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For more information contact:

Solid Waste Management Program
P.O. Box 47600
Olympia, WA 98504-7600
Phone: 360-407-6900

Washington State Department of Ecology – www.ecology.wa.gov

- Headquarters, Olympia 360-407-6000
- Northwest Regional Office, Bellevue 425-649-7000
- Southwest Regional Office, Olympia 360-407-6300
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Advancing Organics Management in Washington State

The Waste to Fuels Technology Partnership 2017-2019 Biennium

By

*Karen Hills¹, Manuel Garcia-Perez², James E. Amonette^{1,3}, Michael Brady⁴, Tom Jobson⁵,
Douglas Collins^{1,6}, David Gang⁷, Embrey Bronstad⁸, Markus Flury⁹, Steven Seefeldt¹⁰,
Claudio O. Stöckle², Michael Ayiania², Anna Berim⁷, Wendy Hoashi-Erhardt⁶,
Neda Khosravi⁵, Sohrab Haghighi Mood², Roger Nelson², Yaime Jefferson Milan²,
Nigel Pickering¹¹, Nathan Stacey¹, Abid H. Tanzil², Jun Zhang^{9,12},
Brooke Saari¹, Georgine Yorgey¹*

¹WSU Center for Sustaining Agriculture and Natural Resources; ²WSU Department of Biological Systems Engineering; ³Physical Sciences Division, Pacific Northwest National Laboratory; ⁴WSU School of Economic Sciences; ⁵Laboratory for Atmospheric Research, WSU Department of Civil and Environmental Engineering; ⁶WSU Puyallup Research and Extension Center; ⁷WSU Institute of Biological Chemistry; ⁸WSU Center for Sustaining Agriculture and Natural Resources (formally WSU Department of Biological Systems Engineering); ⁹WSU Department of Crop and Soil Sciences; ¹⁰WSU Northwestern Washington Research and Extension Center; ¹¹Washington Stormwater Center, WSU; ¹²Guilin University of Technology

*Compiled and edited by
Karen Hills, Brooke Saari, Georgine Yorgey, and Tara Zimmerman*

Solid Waste Management Program
Washington State Department of Ecology
Olympia, Washington

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List of Abbreviations

AD	anaerobic digestion
ANOVA	analysis of variance
ASP	aerated static pile
BGRAM	Biochar Global Response Assessment Model
C	carbon
C _{eq}	carbon equivalent
CEC	cation exchange capacity
CO	compost overs
CO _{2eq}	carbon dioxide equivalent
CSANR	Center for Sustaining Agriculture and Natural Resources
DMS	dimethyl sulfide
EPA	Environmental Protection Agency
GC-MS	gas chromatography-mass spectrometer
GHG	greenhouse gas
GIS	geographic information system
kWh	kilowatt hour
LFG	landfill gas
ME	moisture equivalent
MSP	minimum selling price
MSW	municipal solid waste
Mt/MT	metric ton; megatonne
MW	megawatt
MWh	megawatt hour
NRCS	Natural Resources Conservation Service
N	nitrogen
N-doped	nitrogen doped

NR	nutrient recovery
PAWC	plant available water capacity
PB	particle board
PM _{2.5}	particulate matter less than 2.5 microns in diameter
ppbv	parts per billion by volume
ppmv	parts per million by volume
PTR-MS	proton-transfer reaction mass spectrometry
RNG	renewable natural gas
s.e.m.	standard error of the mean
SWM	Solid Waste and Materials
USDA	United States Department of Agriculture
VOC	volatile organic compound
WARM	Waste Reduction Model
WHC	water holding capacity
WUI	wildland-urban interface
WSU	Washington State University
WTFT	Waste to Fuel Technology
XPS	X-ray photoelectron spectroscopy

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Executive Summary

Over the last ten years, Washingtonians generated an estimated 1,200 to 1,350 pounds of organic waste per person per year (Figure ES.1). Of this material, 49-60% was composted at a composting facility or recovered for other uses (Washington State Department of Ecology, 2019), and the balance was disposed of in landfills. These disposed materials represent a hidden resource. If organics that are currently disposed of can be effectively recovered and used in anaerobic digestion, pyrolysis, composting, and other processes, they have potential to contribute to the growth of Washington’s economy, generate renewable energy, and provide other sustainable and valuable products. Furthermore, even where organics are successfully diverted from landfills via composting, there may be opportunities for alternate end uses to generate more valuable products.

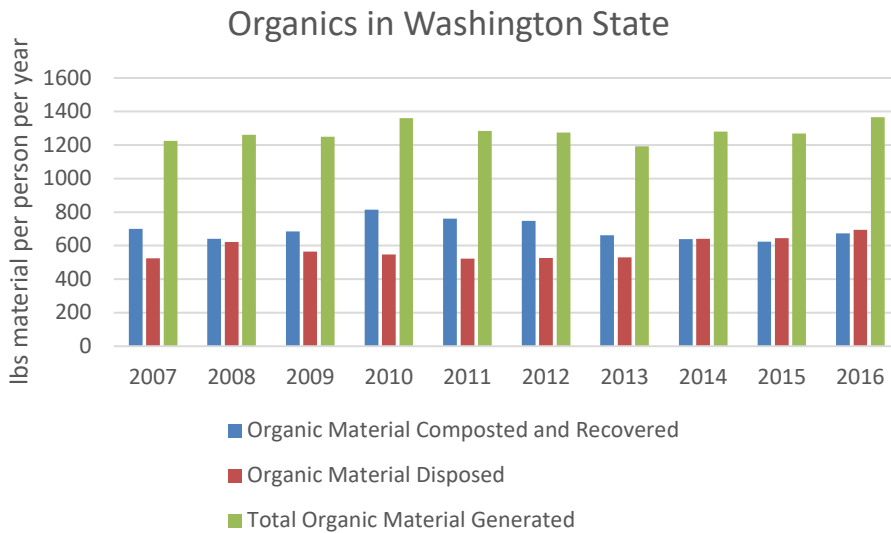


Figure ES.1: Per capita organics generation and disposition in Washington State, 2007-2016. (Washington State Department of Ecology, 2019)

Progress towards recapturing the nutrients, energy, and value in organic wastes will contribute to the achievement of the State Solid and Hazardous Waste Plan, which envisions “a society where waste is viewed as inefficient, and where most wastes and toxic substances have been eliminated (Ecology, 2015).” This approach turns the current paradigm of organic “wastes” on its head and reduces the negative environmental and community impacts that result from disposing and processing organic wastes, including production of greenhouse gases, loss of nutrients to air and water, and use of limited landfill space. If the resulting organic products are land-applied to soils, additional benefits include improved soil quality and structure, increased amounts of carbon sequestered in the soil, and, in some cases, reduced fertilizer and pesticide use. The Waste to Fuels Technology partnership between the Washington State Department of Ecology’s Solid Waste Management Program and Washington State University seeks to further the development

and application of next generation technologies that can enhance the value of Washington’s waste management systems to benefit Washington’s economy, environment, and communities.

