for renewable energy, these offset and drawdown values decrease by about 60%, primarily due to the low carbon intensity of the primary energy supply in Washington State, but also due to the inability of bioenergy to provide the unique benefits associated with soil incorporation of biochar.

In conclusion, the efforts undertaken by the WSU-Ecology partnership are varied, but all are focused on contributing to the achievement of Washington State's long-term waste management goals. This partnership is a piece of a larger picture, but one that has seen increasing diversion of organics (including food wastes) from landfills, ongoing growth in application of anaerobic digesters throughout the state (including the application of small-scale digesters), and the emergence of a small biochar industry, with more than 20 commercial biochar producers currently in Oregon and Washington. Meanwhile, other Washington entities are using insects to process organic wastes, and utilizing wheat straw to produce tree-free paper. There are also ongoing explorations of integrating pyrolysis at commercial composting sites. These varied and innovative efforts are contributing to keeping Washington at the forefront of organics management. Meanwhile, amid signs of progress, new challenges emerge. For example, some industrial facilities have switched from wood-based fuels to natural gas due to low natural gas rates, and are now landfilling wood chips. Throughout, WSU researchers have been available as a resource when needed, and the partnership has continued to explore new production methods and uses for organics that could drive marketability, and to address emerging issues in organics management. Ongoing exchanges of ideas between researchers and practitioners will continue to support a range of efforts across the region, and support progress from research concepts towards more widespread adoption.

# Chapter 1: Extension, Engagement, and Technology Transfer

*Embrey Bronstad, Georgine Yorgey, Karen Hills, and James Amonette*

## 1.1 Extension, engagement, and technology transfer with regional organics management stakeholders and the organics value chain

The goal of the Waste to Fuels Technology partnership is to promote the efficient recovery of energy, nutrients, and other saleable products from organics that are normally considered wastes. Even when renewable energy is not generated, the utilization of recovered products can reduce overall energy inputs, therefore reducing the need for carbon-intensive energy sources. The application of these next-generation technologies and processes depends on adoption by industry and municipal partners, and upon the purchase and use of the generated products (compost, biochar, etc.). Outreach efforts, therefore, targeted stakeholders related to both organics management and product use.

### Building relationships across the organics management and value chain

The team made a number of presentations during the biennium related to the Waste to Fuels Technology (WTFT) partnership – though several opportunities to present were cancelled due to COVID-19-related disruptions, after spring 2020, the remainder of presentations were given utilizing remote, digital technologies. Presentations included discussions of the biorefinery concept, composting, anaerobic digestion, pyrolysis, biosolids, and nutrient recovery. These targeted presentations and interactions offered more than 1,448 opportunities for live interactions with stakeholders, including those who work in the organics management industry, purchasers and users of organic residuals, others working on sustainable organics management at non-profits, county and local governments, students, and other Washington residents. WTFT presenters are in **bold** here and in other listings throughout the chapter:

- **Hills, K. and M. Brady.** Compost Demand: The Role of Agriculture & Other Users. Washington Association of County Solid Waste Managers meeting. Leavenworth, WA (in-person and virtual). June 15, 2021. Attendees: 29
- **Bronstad, E.** It's Garbage CAN, not Garbage Cannot: Energy, Fertilizer, and Other Amazing Things from "Waste." Presentation to Pullman High School, May 10th, 2021. Attendees: 28
- **Stacey, N. and D.P. Collins**. The Elasticity of Biochar Across the Farm: Nutrient Capture, Compost Feedstock, and Soil Amendment. Western Nutrient Management Virtual Conference. March 3, 2021. Estimated attendees: 30

- **Yorgey, G.G.** An Overview of Greenhouse Gas Emissions and Climate Mitigation Opportunities in Washington State, with Examples from Large-Scale Agriculture. Washington Climate Assembly, https://www.waclimateassembly.org/. January 16, 2021. Estimated attendees: 75
- **Collins, D.P., N. Stacey**, T. Tea, **A. Bary**, and **L. Myhre**. 2020. Biochar feedstock influences compost pile temperature and available nitrogen. US Composting Council, Charleston, SC. Attendees: 70
- **Stacey, N.E**. and C. Villa. Biochar: Production and Elasticity of Use on Farms. Tilth Alliance Virtual Conference. November 9-10, 2020. Estimated attendees: 50
- **Bronstad, E.** Detecting Volatile Organic Compound Emissions from Compost Facilities. Pacific and Mountain West Nutrient Cycling, Soil Health and Food Safety Virtual Conference. October 29, 2020. Estimated attendees: 20
- **Brady, M.** A Regional Economic Model of Municipal Compost Supply and Agricultural Demand with an Application to Western Washington. Pacific and Mountain West Nutrient Cycling, Soil Health and Food Safety Virtual Conference. October 29, 2020. Estimated attendees: 50
- **Stacey, N.** Biochar and Co-composted Biochar in Strawberries and Potatoes. Pacific and Mountain West Nutrient Cycling, Soil Health and Food Safety Virtual Conference. October 29, 2020. Estimated attendees: 42
- **Garcia-Perez, M., A. Tanzil**, M. Wolcott, X. Zhang, and J. Hollday. Production of Cheap Sustainable Aviation Fuels (SAFs): A Carbon Balance Problem. Washington Academies of Science Symposium, September 17, 2020 (Estimated attendees: 150), Graduate seminar Biological Systems Engineering, September 24, 2020 (Estimated attendees: 70), National Biochar Week, December 7, 2020 (Estimated attendees: 50)
- **Fuchs, M.** Changing the Biomass to Biochar Paradigm. Biomass to Biochar: Maximizing the Carbon Value Virtual Workshop. April 27, 2020. Organized by **Amonette, J.E., K. Hills, M. Fuchs, G.G. Yorgey**, and J. Dooley. Estimated attendees: 35
- Han, H.S., J. Dooley, **B. Pecha,** K. Wilson, T. Miles, and D. Drinkard. Fundamentals of Biomass Handling and Biochar Production. Biomass to Biochar: Maximizing the Carbon Value Virtual Workshop. April 27, 2020. Organized by **Amonette, J.E., K. Hills, M. Fuchs, G.G. Yorgey**, and J. Dooley. Estimated attendees: 35
- Vallet, J. Climate Change and Biochar Use in Broad-Area Agriculture: The USDA-ARS Perspective. Biomass to Biochar: Maximizing the Carbon Value Virtual Workshop. April 28, 2020. Organized by **Amonette, J.E., K. Hills, M. Fuchs, G.G. Yorgey**, and J. Dooley. Estimated attendees: 35

- Baltar, R. California State Biochar Activity Update: Strategies for Scaling Production and Use. Biomass to Biochar: Maximizing the Carbon Value Virtual Workshop. April 28, 2020. Organized by **Amonette, J.E., K. Hills, M. Fuchs, G.G. Yorgey**, and J. Dooley. Estimated attendees: 35
- **Amonette, J.E.**, D. Page-Dumrose, D. Laird, **M. Garcia-Perez**, and K. Trippe. Biomass Availability and Biochar Properties. Biomass to Biochar: Maximizing the Carbon Value Virtual Workshop. April 28, 2020. Organized by **Amonette, J.E., K. Hills, M. Fuchs, G.G. Yorgey**, and J. Dooley. Estimated attendees: 35
- Rodriguez-Franco, C. Climate Change and Carbon Drawdown in Forestry. Biomass to Biochar: Maximizing the Carbon Value Virtual Workshop. April 29, 2020. Organized by **Amonette, J.E., K. Hills, M. Fuchs, G.G. Yorgey**, and J. Dooley. Estimated attendees: 39
- Graw, R. B. Springsteen, J. Inihara, C. Christoforou, and G. Glass. Environmental Permitting Issues for Biochar. Biomass to Biochar: Maximizing the Carbon Value Virtual Workshop. April 29, 2020. Organized by **Amonette, J.E., K. Hills, M. Fuchs, G.G. Yorgey**, and J. Dooley. Estimated attendees: 39
- Laird, D. Modeling Biochar Systems: Soil-Crop Responses, Technoeconomics, and Economic Viability. Biomass to Biochar: Maximizing the Carbon Value Virtual Workshop. April 30, 2020. Organized by **Amonette, J.E., K. Hills, M. Fuchs, G.G. Yorgey**, and J. Dooley. Estimated attendees: 35
- **Collins, D.P., A. Bary**, A. Nichols. On-Farm Composting Workshop, WSU Puyallup Research and Extension Center, Co-sponsored by Pierce Conservation District. 2 days Feb-March 2020. Coordinator, presenter. Attendees: 29
- **Collins, D.P., N. Stacey**, T. Tea, **S. Seefeldt, W. Hoashi-Erhardt, D. Gang, T. Jobson, M. Garcia Perez, J. Cleary, and M. Fuchs**. 2019. Biochar and Co-composted Biochar in Strawberries and Potatoes. Great Lakes Expo, Grand Rapids, MI. Invited Speaker. Attendees: 30
- **Brady, M., K. Hills,** and **G. Yorgey**. Compost Research Findings of the 2017-2019 Waste to Fuels Technology Partnership: Lessons for Compost Policy from Recycling Policy, and Differentiating the Value and Cost of Compost Across Likely Farm Use Scenarios in Western Washington. Washington Organic Recycling Council Board Meeting, August 15, 2019. Estimated attendees: 12
- **Yorgey, G.** Biosolids Quality: Emerging Contaminants of Potential Concern. Island County Biosolids Community Forum. Coupeville, WA. October 3, 2019. Estimated attendees: 40
- **Hills, K.** Value and Cost of Compost Across Likely Farm Use Scenarios in Western Washington. Washington Organic Recycling Council Conference. Vancouver, WA. November 8, 2019. Estimated attendees: 50

- Additional talks that were given to share the results of the roadmap developed as part of the *Biomass to Biochar: Maximizing the Carbon value* process are described and listed later in this chapter.

## Providing technical support

The team also answered numerous individual inquiries relating to anaerobic digestion, biochar, nutrient recovery, and carbon sequestration from recycled organic matter. Technical support included both formal participation in advisory panels and stakeholder groups, as well as many ad hoc responses to individuals who reached out to Washington State University (WSU) and the Center for Sustaining Agriculture and Natural Resources (CSANR):

### Advisory panel and stakeholder groups

In the first half of 2021, Yorgey and Bronstad were invited by the Center for Sustainable Infrastructure (CSI) to participate on a technical team advisory panel that sought to evaluate wastewater treatment options for a cluster of industrial food processors in the City of Pasco. They spent approximately four months interacting with food processor staff, engineers, and CSI to research and contribute to sustainable waste solutions using the knowledge garnered from WTFT research and investigations. With complementary funding from CSI, Bronstad and Yorgey participated in CSI's value planning process with the City and diverse stakeholders of the City's Process Water Reuse Facility (PWRF). Bronstad also led the development of a final report, *Feeding the Future: City of Pasco, WA* in collaboration with the technical team and CSI. The report summarizes recommended immediate technical and non-technical solutions. It also laid out some longer-term strategies that the City could take to reduce water use and generate more value from the nutrient-enriched water generated. Future collaboration between WSU and CSI in furtherance of the industrial symbiosis/biorefinery concept has been discussed for Pasco as well as other communities around the state.

Bronstad also has ongoing participation in the King County Organics Stakeholder Meetings to stay apprised of solid waste issues in the Puget Sound region and on-going policy and technical efforts to address them.

### Technical support, guidance and resource sharing

Insights and expertise were provided to a number of individuals relating to composting (throughout Washington), biosolids (Island County), and black soldier fly efforts (Chelan County). Several on our team also provided interviews and insights to a research effort led by Zero Waste Washington, *Improving Organic Materials Management in Washington State*.

Yorgey and Bronstad have also been collaborating with Biomethane LLC and The Lands Council (Spokane County) and have provided them support in the form of numerous WTFT reports and documents on a variety of topics.

We provided insight and expertise on the topic of biochar to individuals involved in several nascent biochar-related efforts across the state, including Okanogan, Whatcom, Spokane, and Lewis Counties. There has also been quite a lot of interest, including from Natural Resources Cconservation Service personnel in Washington, in understanding the potential air quality impacts of biochar. A chapter co-authored by Yorgey and other particpants of the Biomass to

Biochar workshop (described below) on *Air Pollutant Emissions and Air Emissions Permitting for Biochar Production Systems* has been distributed to these individuals in draft form.

Many stakeholders across the state have also reached out with questions which are explored in-depth in the extension document *Carbon Sequestration Potential in Cropland Soils in the Pacific Northwest: Knowledge and Gaps*, completed as part of the Waste to Fuels Technology Partnership during the 2017-2019 biennium (currently in revision within the WSU peer review system). A range of environmental groups from across the state are seeking a better understanding of the potential for agricultural croplands to store carbon through amendment and other strategies, as well as agricultural groups, who are interested in the potential to build more resilient soils and/or address climate issues.

## Extension resources

Written extension resources are available online and provide on-demand information to a wide variety of stakeholders interested in improved organics management. These resources are an important complement to in-person presentations and individual conversations. This biennium's efforts included a number of blog posts highlighting the insights learned from last biennium's projects. Posts were cross posted on AgClimate.net (a website with a regional audience, featuring topics related to climate change and agriculture/forestry) and CSANR's "Perspectives on Sustainability" blog. Blog posts were shared via AgClimate and CSANR's social media and there was a continued effort to reach the broader organics community by directly sending posts to representatives of this community.

- **Hills, K.** 2020. Check It Out: Can Biochar be Used for Carbon Dioxide Drawdown in Washington State? (March 13, 2020)
- **Hills, K.** 2020. A New Method for Measuring Plant Available Water Capacity Helps Document Benefits of Biochar-Soil Mixtures (May 18, 2020)
- **Hills, K.** 2020. Boutique Biochars: Exploring Engineering Strategies to Increase Phosphate Adsorption (August 10, 2020)
- **Hills, K.** 2020. Municipal Compost Use in Agriculture: A Question of Cost and Value (September 23, 2020)
- **Hills, K.** 2020. Compost Emissions – More than Just a Matter of Smell. (October 20, 2020)
- **Bronstad, E.** 2021. Developing Biochar Markets in the Pacific Northwest. (January 29, 2021)
- **Brady, M.** 2021. Lessons from Recycling Policy can Inform Compost Policy. (April 8, 2021)
- These blogposts, and older WTFT blogposts from previous biennia, were viewed 5,775 times between July 1, 2019 and June 30, 2021. Beyond the initial readership, posts on biochar markets had also garnered the attention of Pacific NW Ag Network. In February

2021, a reporter from the organization reached out with an interview request to develop a story on the biochar markets work. This resulted in a short story that ran in PNW Ag Network in March 2021, *Biochar Holds Great Potential For Rural Northwest Communities*.

In addition to these blog efforts, other resources including extension documents and talk recordings completed via the Waste to Fuels Technology partnership and a wealth of other information related to organic resource recovery is being maintained on the waste management topic pages of the Center for Sustaining Agriculture and Natural Resources website, which includes subtopic pages for compost, pyrolysis (biochar), anaerobic digestion, biofuels, and nutrient recovery. In total, on-demand products (other than blogs) that were directly relevant to Waste to Fuels Technology (those produced this biennium, and those produced in previous biennia) were viewed a total of 11,112 times during the 2019-2021 biennium.

## Biomass to Biochar regional workshop

Biochar technology has potential to mitigate climate change, improve forest and soil health, decrease wildfire risk, and revitalize rural economies in the Pacific Northwest and beyond. There are now more than 20 companies making and providing biochar in Washington State and the Pacific Northwest (T. Miles, personal communication; see http://www.pnwbiochar.org/producers/). Within this context, a workshop called *Biomass to Biochar: Maximizing the Carbon Value* was held to work with current industry, research, and other stakeholders to identify current barriers and chart a roadmap for future development of biochar technology in the Pacific Northwest and beyond. Though originally envisioned as an in-person event, due to COVID-19, this virtual workshop took place from April 27-30, 2020. Workshop participants represented a wide breadth of regional expertise in biochar including participants from industry, NGOs, government, and researchers.

The main objectives of the workshop were to:

Explore five of the most promising contexts for biochar production and use in the Pacific Northwest, identifying current barriers and the most impactful strategies for moving each sector forward, and define the key elements of an overall strategy for investors, philanthropists, policy makers and others looking to help transform biochar technology into a widespread, effective method for addressing climate change while maximizing its beneficial impacts on managed ecosystems and rural communities.

During the course of the four-day workshop, keynote talks were given on big picture topics related to biochar and climate change, biochar use in agriculture, carbon drawdown in forestry, and California's strategy for biochar and fire reduction. In addition, participants presented on topics related to biomass availability, biomass handling, biochar production, and biochar properties, and environmental permitting issues.

Ecology funding via the partnership supported two of the five workgroups exploring different contexts, specifically the production of biochar from municipal resources co-located with commercial composting facilities, and a second scenario focused on distributed smaller-scale systems. This investment was complemented by funding from U.S. Forest Service (USFS) region 6, which supported the other workgroups. Workgroups met initially during the workshop, and continued to meet over several months to consider the opportunities that conversion of biomass to biochar offers as a way to mitigate climate change, improve forest and soil health, decrease wildfire risk, bolster ecosystem services, and revitalize rural economies. They explored how biomass is harvested, converted to biochar and applied, and where operational changes and funding could significantly magnify biochar's contributions. The five workgroups (and the source of financial support) were:

- Place-Based Biochar Production (Ecology support)
- Moderate Scale Biochar Production Across Forested Landscapes (USFS support)
- Centralized Biochar Production Facilities (USFS support)
- Biochar Produced and Utilized at Municipal Compost Facilities (Ecology support)
- Agricultural Use (USFS support)

In a post-event evaluation, 79% and 74% of participants reported that the workshop achieved its objectives well or very well for objectives 1 and 2, respectively. In addition, 84% of participants anticipated that new knowledge or new connections that they made as a result of the workshop would impact their work relating to biochar over the next year.

After the workshop, the five working groups continued to meet to write and finalize report chapters, and the core team (Amonette, Yorgey, Hills, and Jim Archuleta from USFS) drafted additional chapters to provide background. Workshop findings were presented through a number of different presentations at the *Scaling Biochar Forum* which took place October 13 and 14, 2020, and hosted a range of West Coast individuals and entities interested in learning more about strategies for encouraging broader development of the biochar sector. Presentations that were made based on the Biomass to Biochar workshop included:

- **J.E. Amonette**, Major Funding Priorities Identified by the Biomass to Biochar: Maximizing the Carbon Value Workshop. (Estimated 55 attendees; 114 views of recording.)
- K. Wilson, A Carbon Conservation Corps to Restore Forests with Biochar. (Estimated 35 attendees; 189 views of recording.)
- J. Archuleta, Toward Sustainable Forest, Farm, and Urban Management: Biochar's Place within Landscape Reallocation of Organic Matter. (Estimated 35 attendees; 69 views of recording.)
- D. Dumroese and C. Rodriguez-Franco, Using Biochar for Abandoned Minelands and Forest Restoration. (Estimated 35 attendees; 69 views of recording.)
- G. Flora, Building a Sustainable Biochar Industry. (Estimated 35 attendees; 88 views of recording.)

- J. Dooley, Maximizing the Net Carbon Content of Biochar at Distributed and Community-Scale Production Sites. (Estimated 35 attendees; 71 views of recording.)
- M. Fuchs, Integrating the Biochar and Compost Industries: Carbon Drawdown for Profit. (Estimated 35 attendees; 108 views of recording.)
- T. Miles, Building a Large-Scale Biochar Industry. (Estimated 35 attendees; 3539 views of recording.)
- K. Trippe, Developing a Nationwide Framework for Evaluating and Predicting Agronomic Responses to Biochar-Based Amendments. (Estimated 35 attendees; 71 views of recording.)

The workshop also helped catalyze Jim Amonette and two other workshop participants (David Laird and Deborah Page-Dumroese, with other co-authors) to collaborate on an article, *Integrated Biochar Research—A Road Map* that was published in the Journal of Soil and Water Conservation.

## 1.2 Leveraging Waste to Fuels Technology funds to increase impact

The Waste to Fuels Technology partnership plays an important role in engaging researchers at WSU and beyond in next-generation waste processing issues. Many partners use their work within the partnership to leverage additional funding that enhances their work and impact in the region. Partnership researchers and extension professionals were able to leverage the Waste to Fuels Technology work to successfully obtain $2,689,778 in additional funds during the 2019-2021 biennium. Since 2017, the partnership has obtained a cumulative total of $3 million. Tables 1 and 2 provide details.

Table 1: External funds obtained during the 2019-2021 biennium by partnership members to work on issues relating to the issues explored under the Waste to Fuels Technology partnership.

| **Award Details** | **Dollars Awarded** |
|---|---|
| **Current Biennium (2019-2021)** | |
| O'Neil, T. and **T. Jobson** (2020), Research to Improve Compost Facility Air Emissions Permitting, Environmental Education & Research Foundation, 2021-2022 | $225,000 |
| **Yorgey, G.G.** and **E. Bronstad**. (2021). Smaller Cohorts for Bigger Advancements: Honest Admissions About Anaerobic Digestion Project Risks and Rewards. United States Environmental Protection Agency. 2020-2022. | $129,727 |
| **Yorgey, G.G.** and **J.E. Amonette**. (2020). Biomass to Biochar: Maximizing the Carbon Value. United States Forest Service, Region 6. 2020. | $46,000 |
| **J.E. Amonette**, **D.P. Collins**, **N. Stacey**, and A. Dhingra. (2021). Quantifying Synergy Among Soil-Based Carbon-Drawdown Approaches. WSU CSANR BIOAg. 2021-2022 | $40,000 |
| **J.E. Amonette** and **G. G. Yorgey**. (2021). A Plus Tree Biochar and Milling Project. Sonoma Ecology Center (funds from the California Climate Investment Grant Program through Cal Fire). 2021-2023 | $15,000 |
| **Chen, S.** (2020). Develop an Efficient and Cost-Effective Novel Anaerobic Digestion System Producing High Purity of Methane from Diverse Waste Biomass. US Department of Energy. | $2,234,051 |
| **Current Biennium (2019-2021) Total** | **$2,689,778** |

Table 2: External funds obtained during the 2017-2019 biennium by partnership members to work on issues relating to the issues explored under the Waste to Fuels Technology partnership.

| **Award Details** | **Dollars Awarded** |
|---|---|
| **Previous Biennium (2017-2019)** | |
| **Garcia-Perez M**., G. Moller, and M. Strawn (2019) Engineered Biochars to Enhance the Profitability of Distributed Energy Systems to Reduce the Environmental Impact of Anaerobic Digesters. | $246,000 |
| **Yorgey, G.G., D. Gang, D.P. Collins**,. and **S. Seefeldt** (2019) Integrating Municipal Compost and Biochar for Production of High-Value Crops. United States Department of Agriculture, McIntire Stennis Capacity Funding (Internal competitive allocation of funds). | $50,000 |
| **Collins, D.P.**, A. Siegner, and **N. Stacey** (2019) Investigating the Elasticity of Biochar: Manure Handling, Compost Feedstock, Soil Amendment and Carbon Storage. United States Department of Agriculture, Western SARE, Professional + Producer Grant. | $49,988 |
| **Amonette, J.E.** (2019) Impact of Process Emissions on Climate Offsets by Different Biochar Production Methods, Washington State University BIOAg. | $11,670 |
| **Past Biennium (2017-2019) Total** | **$357,658** |

## 1.3 National and international reach

Though not the primary goal of the Waste to Fuels Technology partnership, sharing the results of partnership research with researchers across the U.S. and the world, via presentations and journal publications, is an important avenue for generating interest and momentum relating to next-generation organics processing.