The Waste to Fuels Technology partnership is working towards the development and appropriate implementation of municipal *biorefineries* – meaning facilities that sustainably convert biomass to energy and other beneficial products. 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, and climate and air quality impacts. Figure ES.2 shows one possible example of a biorefinery, in which composting, the core waste conversion technology, is complemented by pyrolysis, anaerobic digestion, biogas upgrading to renewable natural gas, nutrient recovery and fertilizer production, and greenhouse production utilizing waste heat. Within these seemingly distinct handling trains are synergies: the biochar produced from wood waste benefits the composting process and could be used to scrub the digester gas of sulfur and other contaminants. The renewable natural gas could be used in waste delivery trucks, or could be used to offset energy costs associated with drying feedstock for the pyrolysis unit, which in turn is another potential source of feedstock to supplement the anaerobic digester.



Figure ES.2: One possible realization of a municipal biorefinery. In this particular example, a core composting facility is transformed into a biorefinery through the addition of pyrolysis for treating a fraction of the wood wastes, and anaerobic digestion (AD) for treating food wastes. Nutrient recovery (NR) technologies and biogas upgrading to renewable natural gas (RNG) generate value-added products, while waste heat from pyrolysis is used for greenhouse production. Figure: Nick Kennedy

The biorefinery vision is modular rather than prescriptive, and the specific technologies that make sense will vary depending on the context. Furthermore, as biorefinery ideas continue to develop within and outside our region, they may not include some of the technologies envisioned here – and will almost certainly include technologies not currently proposed.

Significant remaining barriers that have prevented wide adoption of a municipal biorefinery in the Pacific Northwest and elsewhere in the US include the presence of inexpensive hydroelectric power in the Northwest (which impacts project economics), contamination of the waste stream, scale issues, and the need for additional technology development. The applied research and extension efforts carried out through the Waste to Fuels Technology partnership aim to reduce these barriers and provide additional options for organics management throughout Washington.

In the 2017-2019 biennium, the Waste to Fuels Technology partnership carried out a diversity of work. While extension efforts took a broad view and encompassed the range of waste conversion processes including composting, anaerobic digestion, and pyrolysis (Chapter 1), a number of projects explored potential improvements and issues related to composting facilities, currently the most widespread organics processing method utilized (Chapters 2-5). Additional projects (Chapter 6-10) explored pyrolysis and the resulting biochar products. This report summarizes the results and implications of these projects, while important additional detail, including specifics related to methods and additional results, is provided in technical reports associated with each project. Technical reports are available from Washington State University's Center for Sustaining Agriculture and Natural Resources, on the [Waste to Fuels Technology partnership 2017-2019 webpage](#).

Within this executive summary, we also indicate which portions of the State Solid and Hazardous Waste Management Plan are supported by each project. (See Ecology, 2015, for this plan in its entirety.) The Solid Waste and Materials (SWM) goals and actions supported by projects in this report are listed below.

SWM 15: State and local governments will have a better understanding of solid waste energy and material recovery technologies.

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 20: The value of recycled organics as storm and surface water filtration media will be better understood, resulting in increased use.

SWM 21: Soil organic carbon sequestration using recycled organics will increase based on research recommendations.

SWM 22: More diversified organics processing infrastructure will exist in the state.

SWM 23: Composting facilities will produce clean end products.

SWM 24: Diversified end-use markets will be in place for recycled organic products.

Chapter 1, Extension, Engagement, and Technology Transfer, describes the team's extension efforts. This work broadly supports the State Solid and Hazardous Waste Management Plan goals, with particular relevance to SWM 15, SWM 18, SWM 21, SWM 22, and SWM 24. In-person engagement opportunities related to the biorefinery vision, current and past Waste to Fuels Technology research results, and organics processing technologies including composting, anaerobic digestion, pyrolysis and nutrient recovery were provided to an estimated 754 stakeholders. These stakeholders represented a diversity of stakeholder types, and stakeholder engagement included activities such as presentations and discussions with a number of solid waste advisory committees (SWACs) across the state, at King County organics management workshops, and one-on-one conversations with a number of counties and non-profits working on climate- and waste-related planning efforts. The Center for Sustaining Agriculture and Natural Resources also provided access to relevant online resources and publications (developed with Waste to Fuels Technology and complementary funding) that were viewed over 18,000 times. The partnership's work led to \$357,658 of additional leveraged funds acquired during this biennium for work relevant to the priorities of the Waste to Fuels Technology partnership by researchers who have received funds via the Waste to Fuels Technology partnership. These funds will enable further research to address challenges related to Waste to Fuels Technology areas of interest. While the ultimate outcome of engagement activities is likely to be realized in the long-term, there were short-term indications of impact: stakeholders were excited to find that Washington State University is engaged in work related to sustainable organics management, and several stated that they found our extension efforts useful. Engagement activities strengthened and formed productive, ongoing connections between researchers and statewide stakeholders, and raised stakeholder awareness about ongoing partnership activities.

Chapter 2, Emissions from Washington State Compost Facilities: A Review of Volatile Organic Compound Data, and an Estimation of Greenhouse Gas Emissions, reviewed existing information relevant to emissions from composting facilities in Washington State (supporting SWM 23). The work involved three principal analyses. First, a review of existing volatile organic compound (VOC) emission data collected from six Washington composting facilities 2010-2013 found that young compost pile emissions were dominated by light alcohols (methanol and ethanol) and monoterpenes (α -pinene and limonene). The VOC emission profiles varied significantly from the US EPA Speciate profile for green waste compost, suggesting that the EPA Speciate profile for green waste compost is of limited utility for describing Washington's compost emissions. Inconsistencies in which compounds were analyzed in the dataset, and differences in methodologies made comparisons between facilities difficult, and suggested that additional data collection may be needed to support a better understanding of VOC emissions from commercial compost facilities in Washington, including data collected over the life of individual compost piles. Choice of methodology should be based on a more fully developed understanding of emissions components. For example, if emissions are dominated by light alcohols, a total VOC method would significantly underestimate VOC emission rates.

Second, while the review indicated the need for ongoing work to develop a reliable average Washington compost emissions factor, application of the California average total VOCs emissions factor to known throughputs for Washington commercial composting facilities indicated that eight composting facilities in Washington may have the potential to emit more than 100 tons of VOC per year and thus potentially be subject to EPA Title V permitting, depending on emission control approaches being used.

Third, the EPA's Waste Reduction Model (WARM) was applied, and indicated that composting organics is likely to reduce greenhouse gas emissions compared to landfilling. However, uncertainties exist particularly with regard to nitrous oxide and methane emissions factors for compost, and with regard to quantifying soil carbon storage benefits after compost application.

The project described in *Chapter 3, Differentiating the Value and Cost of Compost Across Likely Farm Use Scenarios in Western Washington* (supporting SWM 18 and SWM 19), explored the possible reasons why demand from west-side Washington farmers for municipally-derived compost has lagged behind the expansion of municipal compost production in western Washington. To better understand why this may be true, reasonable "good case" scenarios were developed to illustrate potential cost and value for municipal compost used in the production of specific crops in western Washington. This work demonstrated two important points. First, under reasonable assumptions, the value of compost appeared likely to exceed cost for some high-value crops grown in the region, including raspberries and direct market mixed vegetables in this analysis. This suggests that efforts to address current barriers to compost use on farms would likely be fruitful - including efforts to substantially reduce contamination in compost and addressing high costs of entry for farmers wishing to use compost, such as via programs for shared compost spreader use. Second, the analysis showed that compost can have a wide range of values, depending on the cropping system to which it is applied, the soil health status prior to compost application, and the application rate needed to see crop impacts. This suggests that additional field research to further refine the estimates of value for a broader range of conditions would also be beneficial.

The work described in *Chapter 4, Policies for Increasing Compost Use in Washington: A Comprehensive Economic Consideration of Standards, Subsidies, and Taxes*, evaluated whether changes to compost program design could enhance the use of compost. The project thus supported primarily SWM 18 and SWM 24. Because the implementation of composting projects is relatively recent, the economics literature on composting programs is sparse. Therefore, the recycling literature was examined. The consensus from this body of literature was that policy intervention at the middle of the waste stream, or the household curbside, is relatively inefficient, even if all social costs are properly priced in. The reasons for this include a lack of incentive to get final recyclable product to end users, and the fact that available empirical evidence from recycling programs suggests that households are fairly non-responsive to pricing (their response is "inelastic"). While it is a significant departure from current policy that would need further study, a tax-based approach (i.e., a landfill tax) might provide a statewide context that would encourage some municipalities to adopt composting (those in which the benefits outweigh the costs), without requiring action for those municipalities where costs would outweigh the benefits.

In addition, recent recycling experience suggests that a focus on the upstream ends of the waste production chain, to ensure that organic waste generation is carried out in ways that facilitate eventual composting, is important. In particular, limiting contamination is likely to be an important strategy. In this regard, standards are one area receiving increased interest lately, including bans on problematic components of the organics waste stream such as plastic bags or straws. Beyond reviewing lessons from the recycling literature, the analysis also considered the possibility of two demand side policy interventions specifically focused on increasing agricultural use of compost. Available evidence suggests that refocusing subsidy from collectors to buyers (farms) is likely to have little effect. Instead, reducing the high up-front costs that farmers would need to invest in specialized equipment in order to experiment with compost via equipment sharing programs or other means would be more likely to influence compost demand from farms.

Chapter 5, Integrating Compost and Biochar for Improved Air Quality, Crop Yield, and Soil Health, describes work that explores the integration of composting with biochar. The project is thus supporting SWM 18, SWM 22, and SWM 24. There are indications that biochar, when added to other feedstocks at the beginning of the composting process, can reduce emission of VOCs during composting. Two field sampling experiments and two laboratory experiments therefore examined the effect of biochar on emission of VOCs from compost. Results of laboratory experiments indicate that 10% biochar can significantly reduce emissions of monoterpenes, dimethyl disulfide, and other compounds that are not yet identified. Reduced emission of these compounds would help reduce compost odor.

Meanwhile, biochar, compost, 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. 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). A greenhouse study with two basil cultivars and three biochar-amended composts produced different results for the two basil cultivars, but overall showed moderate increases in biomass production with biochar amended compost. A greenhouse-based experiment with strawberries indicated productivity increases were observed in some of the biochar-compost treatments but were overall only moderate, and require further studies for reliable conclusions.

Initial results from field trials indicated in general, significant effects on crop yield, with effects that varied by amendment type, crop, and soil type. Because of variability due to weather and other factors, repeated field trials, which are anticipated for coming years, will provide more certainty to observations from the initial year. Amendments to the soil did not significantly affect the phytochemical composition of field- or greenhouse-grown sweet basil, an indication that product quality was not compromised. Co-compost and the compost plus biochar 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 crop.

In *Chapter 6, Assessment of the Local Technical Potential for CO₂ Drawdown using Biochar from Forestry Residues and Waste Wood in 26 Counties of Washington State*, an improved high-resolution scalable method was developed to estimate the technical potential for atmospheric carbon (C) drawdown by biochar in Washington State using forestry residues and waste wood as primary feedstocks and agricultural soil as the C-storage reservoir (supporting SWM 21 and SWM 22). Twenty-six counties in Washington State were selected for application of the method. For each county, seven biomass feedstock and biochar process scenarios were developed including one for waste wood harvested from municipal solid waste (MSW) alone, and six for MSW waste wood combined with forestry residues from timber harvesting operations. Individual results for each county were generated. Summing the results for the 26 counties over a period of 100 years shows that, depending on scenario, biochar could generate between 8 and 411 million metric tons of biochar C, a total immediate offset of between 11 and 354 metric tons of C-equivalent (C_{eq}) and, after accounting for climate-system responses, an ultimate (equilibrium) drawdown of between 2.4 and 77 parts per billion by volume of atmospheric carbon dioxide equivalent (CO_{2eq}). If the same sustainably procured biomass were instead combusted for renewable energy, these offset and drawdown values decrease by 50%. The analysis shows that 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 26-county biochar-C soil storage capacity will be saturated in 54 to 109 years for the scenarios that include timber harvest biomass residues. This limit, however, can be pushed to higher levels with the development of additional storage technologies and reservoirs such as forest and rangeland soils.