Relevant peer-reviewed journal publications from this biennium that resulted from work carried out either in part or in whole as the results of the partnership include:

- **Stacey, N.**, T. Tea, **S. Seefeldt, A. Bary,** and **D.P. Collins**. In review. Biochar-poultry manure compost alters temperature and nitrogen dynamics during composting and improves crop growth following field application. Compost Science and Utilization. Under Review; submitted 25 May, 2021.
- Lehmann, J., A. Cowie, C.A. Masiello, C. Kammann, D. Woolf, **J.E. Amonette,** M.L Cayuela, M. Camps-Arbestain, and T. Whitman. (Revised and resubmitted) Biochar in climate change mitigation. Review article submitted to Nature Geoscience.
- **Haghighi-Mood, S., M. Ayiania**, H. Cao, O. Marin-Flores, **Y. Jefferson Milan, M. Garcia-Perez.** Nitrogen and Magnesium Co-doped Biochar for Phosphate Adsorption. Paper Accepted in Biomass Conversion and Biorefinery, 2021
- **Amonette, J.E.**, H. Blanco-Canqui, C. Hassebrook, D.A. Laird, R. Lal, J. Lehmann, and D. Page-Dumroese. Journal of Soil and Water Conservation January 2021, 76 (1) 24A-29A; DOI: https://doi.org/10.2489/jswc.2021.1115A
- Zhang, J., **J.E. Amonette**, and **M. Flury**. 2021. Effect of biochar and biochar particle size on plant-available water of sand, silt loam, and clay soil. Soil & Tillage Research 212:104992. https://dx.doi.org/10.1016/j.still.2021.104992
- Cao, H., X. Wu, S.S.A. Syed-Hassan, S. Zhang, S.S. Mood, **Y. Jefferson-Milan**, and **M. Garcia-Perez.** Characteristics and mechanisms of phosphorous adsorption by rape straw-derived biochar functionalized with calcium from eggshell. Bioresources Technology, 2020, 318, 124063.
- **Ayiania, M.**, E. Weiss-Hortala, **M. Smith**, J.S. McEwen, and **M. Garcia-Perez**. Microstructural Analysis of Nitrogen Doped Char by Raman Spectroscopy: Raman Analysis from First Principles. Carbon, 2020, 167, 15, 559-574.
- **Haghighi-Mood, S., M. Ayiania, Y. Jefferson-Milan,** and **M. Garcia-Perez.** 2020. Nitrogen doped char from anaerobically digested fiber for phosphate removal in aqueous solutions, Chemosphere. 2020 Feb; 240:124889. doi: 10.1016/j.chemosphere.2019.124889. Epub 2019 Sep 17. PMID: 31563102.
- **Ayiania, M**, **M. Smith**, A.J.R. Hensley, L. Scudeiro, J.-S. McEwen, and **M. Garcia-Perez**. 2020. Deconvoluting the XPS spectra for nitrogen-doped chars: An analysis from first Principles. Carbon, 162:528-544.
- **Ayiania, M.**, A.J.R. Hensley, K. Groden, **M. Garcia-Perez**, and J.-.S McEwen. 2019. Thermodynamic stability of nitrogen functionalities and defects in graphene and

graphene nanoribbons from first principles. Carbon, 152: 715-726, doi.org/10.1016/j.carbon.2019.06.019.

Academic/national and international presentations that occurred during the current biennium included:

- **Khosravi, N.**, and **B.T. Jobson**. Estimation of VOC emission factors for manure composting by PTR-MS. American Geophysical Fall Meeting, Dec 2020.
- **Haghighi Mood, S., M. Ayiania**, and **M. Garcia-Perez**. Phosphate Removal from Aqueous Solution Using Nitrogen-Metal Co-Doped Biochar. Presentation at the Thermal & Catalytic Sciences Virtual Symposium, October 2020.
- **Ayiania, M., S. Haghighi Mood**, J.-S. McEwen, and **M. Garcia-Perez**. Novel Amorphous Carbons for the Adsorption of Phosphates. TCS 2020.
- **Garcia-Perez, M.**, and J. Almodovar. Acid Carbonization Process to Maximize Carbon Efficiency: Novel path for Carbon Storage and Biofuel Production. Presentation to the University of Tennessee, March 29, 2021.
- **Garcia-Perez, M.** Progress in Biomass Fast Pyrolysis and Bio-oil Refining. China Agriculture University virtual presentation, May 24, 2020.
- **Garcia-Perez, M., A. Tanzil,** M. Wolcott, X. Zhang, and J. Hollday. Production of Cheap Sustainable Aviation Fuels (SAFs): A Carbon Balance Problem. Cape Breaton University Canada, February 9, 2021 (Estimated attendees: 50), National University of Colombia, March 15, 2021 (Estimated attendees: 150), DAABON Midterm Summit, Colombia, May 2021 (Estimated attendees: 20)

## 1.4 Impacts of technology transfer, outreach, and extension activities

In total, extension efforts resulted in an estimated 1,448 in-person and virtual opportunities for live interactions with key stakeholders who work either primarily on organics management in the region, or whose work touches on sustainable organics management in various ways. Our efforts also resulted in 16,887 views of Waste to Fuels Technology reports, publications, blog posts, recordings, and other extension resources that were funded (either fully or in part) via the partnership. Work carried out in the previous and current biennium was used to leverage an additional $2,689,778 awarded during this biennium to support work in areas related to Waste to Fuels Technology priorities.

While the ultimate outcome of these interactions and activities is likely to be realized in the long term, there are short-term indications that stakeholders have found them useful. We heard numerous times from both municipal and industry stakeholders in Washington that they were very excited to hear that Ecology was involved in this type of work, support WSU's engagement in this type of research, and appreciate the extension support provided relating to these topics. A few representative comments include:

- "These are really good resources, I look forward to reading all of them in detail!"
- "I was very pleasantly surprised at how well the workshop was executed, people stayed engaged, and progress was made on time. Kudos to the WSU team!"
- "Considering the situation with COVID-19 I truly believe the overall workshop was well planned and executed. Thank you very much."

While we are not always able to capture the direct impacts of our interactions with individuals across the state, our existing survey data suggest that we are having an influence. As one example, 84% of participants in the *Biomass to Biochar* workshop said that they anticipated that new knowledge or connections that they made as a result of the workshop would impact their work relating to biochar over the next year, suggesting that our extension effort was effective.

# Chapter 2: Comparing Methods to Measure Air Emissions from Commercial Compost Facilities

*Tom Jobson*

## 2.1 Introduction

A tension now exists in Washington State between the policy of encouraging diversion of urban food and green waste from landfills to composting facilities, and potentially subjecting these same facilities to more costly and complicated air permitting applications as they expand their facilities to meet demand. Our study was motivated by the need to better understand volatile organic compound (VOC) emission factors to aid air emissions permitting in Washington State. Understanding emission factors for the active composting phase and the efficiency of control technologies to reduce VOC emissions are important for air emissions permitting. The VOC emission factor, defined as pounds VOC emitted per wet ton of feedstock material, and control technologies are used establish a facility's yearly emissions potential (potential to emit, PTE). The PTE value determines the course of the permitting process. If the PTE is greater than 100 tons per year, the facility would be required to apply for a Title V air permit through the U.S. Environmental Protection Agency (US EPA). These factors are required for preconstruction permit programs for new large facilities or for significant changes in capacity to current facilities that would warrant a Prevention of Significant Deterioration (PSD) permit, as well as for ongoing requirements for Title V air permitting.

### VOC Emission Factors

There is limited data available on VOC emission factors from composting of municipal green waste and residential food waste - the major compost feedstocks for Washington State compost facilities. The VOC emission factors that have been developed are based on emissions testing conducted in California on green waste composting using turned window systems. The VOC emission factors represent the total amount emitted over the active and stabilization / curing phases of the composting process.

Variability in compost emission factors would be expected due to differences in process conditions and feedstock materials. Washington State Department of Ecology and other air quality permitting jurisdictions in Washington, such as the Puget Sound Clean Air Agency (PSCAA), and elsewhere, such as Colorado (CDPHE, 2012), have adopted the San Joaquin Valley value of 5.71 lbs VOC / wet ton for uncontrolled VOC emissions from windrow green waste composting if site-specific data is lacking.

Due to the limited nature of the underlying studies, it is possible that this emission factor may not adequately represent emissions from the various types of composting processes and aeration types now being used in California and Washington. Composting facilities in Washington State employ a range of composting methods ranging from turned windrows, mechanically aerated piles with positive, negative, or reversing aeration systems, and positively aerated piles covered with a microporous membrane material. The diversity of processing conditions poses some challenges in developing VOC emission factors that could be broadly used in air emissions permitting.

## Sampling Methods

A key issue in compost VOC emissions testing is the cost of testing and the inherent uncertainties and difficulties in the sampling and test methods. The standard method for measuring VOC emissions from surfaces employs a surface emission flux isolation chamber (US EPA, 1986). This chamber samples only a small surface area (0.13 m2) compared to the total surface area of the pile, and obtaining a representative sample is a basic problem with this method. Air samples are collected from the chamber to determine VOC emission rates.

For the studies used to inform the emission factor used by the San Joaquin Valley Air Pollution Control District, VOCs were sampled and quantified according to a standard analysis method approved in California for emissions testing from combustion sources, SCAQMD Method 25.3. This method is similar in principle to US EPA Method 25. With this method, a total VOC value is determined from the sum of two separate samples collected from a surface flux isolation chamber: an air sample collected into an evacuated canister and a water filled impinger sample that collects water soluble VOCs not recoverable from canisters, such as organic acids. The canister sample is subsequently analyzed by gas chromatography for total nonmethane nonethane nonethene organic compounds. The water from the impinger is analyzed for total organic compounds using a commercial water quality analyzer. The results are combined to give a total VOC amount in units of ppmC (parts per million by carbon). The method does not quantify individual compounds that are emitted, nor does it determine total VOC mass emitted. The method counts carbon, so the emission rates are based on amount of carbon (C) emitted in the VOCs, but does not account for other elements like oxygen. A total VOC mass could be determined if the relative amounts of the various VOCs emitted were known. While some information is available for gases, no such information exists for the types of water-soluble VOCs sampled in the impinger. A general lack of information on the types of VOCs emitted and how these vary with pile age and feedstock material prevents reporting emissions in terms of actual VOC mass emitted.

To provide a better accounting of VOC mass emitted, this study investigated the use of proton-transfer reaction mass spectrometry (PTR-MS) to measure VOCs. The PTR-MS instrument has the potential of being able to measure water-soluble VOCs emitted in composting such as alcohols, acids, and ketones. The PTR-MS instrument can be used in the field to sample directly from flux chambers and negative aeration ducts, thus eliminating the need for sample collection into water-filled impingers and canisters. This would simplify the sampling and analysis of VOC emissions and provide an account of what compounds are emitted rather than total VOC carbon mass as reported by Method 25.3. Knowledge of the speciated emissions profile would be beneficial in developing VOC emissions factors for permitting. VOC emissions are regulated at the state and federal level because some are known to be toxic and, thus, potentially harmful to human health, and because they act as precursors in photochemical reactions that create other federally regulated pollutants such as ozone and particulate matter (PM2.5). Some VOCs are exempt by the US EPA from the VOC category because they have low photochemical reactivity and thus do not participate in local ozone and PM2.5 pollution and have low toxicity. Acetone is an exempt VOC and is commonly observed as a major compost emission, but the ability to discount emissions of acetone from composting facilities is not possible with total VOC measuring methods such as Method 25.3.

This report has two parts:

- The first part describes some laboratory-based composting experiments with feedstock material from the Washington State University (WSU) facility, composed primarily of manure and animal bedding, and a trial with green waste feedstock from a commercial composter. Materials were composted in-vessel using 100-gallon containers with lids and positively aerated. The section describes the types of compounds emitted and what is measurable by PTR-MS and gas chromatography mass spectrometry (GC-MS).
- The second part describes results from emission testing at the WSU compost facility on a 160-ton negatively aerated uncovered pile. In this task, a sampling approach was developed to continuously sample from the negative aeration duct using a PTR-MS instrument over the course of the active composting phase (25 days). VOC measurements were made once per minute, and from the measured concentrations and duct air flow rate a VOC mass emission rate was determined. This is the first report of continuous speciated VOC measurements from an aerated static pile (ASP) system. Canister samples were collected from the duct for comparison with the PTR-MS. Compost pile surface flux densities were also measured using flux isolation chambers. Chambers were sampled by the PTR-MS in addition to canister sample collection and analysis by GC-MS. The types of compounds emitted, VOC emission rates as a function of pile age, the relative proportion of fugitive surface emissions to duct emissions, and a total VOC emission factor for this ASP system are reported. The report concludes with some general recommendations for emissions sampling and testing methods based on results from this work and discussions held with various stakeholders, including technical experts at US EPA.

## 2.2 Compost facility survey

As an initial step to support the development of methodologies for appropriately measuring emissions from diverse Washington composting facilities, the ten largest commercial composting facilities in Washington were surveyed (one of the ten declined to participate). One smaller facility was also surveyed, as it was the site of field testing of volatile organic compound (VOC) air emissions measurement methodologies. Facilities were asked about their feedstocks, composting processes, and air emissions mitigation technologies (Jobson et al., 2020).

### Survey findings

Primary findings from the survey included the following:

- Only one facility operated as turned windrow. Most facilities used aeration; turned aerated piles were a common practice (4 facilities).

- Nine of the ten facilities that participated in this survey have mechanical aeration systems. This includes systems that run only as positive aeration (2 facilities), only as negative aeration (4 facilities), or as an air flow reversing system (3 facilities) whereby the air flow is sometimes positive, sometimes negative, or off. Sometimes different phases of the composting process vary in the aeration that is practiced.
- Three broad groupings in terms of process types were identified: turned windrows with no mechanical forced aeration, Gore fabric covered positively aerated piles, and uncovered aerated piles. The most common process type was uncovered aerated piles using either negative aeration (3 facilities), positive aeration (1 facility), or reversing aeration (2 facilities).
- Two facilities have aeration systems designed by Green Mountain Technology. Five facilities have systems designed by Engineered Compost Systems. Two facilities have systems designed by Jumelet Environmental Engineering (with involvement of the compost company).
- Eight of the ten facilities surveyed have a biofilter. Two facilities utilize Gore covers. Some compost facilities cover the pile with overs (4 facilities) or with finished compost (2 facilities) (though exactly where in the process this covering occurs varies by facility).
- Feedstocks varied between facilities and seasonally. The most common seasonal variation noted was that facilities with a large percentage of green waste or mixed food/green waste reported an influx of grass during March or April through June and an influx of grass and leaves in the fall. Seasonal variation in agricultural waste products (e.g., cherries, pears, apples, grape pomace, pumpkins) was noted by some facilities. One facility mentioned moisture as being significantly different by season. Facilities have strategies for adjusting feedstock mixes (e.g., incorporating more woody materials during the spring when there are increases in nitrogen rich high-moisture feedstocks such as grass) to compensate for these seasonal changes.
- Process parameters varied somewhat between facilities, and have not been summarized, as those are most relevant within in the context of the composting process at each individual facility.

## Considerations for sampling

The results of the survey of Washington compost facilities shed light on important considerations for sampling. The current method of using surface flux isolation chambers could work for sampling diffusive emissions from windrows, and positively aerated piles, but would miss VOC emissions being pulled from the piles under negative aeration flow. Air sampling for determining VOC emission rates will have to be modified to sample air from negative aerated piles and reversing air flow aeration systems.

Figure 1 illustrates a generic compost facility using a negative aeration system to highlight known VOC emission sources. Emission of VOCs can occur from feedstock piles of material initially brought to the facility. Feedstock material may be stockpiled for several days or more, either in an enclosed building or out in the open. The feedstock material is ground and mixed, adjusted for moisture content, then sent to the active composting stage where the majority of the decomposition occurs. After active composting, the material is stored for several more weeks as slow decomposition occurs before it stabilizes and can be stockpiled for sale.

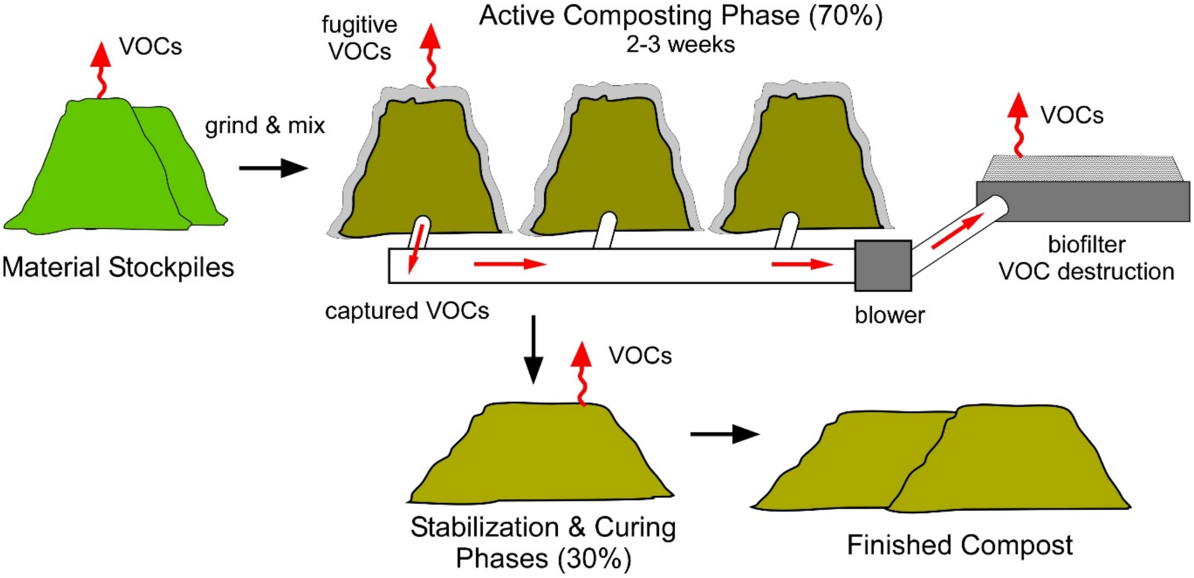

Figure 1: Schematic of VOC emission sources from a compost facility using negative aeration for active phase composting. (Jobson et al., 2020)

The active phase of composting is thought to be the major source of VOC emissions. During this phase piles are aerated by mechanical air flow systems or periodic turning. The active piles may also be covered in a layer of finished compost to provide a thermal insulation cover to ensure that the pile is heated sufficiently to its outer surface. This covering can also act as an adsorbent media for VOC emissions from the pile surface. The active phase may last for three weeks, after which the piles are moved and spend time in a stabilization / curing phase. In this phase, which can last from two to eight weeks, the piles are still biologically active but generate less heat and emit VOCs at a lower rate. These piles may or may not be aerated or turned depending on the facility operator.

It has been estimated that 70% of the VOC mass is emitted during the active composting phase and 30% is emitted during the stabilization / curing phase (CIWMB, 2007). In a negative aeration system, the air pulled through the piles during the active phase is typically sent to a biofilter where the VOCs are consumed by microbes growing in some medium, typically wood chips. Biofilters have been documented to be efficient at removing VOCs and ammonia. This is an important control device in reducing facility VOC emissions for determining PTE. For negative aeration systems the proportion of fugitive VOCs emitted from the pile surface compared to the proportion pulled from the pile by the negative aeration flow has not been clearly established, though data may be available from site-specific testing reports. At facilities using negative aeration with static pile systems, sampling of negative aeration ducts and measurement of biofilter destruction efficiency are the critical process points for VOC emission factor development.

## 2.3 Laboratory experiments

In order to compare PTR-MS and GC-MS techniques under controlled conditions, in-vessel composting (approximately 380 lbs of material in a 100-gallon container) was undertaken in the lab in a series of 5 trials. Trials 1-4 used feedstock materials from the WSU compost facility, consisting primarily of manure and animal bedding. Trial 5 used compost feedstock material from a commercial operator consisting primarily of woody material, green waste, and food waste. Air samples were continuously extracted for analysis by PTR-MS, GC-MS, and other instruments including a methane / carbon dioxide ($CO_2$) analyzer. Typically, the experiment was run for 5-7 days, until VOC abundance in the headspace was insignificant for most compounds. Issues with water vapor condensation in air sampling lines led to the use of an ejector dilutor, a strategy which was subsequently adopted for sampling the negative aeration duct at the WSU facility.

Trial 2 used wet feedstock (water was released from it when “the squeeze test” was performed) to compare performance of wet feedstock materials. Emission of methane and some VOCs, notably methanol, 2-butanone and acetone, were much higher than in other trials. This supports the view that the magnitude and relative abundance of the VOCs emitted will be a function of compost process conditions. This wet material emitted a series of C3-C9 ketones as identified by GC-MS. The emissions of larger ketones may be a useful emissions signature of overly wet composting conditions.

Water soluble organics compounds were emitted at the highest rates from both green waste and manure / animal bedding composting materials. Comparison to GC-MS analysis confirmed that monoterpenes, camphor, acetone, 2-butanone, dimethylsulfide, dimethydisulfide, acetaldehyde were present in the samples, consistent with results from Jobson and Khosravi (2019). Methanol and acetic acid were the most abundant emissions from the green waste material (Trial 5). The abundance of water-soluble compounds like methanol and acetic acid underscores the need for sound sampling procedures to prevent losses to condensed water in surface flux chambers.

## 2.4 Emissions sampling at WSU compost facility

Volatile organic compound emissions rates were successfully measured from the active phase of a negatively aerated 160-ton pile at the WSU Compost facility. Feedstock materials were primarily manure and animal bedding that contained large amounts of straw and wood chips. A sampling approach was developed to continuously sample from the negative aeration duct using a PTR-MS over the course of the active composting phase (26 days). VOC measurements were made once per minute, and from the measured concentrations and duct air flow rate a VOC mass emission rate was determined. Meanwhile, fugitive surface emissions were determined from PTR-MS measurements on 12 different days: July 31, Aug 1, 2, 3, 4, 5, 10, 12, 14, and 19 (Figure 2).

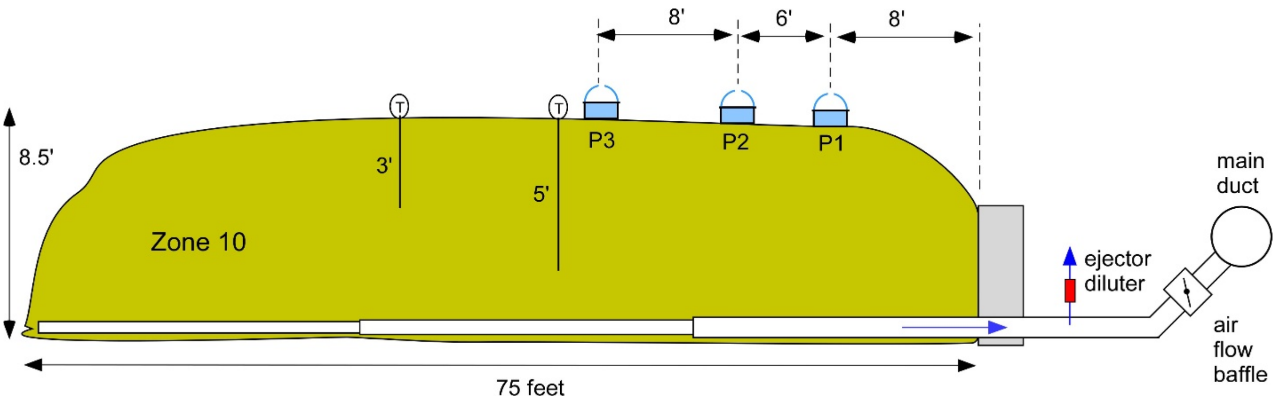

Figure 2: Sketch of surface flux sampling locations on pile ridge (P1, P2, P3). Pile was 75" long and ~ 8.5 feet high. Estimated initial mass was 160 tons. The bottom temperature probe (T) was 5 feet long and located beside sampling location #3. Air flow in the negative aeration duct was sampled using an ejector diluter at the position indicated as the duct exited the ecology block.

The PTR-MS measurements were then compared with VOC measurements made by more standard GC-MS analysis. VOC measurements made by the PTR-MS compared reasonably well with GC-MS measurements for the duct testing. Good agreement was observed for acetone and acetaldehyde. Methanol could not be quantified by our GC-MS system but this is an important compound to target; use of gas chromatography flame ionization detection (GC-FID) as used in EPA Method TO-12 would be a better approach for this compound. In general, the PTR-MS measured higher abundances of dimethyl sulfide, dimethyl disulfide, and total monoterpenes. PTR-MS measurements of 2-butanone had interferences from 2-methylpropanal (same mass) and larger aldehydes that can fragment to yield the m/z 73 ion used to monitor the presence of 2-butanone. The PTR-MS proved useful for continuous monitoring and quantifying oxygenated VOCs, in particular methanol, and very polar compounds like acetic acid.

The total amount of VOC emitted through the duct was 885 lbs. Methanol represented 65% of the total VOC mass emitted. Emission of oxygenated compounds were 92% of the measured VOC mass emitted. Acetone, a VOC that is exempt in terms of considering emissions thresholds for Title V permitting, was 12% of total emissions. The first 130 hours (5.4 days) accounted for 90% of the total VOC mass emitted through the duct, highlighting the need for sample collection in the first 5 days of the process.

VOC emissions rates through the duct were much larger than fugitive emissions from the pile surface. Emission rates through the duct were 5.47 lbs / wet ton (emissions that are subsequently treated in a biofilter), and from the surface 0.084 lbs / wet ton, for a combined emission factor of 5.55 lbs / wet ton (Figure 3). The small fraction of fugitive emissions from the pile surface implies that negative aeration systems have a high potential to control VOC emissions if biofilters downstream of the duct can be operated efficiently. This should be encouraging news for composting facilities.

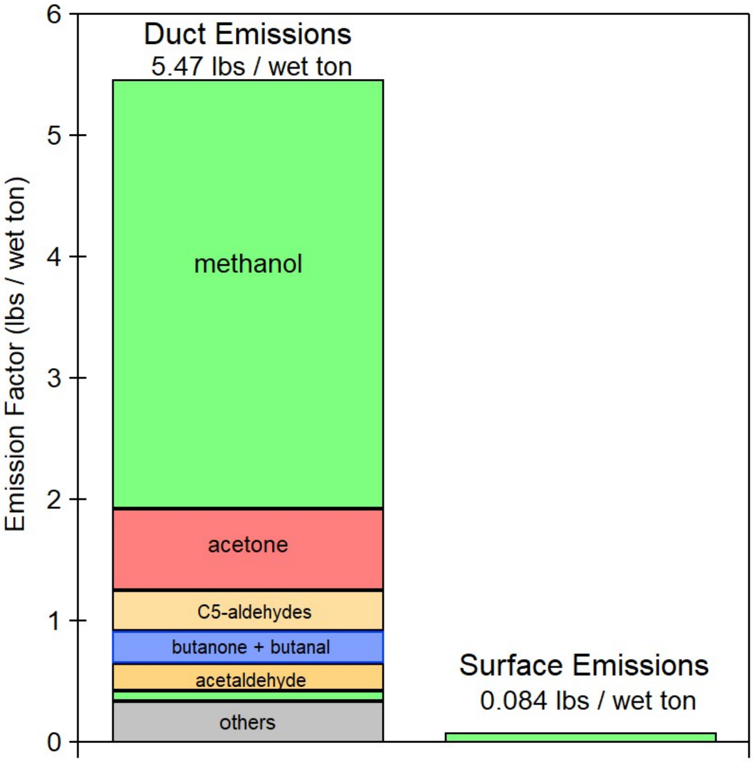

Figure 3: Summary of negatively aerated pile VOC emission factors. Thin green band below acetaldehyde are monoterpenes. Values are actual mass and not carbon mass.

In terms of mass carbon emitted, as reported by SCAQMD Method 25.3, the emission rate is 2.6 lbs C / wet ton. This value is lower than the SJV recommendation of 5.71 lbs C / wet ton for windrow green waste composting, but higher than their 1.78 lbs C / wet ton for windrow composting with manure, biosolids, and poultry litter (SJV, 2010). If emission of methanol and other oxygenated compounds identified here are found to dominate emissions generally from Washington State compost facilities, then Method 25.3 reporting of total VOC emissions as carbon mass would significantly undercount VOC mass emission rates.

Our results can also speak to the potential for the stabilization / curing phase to emit VOCs. Given the VOC emission rate at the end of the active phase (4.5 grams / hour) and applying this as a constant rate through the stabilization / curing phase, it is estimated another 7.1 lbs VOC could be emitted over the next 30 days, less than 1% emitted in the active phase. This ratio is much lower than the 30% estimated for windrow-based composting (CIWMB, 2007), indicating basic differences in material processing rates between windrow composting and aerated static pile systems.

## 2.5 Recommendations

Based on results from this work and discussions held with various stakeholders, including technical experts at US EPA, we make some general recommendations regarding sampling for VOCs at compost facilities.

### Limit surface flux sampling if possible

Surface emissions sampling with flux isolation chambers using Method 25.3 is fraught with potential problems and this motivates reducing its use. The small fraction of VOCs emitted from the negatively aerated pile surface has implications for the necessity of surface flux sampling for such systems. The very low emission rates at the end of active composting also imply that fugitive VOC emissions from the stabilization and curing phases might also be very low. This should be confirmed at another facility using negative aeration. If this finding is generally true, considerable expense and effort could be avoided by using a default fugitive emission fraction rather than performing surface flux sampling with flux isolation chambers. For windrow composting it will still be necessary to measure surface fluxes. Facilities using positive aeration must also measure surface emissions which is potentially problematic given the issues of water condensation in the flux chamber. For reversing systems, it may be sufficient to measure emission rates in the negative aeration duct and simply assume that same emission rate would apply under the same positive air flow.

As discussed, an additional issue with the flux isolation chamber method is that there are no ambient air “blanks” used to correct chamber VOC concentrations for wind dilution. Test reports showing speciated VOC data from TO-15 analysis of canister samples may be reporting compounds not emitted from the compost but rather reflect compounds found in polluted ambient air that diluted the chamber. Air permit writers should be made aware of this basic issue.

### Improve surface sampling representativeness

The second issue with surface sampling is the representativeness of the samples given the small size (0.13 $m^2$) of the surface flux isolation chambers typically used compared to the very large surface area of the compost pile. Recommendations from staff at the US EPA Measurement Technology Group included using a total temporary enclosure that could be built over the pile so that emissions from the whole pile could be captured and sampled (US EPA Method 204). This might be a prudent approach for sampling biofilters and for positively aerated piles. In both cases there is forced air flow coming up to the surface and flows may not be uniformly distributed across the surface due to local differences in underlying porosity. It is not clear how easy this would be to implement on a large positively aerated pile, but should be straightforward to implement on biofilters. Measurement of biofilter collection efficiency is very important for the emission permitting process but there are limited reports documenting performance with well-tested sampling protocols. The issue of water condensing on the enclosure surfaces could still be an issue for positively aerated piles but should be much less a problem for biofilter sampling where the air flow temperature and water vapor concentration are lower. It is recommended that a simple procedure for enclosure sampling of biofilters be worked out with facility owners. It would be important to consider materials for construction of such an enclosure to ensure they have low VOC outgassing rates.

## Test continuous in-situ duct sampling

Sampling from aeration ducts will be necessary to determine emissions rates from compost piles using negative or reversing aeration systems, and for upstream sampling ahead of the biofilter to establish its collection efficiency. A protocol needs to be established for doing this. The US EPA Measurement Technology Group recommended in-situ monitoring of ducts and avoiding use of canister sampling as many of the oxygenated compounds and other heteroatom compounds emitted in compost emissions are not stable in canisters. With Method 25.3 there is the need to determine the collection efficiency of the impinger for the high VOC concentrations in the duct.

Continuous measurements from the duct were relatively straightforward to implement using a heated ejector diluter. This allowed rapid dilution and mixing of the duct air sample with dry air to reduce the humidity and prevent sample line losses of water-soluble VOCs. For measuring the VOCs in this diluted air flow, consideration should be given to testing the commercially available semi-continuous total VOC monitors employing flame ionization detectors. Such VOC monitors use a similar measurement principle as described in US EPA Method 25 (and SCAQMD 25.3) for analyzing VOCs recoverable from canisters. These instruments are available from a number of companies, where the market is continuous emissions monitoring (CEM) from stacks. While they are designed for the analysis of hydrocarbons found in fossil fuel combustion exhaust, modifications or a custom specification may allow for measuring the light alcohols, ketones, aldehydes and monoterpene compounds emitted from compost. It is not clear how well they would perform with organic acids. For pile emissions, getting enough data in the first week to really capture the exponential increase and decrease of VOC emission rates will be important for accurate emission factors. Automated, semi-continuous sampling with these in-situ monitors would allow for around the clock monitoring of ducts and temporary enclosures of piles and biofilters. A less labor-intensive sampling methodology would reduce labor costs for source testing.

## Develop emission factor independent of facility testing

A small scale (50 yd$^3$ pile size) two zone pilot plant is being constructed at WSU by Engineered Compost Systems (Seattle, WA) with funding from Environmental Research and Educational Foundation (EREF) to test VOC emissions as a function of process conditions such as aeration method. An opportunity exists to use this testing facility to help develop VOC emissions factors for green waste, food waste, biosolids composting in Washington State that does not involve specific facility testing. A key issue in air permitting has been the adoption of California emission factors based on windrow composting. Much of Washington state composting by the biggest facilities is not windrow composting but an aerated static pile process (Jobson et al., 2020). Windrow emission factors may not accurately reflect VOC emissions from aerated static piles, where better control of temperature and oxygen levels in the pile may decrease emission rates compared to windrows. This is something that should be determined experimentally with the pilot plant in side-by-side trials. The data would be extremely useful to the air emissions permitting process for the composting systems widely used in Washington state. The pilot plant could also be a test bed to develop enclosure sampling and in-situ duct sampling methods recommend by the US EPA Measurement Technology Group. A partnership with stakeholders (compost facility operators, air permit writers, equipment suppliers, air quality chemists) to guide development of more accurate, reliable, and cost-effective VOC sampling and analysis methods for the state is recommended.

Additional detail is available in the two technical reports *Survey of Large Commercial Compost Facilities in Washington* and *Comparing Methods to Measure Air Emissions from Commercial Compost Facilities* on the WTFT 2019-2021 webpage of Washington State University's CSANR.

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# Chapter 3: Development of Engineered Biochar Cocktails for Odor Removal in Composting Facilities

*Yaime Jefferson Milan, Manuel Garcia-Perez*

## 3.1 Introduction

Solid organic waste management is a challenge worldwide. Several technologies, including anaerobic digestion and composting, utilize microorganisms to process solid organic waste into more stable and environmentally friendly products that may be utilized by diversified end-markets including landscaping and agriculture (Font et al., 2011).

Composting is widely used in Washington State and throughout the U.S. to sustainably manage organics. In 2019, there were approximately 66 compost facilities in Washington State, composting a total of nearly 1.4 million tons of material (Ecology, 2019). As the amounts of organic and food wastes diverted to composting facilities has risen in recent years, some facilities have increased the emission of odors. Emissions of concern can include nitrogen-based compounds, sulfur-based compounds, and volatile organic compounds (VOCs) (Font et al., 2011; Eitzer, 1995). VOCs are organic chemicals with higher vapor pressures and malodorous and hazardous properties (Dhamodharan, 2019; Komilis et al., 2004). Certain VOCs are carcinogens, and can directly affect human health.

The composting process is aerobic; however anaerobic conditions exist in some parts of the piles (Dhamodharan, 2019). During composting, carbon dioxide ($CO_2$) is released under aerobic conditions, while $CH_4$ (methane), $H_2S$ (hydrogen sulfide), and $N_2O$ (nitrous oxide) are generated under anaerobic conditions. The $CO_2$ from composting is not normally considered a greenhouse gas emission because it is of recent origin from living material. However, its retention in finished compost could enhance carbon sequestration, an important avenue for reducing greenhouse gas emissions. Methane and nitrous oxide are powerful greenhouse gases that contribute to climate change. Meanwhile, $H_2S$ is a common, dangerous, and odorous compound. Ammonia ($NH_3$) is a malodorous and potentially toxic air pollutant. Organic wastes emit variable amounts of $CO_2$, $CH_4$, and $N_2O$ based on their carbon and nitrogen content and the conditions under which the composting process is managed (Swati and Hait, 2018).

Composting operations have a handful of methods available to control odor (Ma et al., 2013). Among these methods, it is critical to ensure that conditions in the composting piles remain aerobic to avoid the formation of malodorous products released under anaerobic conditions. This is achieved by blending adequate lignocellulosic materials (e.g., wood, dry plant matter, other bulking materials) with wet materials (e.g., food wastes). Appropriately controlling the air supply can also be important to maintaining aerobic conditions.

As an additional control, some facilities also use biofilters and biocovers to adsorb offensive odors. Within these filters, molecules responsible for unpleasant odors are metabolized by bacteria. The main challenge of existing biofilters is the need to keep them wet, and maintaining the filters to work effectively.

If appropriately engineered, biochar may provide an additional tool in the future for managing gaseous emissions (Sánchez-Monedero et al., 2019). Biochar is a carbonaceous solid product derived from the thermochemical decomposition of wood or other organic matter in the absence of, or with restricted amounts of, oxygen (Lehmann and Joseph, 2015). In contrast to biofilters, biochar use could be applicable in situations in which the air pipe and the fans deliver air into the pile and the odors are released to the surrounding environment. Blending the right quantities of engineered chars with the composting materials should be able to retain most of the odors released by the composting facility.

The objective of this research was to identify production conditions for creating biochar with an enhanced capacity to adsorb some common air pollutants released during biomass composting in municipal solid waste composting facilities. Because of the diversity of the contaminants released during biomass composting, it is unlikely that a single type of biochar will be able to adsorb all of the contaminants, so the development of biochar mixtures was expected to be more effective. This research focused on three common pollutants – $CO_2$, $H_2S$, and $NH_3$ – as a first step towards understanding how biochar, and cocktails of various types of biochar, can be used to adsorb emissions.

This research involved first producing and then characterizing 12 types of engineered biochar (elemental composition, proximate analysis, gas physiosorption analysis, Fourier-transform infrared spectroscopy, and a pH analysis). Then $H_2S$, $CO_2$, and $NH_3$ adsorption studies were performed to evaluate the ability of each type of biochar to adsorb these targeted pollutants. Finally, we explored how the diverse biochars could be blended to remove the pollutants in a hypothetical composting facility.

## 3.2 Biochar preparation, production, and activation

Wheat straw and Douglas fir (*Pseudotsuga menziesii)* biomass feedstock were chosen to produce the biochars based on their availability in Washington State, and their common use for biochar production. The feedstocks, which were obtained from Green Stripe brand wheat straw (WS) and forestry residuals (DF), were washed and oven-dried, and particles were separated. Twelve different samples of biochar were produced from WS and DF pyrolyzed at two different temperatures (400 °C and 600 °C). The 12 samples, and their production temperatures, are shown below in Table 3.

To produce biochar samples doped with nitrogen (N), the following pyrolysis process was used: Approximately 3 grams of biomass (DF or WS) were placed in the reactor in a nitrogen ($N_2$) atmosphere (oxygen-free) as the temperature was raised to 400 °C or 600 °C, then maintained at that temperature for one hour under $N_2$ gas (to allow the carbonization process to occur). The biochar was then treated at the same temperature with $NH_3$ for 1 hour to produce nitrogen-doped (N-doped) biochar (the doping process). Samples were then cooled to 25 °C in nitrogen gas. The flow rates of gas used in the process were of 500 mL $min^{-1}$ for both $N_2$ and $NH_3$.

The magnesium (Mg)-impregnated biochar was prepared by first mixing 10 grams of DF or WS biochar with 40 mL of magnesium chloride hexahydrate ($MgCl_2$•$6H_2O$) solution, prepared by dissolving 16.75 g of $MgCl_2$•$6H_2O$ in 100 mL of deionized water at room temperature. Then, the steps outlined above for N-doping were followed.

The biochar yield was recorded, and the biochars obtained were then characterized and used for the adsorption studies.

Table 3: A summary of the 12 types of biochar samples produced from wheat straw (WS) or Douglas fir (DF) biomass.

| Biochar sample | Temperature (℃) |
|---|---|
| WS Raw | 400 |
| WS Raw | 600 |
| WS N-doped | 400 |
| WS N-doped | 600 |
| WS N-Mg doped | 400 |
| WS N-Mg doped | 600 |
| DF Raw | 400 |
| DF Raw | 600 |
| DF N-doped | 400 |
| DF N-doped | 600 |
| DF N-Mg doped | 400 |
| DF N-Mg doped | 600 |

The yield of biochar depends on feedstocks and production conditions. Yield decreased as the production temperature increased. WS biochars produced from raw biomass and doped with nitrogen resulted in a higher yield than the biochar obtained from DF at the same conditions, because WS feedstock has more ash content than DF, which contributed to biochar formation. The yield of DF nitrogen and magnesium doped (N-Mg doped) biochar is greater than that of WS N-Mg doped, mainly due to the presence of alkali and alkaline earth metals (Mg, calcium, potassium) that can catalyze biomass decomposition and promote the formation of biochar (Zhang et al., 2018).

## 3.3 Biochar characterization results

The following characterization analyses took place. For more information about equipment used and methods or procedures that were followed, as well as the statistical analysis, please see Milan and Garcia-Perez (2021).

### Elemental analysis

The elemental analysis determined the C, hydrogen (H), N, and oxygen (O) composition of the biochar samples. The amounts of these elements in each biochar is available in Milan and Garcia-Perez (2021). Increasing the pyrolysis temperature from 400 °C to 600 °C increased the C content for all biochar, however, the hydrogen and oxygen mass fractions decreased due to the bond-breaking reactions that form volatile species, which escape with the increase of the temperature.

### Proximate analysis

The proximate analysis determined the moisture content, fixed carbon, volatiles, and ash content of the biochar samples. Complete results are shown in Milan and Garcia-Perez (2021). The ash content increased as production temperature increased, due to the accumulation of inorganic elements during the reduction of organic constituents (Enders et al., 2012). Fixed carbon also increased with increasing temperature due to the removal of volatile matter, leaving the more stable carbon in the biomass (Yang et al., 2020). Meanwhile, the volatile matter decreased with increasing temperature, because at higher temperatures more organic compounds from the biochar are released. The ash content is significantly higher in N-Mg doped char than in raw biochar, a consequence of the impregnation of Mg ions in the biochar.

### Gas physisorption analysis

This analysis used $CO_2$ to determine the biochar surface area and pore size. Biochar samples were analyzed by $CO_2$ adsorption to determine the porous structure of the biochar. The $CO_2$ adsorption isotherms and pore size for WS and DF, and the results of the specific surface area analysis and pore volume are available in Milan and Garcia-Perez (2021). The surface area and the pore volume of all biochar samples increased as pyrolysis temperature increased for the $CO_2$ adsorption, due to the removal of volatile compounds which allowed formation of micropores on the biochar surface.

### Fourier Transform Infrared Spectroscopy (FTIR)

A Fourier-transform infrared spectroscopy (FTIR) analysis identified functional groups on the surface and within the biochar samples to analyze the potential ability to adsorb various pollutants. The FTIR spectra are available in Milan and Garcia-Perez (2021), but the results suggested the biochar samples have many important O and N containing functional groups, some of which may play an important role on the removal of odorous compounds.

### pH analysis

The pH of the biochar samples (an important factor in the ability to adsorb pollutants) was determined, and is available in Milan and Garcia-Perez (2021). The pH of all biochar samples in water ranged from 6.73 to 11.54, and were higher for biochars produced at higher temperatures. Based on the pH values obtained, all the biochars were alkaline, except for the DF raw biochar (6.73), where the low ash content reduced the alkalinity.

## 3.4 Adsorption studies and results

Adsorption tests were performed in vertically oriented polycarbonate tubes, at atmospheric pressure and room temperature conditions. For additional information about the methods used, including calculation of the adsorption capacities, and a full suite of breakthrough curves, please see Milan and Garcia Perez (2021).

## $H_2S$ adsorption studies

Simulated biogas containing 2000 parts per million (ppm) of $H_2S$ was passed through 0.3 grams of biochar. The breakthrough curves for biochar samples adsorbing $H_2S$ are presented in Figure 4. The breakthrough time was defined as the time when the first non-zero $H_2S$ concentration was measured in the column exit. WS raw 600 showed the longest breakthrough time for $H_2S$ at 160 minutes. Some of the biochar characteristics that have an important role in the adsorption of $H_2S$ are the presence of ash, surface area, pH, pore size and surface chemistry (Ayiania et al., 2019).

None of the DF biochars performed as well as the best WS biochars, but among the DF biochars, DF N-doped 600 biochar had the best performance. The DF N-doped 600 biochar showed the highest surface area which indicates a greater number of adsorption sites and space are available for $H_2S$ adsorption; this factor influenced the adsorption capacity of $H_2S$ for this biochar.

The pH values for the biochars with higher breakthrough time ranged from 7.91 to 11.54. A pH in the basic range promotes the dissociation of $H_2S$ and has a positive influence on $H_2S$ adsorption (Bagreev, 2001). The moisture content of biochar also facilitates the dissociation of $H_2S$, which can be oxidized to sulfur and sulfur dioxide (Yan et al., 2002). The moisture fraction of the biochar samples varied from 0.94 to 4.8 % by weight.

The $H_2S$ adsorption capacity of the best performing biochar (WS raw 600) is 27.7 mg/g. This value is comparable with the $H_2S$ adsorption capacity of a biochar derived from anaerobic digestion fiber (21-51 mg $H_2S$/g char) (Ayiania et al., 2019). The emission of volatile sulfur compounds (VSCs; e.g., methyl disulfide, methyl sulfide, carbon disulfide, methyl mercaptan, and $H_2S$) in composting units has been reported to be close to 0.561 mg $H_2S$/g waste (Han et al., 2018). To remove such quantities of VSCs, a biochar with capacity to remove 27.7 mg $H_2S$/g is needed, and will be required in a ratio of 0.020 g biochar/g waste (or 2% biochar by weight).

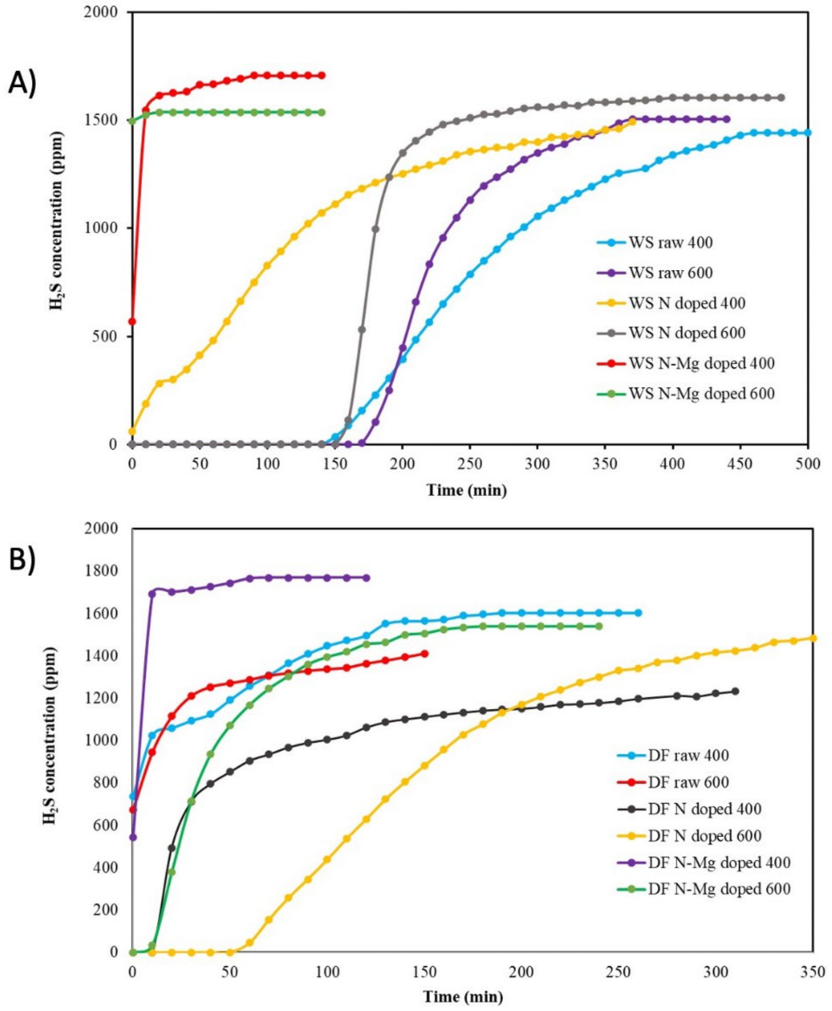

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Figure 4: A. Typical breakthrough curves of $H_2S$ adsorption on WS biochars, B. Typical breakthrough curves of $H_2S$ adsorption on DF biochars.