Chapter 7 investigated methods for creating higher value biochar that could have specialized uses, specifically, the *Production of Engineered Biochars for Phosphate Removal from Waste Lignocellulosic Materials* (supporting SWM 20 and SWM 22). The focus on phosphate as well as hydrogen sulfide removal, and the biochar feedstocks including anaerobically digested fiber, urban wood residuals, and wheat straw, were chosen specifically for their potential to integrate into urban and rural waste processing biorefineries. The impact of engineering on water holding capacity was also examined as this is an important function for biochar incorporated into soils. In the first generation of engineered biochar, a pyrolysis step was followed by an activation step with carbon dioxide. Carbon dioxide-activated char from anaerobically digested fiber had phosphate adsorption capacity of 32.4 mg g⁻¹ biochar. The hydrogen sulfide adsorption capacity of AD fiber-derived chars was 51.2 mg g⁻¹. The breakthrough time for adsorption of hydrogen sulfide for AD fiber-derived char produced at 750°C compared favorably to commercial activated carbon, an important benchmark.

Second generation biochar was produced using two-step or one-step “nitrogen doping” (the process of introducing nitrogen functional groups into a carbonaceous material, abbreviated “N-doped”). Nitrogen-doped char produced using a single step process had a phosphate adsorption capacity nearly double that of char produced using a two-step process (110.3 mg g⁻¹ vs. 63.1 mg g⁻¹). The research team also conducted analysis of water holding capacity with N-doped biochars

produced from urban wood residuals (particle board and compost overs). When raw (non N-doped) char from particle board was blended with Quincy sand soil at a rate of 10% by weight, water holding capacity more than doubled compared to no biochar, from 29.9 to 69.6% by weight. However, N-doping provided little benefit compared to untreated (raw) biochar, and actually reduced the water holding capacity compared to raw biochar at higher application rates.

Last, third generation biochars were produced by impregnating feedstock with metals (magnesium, calcium, or iron) and then using N-doping process to create a metal-N-doped biochar from both wheat straw feedstocks and pure cellulose. Pure cellulose was used in this experiment to provide insight into the action of cellulose versus other components of more complex lignocellulosic compounds. Metal-N-doping using magnesium and nitrogen together were effective at improving phosphate adsorption capacity for wheat straw to 288 mg g⁻¹ and for cellulose char to 335 mg g⁻¹. With further development, these processes hold great promise for integration into a municipal biorefinery. For example, activated biochar derived from AD fiber could be used for hydrogen sulfide removal from AD biogas and phosphate removal from AD effluent, or engineered chars from a number of materials could be sold and used to adsorb phosphate from a variety of wastewaters. Within either of these scenarios, the resulting phosphate-charged biochar could perhaps be sold as a nutrient-rich soil amendment, though more information is needed to determine nutrient availability to plants following soil amendment with these materials.

Chapter 8 concerns the development of *A Rapid Test for Plant-Available Water-Holding Capacity in Soil-Biochar Mixtures* (supporting SWM 19 and SWM 22). Measurement of biochar's impact on soil water holding capacity by standard methods can be time consuming and expensive. However, it is important to characterize this impact, because biochar's ability to increase the plant-available water-holding capacity (PAWC) of soil can increase plant productivity when water is limiting. In this project, a new inexpensive, rapid method for measuring PAWC of soil-biochar mixtures was refined and calibrated. The new method was then applied to 72 combinations of soil and biochar: nine Washington soils of varying textures, four biochars, and two application rates. Application of this method led to the following conclusions regarding the effects of biochar amendments on the PAWC of soils: (1) biochar increases the PAWC of soils, but the contribution of biochar is not linearly proportional to the amount of biochar added; (2) soil texture, and possibly soil mineralogy, in some instances, have a large impact on the degree to which biochar increases PAWC; and (3) inter-particle effects are the largest contributor to the overall impact of biochar on PAWC. This method has great application potential for future work in measuring the PAWC of biochar-soil combinations, particularly as a screening tool, and for use in monitoring of changes in PAWC over time.

In *Chapter 9, Using CropSyst to Evaluate Biochar as a Soil Amendment for Crops*, a computer simulation study was conducted to evaluate the potential effect of biochar addition to a loamy sand soil with low water holding capacity, with the expectation that two biochars (with high and low ammonium adsorption capacities) added at rates of 96 and 288 tons per hectare to a soil depth of 0.3 meters would increase water retention, decrease nitrogen losses, and increase yields of potatoes (supporting SWM 18, SWM 19, and SWM 22). The simulations were performed

using a cropping system simulation model (CropSyst) that considered full irrigation, deficit irrigation, and no irrigation treatments and covered a period of 30 years using weather data from a location near Moses Lake, Washington. The effect of biochar additions on potato yields was negligible under full irrigation. However, when soil water limited crop growth (under deficit and no irrigation treatments), small yield gains of approximately four percent were projected. This result is consistent with dryland cereal crop experiments reported in the literature, conducted in agricultural soils not limited by low pH or low fertility. The addition of biochar to irrigated crops under full irrigation is unlikely to be advantageous unless acidic conditions are improved by biochar additions. Yields of dryland crops grown in the Inland Pacific Northwest might benefit, particularly when applied at high rates, although the economic benefit of biochar addition at these rates to relatively low value per acre crops will require evaluation. Regarding nitrogen, the relatively fast conversion of ammonium to nitrate (nitrification) neutralized the effect of the additional ammonium sorption capabilities from the biochars, providing no advantage. It is unclear if biochar can provide a degree of protection against the microbial activity responsible for nitrification, thus delaying the conversion process.

Chapter 10, Biochar Production in Biomass Power Plants: Techno-Economic and Supply Chain Analyses, examined the potential market for biochar in the Pacific Northwest by estimating production cost and agricultural use values (supporting SWM 22). Techno-economic analysis suggested that there is a scenario where the minimum viable selling price for biochar is in the vicinity of \$150 per metric ton. The potential value of biochar was also analyzed based on carbon sequestration and yield improvement. This analysis suggested that without a climate policy compensating farmers for carbon sequestration, there was only one type of crop (mixed vegetables) which under a very optimistic yield improvement assumption of 30%, could justify the use of biochar. Biochar use in agriculture would become much more feasible if there were a carbon market with prices nearing \$40 per metric ton of carbon dioxide equivalent. While this price is higher than the range seen currently within most carbon markets, it is at the low end of the range that has been suggested may be needed to avoid the worst climate change impacts.

Taken together, this diverse body of work explores multiple avenues through which Washington State could successfully and profitably incorporate advanced waste treatment technologies such as pyrolysis and anaerobic digestion, and produce value-added products from organic waste. The results of this work provide both technical information and economic analyses relevant to Washington State, supporting further development of policy and technology related to organic waste recovery. Meanwhile, the diverse connections formed through the research have led, over time, to additional resources from other sources being directed at problems related to improving solid waste management. Over time, these developments could contribute a wide range of economic, environmental, and social benefits for residents and communities of Washington State.