## $NH_3$ adsorption studies

Simulated biogas containing 15 ppm $NH_3$ was passed through 0.45 grams of biochar. The breakthrough curves for biochar samples adsorbing $NH_3$ are presented in Figure 5. WS N-Mg doped 400 biochar showed the longest breakthrough time at 190 minutes. The most important factor in the adsorption of $NH_3$ was the oxygen content, suggesting that the acidic functional groups present on biochar surface support the increase of adsorption of $NH_3$, due to the basic nature of this gas.

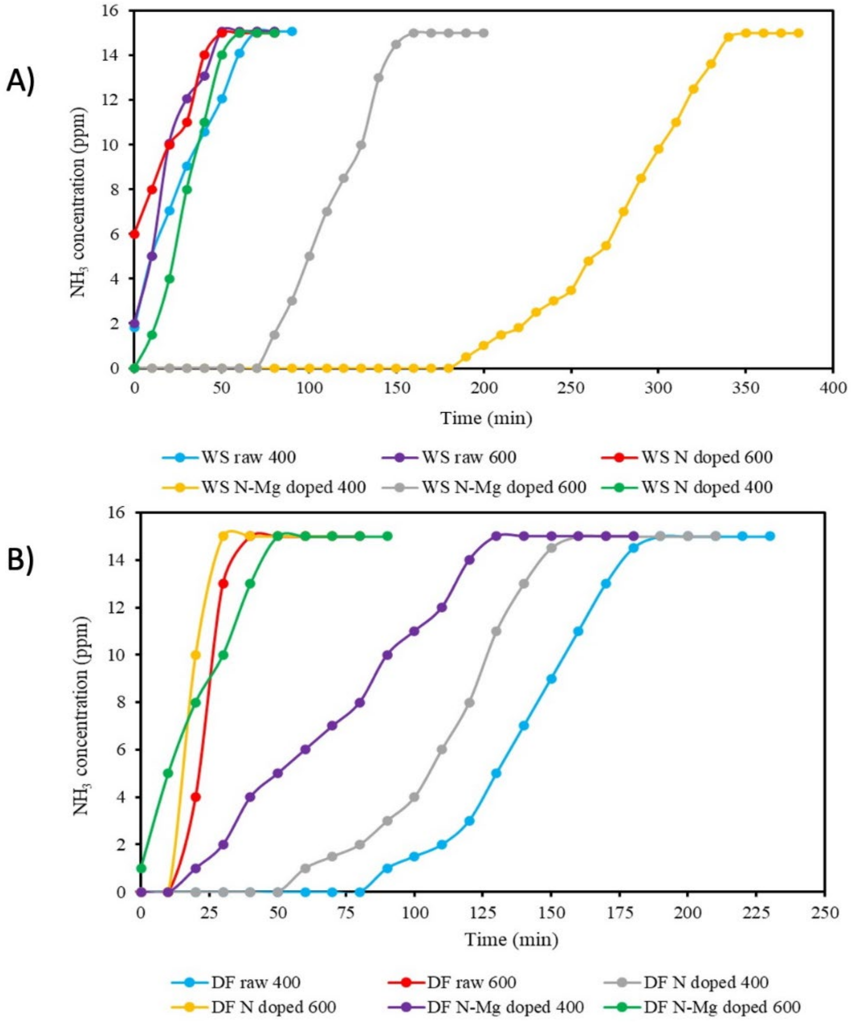

Figure 5: A. Typical breakthrough curves of $NH_3$ adsorption on WS biochars, B. Typical breakthrough curves of $NH_3$ adsorption on DF biochars.

The $NH_3$ adsorption capacity of the best biochar produced (DF raw 400) was 0.47 mg/g. This value is comparable with the $NH_3$ adsorption capacity of a non-activated biochar reported in the literature and is very low (0.15-5.09 mg $NH_3$/g biochar); in that study, activation of biochars with phosphoric acid greatly increased ammonia adsorption (24-53 mg $NH_3$/g biochar) (Ro et al., 2005).

The emission of $NH_3$ in composting facilities reported in the literature is between 0.018 and 1.150 mg/g of waste (Clemens and Cuhls, 2003; Cadena et al., 2009). This means that for a material releasing 0.35 mg $NH_3$/g of waste and a biochar with a capacity to remove 40 mg $NH_3$/g biochar (produced with phosphoric acid), all $NH_3$ released can be adsorbed by adding 0.00875 g biochar/g waste. In the case of a non-activated biochar with low adsorption capacity (0.47 mg $NH_3$/g biochar), 0.744 g of biochar per g of waste will be needed.

This amount of biochar is not practical to use in composting, and clearly shows the importance of continuing to develop engineered materials with a high capacity to adsorb $NH_3$.

## $CO_2$ adsorption studies

Simulated biogas containing 348,000 ppm $CO_2$ was passed through 5 grams of biochar. The breakthrough curves for biochar samples adsorbing $CO_2$ are presented in Figure 6. The adsorption study results show that all biochar samples adsorbed $CO_2$. The longest breakthrough time was WS N doped 600, at 120 minutes. The pH, ash and nitrogen content are the most influential factors in the $CO_2$ adsorption process. These factors contribute to making the biochar more alkaline, which better adsorbs $CO_2$.

The $CO_2$ adsorption capacity of the highest performing biochar (WS raw 400) was 0.49 mg/g. This value was very low compared with $CO_2$ adsorption of biochar reported in the literature: 57-176 mg $CO_2$/g biochar (Li and Xiao, 2019). The emission of $CO_2$ in composting facilities has been reported to be 150-370 mg/g waste (Komilis and Ham, 2006). This means that to remove 200 mg $CO_2$/g waste with a biochar that has the capacity to remove 57-176 mg $CO_2$/g biochar, more than 1g biochar/g waste will be needed.

Because of the high amount of $CO_2$ produced by composting, it is very unlikely to a biochar could be developed with sufficient adsorption capacity to remove all the $CO_2$ released in the facility at an economically viable rate. However, in the course of utilizing biochar to treat other emissions, adsorption of some $CO_2$ may lead incidentally to the capture of some portion of the $CO_2$ that would otherwise be lost, benefitting carbon sequestration.

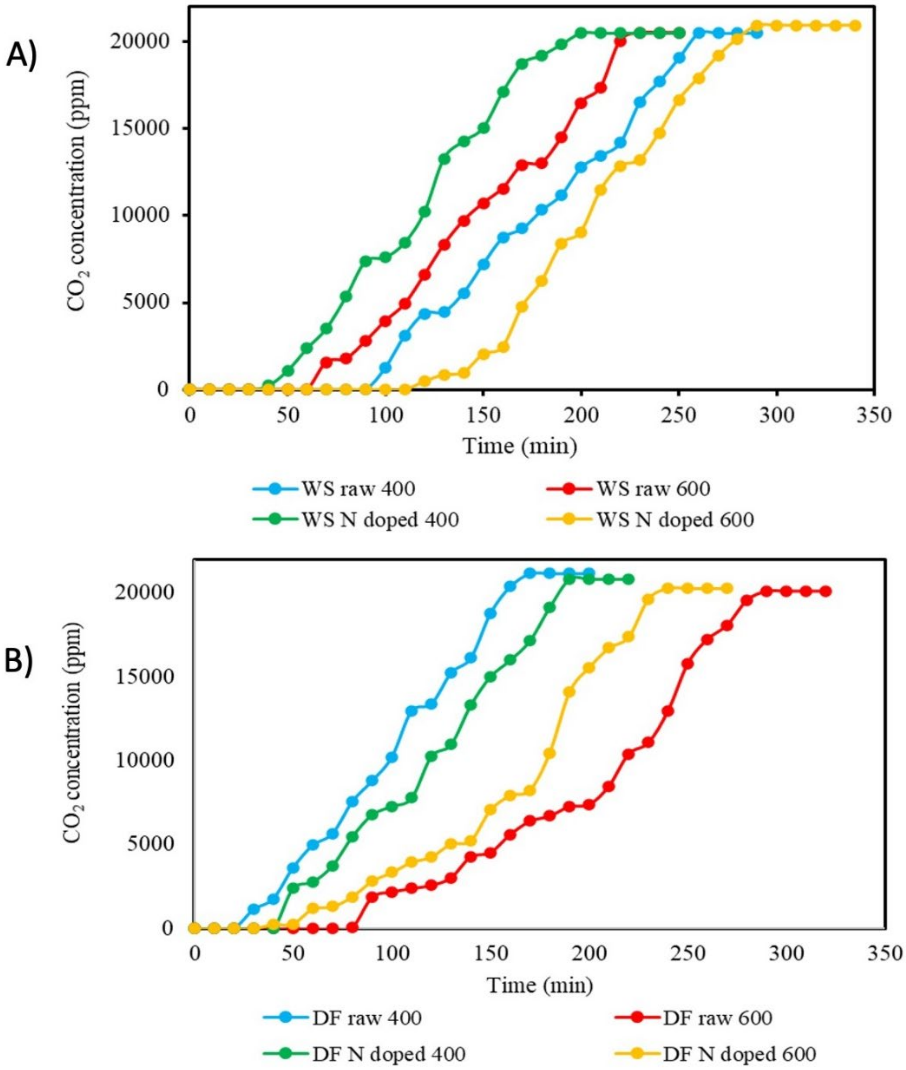

Figure 6: A. Typical breakthrough curves of $CO_2$ adsorption on WS biochars, B. Typical breakthrough curves of $CO_2$ adsorption on DF biochars.

## 3.5 Formulation of an example engineered biochar cocktail for odor emission/VOC removal in a hypothetical compost facility

Table 4 shows a range of estimated emissions factors from compost for five common compounds as described in the literature. It also shows a range of adsorption capacities of engineered biochars for those same contaminants, including those reported on in this study ($H_2S$, $NH_3$, $CO_2$), also as described in the literature. The amount of biochar needed to treat the emissions is also calculated. Based on the target compounds for which treatment is desired, a biochar cocktail (a blend of engineered biochar samples) could be developed to treat a suite of contaminants.

Table 4: Amount of biochar needed to treat each contaminant.

| Target Compound | Emission factor (mg/g waste) | References | Adsorption capacity[1] (mg/g biochar) | References | Biochar needed to treat contaminants (g char/g waste) |
|---|---|---|---|---|---|
| $H_2S$ | 0.561 | Han, et al., 2018 | 21-51 | Ayiania et al., 2019 | 0.011- 0.027 |
| $NH_3$ | 0.018 - 1.150 | Clemens & Cuhls, 2003<br>Cadena, et al., 2009 | 24-53 | Ro et al., 2005 | 0.00034-0.048 |
| $CO_2$ | 150-370 | Komilis & Ham, 2006 | 57-176 | Li & Xiao, 2019 | 0.85-6.5 |
| $CH_4$ | 0.05-0.49 | Amlinger et al., 2008 | 6.5 | Song et al, 2021 | 0.0076-0.075 |
| $N_2O$ | 0.074-1.57 | Zheng, et al., 2020 | 300 | Cha & Kong, 1995 | 0.00024 - 0.0052 |

[1] Literature values

The results shown in Table 4 suggest that for compounds other than carbon dioxide, there are strategies reported in the literature that could be used to produce biochars with capacities sufficiently high to justify their use in composting facilities at concentrations below 7% by weight. For several of the compounds described here ($H_2S$, $NH_3$ and $N_2O$), the amount needed is much less, roughly 2% or less by weight. However, in our laboratory experiments we were not able to obtain biochars with adsorption capacity sufficiently high to be economically used in composting facilities. Our results for $H_2S$ indicate an adsorption capacity in the range of those values from the literature, but our results for $NH_3$ and $CO_2$ were lower than those found in other studies. In this work we studied a number of standard activation strategies with two feedstocks, but observed poor adsorption results for $CO_2$ and $NH_3$. In the future we will need to explore other activation strategies reported in the literature, using feedstocks available in the Pacific Northwest.

The removal of $CO_2$ with biochar is unlikely to be economically viable due to the high quantities of $CO_2$ released during composting, which necessitates an impractically large amount of biochar for adsorption. Because the results found elsewhere (see Table 4) indicate potential higher adsorption capability, a biochar cocktail is still a promising concept. However, more research is needed on feedstocks available in Washington State to produce chars with high adsorption capacities.

## 3.6 Conclusions

The results of this project show that biochar produced from the same feedstock, pyrolyzed at different temperatures (400 $^{o}$C and 600 $^{o}$C) and either raw, N doped, or Mg-N doped, have different capacities to adsorb $H_2S$, $NH_3$, and $CO_2$.

- The thermogravimetric analysis results show a significant amount of ash in WS, an important property which contributes to high levels of $H_2S$ retention as it helps to increase the pH of the biochar.

- Surface area is another important metric which enhances gas adsorption. The biochar produced in this project are mostly dominated by micropores. Biochar produced at 600 $^{0}$C showed a higher surface area compared to those produced at 400 $^{0}$C.
- Nitrogen content has a significant influence on the removal of $H_2S$ and $CO_2$. Biochar pyrolized at 600 $^{0}$C has more nitrogen functional groups, which makes the biochar more alkaline, contributing to the adsorption of acidic pollutants such as $CO_2$.
- The FTIR analysis showed the presence of oxygen functional groups on the majority of the biochar samples. This element (oxygen) is the main factor supporting $NH_3$ adsorption.
- An X-ray photoelectron spectroscopy (XPS) analysis is recommended to determine the elemental composition of the biochar surface with greater certainty.

The adsorption capacities obtained in this project are still below those reported in the literature. The results suggests that although biochars with adsorption capacity sufficiently high to be economically used in composting facilities were not produced, there are strategies reported in the literature to produce biochar with sufficiently high capacity to justify their use in composting facilities at concentrations below 10% by weight. These strategies need to be further explored with feedstocks available in Washington State to produce biochar with high adsorption capacities.

This is a complex problem that warrants further research. While this project examined engineered biochar cocktails to address three common pollutants ($H_2S$, $NH_3$, $CO_2$), there is a wide range of emissions, including VOCs, that will require further research and development of targeted biochar cocktails. This research demonstrates the efficacy of biochar as a means to address noxious gases and illustrates the potential for engineered biochar cocktails.

Additional detail is available in the technical report *Development of Engineered Biochar Cocktails for Odor Removal in Composting Facilities* on the WTFT 2019-2021 webpage of Washington State University's CSANR.

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# Chapter 4: Impact of High Rates of Biochar on the Composting Process and Resulting Products

*Nathan Stacey, Douglas Collins, Andy Bary, Elizabeth A. Mhyre, Steven Seefeldt*

## 4.1 Introduction

Composting is an aerobic process that transforms organic waste via decomposition into stable organic matter, which can be used as a nutrient source and soil conditioner: a valuable downstream product for use in agriculture or other settings. Much research has focused on understanding the complex chemical and physical changes that occur within each stage (beginning, middle, and end), so that the compost can be optimized for agricultural use (Nafez et al., 2015; Oviedo-Ocaña et al., 2019). For example, it is well understood that properties like the carbon to nitrogen (C:N) ratios of the feedstock influence the composting process and final product.

Biochar is the solid, carbon-rich by-product of thermochemical conversion and results when biomass (such as forestry or agricultural by-products like Douglas fir or wheat straw) is heated at high temperatures in oxygen starved environments. Because of the high carbon content of biochar, and its porous structure with large surface area, biochar is an appealing product for use in industrial and agricultural contexts because it readily ad- and absorbs chemical compounds (Aller, 2006; Zheng et al., 2013; Xiang et al., 2020). It also shows promise as a means for carbon sequestration (Smith, 2016).

Biochar is not a panacea, however, and can be difficult to work with because of its high variability. Many factors can influence the composting process, the end result, and how the compost interacts with the environment (via emissions or leaching): Different feedstocks (e.g., hardwood vs. herbaceous biomass) and production temperatures alter the physical and chemical characteristics of the biochar in functionally important ways that affect how well it works as a nutrient source (Wang et al., 2015; Tomczyk et al., 2020). Different types of biochar, including those sourced from crop residues, animal manures (e.g., chicken manure), wood, and bamboo have been tested at different incorporation times (i.e., beginning vs. end of composting) and rates (Zhang et al., 2016; Vandecasteele et al., 2016; Agyarko-Mintah et al., 2017; Liu et al., 2017b).

Biochar incorporation causes physical changes in the compost pile, like lower bulk density. It also causes chemical changes (including loss of nitrogen [N]). These changes and potential improvements that result from biochar incorporation are neither uniform nor ubiquitous.

Based on work in the previous biennium (see Stacey et al. 2021) we wanted to assess incorporation of biochar at much higher rates (20% and 40% by volume) than previously used (5% and 10% by volume). The work covered in this Chapter is related to our previous work and helps us make better recommendations for producers experimenting with varying rates of biochar incorporation. To do so, we designed two experiments:

1. A compost experiment, which includes compost preparation and mixing, sampling, and a maturity bioassay, to evaluate the use of biochar in the composting process.

2. A field experiment to assess the resulting co-compost products as soil amendments in potato production.

We used a regionally produced biochar made from forestry residuals (Douglas fir [*Pseudotsuga menziesii*] and pine [*Pinus* spp.]), similar to what a local compost operator might purchase. The biochar was purchased from Oregon Biochar Solutions (White City, OR). (For select physical and chemical properties for the forestry residual biochar product, see Stacey et al. 2021.) We tested three compost mixtures: an unamended control compost (made of chicken manure and wood chips) and two different biochar incorporation rates - moderate (20% by volume) and high (40% by volume) to better understand the effects biochar has on the composting process and the resulting product, and to evaluate the downstream agronomic benefits of these products.

## 4.2 Compost preparation and analysis

To create the biochar amended co-composts, we first developed an optimal base compost mixture which was made from locally procured chicken manure and wood shaving feedstocks. The compost feedstock was composed of a ratio of 4:1 wood shavings to chicken manure (by volume), resulting in a C:N ratio of 25:1 with 34.6% free air space. This was utilized as the control treatment in the experiment. To evaluate the composting process, each of three treatments (control compost, biochar 20%, and biochar 40%) were composted in triplicate in nine, 1.43 $m^3$ vessels, simultaneously, for 34 days. See Stacey et al. (2021) for greater detail. Compost was supplemented with forced air, and temperature data was collected throughout the 34 day duration of compost production. Samples of material were taken at days 1, 14, and 34.

A compost maturity bioassay (indicating stable organic matter and a lack of phytotoxic compounds or plant and animal pathogens) using a cucumber seedling growth and emergence test, indicated that all composts had reached maturity by day 34.

The compost samples were analyzed for bulk density and percent moisture, temperature, and electroconductivity (EC), pH, total carbon (C) and N, ammonium nitrogen ($NH_4$-N), and nitrate nitrogen ($NO_3$-N).

### Compost bulk density and moisture content

As biochar incorporation rates increased from 0 to 20% to 40%, bulk density values decreased slightly, and moisture content increased slightly. However, changes in bulk density and moisture content average values were small, and significant differences were only observed in bulk density pre-samples (Day 1) and moisture content end-samples (Day 34).

Similar results for bulk density have been reported in laboratory-scale compost reactors (Ravindran et al., 2019). Our results likely reflect the physical properties of biochar, such as its high surface area and porosity. The inherently low bulk density of the biochar likely diluted the original material, causing a reduction in bulk density, and this effect was diminished as the volume of the compost shrunk due to chemical transformations. It is less clear why the moisture content is affected. It may be that pores in the biochar are filled with water, or it could be that water is held tightly between the surface of the biochar and other materials in the compost. Likely, the effect is a combination of the two.

## Compost temperature

All composts reached 55 °C for a minimum of three days which meets the compost industry standard to degrade phytotoxic compounds and reduce pathogens. Composts with 20% biochar incorporation showed no significant differences from the control, and composts with 40% biochar incorporation reached 55 °C nine hours faster than the control, but also lost heat at a greater rate after reaching peak temperature (Figure 7). This may be due to increased microbial activity, which then exhausts nutrient stores, or could be a result of increased aeration and therefore heat exchange from incorporation of biochar.

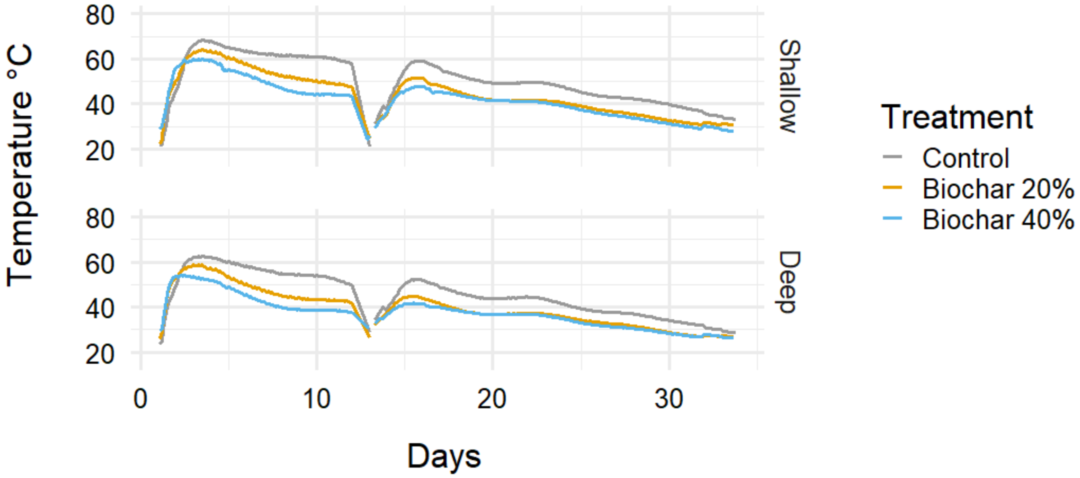

Figure 7: Temperature profiles for each of three treatments over the 34-day composting trial. The break in temperature at day 13 indicates when the compost was turned and each treatment line (colored grey, orange, and blue) represents the average of three replications.

If the differences in temperature profiles among treatments are transferable to a larger industrial scale, this may have implications for compost producers considering this strategy for several reasons. In order to maintain a suitable C:N ratio, incorporating biochar at higher rates may require more N, proportionally, than what was used here. Additionally, if the heat loss continues to occur more quickly at higher rates of biochar incorporation, at some point the compost process could be negatively impacted. On the other hand, if the rate of temperature increase was consistent (when incorporating biochar), this may reduce the time required to reach a mature compost product. Differences in scale can have considerable effect on the physical and chemical properties of a compost and thus, extrapolations from these data should be made with this in mind.

## Compost chemical parameters

Table 5 lists average values for various chemical properties evaluated before (Pre) and after (End) composting. At the end of composting, significant differences were observed in mean values for total C, C:N, EC, and $NO_3$-N.

Table 5: Mean values for chemical properties in each of three treatments, control (C), biochar 20% (B20), and biochar 40% (B40) at two collection times, before (Pre, Day 1) and at the end (End, Day 34) of composting.

| | Pre | | | End | | |
|---|---|---|---|---|---|---|
| **Property** | C | B20 | B40 | C | B20 | B40 |
| Total C (%)* | 35.9a** | 39.4ab | 45.6b | 30.2c | 38.0d | 42.3d |
| Total N (%) | 1.56 | 1.56 | 1.54 | 1.13 | 1.17 | 1.22 |
| C:N | 23.0 | 25.6 | 29.3 | 26.8a | 32.6ab | 34.8b |
| pH | 8.1 | 8.8 | 8.8 | 7.9 | 7.9 | 7.6 |
| EC (dS $m^{-1}$) | 10.9 | 9.29 | 8.0 | 7.7a | 6.8b | 6.6b |
| $NO_3$-N (mg $kg^{-1}$) | 9.0 | 8.3 | 9.0 | 1328a | 1524a | 1955b |
| $NH_4$-N (mg $kg^{-1}$) | 6014 | 4848 | 4674 | 268 | 29 | 41 |

*Percent total C and N are reported on a dry weight basis.

**Mean values within a collection period (Pre and End) and variable row (e.g., Total C) followed by different letters are significantly different according to Tukey's HSD test ($P<0.05$).

## C, N and C:N

The nutrient content of compost helps define its economic and agronomic value. To assess whether these rates of biochar incorporation influenced nutrient content, we calculated the nutrient loss or gain, comparing the nutrients before and after composting.

In pre-compost samples, the only significant difference was in total C mean values observed between the control and biochar 40% treatment (Table 5). This likely reflects the additional C that was added as biochar, which is resistant to degradation. These increases may reflect biochar's ability to ad- and absorb soluble compounds like dissolved organic C, which could alter C cycling and the resulting losses of carbon dioxide ($CO_2$). C:N ratios in pre-compost samples were close to significant and are indicative of the differences in feedstock proportions.

Preliminary data from nutrient content analysis suggests that biochar incorporation does reduce N loss, but only at the highest rate of incorporation. When biochar N is accounted for in the nutrient analysis, however, the potential reduction in N loss becomes less significant. This indicates that biochar N retention in these composts is minimal.

## Electroconductivity, and $NO_3$-N and $NH_4$-N

Electroconductivity (EC) is a measure of soluble salt content which at elevated levels can injure plants. EC values in composts generally decline over time. We measured additional declines in EC values that followed the increased biochar incorporation rates, though this may reflect the dilutive effect of the biochar material. Interestingly, in the biochar 40% treatment, we observed high levels of $NO_3$-N, a soluble salt, and low EC values. The cause of this inverse relationship, however, is unknown.

$NO_3$-N and $NH_4$-N are forms of plant available N and are immediately available in the soil environment. The increases in $NO_3$-N, at the highest rates of biochar incorporation in comparison with the control could reflect processes of ad- and absorption (processes that may affect gaseous or leachate N loss) or it could be the result of increased microbial activity. Even with the increased $NO_3$-N concentrations, the highest rates of biochar used here are likely cost prohibitive and would not improve the soil amendment value of this compost.

## 4.3 Potato field trial

### Experimental design

Potato plots were established at the Washington State University Mount Vernon Research and Extension Center in Mount Vernon, Washington. Prior to amendment, research plots were fertilized with standard amounts of phosphate (11-52-0, 163 kg $ha^{-1}$), langbeinite (0-0-22, 172.6 kg $ha^{-1}$), muriate of potash (0-0-62, 168.1 kg $ha^{-1}$), ammonium sulfate (20-0-0, 103.1 kg $ha^{-1}$), and urea (46-0-0, 196.1 kg $ha^{-1}$).

Then, in a randomized and replicated complete block design, we amended soils (15 June 2020) with seven treatments: an unamended control, a control compost at high and low rates, biochar 20% compost at high and low rates, and biochar 40% compost at high and low rates (Table 6). Treatments were applied by hand to meet target rates of 15 and 7 Mg dry C $ha^{-1}$ (high and low, respectively).

Table 6: Amendment rates for the potato field trial.

| | Unamended Control | Control Compost | | Biochar 20% | | Biochar 40% | |
|---|---|---|---|---|---|---|---|
| Rate | 0 | High | Low | High | Low | High | Low |
| Wet Weight (t $ha^{-1}$) US tons | n/a | 51.34 | 24.05 | 44.02 | 25.85 | 43.06 | 20.07 |

Following amendment, on 18 June 2020, plots were tilled to 15 cm, planted with cut potato (*Solanum turberosum* L. var. *Chieftain*) and hilled 21 days later (9 July 2020). One hundred and thirty-two days later (28 October 2020), from the center row of each treatment, three plants were harvested for above- and below-ground biomass. Leaves and tubers from these plants were collected, counted, and weighed. Two bulk density cores and an additional 10-12 soil cores were collected, homogenized, and dried for later analysis.

The potato field trial was conducted over one growing season and therefore extrapolation from this limited data set should be conducted with caution.

### Results

Soil bulk density and total N mean values were significantly affected by amendment, but only in comparisons with the control compost and 40% amendment high rates, respectively (Table 7). Bulk density values decreased while total N concentrations increased; all other variables were unaffected by the various compost products at low and high rates (Table 7).

Table 7: Soil physical and chemical properties following compost amendments at low and high rates and one growing season.

| | Control* | Control Compost | | B20 | | B40 | |
|---|---|---|---|---|---|---|---|
| Property | | Low | High | Low | High | Low | High |
| Bulk Density (Mg $m^{-3}$) | 1.91a** | 1.83ab | 1.79b | 1.86ab | 1.84ab | 1.85ab | 1.82ab |
| Total $C^{\dagger}$ (%) | 1.07 | 1.13 | 1.45 | 1.17 | 1.26 | 1.28 | 1.23 |
| Total N (%) | 0.137a | 0.145ab | 0.169ab | 0.156ab | 0.168ab | 0.163ab | 0.174b |
| $NO_3$-N (mg $kg^{-1}$) | 230 | 246 | 273 | 300 | 312 | 376 | 369 |
| $NH_4$-N (mg $kg^{-1}$) | 19.2 | 16.5 | 21.8 | 22.5 | 19.6 | 17.9 | 19.6 |

*Field amendments are as follows: Control is the unamended control, Control compost (compost without biochar) at low and high rates; B20 is the biochar amended compost 20% at low and high rates; B40 is the biochar amended compost 40% at low and high rates.

**Means within a variable row (e.g., Bulk Density) followed by different letters are significantly different according to Tukey's HSD test ($P<0.05$).

†Total C and N values are reported as dry weight.

Soil bulk density can be altered by the rate and physical properties of an amendment material. The control compost material was amended at the highest rate of application in our field experiment so the decrease in soil bulk density is expected. It is interesting, however, that no other material decreased soil bulk density. In potato production, the soil is initially tilled, planted, and re-disturbed by creating soil hills over the potato plant. This second soil disruption may account for the unchanged soil bulk density.

Soil total N represents both organic and inorganic forms of N. Though a clear trend was obvious where increasing $NO_3$-N values followed the increase of biochar incorporation and amendment rates, in comparisons between treatment, no differences were detected. This means that the significant increase in total N (B40 high treatment > control) is likely primarily due to an increase in organic N. Our plots were amended with supplemental N (from synthetic fertilizer) which removes N limitations that may restrict soil microbial growth and reproduction. The observed increase in total N, and thus organic N, may reflect an increase in soil microbial activity, but because we did not evaluate microbial properties, it is unknown whether microbial populations were affected by the B40 high treatment.

Comparable results were observed in collections of potato plant biomass as tuber weight significantly increased in plots amended with the B40 high treatment (2.11 kg $ha^{-1}$), but this was observed only in comparisons with the control (1.49 kg $ha^{-1}$). However, there was no significant difference in leaf weight and tuber number in comparisons between treatments.

In potato production, composts are typically used as soil conditioners, not as the primary plant nutrient source. Therefore, the limited responses we observed in potato soil and plant biomass following compost amendment are not unexpected. Additionally, the fertilizer application made prior to compost amendment likely masked potential soil and plant responses. Because compost products supply N at much slower rates than quick release synthetic fertilizers, a second season of growth in these plots without fertilizer application could potentially reveal additional treatment differences.

## 4.4 Conclusion

Woody waste biochar incorporation at 20% and 40% (by volume) into chicken manure and wood chip compost impacted compost nutrient status, moisture content and temperature profiles, but only minimally so. The linear increase in biochar rate did not result in consistent and significant observations in the response variables we measured. Reductions in nitrogen (N) loss were observed in the 40% biochar amended composts and two things likely contributed to these measurements: differences in starting N, and N retention by biochar, though the latter was likely minimal. This N retention, in fact, did not result in drastic differences in potato soil and plant biomass, except in comparisons with the unamended control.

For the composting experiment, all composts reached 55 °C for a minimum of three days which meets the compost industry standard for pathogen reduction. Composts with 20% biochar incorporation showed no significant differences from the control in terms of when they met this temperature requirement. Composts with 40% biochar incorporation reached 55 °C nine hours faster than the control, but also lost heat at a greater rate, and had 6% greater moisture content in comparisons with the control. The compost with 40% biochar also had the highest concentrations of $NO_3$-N, indicating some impact on the composting N cycle.

For the potato field experiment the control compost (amended at the highest rate) reduced soil bulk density and increased soil total N. Potato tuber yield was increased in comparisons between the control and the high rate of compost with 40% biochar. Additionally, this experiment looked at one year of growth; a second season of growth would likely reveal more differences between treatments, particularly to understand the longevity and performance of N that may become slowly available in soils following biochar co-compost amendment.

In summary, biochar incorporation did impact the composting process in terms of both physical and chemical responses. The greatest impacts were measured when biochar was incorporated into the chicken manure and wood chip compost at the highest rate, 40%.

Compost producers considering this type of compost strategy need to carefully evaluate the feedstock and production process for a chosen biochar so that their expectations align with the most likely outcomes. Similarly, future work should also carefully consider the biochar feedstock and production temperature with additional experiments that evaluate the potential for greater reductions in compost N loss.

Additional detail is available in the technical report *Impact of High Rates of Biochar on the Composting Process and Resulting Products* on the WTFT 2019-2021 webpage of Washington State University's CSANR.

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# Chapter 5: Integrating Compost and Biochar for Improved Air Quality, Crop Yield, and Soil Health

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## 5.1 Introduction

Production of compost often causes odor and greenhouse gas emissions. Application of biochar, defined as "a solid material obtained from thermochemical conversion of biomass in an oxygen-limited environment" by the International Biochar Initiative (Agegnehu et al., 2017) to reduce gas emission during and after the composting process is a promising efficient low-cost solution to this problem (Godlewska et al., 2017; Sanchez-Monedero et al., 2018). Furthermore, numerous investigations have been and are being conducted to evaluate the potential of biochar application for the improvement of soil quality and crop performance (Agegnehu et al., 2017). Our previous studies (Gang et al. 2018; 2019) found beneficial effects of co-composting biochar in terms of reducing volatile organic compound (VOC) emissions during composting and increasing biomass accumulation of a specialty crop, sweet basil. This biennium's work built on the results from the past two biennia by reproducing and expanding them. To extend the knowledge of biochar's impact on compost production and quality, biochar from a single source was co-composted in 2018 at two different facilities, Lenz Enterprises Inc. (Stanwood, Washington) and the Washington State University (WSU) Compost Facility, (Pullman, Washington) and gas emissions were sampled. Emissions were also sampled from laboratory co-composting experiments under more controlled conditions. In this biennium, we added a new compost made in 2019 at the Puyallup Research and Extension Center in Puyallup, Washington. The different co-composts, along with the composts, and biochar were used to amend soils for greenhouse and field trials (see Table 8 for more information). Here we add to results reported on for the 2017-2019 biennium (Gang et al., 2019) to provide a comprehensive summary of the full body of work. Additional detail including biochar and compost characteristics is provided in Gang et al. (2021).

The aim of this project was threefold:

1) To provide measured data through field and laboratory tests in order to identify emitted VOCs and odorants and to quantify their emission fluxes from composting processes, as well as identify the effects of biochar addition on these emissions;

2) To evaluate the effect of compost, biochar, co-compost, and compost plus biochar (not co-composted) amended to soil on crop production and quality in greenhouse (basil and strawberry) and field (basil, strawberry, potato) settings; and

3) To examine the effect of these soil amendments on soil physicochemical properties.

Table 8: Summary of biochar and compost sources used in the experiments described in this chapter.

| Experiment | Dates(s) of trial | Biochar source | Compost source | Measurements |
|---|---|---|---|---|
| Emissions | Feb-Mar 2018 (Lenz)<br>Jun-Jul 2018 (WSU) | OBS | WSU & Lenz (on-site) | Air emissions |
| Basil – field (Colbert) | Aug 2018 | Amaron Energy | WSU 2017 Footehills | fresh plant mass<br>phytochemical composition |
| Basil – greenhouse | Feb 2019 (both)<br>Oct 2019 (Eleanora)<br>Apr 2021 (TSQ) | Amaron Energy<br>OBS | WSU 2017<br>WSU 2018<br>Lenz | fresh plant mass<br>phytochemical composition |
| Strawberry – greenhouse | Aug-Oct 2018<br>Sep-Dec 2020 | Amaron Energy<br>OBS | WSU 2017<br>WSU 2018<br>Puyallup 2019<br>Lenz | yield<br>berry number<br>single berry mass<br>Trial 2020: phytochemical composition |
| Strawberry – field (Puyallup) | July-Oct 2018<br>May-Sep 2019 | OBS | Lenz | yield<br>soil properties |
| Potato – field (Mount Vernon) | June-Sep 2018<br>May-Sep 2019 | OBS | Lenz | yield<br>soil properties |

## 5.2 Effect of biochar on gas emissions during composting

In the first part of this project, the potential of biochar to reduce emission fluxes of greenhouse gases and odorant VOCs from composts was evaluated.

Two field samplings were conducted using the flux isolation chamber method at Lenz (February-March, 2018) and at WSU (June-July, 2018). Two large (200 yd$^3$) aerated static piles were sampled at Lenz (a control pile and a pile with 5% biochar by volume) and air samples were collected on days 3, 7, 11, 20, and 30. At the WSU compost facilities, twelve static piles (10 yd$^3$) were constructed; three of the piles contained biochar mixed at 2.5%, three at 5%, and three at 10% by volume as well as three control piles (no biochar). Air sampling was scheduled on days 3, 7, 11, 16, 22 and 31. For both field tests, canisters filled with sampled air were sent to the laboratory for gas chromatography-mass spectrometer (GC-MS) analysis (Figure 8).

Figure 8: Photograph of flux isolation chamber at top of a 5% biochar pile.

In addition to the field experiments, a laboratory-scale setup was used for continuous measurement of VOCs and trace gases during composting in two 100 gallons tanks: one filled with compost amended with 10% biochar by volume and the other was a control tank (no biochar). Laboratory experiments were repeated twice (March to April 2019) and utilized proton-transfer reaction mass spectrometry (PTR-MS) to continuously measure VOCs (Figure 9). In the first trial the material contained no food or green waste and was mostly manure, whereas the second trial contained a mixture of food waste, green waste, and manure.

From the measured VOC concentrations in the samples a VOC emission rate given by a flux density ($\mu g \cdot m^{-2} \cdot hr^{-1}$) was calculated. Analysis of the field samples from Lenz and WSU revealed that emitted monoterpene compounds, principally α-pinene and limonene, were a large faction of the total VOC emissions flux density. The WSU compost also had significant emissions of sulfur-containing compounds, principally dimethyl disulfide (DMDS) and dimethyl sulfide (DMS), which in some samples were larger than the monoterpene emission rate. High variability in VOC emission rates among the control and biochar samples were observed for both Lenz and WSU piles, and this made it difficult to discern the impact of biochar. Factors contributing to the high variability may have included the forced air flow through the Lenz piles, non-homogenous surface emissions, and problems with sampling the high humidity air inside the flux chamber.

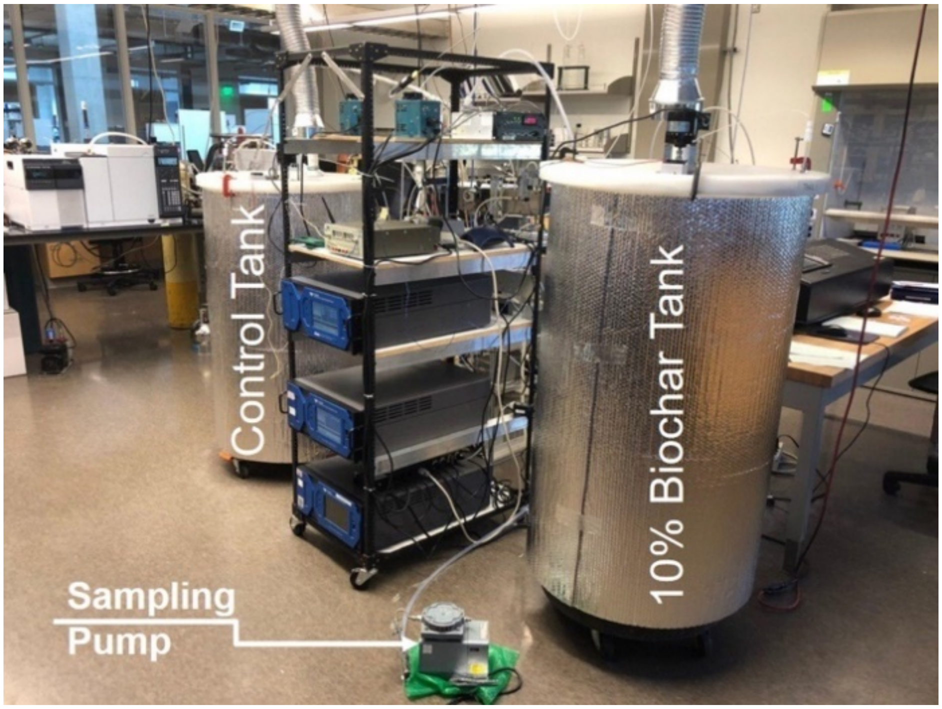

Figure 9: Photograph of compost lab test setup.

The WSU piles displayed differences in the composting process, indicated by considerable variation in temperature between piles of similar type (i.e., control piles). Factors such as ambient atmospheric conditions (wind, air temperature), pile shape and position, placement of the monitoring dome, and the granular nature of compost pile composition may have also contributed to pile variation in VOC emission rates. Despite best efforts to mix the compost uniformly, the nature of the feedstock is such that differing microenvironments within piles cannot be avoided, and dome placement can therefore tap into different microenvironments from pile to pile.

The laboratory-based experiments, performed under more controlled conditions, showed that 10% biochar was effective in reducing emissions of monoterpenes, DMDS, and several other compounds that have not yet been identified (PTR-MS ion signals at *m/z* 69, 83, and 135) (Table 9). Biochar was not shown to be as effective at reducing emissions of hydrogen sulfide ($H_2S$) and DMS. The reduction of monoterpene and DMDS emissions should help reduce compost odor but this has yet to be quantified by actual odor measurements. The second trial had much larger emissions of ethanol, methanol, and acetone, presumably because the starting materials contained food waste and green waste. Ethanol emissions were significantly lower for the biochar tank in the second trial. In addition, emission rates of the greenhouse gases methane and nitrous oxide were measured in the first trial. Emission of methane from the biochar-co-composted tank was substantially lower than the control tank, while nitrous oxide emission was only slightly lower. This experiment indicates that biochar might also be effective in reducing greenhouse gas emissions, an interesting and important co-benefit to reducing emission of VOC odor compounds.

Table 9: Summary of VOC emissions from compost for both trials in control and biochar co-composted tanks

| | | 1st Trial | | 2nd Trial | |
|---|---|---|---|---|---|
| **Compound** | **Sampled Tank** | **Emitted mass (ug)** | **Flux reduction in Biochar Tank (%)** | **Emitted mass (ug)** | **Flux reduction in Biochar Tank (%)[1]** |
| **Ammonia** | **Biochar** | **521,040** | **24** | **244,252** | **-78** |
| | **Control** | **681,691** | | **137,492** | |
| **Monoterpene** | **Biochar** | **3,110** | **74** | **516,925** | **46** |
| | **Control** | **11,779** | | **965,948** | |
| **m/z 69** | **Biochar** | **1,249** | **60** | **7,400** | **38** |
| | **Control** | **3,151** | | **11,872** | |
| **m/z 83** | **Biochar** | **243** | **63** | **1,720** | **74** |
| | **Control** | **653** | | **6,589** | |
| **Ethanol** | **Biochar** | **27,443** | **-8** | **162,518** | **48** |
| | **Control** | **25,416** | | **312,779** | |
| **DMS** | **Biochar** | **17,324** | **7** | **40,203** | **17** |
| | **Control** | **18,609** | | **48,385** | |
| **DMDS[2]** | **Biochar** | **215** | **29** | **2,320** | **60** |
| | **Control** | **302** | | **5,842** | |
| **$H_2S$** | **Biochar** | **322** | **18** | **507** | **15** |
| | **Control** | **391** | | **597** | |
| **Methanol** | **Biochar** | **16,649** | **-1** | **137,007** | **17** |
| | **Control** | **16,492** | | **164,722** | |
| **Acetone** | **Biochar** | **6,980** | **-2** | **35,317** | **23** |
| | **Control** | **6,871** | | **45,675** | |
| **m/z 135** | **Biochar** | **87** | **66** | **728** | **83** |
| | **Control** | **255** | | **4,334** | |

[1] negative values indicate emissions increased

[2] dimethyl disulfide

Continuous measurements of VOCs using PTR-MS clarified that for some compounds large emissions occurred during the first days of composting. Thus, starting measurements on day 3, as was done during field sampling, misses large emission rates of ammonia, alcohols, and monoterpenes on the first few days. To quantify how this affects total emitted fluxes, emission profiles of some compounds were estimated assuming discrete sampling started at day 3 and day 3 fluxes applied to days 1 and 2. Estimated profiles were compared with those obtained by continuous measurement for trial 2. This analysis suggests that the discrete sampling approach starting on day 3 underestimated total flux values over the first twelve days for monoterpenes by 47%, methanol by 81%, ethanol by 90%, acetone by 60%, and ammonia by 40% (Figure 10). This is a significant error in determining VOC emission rates and suggests that sampling early and often in the first week is required to determine accurate VOC emission factors for composting.

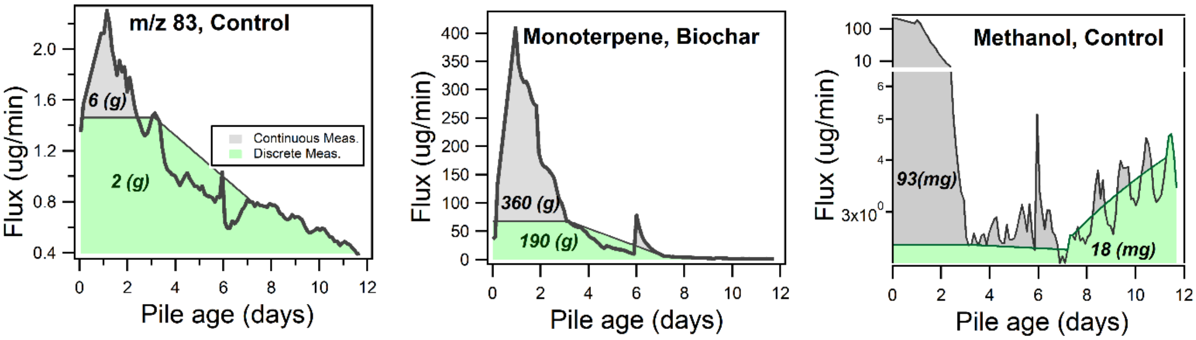

Figure 10: Comparison of emission profiles of m/z 83, monoterpene, and methanol estimated by continuous measurement and discrete measurements. Black trace shows measured fluxes. Area under this curve is the mass emitted. Green shading illustrates the area of the flux profile if discrete sampling at day 3, 7, and 11 was done to determine fluxes. The grey shading shows the amount of mass that is under reported by discrete sampling.