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Chapter 1: Extension, Engagement, and Technology Transfer

Embrey Bronstad, Georgine Yorgey, Karen Hills, and Brooke Saari

The Waste to Fuels Technology (WTFT) partnership promotes the efficient conversion of organic wastes into energy, nutrients, and other valuable materials. Targeted outreach to stakeholders helps develop an understanding of the biorefinery concept and its potential benefits while promoting appropriate consideration and adoption of biorefinery-related technologies. This chapter describes the extension and engagement efforts related to the WTFT partnership. These efforts provided an estimated 754 contacts through direct engagement with stakeholders, and access to online resources and publications that were viewed over 18,020 times. Current and past work under the Waste to Fuels Technology partnership also led to \$357,658 of additional leveraged funds acquired during the current biennium. These funds will contribute to continued work on Waste to Fuels Technology topics.

1.1 Technology transfer and engagement with regional organics management stakeholders and the organics value chain

In-person presentations led to the engagement of 754 stakeholders through 22 events. These presentations targeted key stakeholders within the organics management industry, purchasers, organic residuals users, students who may work in areas centrally or peripherally related to organics management after graduation, and others related to sustainable organics management.

- Program members provided technical support to solid waste industry and municipal stakeholders through calls and emails to approximately 60 people. Through these interactions, non-biased scientific information and decision-making assistance around biorefinery-related issues was provided.
- Support was given on many topics including organics diversion and composting, cannabis waste management, workshop planning assistance, and university research partnership opportunities.
- Technical support relating to sustainable management of organics was provided to ongoing efforts in Washington State including: 2040 Sustainable Materials Management (SMM) pathway regional plan, Whatcom County Community Research Project, and agriculture strategies for the Methow Valley climate action plan.
- In King County, two team members were invited participants to the King County Organics Recycling Two-Day Summit, which collected stakeholder input to identify and prioritize efforts for organics recycling in King County (KC) and surrounding areas. The summit topics led to discussion of specific Waste to Fuels Technology work around compost valuation, biochar co-composting, anaerobic digestion, as well as many other WTFT subjects. As part

of on-going communication between WSU and KC, the County solicited specific assistance and advisory work from WSU on developing a request for proposals (RFP) for a composting feasibility study.

1.2 Extension products

Written extension resources that are continuously available and provide on-demand information are an important complement to in-person presentations and individual conversations. They also provide a variety of products to meet a range of information needs. Work on extension products this biennium included providing ongoing access to documents developed in previous biennia as well as work on new documents relevant to ongoing Waste to Fuels Technology work. In total, all extension products relevant to Waste to Fuels Technology that were produced in the current and previous biennia were viewed a total of 18,020 times in the 2017-2019 biennium.

- In terms of product development, our major effort was the updating of a white paper on carbon sequestration in agricultural systems to include information about municipally-derived organics. The original paper was developed by Georgine Yorgey and colleagues in response to questions raised by the Washington non-profit community related to the potential for agricultural systems to help in climate change mitigation efforts. Strategies such as tillage and perennial plants, among others, were discussed in the white paper, but the paper did not include soil amendment with municipally-derived organics. The work within the Waste to Fuels Technology partnership thus included a review of the literature related to climate change mitigation opportunities provided by municipally-generated organics, including biosolids, compost, and biochar. The white paper was also submitted to the Washington State University Extension system, where it will undergo peer review and then be available to stakeholders throughout Washington and the broader Pacific Northwest.
- WTFT collaborators also recruited and coordinated blog content that resulted in multiple articles for the agclimate.net initiative and Washington State University Center for Sustaining Agriculture and Natural Resources (CSANR) website. These efforts resulted in six articles focusing on biochar, soil, and co-composting.
- In addition, topic pages housing up-to-date online content related to WTFT findings, current and previous efforts, resources and related work relevant to sustainable organics management were maintained on the CSANR website.
- Associated work by partners and other colleagues (carried out with complementary funding) incorporated Waste to Fuels Technology concepts and strategies and resulted in multiple publications, presentations, and activities during this biennium. Relevant products included peer-reviewed journal and extension publications, book publication, a conservation tillage case study, and a multi-day/multi-state workshop.

1.3 Leveraging Waste to Fuels Technology funds to increase impact

The Waste to Fuels Technology partnership plays an important role in engaging faculty across Washington State University in next-generation waste processing issues. Many partners use their work within the partnership as the basis to successfully leverage additional funding that enhances their work and impact in the region. During the 2017-2019 biennium, partners obtained leveraged funding through four grants that provide research focus on various aspects of biochar production and use, successfully obtaining \$357,658 in additional funds. These efforts expand the reach of the WTFT program and allow for the development of timely research on ongoing and new technologies and efforts.

1.4 National and international reach

Research efforts of the WTFT program were shared by partners through publications and presentations that spanned state, regional, national, and international areas. Current biennia peer reviewed publications can be found in multiple journals with wide reach including Biomass and Bioenergy, Waste Management, Carbon, and Science of the Total Environment.

The Waste to Fuels Technology partnership has provided expertise, research, outreach and resources throughout each biennium. Partners continue to disseminate key ideas and findings from the partnership, developing depth in existing relationships, and expanding the number of individuals who are aware of the work. As demonstrated in this summary chapter this work has great reach both in Washington State and beyond, providing support for more sustainable organics management strategies via technical expertise and science-based research knowledge.

Additional detail on available products and efforts are available in the technical report *Extension, Engagement, and Technology Transfer* on the [WTFT 2017-2019 webpage](#) of Washington State University's CSANR.

Chapter 2: Emissions from Washington State Compost Facilities: A Review of Volatile Organic Compound Data, and an Estimation of Greenhouse Gas Emissions

Tom Jobson and Neda Khosravi

2.1 Introduction

Commercial compost facilities are known emitters of a wide range of volatile organic compounds (VOCs), which are a potential concern for air quality managers. The air quality impacts of VOC emissions from composting include the impact of chemically reactive monoterpenes on the formation of ozone and particulate matter, the emission of sulfur compounds which have unpleasant odors, and the emission of compounds that are listed as air toxics by the EPA such as acetaldehyde. The state of California has led the US in studies and regulations focused on reducing VOC emissions from composting facilities. This has been part of a general effort to control photochemical ozone and PM_{2.5} formation in several regions of California that are in non-attainment for these criteria air pollutants. The Washington State Department of Ecology is considering using these California studies to estimate VOC emissions from commercial composters in Washington. These estimates would be used to determine the need for air operating permits.