## 5.3 Effect of biochar on crop productivity and quality

In the second part of the project, the Lenz compost and co-compost, as well as biochar and compost plus biochar (not co-composted) were evaluated in field trials with potatoes, strawberries, and sweet basil and in greenhouse trials with sweet basil and strawberries. In field trials (sweet basil) and greenhouse trials (strawberries), composts evaluated (alone and as part of co-compost and compost plus biochar amendments) included WSU 2017, WSU 2018, Puyallup 2019, and Lenz composts. Biochar and compost sources are summarized in Table 8. Note that this biennium's work builds on work from the last biennium (2017-2019), allowing for a total of three years of data collection for most trials, which captured more variability due to weather conditions and other factors.

The sweet basil field trial was conducted on an organic farm in Colbert, Washington (Spokane County). Compost-biochar mixtures amended to soil suggested growth benefits for the plants supplemented with co-compost (Figure 11). The amendments tested had no significant qualitative and quantitative effects on the main antioxidative phenolic and aroma compound production in that field trial (data shown in Gang et al., 2021).

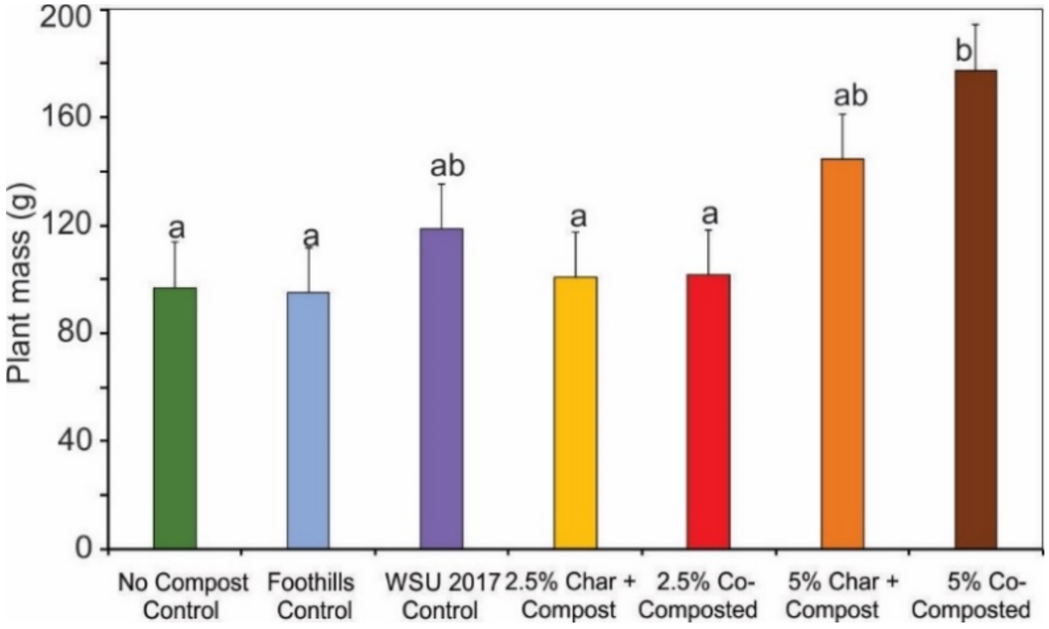

Figure 11: Effect of amendments on fresh mass of Genovese sweet basil grown at Footehills Farm (Colbert, Washington). Average plant mass ± standard error of the mean (n=10) are shown. Same lower case letters indicate there was no significant difference between treatments ($p<0.05$). Note: "char + compost" indicates char added post-composting, while "co-composted" indicates char added prior to composting.

Greenhouse studies with two basil cultivars (Thai Siam Queen, Eleanora) and three biochar-amended composts produced different results for the basil cultivars, but overall showed only moderate increases in biomass production (Gang et al., 2021). Taking into account faster growth, plants were harvested earlier than the previous trials, and new compost mixtures were used due to depletion of the WSU 2017 compost. In the most recent trials, significantly higher biomass was produced by the cohorts treated with WSU 2018 compost amended with 5% biochar, both before and after composting process. Furthermore, WSU 2018 mixed with 10% biochar post-composting also yielded higher average biomass. Lenz compost mixed with 5% biochar post-composting resulted in significantly lower biomass accumulation. These results suggested that growth of basil was likely expedited by addition of co-compost, potentially benefiting the fresh cut market, but that seed production would not be impacted, as the mature plants at seed set were essentially indistinguishable regardless of treatment.

Two seasons of field trials involving strawberry and potatoes were conducted at Washington State University Research and Extension Centers in Puyallup and Mount Vernon, Washington, in the 2018 and 2019 growing seasons. In these trials, added nitrogen was also included as subplots in a split-plot design. Neither amendment nor fertilizer affected strawberry yield during the first establishment year (Gang et al., 2021). Potato yield was strongly affected by nitrogen addition. For potatoes, in fertilized plots, co-compost significantly increased yield compared to the no amendment control (Figure 12).

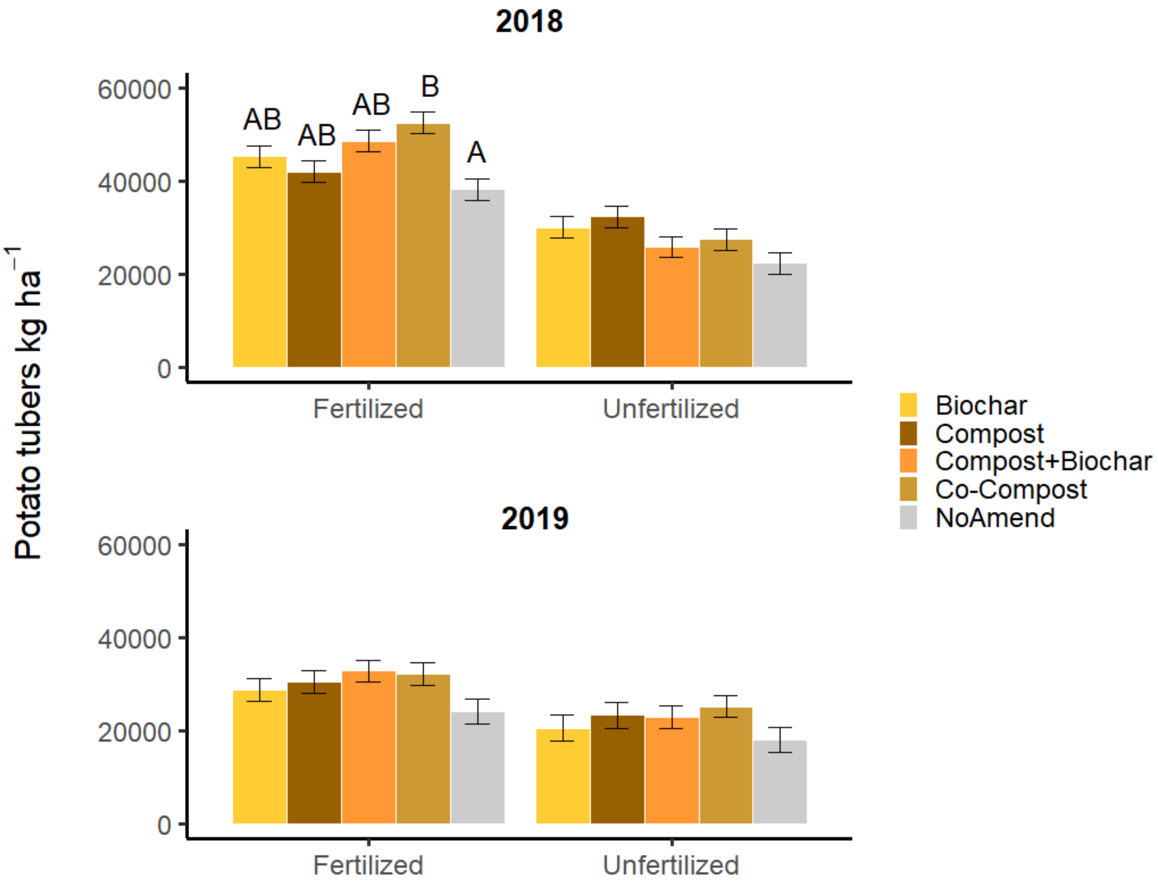

Figure 12: Potato yield (tuber weight) at Mount Vernon as affected by fertilizer and amendment in 2018 (top) and 2019 (bottom). Bars capped by different letters are significantly different according to Tukey's HSD test $p<0.05$.

Two greenhouse-based experiments with strawberries (2018, 2020) took place: initial results indicated productivity increases in some of the biochar-compost treatments, but differences were only moderate and not statistically significant (Figure 13). In 2020, in contrast to the harvest conducted in 2018, yields from individual plants varied strongly, resulting in large variation of average fruit number and total yield per plant in all treatments, but none of the observed differences of the average numbers were statistically significant, with the exception of the Lenz control vs. Lenz at 10% for total and single berry mass, though the cause of variation was unknown. This trial used a modified set of composts: a control and a co-compost with 40% by volume OBS biochar. A mixture of compost with 40% OBS biochar (post-composting) was used for comparison. This time, all plants were fertilized as this is a common practice in commercial farming.

The total sugar content evaluation suggested that fruit from plants grown with WSU 2018 compost plus 10% biochar mixed after composting had higher levels of sugars. An important caveat is that strawberries collected for this analysis differed in size and degree of ripeness. A more careful selection of fruit for chemical analysis may produce more consistent results. We also looked at organic acids to assess characteristics of flavor which showed no statistically significant variations; further replicates are needed to make conclusions about the effect of these treatments on quality.

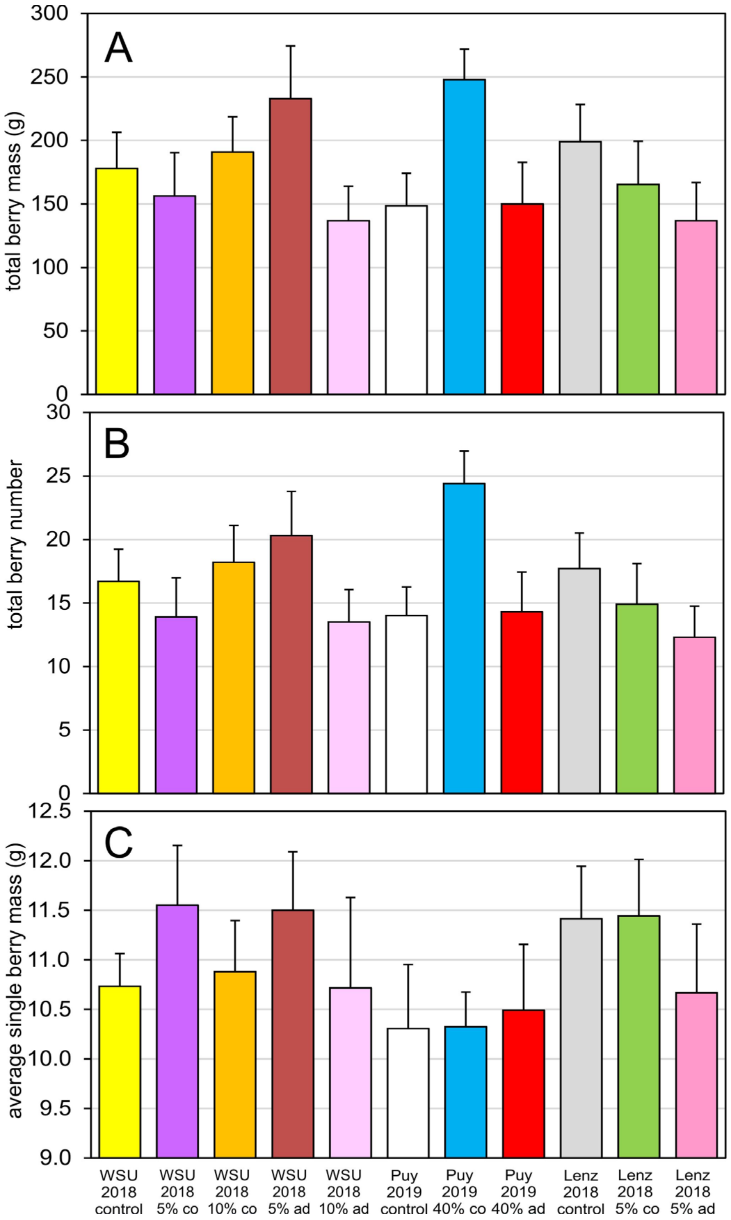

Figure 13: Effect of compost and biochar amendments on productivity of Albion strawberries. Plants were subjected to 11 treatments. Fruit was harvested as it ripened over 16 harvests from each plant. Total berry mass (A) and berry number (B) per plant were recorded. Average single berry mass (C) was calculated from the totals. In these figures, "+5%" indicates 5% biochar added after composting and "5% co" indicates 5% biochar co-composted (added before composting). Results are means ± s.e.m. (n=10).

## 5.4 Effect of biochar on soil physicochemical properties

Soil physicochemical properties including bulk density, total and particulate carbon and nitrogen, cation exchange capacity, and levels of available nitrogen (N), potassium (K), magnesium (Mg), sulfur (S), and zinc (Zn) were measured in the Mount Vernon and Puyallup field trials. Soil physicochemical properties vary naturally from site to site and also from previous and current management practices. Repeating experiments across multiple sites and years provides more robust data about the effects of amendment on soil physicochemical properties and how consistent effects are. This report provides preliminary results from three years of study at two sites. At Mount Vernon, soil bulk density was reduced and total and particulate carbon were increased relative to the control with both compost and co-compost. Cation exchange capacity was increased with biochar, compost, and co-compost in Puyallup soils, but not in Mount Vernon soils. In unfertilized plots, treatments with compost increased available N, K, Mg, S, and Zn more than biochar alone (Gang et al., 2021).

This work also was expanded this biennium as described in Chapter 4 by experimenting with high rates of incorporation of biochar (at 20 and 40% by volume) and additional information is available in Stacey et al. (2021). Our results suggest that blending compost with biochar, especially prior to composting, may optimize the physical and chemical properties of each. Compost provides a nutrient addition that is not provided with biochar alone, but biochar, perhaps because of its high surface area, may increase availability of nutrients added as fertilizer or compost.

## 5.6 Conclusion

The results of this study support the potential for using biochar as an addition to the composting process to reduce emissions of VOCs and greenhouse gases during the composting process and provided insight regarding methodology that will inform future work. Because of large variability in VOC flux densities during field sampling at both Lenz and WSU compost facilities, it was not possible to conclude whether biochar reduced VOC emissions from composting processes through field sampling. However, laboratory-based composting experiments provided evidence that the addition of 10% biochar can reduce emissions of monoterpenes, DMDS, and other compounds that are not yet identified. Since monoterpenes were the most abundant VOC at Lenz and WSU compost facilities, the reduction in emission of monoterpenes has the potential to be useful in reducing total VOC emissions for regulatory compliance. Continuous measurements demonstrated that biochar had little effect on emissions of alcohols, ketones and sulfur-containing compounds ($H_2S$ and DMS). Analysis of greenhouse gas emissions for the first trial revealed that biochar reduced greenhouse emissions. In addition, results of the laboratory study suggest that discrete measurement is likely to underestimate emissions. Therefore, for future sampling either in the field or laboratory, continuous measurements should be used for estimation of VOC emission fluxes from compost.

Likewise, the addition of biochar to compost, either at the beginning of the composting process or after composting, provides a way to add value to compost as a soil amendment. Amendment with compost, co-compost, or biochar plus compost resulted in some productivity increase in sweet basil and strawberries. However, the effects were not uniform and varied by amendment, crop, and specific experimental conditions. The same amendments to the soil did not significantly affect the phytochemical composition of field- or greenhouse-grown sweet basil, indicating no detrimental impact on basil quality from amendments. For the potato field trial, co-compost amendments were the only amendment whose application resulted in significant crop yield increases.

For one site-year (Mount Vernon, 2018), there were significant improvements to soil physical properties with both compost and co-compost applications. Though all materials were applied at similar organic carbon rates, the compost and co-compost materials had lower C concentrations and were therefore applied at the largest rates, creating a more porous (less dense) soil environment. Repeated yearly application at the same rate or a larger one-time rate would likely lead to more significant and consistent changes.

Co-compost and compost also influenced nutrient availability. These two treatments tended to increase K, Ca, and Mg in soils. During the composting process, nitrification is often increased when biochar is added as a feedstock (see Chapter 4) and biochar may also increase nitrate capture. The ability for biochar to increase nitrification or to potentially hold onto mineralized nitrogen increases the fertility benefit and adds value to compost. While compost provides nutrients to crops, compost plus biochar may provide synergy between nutrient availability and crop nutrient needs beyond what is provided by compost alone. On the other hand, biochar applied alone did not tend to increase soil nutrient status or crop yield. There may be some positive effects to soil properties when biochar is applied alone (e.g., increased cation exchange capacity as we measured in one site-year) but soil fertility was not positively impacted.

The data presented here are from two or three growing seasons, supporting conclusions made in previous biennia as to the potential use of biochar and co-composted products as soil amendments. Because the characteristics of the biochar and compost impact chemical and biological processes, different types of biochar or compost could yield different results.

Additional important detail on this work is available in the technical reports *Integrating Compost and Biochar for Improved Air Quality, Crop Yield, and Soil Health* and *Impact of High Rates of Biochar Incorporation on the Composting Process and Resulting Products* on the WTFT 2019-2021 webpage of Washington State University's CSANR.

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# Chapter 6: Production of a Biochemical from Food Waste Through Integration of Anaerobic Digestion and Fermentation Processes

*Xiaochao Xiong, Shulin Chen*

## 6.1 Introduction

The re-use of materials currently considered wastes is a critical strategy for sustainable growth in a resource-constrained world. The US has the potential to use 77 million dry tons of wet waste per year, which could contribute about 1.079 quadrillion British thermal units (Btu) of energy (USDOE, 2017). These waste streams include biosolids, animal manure, food waste, and fats, oils, and greases. Utilizing these resources will contribute to sustainability by supplying energy, fostering economic growth, reducing greenhouse gas emissions, and contributing to food waste diversion goals. While anaerobic digestion (AD) is well developed practical technology for wet waste utilization at the commercial level, the economics of AD have proven challenging because of the low price for biogas. Integration of processes to produce a high value biochemical with AD would improve the economics of AD and provide a sustainable route for biochemical production.

One promising approach that has been identified for production of high-value biofuels and bioproducts from wet waste streams involves arresting methanogenesis. In arrested methanogenesis, the microbial processes that normally occur during AD are “arrested” after volatile fatty acids (VFAs) are produced from complex organic wastes. The VFAs can then be converted into the desired product (USDOE, 2017). With the funding received through the Waste to Fuels Technology partnership during the 2015-17 biennium, the team made substantial progress to initiating a VFA platform for waste bioconversion. During the current biennium, the team further refined the key process components, including VFA production, yeast cell factory engineering, and production glycolic acid, a high-value bioproduct. Glycolic acid (hydroxyacetic acid) has important industrial applications for making cleaning agents, polymers, and personal care products. The global glycolic acid market is projected to grow from $288.9 million in 2017 to $406.4 million in 2023, with a compound annual growth rate of 6.83% (Research and Markets, 2018). The current chemical processes through which glycolic acid is produced have disadvantages, namely, use of toxic materials, generation of undesirable by-products, and operation under harsh conditions (Salusjärvi et al., 2019).

As shown in Figure 14, the technological approach explored in this study integrates two processes:

- Converting complex waste materials into VFAs, mainly acetate, through AD, and
- Transforming the resultant VFAs, a group of simple molecules, to the target products (in this case, glycolic acid) in a separate fermentation bioreactor using a metabolically engineered yeast strain.

This novel pathway benefits both from the capacity of the anaerobic microbial consortia to handle complex waste and from ability of the engineered cell factory to biosynthesize the target molecule.

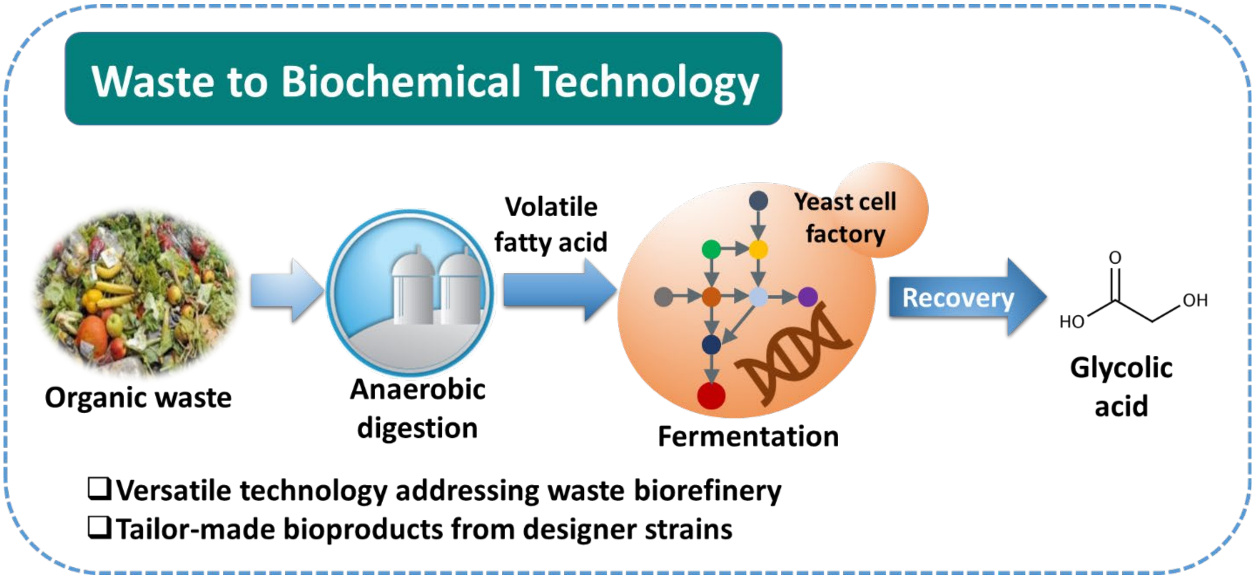

Figure 14: Production of bio-based glycolic acid from waste by integration of volatile fatty acid (VFA) generation in anaerobic digestion and fermentation processes.

## 6.2 Approach and state-of-the-art technology

Anaerobic digestion is a well-established waste management and treatment process for converting organic wastes to bioenergy (Zamri et al., 2021). When fully implemented, the AD process involves the degradation of organic matter to methane ($CH_4$), carbon dioxide ($CO_2$), and mineralized nutrients using a mixed culture of symbiotic bacteria (Rasapoor et al., 2020). During AD, complex organic molecules are broken down into VFAs, simpler organic molecules. Then, VFAs are consumed by methanogens, who generate biogas.

The production of VFAs through arrested methanogenesis has been extensively explored. In this platform, AD is “arrested” after the production of VFAs. VFAs can either be end products or precursors for production of biofuels or industrial chemicals (Lu et al., 2020). VFA production can be improved by enhancing the hydrolysis of waste through physical or chemical pretreatments and supplement of enzymes, and by enhancing acidogenesis rates through pH control, temperature control, redox potential, and by optimizing inoculum. In addition, chemicals such as 2-bromoethanesulfonate (2-BES) can be added to inhibit methanogenesis (Lukitawesa et al., 2020). Previous studies have shown that arresting methanogenesis results in production of more VFAs and less methane in AD using a variety of organic wastes including wheat straw (Awasthi et al., 2018), corn stover, fruit and vegetable waste, food waste (Jones et al., 2021), and manure.

Although VFAs themselves are useful chemicals, separating each of these chemicals from the bulk liquid presents a significant cost barrier (Shi et al., 2018). To bypass this barrier, another logical approach is the utilization of the VFA in bulk, bypassing the need to separate a single target VFA chemical.