This review project had three principle components: reviewing existing emissions data collected at Washington commercial composting facilities, reviewing the criteria established for Title V permitting, and comparing greenhouse gas emissions rates from composting to organic debris disposal in landfill.

2.2 Review of VOC emissions data from Washington composting facilities

In the first component, existing emission data collected at commercial composting facilities in Washington by the Solid Waste Program was reviewed in order to:

- Estimate emission rates of VOCs in consultation with the Solid Waste and Air Quality Programs within the Washington State Department of Ecology;
- Evaluate the types of compounds emitted and their potential impact as air toxics and as precursors for photochemical ozone and PM_{2.5}; and
- Evaluate whether total VOC emission rates can be used as reliable estimates of VOC and air toxic emission rates across Washington facilities.

Flux density emission data was collected from six commercial compost facilities in Washington State by Ecology’s Solid Waste Management program from 2010 to 2013 and analyzed by a number of commercial contract labs. The data show that VOC emissions from young windrows were dominated by two compound classes: light alcohols (methanol and ethanol) and monoterpenes (α -pinene and limonene) (Table 1). However, comparisons between facilities were difficult, because different methodologies were used by the different contract labs performing the VOC analysis and individual compounds were not always reported. Methanol, in particular, was not typically analyzed and information about this compound is lacking. Sulfur compound emissions (principally dimethyl sulfide) and emissions of ketones (acetone and 2-butanone) and some aldehydes (acetaldehyde) were also sometimes significant and important contributors to total VOC emissions. Largest flux densities were often observed from the youngest piles, as expected, and flux densities from some sampled older piles were very low. Unfortunately, there was no data collected from a single pile over its life cycle from which a VOC emissions factor could be derived. This should be done for one or more Washington compost facilities to determine a relevant VOC emissions factor for permitting purposes.

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

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

¹ NR = Not reported (no analysis for the compound);

² ND = Not detected; ³ < MRL = detected but below method reporting limit;

* reported as toluene equivalent; ASP = aerated static pile; DMS = dimethyl sulfide

The mixture of VOCs emitted from these compost piles was significantly different from the emissions profile listed in the US EPA Speciate database. The US EPA Speciate profile for green waste compost (#8933) (Figure 1) uses data reported from a California study where monoterpenes comprised only a small fraction of the total emissions (6%), and emissions were dominated by i-propanol (42%), a compound rarely detected in the Washington data. The EPA Speciate profile is different enough from the Washington compost data that it does not appear useful for describing Washington compost emissions (compare Figure 1 and Figure 2), notably the near absence of i-propanol in the Washington dataset while it was the dominant emission in California, and the near absence of monoterpenes in the California data while they were the largest fraction in some of the Washington compost facilities (Table 1).

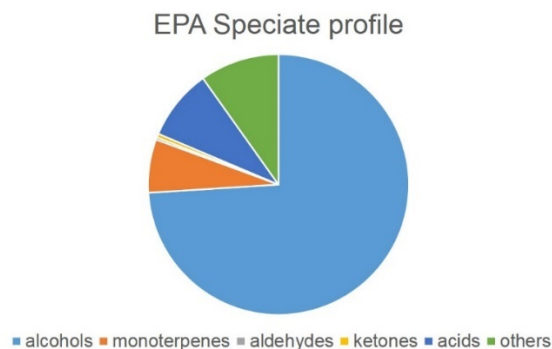


Figure 1: Green waste compost emissions profile from EPA speciate data base, profile number 8933 (EPA, 2016)

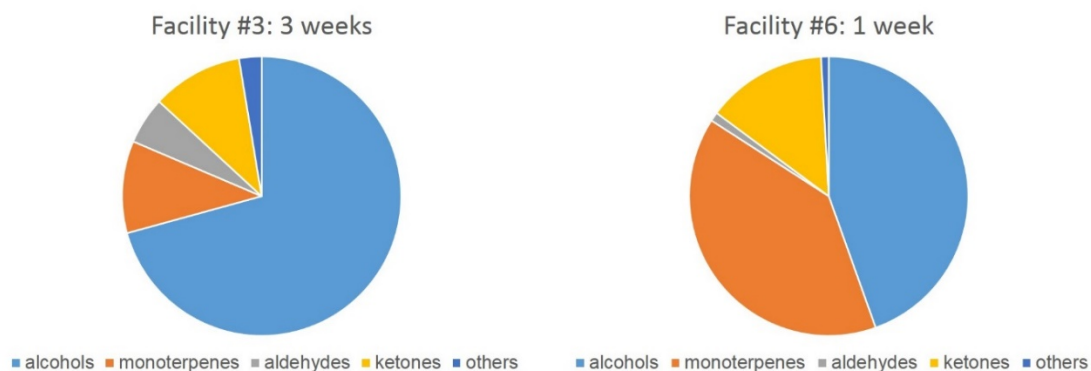


Figure 2: Fractional composition of VOC emissions from the highest flux density samples collected at composting facilities #3 and #6

Future work on VOC compost emissions factors in Washington requires a VOC analysis method that can quantify both light alcohols (methanol, ethanol, i-propanol) and the various monoterpene compounds to get an accurate assessment of total VOC emissions. For the highest emitting samples collected at facility #3 and facility #6, just five compounds (ethanol, α -pinene,

limonene, acetone, and 2-butanone) made up more than 90% of the emissions. This situation was also true at the other facilities; only a few compounds accounted for most of the VOC mass emission rate. Applying a total VOC analysis method, such as EPA Method 25 that detects VOCs as methane, requires an accurate assessment of the VOC emissions composition to convert measured methane back to a mass of emitted VOCs. Method 25 works by converting all VOCs into CO₂ through catalytic oxidation, then reduces the CO₂ into methane and detects methane using a flame ionization detector. Thus, the method is a carbon counter and accounts for carbon mass. It does not detect oxygen or sulfur atoms associated with some VOC compounds. To convert the mass of methane detected back to a mass of VOC requires knowledge of the VOC molecular composition. Hexane is used as a default surrogate compound if there is no information on the VOC composition. If the VOC contains oxygen atoms these must be accounted for to properly convert the measured methane mass into a VOC mass. For compost emissions dominated by light alcohols, such as methanol where oxygen is half the molecular mass, Method 25 would significantly underestimate VOC mass emission rates using the hexane default. The large difference in flux densities observed for piles of different ages at facility #5 (Figure 3) also suggests the need for continuous VOC monitoring or more frequent grab sampling of VOC emissions during the first week to properly capture rapid changes in flux density with pile age.

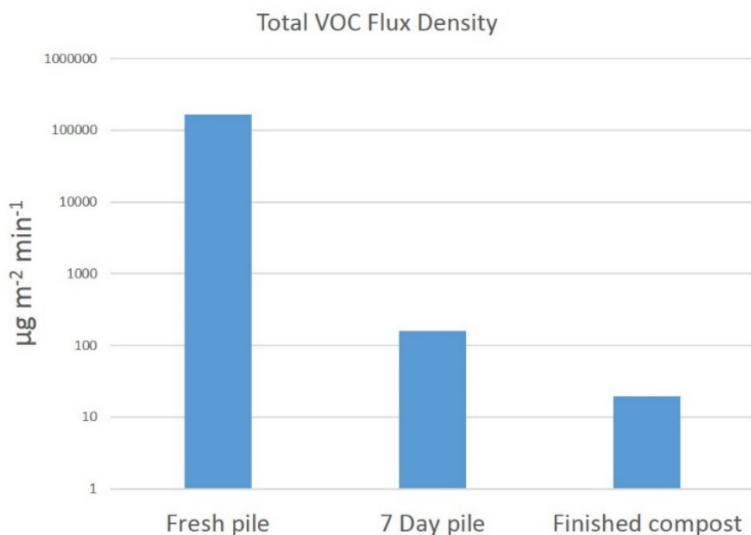


Figure 3: Comparison of total VOC flux density from windows at different ages at facility #5

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

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

defined as 100 tons of VOCs per year. An estimate of annual VOC emission rates for Washington composting facilities was made based on annual reported waste material throughputs and VOC emission factors derived from California studies of green waste composting. For California composts, an average emission factor of 3.58 lbs VOC per wet ton has been determined (CARB, 2015). While it would be more prudent to apply a Washington average emissions factor, due to the issues already discussed, in the absence of such information, the California average can give some sense of the number of facilities that may be near the threshold, and for whom additional information may be needed. Using the California average total emissions factor, it was calculated that eight composting facilities in Washington have the potential to emit more than 100 tons of VOC per year and thus potentially be subject to EPA Title V permitting. However, assuming the simplest emission control approach—covering the windrow with a cap of finished compost, with an estimated 75% efficiency, can significantly reduce VOC emissions (Table 2).

Table 2: Estimated annual total VOC emission rates assuming California Air Resource Board green waste emission factors and 2017 annual throughputs. Uncontrolled emissions assumes no emissions control technology, while controlled emissions assume that emissions are reduced by 75%, compared to uncontrolled emissions

Rank	Facility	Location	Estimated VOC emissions (tons / yr)	
			uncontrolled	controlled
1	Cedar Grove Composting Co. Maple Valley	Maple Valley	437.8	126.8
2	Cedar Grove Composting, Inc.	Everett	264.3	76.6
3	WSU Compost Facility	Pullman	180.7	52.3
4	Barr-Tech Composting Facility	Sprague	162.0	46.9
5	LRI Compost Factory	Puyallup	139.7	40.5
6	Lenz Enterprises Inc	Stanwood	138.6	40.2
7	Silver Springs Organics Composting LLC	Rainier	121.0	35.1
8	Pacific Topsoils - Maltby	Woodinville	102.2	29.6
9	Boise White Paper LLC	Wallula	95.3	27.6
10	Natural Selection Farms Composting	Sunnyside	93.9	27.2
11	Pierce County (Purdy) Composting	Gig Harbor	81.0	23.5
12	Dirt Hugger LLC	Dallesport	69.1	20.0
13	La Conner WWTP Skagit Co SD #1	La Conner	61.4	17.8
14	Sunnyside Dairy	Sunnyside	56.7	16.4
15	Green Earth Technology (Compost)	Lynden	50.4	14.6
16	Royal Organic Products	Royal City	44.0	12.7
17	Ovenell Farms Composting Facility	Quincy	40.2	11.7
18	Thomas Farm Agricultural Composting	Snohomish	37.8	11.0
19	Bailand Farms Yardwaste (Bailey)	Snohomish	32.1	9.3
20	Stemilt World Famous Compost Facility	Wenatchee	31.3	9.1
21	North Mason Fiber Co.	Belfair	29.9	8.7
22	Olympic Organics LLC	Kingston	28.4	8.2
23	Cheney WWTP & Compost Facility	Cheney	23.6	6.8
24	Skagit Soils Inc	Mount Vernon	19.4	5.6
25	Lawrence Farms LLC Compost Facility	Royal City	12.9	3.7

2.4 Comparison of greenhouse gas emission rates from landfilling and composting

The third task involved comparing greenhouse gas emissions rates from composting to organic debris disposal in landfill. A simple comparison was made comparing estimated methane (CH₄) and nitrous oxide (N₂O) emissions from composting or landfilling organic waste using the EPA’s Waste Reduction Model (WARM). Landfills equipped with landfill gas (LFG) collection systems were analyzed separately from landfills without LFG recovery systems to account for power production benefits. It was concluded that composting as an alternate waste management strategy will likely decrease greenhouse gas emissions from organic waste compared to landfilling (Table 3), with most of the reduction attributed to two factors: removing food waste emissions of methane from landfills and the soil carbon storage benefit of applying compost to soils. The analysis contains uncertainties related to N₂O and CH₄ emission factors related to compost, which vary widely in the literature, and to quantifying a soil carbon storage benefit when compost is applied to agricultural soils. When applied to soils, the model assumes some fraction of the compost carbon is stable in the soil for many years but this fraction may vary depending on soil type and agricultural land management practices. Establishing emission factors for N₂O and CH₄ from Washington State compost facilities would be needed to more accurately quantify greenhouse gas reduction benefits of composting compared to landfilling.

Table 3: Net CO₂ equivalent emissions in mega tonnes of carbon dioxide equivalent (MT CO_{2eq}) from diversion of organic waste from landfills to composting for landfills with and without landfill gas (LFG) collection systems

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

Additional important detail on this work is available in the technical report *Emissions from Washington State Compost Facilities: A Review of Volatile Organic Compound Data, and an Estimation of Greenhouse Gas Emissions* on the [WTFT 2017-2019 webpage](#) of Washington State University’s CSANR.