Microorganisms can use VFAs to synthesize more complex molecules through aerobic cultivation. For example, microbial lipids are precursors for producing biofuels with high energy density (e.g., biodiesel), and can be generated from VFAs, including acetic acid. However, the drawback of this approach is that long-chain hydrocarbon molecules, including lipids, are reduced substances with much lower oxygen content than the substrate, acetic acid. Consequently, both theoretical and empirical lipid yield from acetate is low (around 0.25 g/g) (Hu et al., 2016).

Biosynthesis of glycolic acid can potentially overcome the limitations and disadvantages encountered with production of long-chain hydrocarbons. Genetic engineering of *Escherichia coli, Saccharomyces cerevisiae, Kluyveromyces lactis,* and *Corynebacterium glutamicum* have each been explored for the biosynthesis of glycolic acid from a number of cellulosic sugar precursors (Cabulong et al., 2018; Salusjärvi et al., 2017; Koivistoinen et al., 2013). Our approach was developed to use organic waste with negative or low value instead of cellulosic sugars as feedstock, potentially lowering production costs.

*Y. lipolytica* has several advantages as a microbial host for metabolic engineering in this study, including tolerance to a high concentration of VFAs (100 g/L of propionate) and the capability to utilize VFAs, *Y. lipolytica* has been cultured and genetically engineered to biosynthesize various bio-based chemicals, including the organic acids α-ketoglutaric acid, citric acid, succinic acid, and itaconic acid. However, the production of glycolic acid from acetate, especially acetate generated from organic wastes, has not been fully explored.

As a Generally Recognized As Safe (GRAS) organism, *Y. lipolytica* has been widely used for industrial production of a suite of chemicals and this knowledge has been harnessed in the current project. Previous work by Liu et al. (2016) provided a deep insight into the metabolism of *Y. lipolytica* cultured on acetate. Our previous research also developed a comprehensive set of molecular biology tools for genetic manipulation of *Y. lipolytica* (Xiong and Chen, 2020). We have successfully genetically engineered *Y. lipolytica* to produce lipid, free fatty acid (Ghogare et al., 2020), long-chain dicarboxylic acid, fatty alcohol (Wang et al., 2016), and wax ester.

In addition, our approach builds upon sound rationale because (1) the AD process is very effective in converting various organic wastes to VFA, and (2) VFA is particularly suitable for the production of glycolate because expression of the enzymes involved in the glyoxylate shunt, which plays a role in the metabolism of two-carbon substrates (e.g., acetate) and the replenishment of tricarboxylic acid (TCA) cycle intermediates essential for the production of biomolecules. The glyoxylate shunt, the biochemical pathway for generating the precursor, is known to be up-regulated (expression is increased) in yeast when the strains are grown on a VFA, such as acetic acid (Walsh and Koshland Jr, 1984).

## 6.3 Innovations in the waste to biochemical approach

We devised a unique technical route with multiple innovations, including arresting methanogenesis in the AD process for the production of VFAs at a high rate, selecting *Y.*

*lipolytica* as a desirable host that is both tolerant to high concentrations of VFAs, engineering *Y. lipolytica* for production of glycolic acid, and compartmentalizing metabolic pathways in the yeast cell factories.

(1) Arresting methanogenesis in the AD process for the production of VFAs

We developed a novel thermophilic AD operating at 70 °C to produce VFAs from waste streams. Aside from the generally accepted advantages of AD processes (no sterile conditions or expensive enzymes required, mixed microbial communities that can handle complex and variable organic waste streams), thermophilic AD adds unique benefits for producing VFAs (Saber et al., 2021). At such high temperatures, methane production ceases and more complete digestion of the feedstock occurs, leading to higher VFA yields and decreased solid retention times (Qiao et al., 2013). The higher temperature also leads to greater reductions in pathogens and antibiotics in the effluent, generating materials that are quite suitable for agricultural field application.

(2) Pathway design for producing bio-based glycolic acid

The problem of low yield represents a significant barrier in most biofuel and biochemical production from VFAs. Selecting glycolic acid as the target product overcomes the barrier of low yield because = the similar carbon and oxygen contents between the product (glycolic acid) and substrate (acetic acid) improves yields. Due to the similarities between substrate and product, only one heterologous gene encoding glyoxylate reductase (GR) needs to be introduced into *Y. lipolytica* to produce glycolic acid from acetate through the glyoxylate shunt in this designed pathway (Koivistoinen et al., 2013).

(3) Compartmentalizing metabolic pathways for biosynthesis of glycolate

We employed the strategy of pathway compartmentalization to isolate and concentrate the substrates, regulate pathway fluxes and eventually generate a high yield of the target product. In yeast, the glyoxylate shunt and TCA cycle reactions are highly connected, involving different cellular compartments, including cytosol, peroxisomes, and the mitochondria. Pathway compartmentalization as a novel metabolic engineering strategy allows for enrichment of the precursors and avoidance of undesirable consumption of the precursors and intermediates by other pathways (Avalos et al., 2013). While traditional pathway engineering uses cytoplasm, both mitochondria and peroxisomes can also be engineered as the production units in yeast cells.

## 6.4 Methods and materials

Methods and materials are summarized here. For full details on these and analytical procedures refer to Xiong and Chen (2021).

### Producing VFAs from organic waste

Two common types of organic wastes, food waste and dairy manure, were tested for the production of VFAs in this study. We collected the anaerobic sludge inoculum from a primary sedimentation tank at the wastewater treatment plant in Pullman, Washington. The sludge was transferred into sterile bottles purged with nitrogen gas to ensure anaerobic conditions and then stored at 37 °C for one week to minimize the degradation of organic compounds in the sludge. Food waste was obtained from a student cafeteria at Washington State University (WSU) in Pullman, Washington. The food waste consisted of rice, noodles, meat, and various vegetables and fruits. The dairy manure was collected from Knott Dairy Center at WSU. The sludge inoculum and waste materials were characterized in terms of total solid (TS) and volatile solid (VS) contents as detailed in Xiong and Chen (2021).

Methanogenesis has to be blocked in order to produce VFAs rather than biogas from organic wastes during the AD process. VFA production was conducted in a 7.5-liter fermenter (NBS Bioflo-110) with a 5-liter working volume. The mixture contained 2,500 g of waste materials (e.g., food waste) and 2,500 g of anaerobic sludge, and was adjusted to contain 15% total solids. The mixture was purged with nitrogen for 20 minutes and capped tightly with butyl rubber to maintain anaerobic conditions. The AD process was carried out at a controlled temperature of 70 °C, agitation speed at 300 rpm, pH at 7.0, and without aeration. Samples were collected from the bioreactor each day and analyzed to determine the content and composition of VFAs produced.

## Strains, culture conditions, and DNA techniques

In this project, *Y. lipolytica* PO4f previously developed by our lab from the strain *Y. lipolytica* PO1f (ATCC MYA-2613) was used as a microbial host for metabolic engineering to produce glycolic acid (Wang et al., 2016). The general procedures for conducting molecular biology experiments, such as PCR and gene cloning, are described in previous publications (Wang et al., 2016; Xiong and Chen, 2020).

## Strains engineered for glycolic acid production

For the pathway developed for this project, only one heterologous gene encoding glyoxylate reductase (GR) needed to be introduced into *Y. lipolytica* for producing glycolate from glyoxylate, which can be biosynthesized from acetate through the glyoxylate cycle (Li et al., 2019). However, glyoxylate can alternatively be consumed by the native enzyme malate synthase (MS) in *Y. lipolytica* (Figure 15). Deleting the genes encoding MS thus prevents the cell from consuming glyoxylate and allow the organism to funnel all glyoxylate towards glycolate synthesis, improving yields.

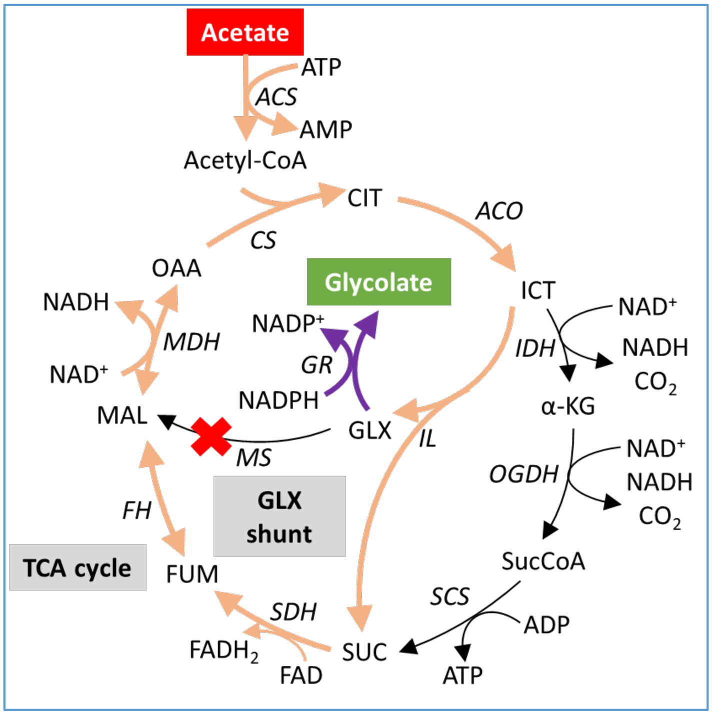

Figure 15: Pathway design for the biosynthesis of glycolic acid (glycolate) from acetic acid (acetate) in yeast cells.

The specific steps in engineering *Y. lipolytica* for glycolic acid production are listed below and details on how these steps were carried out are provided in Xiong and Chen (2021).

1. Deletion of two genes encoding malate synthase (MS) in *Y. lipolytica,*
2. Expressing glyoxylate reductase (GR) in *Y. lipolytica* ΔMS1ΔMS2 for producing glycolic acid, and
3. Investigation of the role of the gene *acs2* in the utilization of acetic acid.

## 6.3 Results and discussion

### Metabolic engineering of yeast strains

We designed a pathway for the production of glycolic acid from acetic acid (Figure 15) by engineering the yeast cell factory of *Y. lipolytica*. The double knockout strain ΔMS1ΔMS2 did not show observable defects of growth on acetate as a sole carbon source. Thus, this represented a successful first step towards the engineering of strains for the production of glycolic acid.

To produce glycolic acid, we individually expressed two genes encoding glyoxylate reductase from different organisms (*E. coli* and *A. thaliana*) in yeast with two disrupted genes ms1ms2 . Only the second gene worked well and glycolic acid could be detected by the transformant bearing *GLYR1* from *A. thaliana*. Expression of *GLYR1* could be optimized to improve glycolic acid production by *Y. lipolytica*.

As a eukaryotic cell, yeast has different cellular compartments, including cytosol (the water-based solution inside cells, in which the organelles, proteins, and other cell structures float), peroxisome, and mitochondria (two important organelles). Previous studies indicated that the target product, glycolic acid, could be produced from glyoxylate, which was the primary metabolite produced in the glyoxylate cycle. The reactions for the glyoxylate cycle mainly occur in yeast peroxisome. In contrast, the steps of TCA cycle for the generation of isocitrate, the substrate for the formation of glyoxylate, localize in the mitochondria (Figure 15). Therefore, it was necessary to take the pathway compartmentalization into account for the engineering of a yeast cell factory for the biosynthesis of glycolic acid.

To target an enzyme in yeast mitochondria, the expression gene of interest needs to be fused with a functional MTS. We evaluated the location of the expression gene of interest in *Y. lipolytica* using enhanced Green Fluorescent Protein (eGFP). As shown in Figure 16, mitochondrial localization of GFP fusion protein was confirmed by merging red and green fluorescence images.

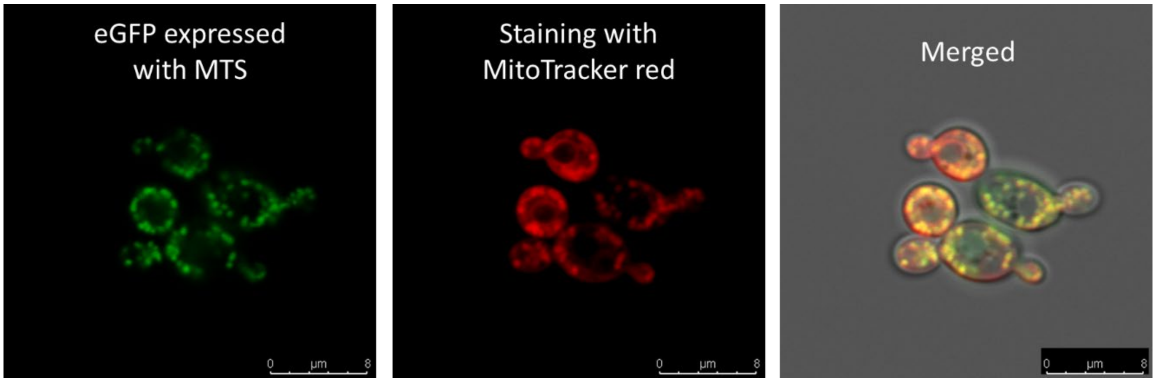

Figure 16: Subcellular localization of green fluorescent protein (GFP) in Y. lipolytica mitochondria with leading peptide from OGDC1 observed under a fluorescence microscope. The fluorescence between GFP and MitoTracker Red, which specifically stained the yeast mitochondria, was merged to verify the mitochondrial localization of the GFP.

We found that the production of the enzyme GYLR1 in different organelles, including cytosol, peroxisome, and mitochondria, resulted in the production of glycolic acid at different concentrations. Using 30 g/L of acetate, one of the major components in VFAs, as the sole carbon source, the strain *Y. lipolytica* ΔMS1ΔMS2 expressing GYLR1 in peroxisome produced the highest concentration, 3.4 g/L of glycolic acid, after cultivation for 96 hours at 28 $^{o}$C in a shaking flask. Under the same culture conditions, the stain bearing GYLR1 in mitochondria produced 2.5 g/L glycolic acid, whereas expression of GYLR1 in cytosol only led to the production of 1.8 g/L glycolic acid. The strains secreted the glycolic acid to the supernatant of culture media. During cultivation, the pH in the culture media increased from 6.6 to 9.2, mainly due to the utilization of acetate.

We further investigated the cell growth of *Y. lipolytica* ΔMS1ΔMS2 expressing GYLR1 in peroxisome on 30 g/L of acetate, and there was no noticeable difference after genetic engineering compared with the parent strain. This result highlights that pathway compartmentalization in different cellular compartments can be used as an essential strategy for designing and engineering a yeast cell factory to produce glycolic acid. Furthermore, we can improve glycolic acid production from acetate by both pathway engineering and by optimizing fermentation by controlling pH and other factors.

*Y. lipolytica* can utilize acetate as a sole carbon source for its growth, and we can further engineer the strain to improve the capability of acetate utilization. To understand the role of the gene *acs2* of *Y. lipolytica* in acetate utilization, we first deleted this gene, and then overexpressed *acs2* in this knockout strain *Y. lipolytica* ΔACS2. The result indicated that *acs2* of *Y. lipolytica* played an essential role in acetate utilization. The single knockout *Y. lipolytica* ΔACS2 still could use acetate as a sole carbon for growth. This suggested that there might be other enzymes, such as short-chain fatty acyl-CoA synthetase, responsible for converting acetate into acetyl-CoA. We can overexpress *acs2* to improve acetate and other VFA compound utilization by *Y. lipolytica*.

The next step was to test the use of the engineered strains of *Y. lipolytica* for production of glycolic acid from VFAs generated from waste materials. Because the use of food waste as feedstock led to higher concentrations of VFAs than the use of dairy manure, we used VFAs generated from food waste to culture the engineered strain expressing peroxisomal GYLR1to produce glycolic acid. The liquid effluent was separated from the product of food waste digestion. The effluent enriched with VFA was used to culture the glycolic acid-producing strain. As shown in Figure 17, the strain produced more than 3.6 g/L glycolic acid in a shaking flask for 120 hours. During cultivation, pH increased from 7.0 to 9.5. This hybrid process succeeded in production of glycolic acid from organic waste by incorporating the AD process with yeast cultivation.

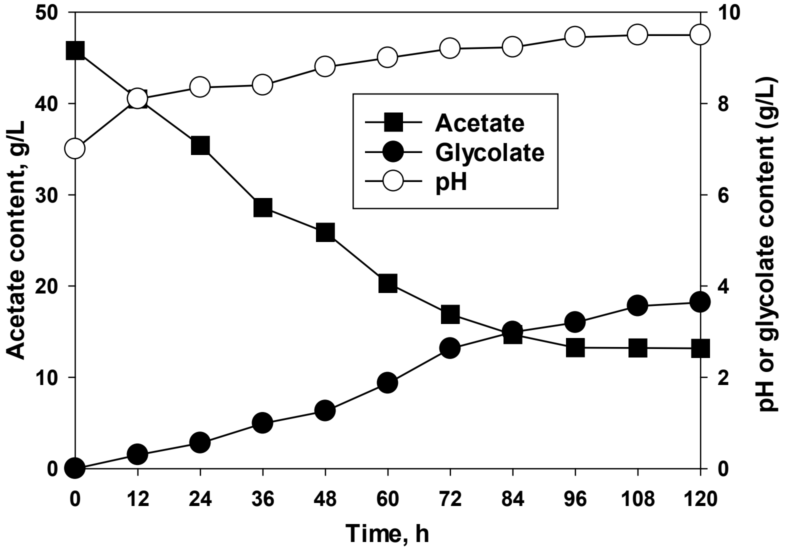

Figure 17: Production of glycolic acid from VFAs generated from food waste by the genetically engineered yeast strain.

## 6.4 Conclusions & future direction

In this project, we demonstrated the feasibility of producing glycolic acid, a biochemical with broad industrial applications, from organic waste using an innovative technological route. The AD process was operated at a high temperature to produce VFAs from organic wastes. The concentration of acetate, the primary component in VFAs, reached 50 g/L by using food waste as a feedstock. We successfully engineered a yeast, *Y. lipolytica,* as a cell factory to biosynthesize glycolic acid from acetate as a sole carbon source.

Future efforts will focus on optimizing the developed processes and incorporating them into a functional biorefinery to achieve necessary technical and economic performance targets. Engineering the microorganisms to reach high TRY performance (Titer [concentration], production Rate, and Yield), remains a top priority. We will employ the strategies of overexpressing genes for enhancing precursors supply ("push") and product formation ("pull") to develop a productive strain for generating glycolic acid.

After further improvement of performance of the strains developed in this project for glycolic acid production, their potential for use in a bioreactor setting will be enhanced by further work in four areas:

1. Further optimization of the thermophilic AD process for generation of VFAs,
2. Developing an optimal fermentation process, including a feeding strategy,
3. Optimization of product separation and purification, and
4. Development of a techno-economic analysis.

Although this project focuses on the production of glycolic acid, the technical platform can be tailored to deliver a variety of commodity chemicals by constructing different metabolic pathways in the microbial host. Therefore, developing and demonstrating this technology has the potential for broad impacts for the biofuel and bioproduct industries.

Additional detail is available in the technical report *Production of a Biochemical from Food Waste Through Integration of Anaerobic Digestion and Fermentation Processes* on the WTFT 2019-2021 webpage of Washington State University's CSANR.

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# Chapter 7: Technical Potential for $CO_2$ Drawdown Using Biochar in Washington State

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## 7.1 Introduction

As outlined in Amonette et al. (2016a,b), production of biochar from waste wood in Washington State using modified biomass boilers has the potential to yield many benefits, including improved biomass productivity, decreased irrigation costs, and, perhaps most importantly, drawdown of atmospheric carbon dioxide ($CO_2$; Amonette et al., 2016a,b). Although Amonette et al. (2016a,b) used the results of an earlier global model (Woolf et al., 2010) to estimate that on the order of 500-600 metric tons (megatonnes; Mt) atmospheric $CO_2$ could be offset in Washington State over the course of a century (before accounting for releases of carbon [C] currently in the oceanic and terrestrial pools), they recommended further analysis be made to refine and solidify this estimate.

Amonette (2018) took the first step along this path, by developing and demonstrating a high-resolution scalable method for estimating the net 100-year $CO_2$ drawdown technical potential of biochar for Spokane County with the aim to apply the method to the entire state in subsequent work. The method took into account local, site-specific factors such as (1) the availability and distribution of waste-wood biomass, (2) the locations of existing biomass boilers, (3) the soil types and land-use categories receiving biochar amendments, and (4) the expected primary productivity responses to biochar amendments (a positive-feedback loop). Global climate system responses to drawdown, such as net losses of non-pyrogenic soil organic C (npSOC, which is the existing soil organic matter, distinguished from the pyrogenic organic C added in the biochar) and the long-term equilibration of atmospheric and oceanic $CO_2$ reservoirs, were also considered.

The second report of the series (Amonette, 2019), which expanded the analysis to include 26 counties, strengthened this approach in several ways. First, land capability classes and cropping systems were explicitly related at a 1-hectare (ha) spatial resolution for use in estimating primary productivity responses to biochar amendments. Second, soil priming effects (i.e., the change in npSOC levels expected from additions of biochar) were updated to reflect recent literature suggesting a small enhancement of npSOC by biochar amendments to agronomic soils. This effect was treated separately from the decreases in forest npSOC levels expected when forestry residues were harvested to make biochar. Third, explicit time-dependent tracking of both biochar production levels and biochar soil-storage capacities was incorporated. This was to account for the exports of biochar from counties that have exceeded their soil-storage capacities to counties that have more storage capacity than biochar-production capacity. This tracking provided the first assessment of the relative levels of production and consumption over time among the 26 counties included in the study and set the stage for a future economic assessment that includes transportation costs as a factor.

The present work builds on the previous results by adding a new feedstock scenario to account for the contributions from agricultural crop residues (chiefly straw from cereal production) and two forestry feedstock scenarios that account for biomass from thinning operations associated with potential wildfire hazard-reduction efforts. The present work also significantly improves the estimates of soil priming effects by incorporating first-order kinetic models to account for the rate of npSOC increase over time, the loss of biochar to oxidation, and a saturation level of npSOC per unit of biochar added. These priming effects are then scaled according to the initial npSOC stocks, which account for various site-specific factors such as temperature, rainfall, mineralogy, etc. that affect equilibrium npSOC levels. Thus, for the same level of biochar amendment, a smaller increase in npSOC would be predicted for a low-npSOC soil in the hot and dry regions of the state than for a high-npSOC soil in the cooler and wetter regions.

Finally, the present work applies the method to all 39 counties in Washington State, individually and collectively, thereby providing a more detailed and scientifically defensible estimate of the statewide potential of biochar technology to draw down atmospheric $CO_2$ over a century than that provided earlier by Amonette et al. (2016a,b).

## 7.2 Biochar Global Response Assessment Model

The algorithm used to perform the assessment is a modification of the Biochar Global Response Assessment Model (BGRAM) implemented in spreadsheet form by Woolf et al. (2010). This algorithm considers biomass composition, pyrolysis and combustion process parameters, energy production, C intensity of energy being offset, rate of technology adoption, biochar properties, biomass growth response, biomass and biochar transport, biochar decomposition rates, and greenhouse gas emissions at every stage of the cycle from biomass harvest to 100 years after biochar has been added to the soil. The original version was developed for a global analysis based primarily on the use of agricultural-biomass residues, and required modest revisions to be able to work with smaller national, regional, and local datasets. Extensive details about the original BGRAM algorithm can be found in the online supplemental information file associated with the Woolf et al. (2010) publication.

The BGRAM algorithm performs calculations for a specific input scenario, which basically consists of estimates of the amount and composition of sustainably available biomass for each feedstock being considered, coupled with information about whether the biomass is processed in the field by a mobile unit or at a central location, whether pyrolysis (for biochar) or combustion (for bioenergy) processes are to be used, and the travel distances required to get the biomass to the processor and the biochar to the land where it is to be applied. For this study, four primary feedstock streams were used: agricultural crop residues (straw from cereal crops), residual forest biomass from timber-harvesting operations, wood reclaimed from municipal solid waste (MSW; dimensional lumber, engineered wood, pallets and crates, natural wood, and other non-treated wood), and green waste also reclaimed from the MSW stream. In addition, a fifth secondary feedstock stream, based on the additional drawdown stemming from biomass response to biochar amendment (i.e., enhanced yield), was considered in each scenario. The enhanced-yield secondary feedstock stream in BGRAM required input data for the initial (i.e., pre-biochar) crop type and yield and for the soil productivity potential for each parcel of cropped land in the county.

BGRAM involves time-dependent tracking of biochar production and soil storage capacity for each county. This allows estimation of the point at which the agricultural soil in the county becomes “saturated” with biochar (currently assumed to be incorporation of 50 t biochar C/ha to a nominal 15-cm depth). At that point, the county then must export biochar to other counties in the state with available storage capacity. As will be shown in the results, assembling and analyzing the data collected across the state allows a better grasp of the biochar export/import economy and helps provide input to future techno-economic studies of the probable development path of the biochar industry in Washington State.

This present work involves two significant updates to BGRAM that are discussed further below:

1. A new algorithm to estimate the losses and gains in npSOC, and
2. Addition of a fourth major source of biomass feedstock (agricultural crop residues).

## Non-pyrogenic soil organic carbon

The most important change to BGRAM in the present work was to develop a mechanistically based algorithm to estimate the losses (positive priming) in npSOC stemming from harvesting of biomass for production of biochar as well as the gains (negative priming) in npSOC, stimulated by amendments of biochar to agronomic soils. These changes replaced the relatively simple approach taken in Amonette (2019) for estimating the priming effects of biochar technology.

For more information about the body of work exploring how biochar amendments affect npSOC stocks, and development of the updated algorithm, see Amonette (2021).

## Biomass and processing scenarios

This work updates previous work (Amonette 2016, 2019) by incorporating agricultural crop residues as feedstock streams into BGRAM, in addition to those of woody biomass from MSW and timber harvest. Methods are summarized below with more detail provided in Amonette (2021).

Agricultural crop residues from cereal crops were modeled as a feedstock stream in BGRAM. These crops include wheat, corn grain, barley, buckwheat, hemp, quinoa, rye, sunflower seed, triticale, oats, and any crop grown for commercial seed production (grasses, vegetables, legumes). Total residue production values for each county were obtained from the cereal-crop yield data. A residue availability factor of 66% was chosen, which is considerably higher than that used in the original version of BGRAM (8%) and the values recommended in the literature (generally below 35%, Woolf et al., 2010; Lal, 2005). The higher residue removal rate is justified by the new ability in BGRAM to estimate the beneficial long-term impact of biochar on formation of npSOC (i.e., negative priming), results for which show a net gain in npSOC for most agronomic soils amended with biochar when residue removal rates to make the biochar *exceed* 25% to 40%.

Two woody biomass feedstock streams recovered from MSW were modeled in BGRAM, green waste and reclaimed waste wood. Estimated quantities for these in each county were developed from a survey conducted in 2015-2016 by the Washington State Department of Ecology (Ecology, 2016) and updated to 2019 based on official county population estimates. A third woody biomass feedstock stream consisted of timber-harvest residues. Six estimates of harvestable woody biomass (i.e., the trimmings from tree stems harvested for lumber that were brought to the landing) were generated for each county using the Washington State Department of Natural Resources online biomass calculator for two 5-year planning periods (2021-2025 and 2026-2030). The estimates assumed conservative, average, or aggressive timber-harvest scenarios.

Each of these timber-harvest scenarios was further divided into three processing scenarios: scattered, roadside, and market. Scattered biomass was left at various locations on the harvest site where the trees were cut and trimmed. Roadside and market biomass was gathered and brought to a roadside "landing" at the harvest site for possible loading and transport to a central facility. Market biomass was actually loaded and transported, whereas roadside biomass was not transported. Of these three categories, roadside and market biomass were considered available for processing into biochar. Roadside biomass could be processed using a mobile pyrolysis platform brought to the landing. Market biomass could be processed at a central pyrolysis facility identified by the biomass calculator. As the estimates of available harvestable biomass did not differ greatly among the two planning periods, mean data from the two periods were calculated for use in the input scenarios.

Table 10 shows the potential annual biomass inputs of all four feedstock streams (harvested crop residues, harvested forestry residues, MSW recovered wood, MSW green waste) and the ten total processing scenarios.

Table 10: Annual biomass inputs by harvest scenario for Washington State.

| Harvest scenario | Processing location |  | Biomass inputs |  |  |  |  |
|---|---|---|---|---|---|---|---|
|  | Facility | Field | Harvested crop residues | Harvested forestry residues | MSW recovered wood | MSW green waste | Total biomass processed |
|  |  |  | --------------------- thousands of green tonnes ---------------------- |  |  |  |  |
| Feedstock-Specific Scenarios |  |  |  |  |  |  |  |
| Crop Residues | X | X | 2020 | n/a | n/a | n/a | 2020 |
| MSW | X |  | n/a | n/a | 311 | 43 | 354 |
| Full Scenarios with Facility Processing |  |  |  |  |  |  |  |
| Conservative | X |  | 2020 | 6,360 | 311 | 43 | 8,730 |
| Average | X |  | 2020 | 9,190 | 311 | 43 | 11,600 |
| Aggressive | X |  | 2020 | 11,100 | 311 | 43 | 13,400 |
| Average w/Thin | X |  | 2020 | 9,780 | 311 | 43 | 12,200 |
| Full Scenarios with Facility and Field Processing |  |  |  |  |  |  |  |
| Conservative | X | X | 2020 | 13,500 | 311 | 43 | 15,900 |
| Average | X | X | 2020 | 19,200 | 311 | 43 | 21,600 |
| Aggressive | X | X | 2020 | 23,000 | 311 | 43 | 25,400 |
| Average w/Thin | X | X | 2020 | 20,800 | 311 | 43 | 23,100 |

Because the focus of this study is on the technical potential and a goal was to estimate the highest possible potential, only one set of economic conditions was specified: low biomass harvest costs and high ($100 per bone-dry ton) biomass price paid at facility. All existing and potential biomass-processing facilities within four hours driving time of the harvest location were selected for consideration of economic viability.

Of the total amount of residues produced from timber-harvest operations in Washington State, roadside biomass accounts for 36%, and market biomass for 32%. Thus, residue-removal rates for harvested timber lands are 68%, which is comparable to the 66% assumed for agricultural cropped lands. In the current version of BGRAM, however, the biochar produced with this residual biomass is assumed to be applied only to agricultural croplands.

## Biochar supply, production and storage capacity

Calculations were performed using input generated at both the individual county level and at the state level. The state-level calculation, which used the same biomass inputs summed across all counties but an average soil-property and crop-response parameter set, provides a more realistic assessment because it implicitly allows export and import of biochar across county lines to achieve a greater degree of soil incorporation than is possible when all biochar remains in the county in which it was produced. Although they tend to underestimate the total impact of biochar technology by 21% to 26%, the county-level calculations nevertheless provide key insights into local biomass supply and biochar production levels, as well as biochar storage capacities. See the Appendices of Amonette (2021) for county-by-county results.

The biomass inputs for the ten scenarios considered ranged from a total of 354 Mt for the MSW (Facility) scenario to 25,400 Mt for the Aggressive (Facility + Field) scenario, a factor of 72 (Table 10). The amount of biomass from crop residues is about 6 times larger than that from MSW and, after accounting for the 66% residue availability factor, it represents 8% to 23% of the total biomass for the eight full scenarios. As with MSW, the proportion of biomass from crop residues varies significantly across counties—from no residues available in several heavily timbered west-side counties to above 97% in several rural eastern counties where no timber is harvested.

The vast majority of the available biomass in Washington State comes from timber-harvesting residues, which account for 73% to 91% of the total in the eight full scenarios that include all three sources of biomass (i.e., crop residues, MSW, and timber-harvesting residues). On average, the proportion of the biomass coming from MSW is small, ranging from 1.3% to 3.6% of the total for the eight full scenarios. For individual counties, however, the MSW proportion ranges more widely. For example, the MSW proportion for the full scenarios in King County ranges from 9% to 30%, whereas the range for Grays Harbor County is 0.14% to 0.49%, reflecting the large differences in the types of biomass available in urban population centers compared to heavily timbered rural counties.

As logic would dictate, the cumulative 100-year biochar-C gross production levels follow the same trends as the biomass input levels. With progressively higher biochar production levels, however, the years of available storage capacity drop quickly—to 106 years for the

Conservative (Facility) scenario and as little as 62 years for the Aggressive (Facility + Field) scenario. Further discussion of these trends can be found in Amonette (2021).

A plot of the biochar-C storage capacities of all 39 counties, ranked in the same order as for the maximum biochar-C production levels shown in Figure 18, shows that the counties with small woody-biomass biochar production capacities generally have large biochar-C storage capacities (Figure 19). In fact, the counties having the largest biochar-C production, such as Grays Harbor and Lewis, will generally exceed their intra-county storage capacity within the first two decades of production and will become biochar exporters for the remainder of the century. Large-scale adoption of biochar technology, therefore, will require a substantial effort to transport not only biomass to processing facilities, but also biochar to storage sites that may be 100 to 200 miles distant. Although the climate impact of this transportation effort is relatively small compared to the overall benefit, the economic impact will likely be very large. Further techno-economic study of the problem is needed to refine the overall C-drawdown potential of biochar technology in Washington State and to identify the locations where it is most likely to be economically viable.

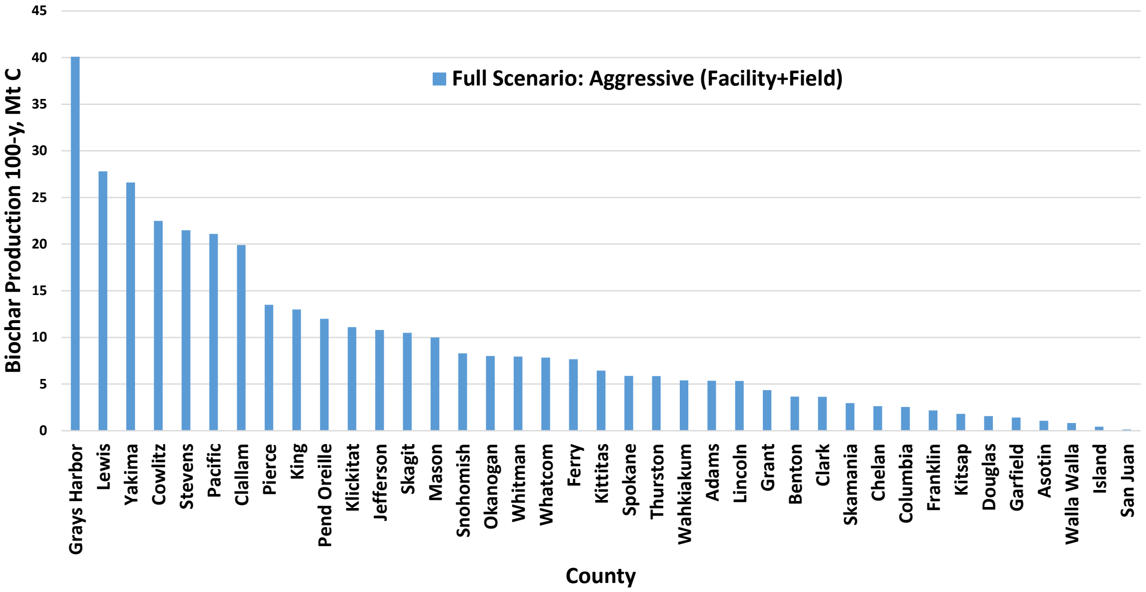

Figure 18: Maximum cumulative 100-year biochar production for all feedstocks in Washington State ranked by county.

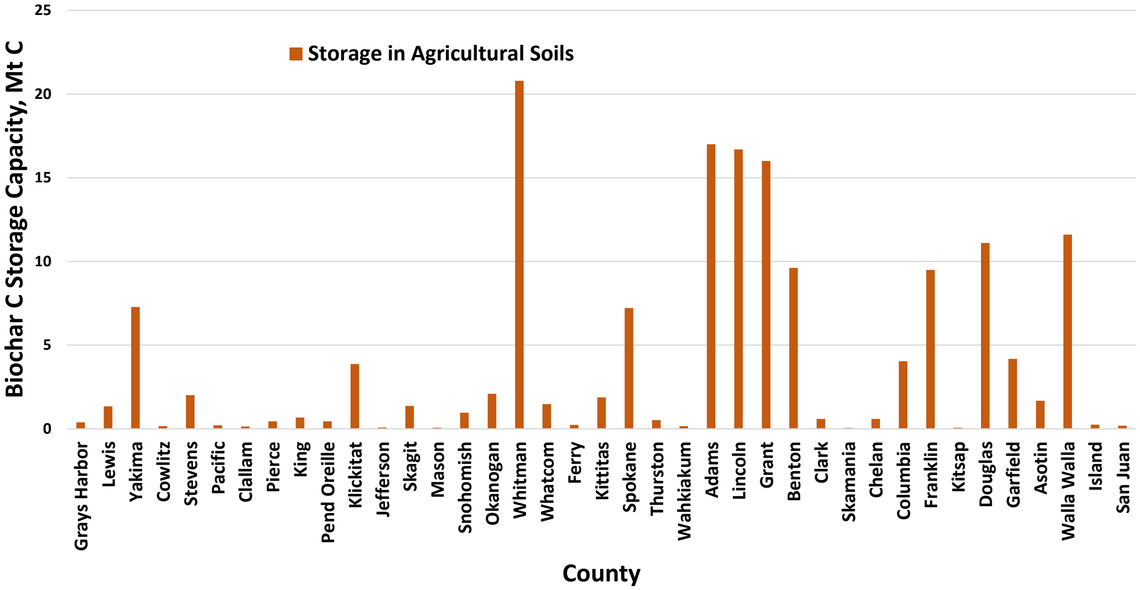

Figure 19: The initial biochar storage capacity in agricultural soils for each county in Washington State ranked by maximum cumulative 100-year biochar production for all feedstocks.

Another, somewhat more tractable issue, relates to the overall statewide biochar-C storage capacity. A timeline comparison of the net cumulative biochar-C stored, which is the difference between the gross biochar produced and that which is oxidized once in soil, shows that five of the eight scenarios fully saturate the available storage capacity during the first 100 years. This seemingly dire limitation to the overall C-drawdown potential of biochar, however, can be addressed in part by developing additional locations and mechanisms for storage, of which there are several. This limit can be pushed to higher levels with the development of additional storage reservoirs (e.g., forest and rangeland soils) and technologies (e.g., incorporation into construction materials). Fortunately, the current results suggest that we will have several decades at least to develop alternative storage options.

## 7.3 Results

### Climate offsets

To assess the climate impact of a given scenario, BGRAM calculates a variety of offsets for each feedstock stream, which are summed for the individual feedstock stream, and then over all feedstock streams to obtain a total offset. In addition to results for biochar, which assume slow pyrolysis, BGRAM also calculates results for complete combustion of the same biomass to generate bioenergy. These two sets of results bracket the range of offsets possible by different methods for making biochar, such as slow pyrolysis, fast pyrolysis, gasification, etc., with slow pyrolysis being the most C-efficient process for making biochar and combustion being the extreme case in which no biochar is produced. They also highlight the different contributions to the climate offset, with biochar-C added being most important for biochar and fossil-fuel emissions offset being the most important for bioenergy.

The total 100-year offsets for biochar and bioenergy in the ten scenarios are listed in Table 11. The results can be interpreted in two ways: the immediate offset (Mt carbon equivalent [Ceq]), which accounts for the initial C drawdown, and the ultimate offset (ppbv $CO_2$eq), which is expressed here in terms of atmospheric $CO_2$ levels and adds the long-term buffering response of the earth's climate system to the initial C drawdown. Put simply, to lower the ultimate (equilibrium) concentration of $CO_2$ in the atmosphere by 1 parts per million by volume (ppmv), 2.17 ppmv of $CO_2$ need to be removed, due to equilibrium dynamics between $CO_2$ reservoirs in the atmosphere and ocean.

The 100-year climate offsets generally follow the expected trend established by the size of the biomass inputs (Table 11). Thus, addition of biochar and bioenergy production in the field (i.e., Facility + Field scenarios) increases the climate offsets by 70% to 80% over those obtained when only centralized facilities (Facility scenarios) are used for processing. For biochar, the immediate offset ranges from 8.5 Mt Ceq for the MSW Only (Facility) scenario to 430 Mt Ceq for the Aggressive (Facility + Field) scenario. The ultimate offset ranges from 1.8 parts per billion by volume (ppbv) carbon dioxide equivalent ($CO_2$eq) to 93 ppbv $CO_2$eq for the scenarios analyzed. Implementation of forest-thinning operations to reduce wildfire risk is predicted to increase the available biomass by 5% to 7% (Table 10) and the net offset for biochar by a similar percentage (Table 11).

Table 11: Total 100-year offsets for production of biochar and bioenergy summed by harvest scenario, and the ratios of the bioenergy offsets to the biochar offsets for Washington State.

| Harvest scenario | Processing location | | Total 100-year offsets[a] | | | | |
|---|---|---|---|---|---|---|---|
| | Facility | Field | Biochar | Bioenergy | Biochar | Bioenergy | Bioenergy / Biochar |
| | | | Mt $C_{eq}$ (immediate) | | ppbv $CO_{2(eq)}$ (ultimate) | | |
| Feedstock-Specific Scenarios | | | | | | | |
| Crop Residues | X | X | 44 (45) | 18 (18) | 10 (10) | 3.9 (4.0) | 0.41 (0.37) |
| MSW | X | | 8.5 (10) | 4.5 (4.5) | 1.8 (2.2) | 1.0 (1.0) | 0.53 (0.52) |
| Full Scenarios with Facility Processing | | | | | | | |
| Conservative | X | | 174 (144) | 66 (53) | 38 (31) | 14 (11) | 0.38 (0.38) |
| Average | X | | 223 (180) | 85 (67) | 48 (39) | 18 (15) | 0.38 (0.38) |
| Aggressive | X | | 253 (203) | 98 (77) | 55 (44) | 21 (17) | 0.39 (0.38) |
| Average+Thin | X | | 232 (188) | 89 (71) | 50 (41) | 19 (15) | 0.38 (0.38) |
| Full Scenarios with Facility and Field Processing | | | | | | | |
| Conservative | X | X | 291 (232) | 115 (88) | 63 (50) | 25 (19) | 0.39 (0.39) |
| Average | X | X | 376 (299) | 154 (117) | 81 (65) | 33 (25) | 0.41 (0.39) |
| Aggressive | X | X | 430 (343) | 179 (137) | 93 (74) | 39 (30) | 0.42 (0.39) |
| Average+Thin | X | X | 398 (319) | 164 (127) | 86 (69) | 35 (28) | 0.41 (0.39) |

[a] First value in each cell is calculated using the state-average input parameters. Second value (in parentheses) is the sum of individual county-level calculations and does not consider exports or imports of biochar among counties to alleviate soil-storage capacity limitations

Recognizing that the current levels of atmospheric $CO_2$ are on the order of 415 ppmv (NOAA-ESRL-GMD, 2021) as compared to pre-industrial levels of 270 ppmv and a recommended target level of 350 ppmv to avoid significant climate disruption (Hansen et al., 2008), it is clear that the maximum potential contribution of biochar produced from biomass in Washington State, while large, addresses only 0.14% of the needed global drawdown (assuming no further increase in atmospheric concentrations). Fortunately, this is roughly in proportion to the fraction of the earth's unglaciated land surface occupied by Washington State (0.13%) and further supports the concept that global adoption of biochar technology can make a significant contribution to the drawdown effort. The magnitude of the drawdown effort required to address climate change is truly significant and requires a comparably sized contribution from every region of the planet.

### Impact on non-pyrogenic soil organic carbon

As suggested by the npSOC priming ratio results, discussed further in Amonette (2021), the total impact of biochar technology on npSOC stocks in Washington State is positive. Current npSOC stocks, estimated at 115 Mt in the top 23 cm of agronomic soils, are predicted to increase by 13 Mt (11%) for the Crop Residues scenario and to nearly triple (i.e., by 224 Mt [195%]) for the Full Aggressive Facility + Field scenario. To validate these predictions, additional field research into the long-term impact of biochar amendments on npSOC stocks should be a high priority.

### Bioenergy

In general, the climate offsets from bioenergy in Washington State are about 40% of those estimated for biochar (Table 11). This is largely due to the low C intensity of Washington's primary energy supply (10.16 kg C / GJ, U.S. Energy Information Agency, 2019) stemming from the large contributions of hydro- and wind-power to the electrical grid, but also to the degree of enhanced yield obtained when biochar is applied to soils (see Woolf et al., 2010 for further discussion). In most scenarios, bioenergy also forgoes the increases in npSOC content stimulated by biochar amendments while still paying the penalty for removal of residual biomass from soils. For the MSW scenario, however, no penalty for removal of residual biomass from soils is applied and, as a result, the relative offset for bioenergy increases to 52%. Given the relatively small contribution of MSW biomass to the full scenarios, however, the general observation that biochar is 2.5 times more effective than bioenergy as a climate mitigation option in Washington State still applies.

## 7.4 Conclusions

This assessment of the C-drawdown potential of biochar technology when implemented in Washington State over the course of 100 years shows that a wide range in drawdown potential exists, depending primarily on the size of the woody biomass supply.

- Use of recovered woody biomass from MSW yields a total immediate greenhouse gas offset of 8.5 Mt Ceq.
- Use of cereal crop residues yields a total immediate greenhouse gas offset of 44 Mt Ceq.

- Addition of timber-harvest residual biomass to the MSW and crop-residue biomass results in 170 to 430 Mt Ceq depending on the harvest scenario and process facility location.
- Addition of field processing of biomass to that done in centralized facilities roughly doubles the available biomass and increases the C-drawdown potential by 70% to 80%.
- When equilibrium with the climate system reservoirs is considered, an ultimate greenhouse gas offset can be calculated in terms of decreases in atmospheric $CO_2$ levels. This metric yields a drawdown potential range from 38 to 93 ppbv $CO_2$eq for scenarios that include all potential sources of biomass. The highest drawdown level corresponds to 0.14% of what is needed to stabilize the earth's climate system from today's levels of $CO_2$. This is roughly in proportion to the fraction of the earth's unglaciated land surface occupied by Washington State (0.13%), confirming that global adoption of biochar technology can make a significant contribution to the drawdown effort.
- With residue-harvesting rates of 66% (crop residues) to 68% (forestry residues), biochar technology is expected to increase npSOC stocks when the crop-residue production rate is greater than 1.5 t/ha.
- Use of the same biomass to generate bioenergy instead of biochar yields about 40% of the climate drawdown potential obtained with biochar.
- The biochar-C storage capacity is lowest for counties that generate large amounts of woody biomass, and consequently, after a few decades they will need to export their biochar to agricultural counties, located primarily in the southeast quadrant of the state.
- Under current storage-potential assumptions, the biochar-C soil-storage capacity will be saturated in 62 to 106 years for the full scenarios that include crop residues, MSW, and timber-harvest biomass residues. This limit, however, can be pushed to higher levels with the development of additional storage reservoirs (e.g., forest and rangeland soils) and technologies (e.g., incorporation into construction materials).

At the maximum biomass-utilization rate, which is achieved after five decades, biochar production could offset between 8% and 19% of the greenhouse-gas emissions in Washington State (taken at 2018 levels). Because this study examined technical potential with the aim to identify the upper limit, only one set of economic conditions was specified: low biomass harvest costs and high ($100 per bone-dry ton) biomass price paid at the facility (all existing and potential biomass processing facilities within a four hour drive of the harvest location). While actual offsets may be lower than the 9% to 20% range because economic feasibility in an emerging market is complex to predict, the result of this analysis suggests that ongoing work to improve biochar economics and performance are well worth the effort – and confirms that implementation of policies (e.g., C-sequestration credits) to promote adoption of biochar technologies could make a substantial contribution to the effort to address climate change.

Additional detail is available in the technical report *Technical Potential for $CO_2$ Drawdown Using Biochar in Washington State* on the WTFT 2019-2021 webpage of Washington State University's CSANR.

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