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Chapter 3: Differentiating the Value and Cost of Compost Across Likely Farm Use Scenarios in Western Washington

Karen Hills, Michael Brady, Georgine Yorgey, and Doug Collins

Despite the expansion of municipal compost production in western Washington, demand from agricultural end users for this compost has lagged. This could be for two potential reasons:

- The value of compost to agriculture is underestimated or inadequately understood, and/or
- The value of compost in farming is lower than the cost of producing it.

Ideally, results from extensive field trials and long-term agronomic and horticultural research would provide evidence that could quantify the production relationship between compost and yield. However, while there are published studies of this nature, they are not extensive enough to develop precise compost value estimates.

The value of compost to a certain user depends on both agronomic (i.e. crop and soil) and economic factors. This project focused on quantifying the extent to which compost values vary depending on economic factors, which includes prices for the crop and other inputs. Information is available on price ranges for most inputs and crops, so it is easy to run realistic best and worst case scenarios. Uncertainty in agronomic relationships is handled by assuming that the low end of the range may be near zero for compost, so therefore focused on establishing a realistic “good” estimate that can elucidate whether or not there are likely to be situations in which the value exceeds the cost.

3.1 Compost use scenario development

Scenarios were developed to illustrate potential cost and value for compost used in the production of four specific crops in western Washington: winter wheat, blueberries, raspberries, and direct market mixed vegetables (Table 4). The purpose of these scenarios is to examine the value of compost in specific situations where there is likely to be an impact, rather than assuming it has equal value across all agricultural production areas. These scenarios represent realistic “good case” scenarios - those combinations of location (Carnation, Enumclaw, and Marysville, which are close to Seattle metro area for lower transportation costs), high application rates, and, with the exception of wheat, high value crops that offer the best chance of maximizing value for the compost. Compost application rates and projected yield increases were chosen that seemed possible based on the available literature for each crop type. The scenarios are meant to illustrate a potential range of compost value for different cropping systems based on varying crop values, not to prescribe specific application rates or predict associated yield effects. Fertilizer

replacement value was calculated at \$18.84 for one yard of compost, based on conversations with regional suppliers.

Compost, Transportation, and Spreading Costs

Cost of compost was estimated by getting quotes from a local compost producer on the cost of compost, delivered, to farmers in the specific locations chosen. The cost was the same for each of the three locations, and was \$20 per yard, delivered. Spreading cost was assumed at \$15 per ton (wet weight). Eastern Washington spreading costs are typically \$9-10 per ton for compost or other bulky material with a minimum of 120 acres (Thad Schutt, Cedar Grove, personal communication). Though some economy of scale could be achieved if custom spreading was more common in western Washington, it's unlikely that the cost would ever get as low as it is in eastern Washington because of the smaller field size that is common west of the Cascades.

Calculation of Net Returns

Crop enterprise budgets were used to calculate per acre net returns without compost and with compost, taking into account assumed yield increases and increases in variable costs (harvesting) associated with these yield increases. The difference in net returns was used to calculate the value of a yard of compost. See the narrative section for specific details on the crop enterprise budgets used for each scenario. For the wheat, raspberry, and blueberry scenarios, net returns are calculated in terms of prices received for a conventional crop sold through standard marketing channels. The budgets for the mixed vegetable farms are based on direct market sales.

Additional Assumptions

Along with the assumptions outlined in the above section, these assumptions were made in the scenarios in Table 4:

- 1.) The compost being applied is generic and does not account for value changes from more nutrient rich or specialty compost properties; and
- 2.) Compost is applied to soils that will show a yield response. We do not consider how compost value varies with soil conditions, though we recognize that soil conditions and past management history will play a major role in the potential value of compost in a given crop production setting.

Table 4: Compost application scenarios for different crop types in western Washington

	Crop	Application rate ¹ (dry tons/acre)	Per acre net returns without compost (standard fertilizer regime)	Assumed yield increase	Per acre net returns with compost	Per acre increase in net returns	Compost value ² (\$/yard)	Compost cost ^{3,4} (\$/yard; including delivery and spreading)
Low Value	Soft white winter wheat	20	\$644	+10%	\$708	+\$64	\$0.85	\$27.05
High value	Blueberry	20	\$2,719	+10%	\$4,233	+\$1,514	\$19.93	\$27.05
High value	Raspberry	7.5	\$3,913	+10%	\$5,066	+\$1,153	\$38.43	\$27.05
High value	Direct market mixed vegetable ⁵	20	\$12,549	+20%	\$17,398	+\$4,849	\$63.80	\$27.05
High value	Direct market mixed vegetable ⁶	20	\$16,144	+20%	\$20,581	+\$4,437	\$58.38	\$27.05

¹ Relationship between tons and yards of compost: Assumed compost bulk density of 44.5 lbs/cubic foot or 1200lbs/cubic yard and moisture level of 44%. (Moisture from Cedar Grove analysis: 39.8% [June], 48.6% [January]. average = 44%). 20 dry tons = 76 cubic yards; 7.5 dry tons = 30 cubic yards.

² Compost value is based on assumed yield increase. Actual compost value will be dependent on soil type and management history, which are not accounted for in this scenario.

³ Cost quoted by Cedar Grove Compost as price for standard fine grade compost at cost that they could provide to farmers (\$15 per yard FOB at Cedar Grove). Freight to Enumclaw, Marysville, or Snoqualmie is an average of \$5/cubic yard based on a full load of 50 cubic yards. Total cost delivered is \$20/yard.

⁴ Spreading cost is assumed at \$15 per ton (wet weight). Eastern Washington spreading costs are typically \$9-10 per ton for compost or other bulky material with a minimum of 120 acres (Thad Schutt, Cedar Grove, personal communication). Though some economy of scale could be achieved if custom spreading was more common in western Washington, it's unlikely that the cost would ever get as low as it is in eastern Washington because of the smaller field size that is common west of the Cascades.

⁵ Net returns based on Colorado enterprise budget.

⁶ Net returns based on British Columbia enterprise budget.

3.2 Discussion

When the crops are soft white winter wheat and blueberry, the value of the compost is less than the cost (\$0.85 and \$19.93 per yard, respectively, compared to \$27.05 per yard estimated cost, including delivery and spreading). For the raspberry and mixed vegetable scenarios, the situation is different, with the value of compost exceeding the cost (\$38.43, \$63.80, and \$58.38 per yard for raspberries, and the two mixed vegetable scenarios, respectively, compared to a cost of \$27.05 per yard, including delivery and spreading).

The scenarios above represent “good case scenarios” for a number of crops grown in western Washington, and provide a framework for farmers to adjust the assumptions based on their own situation and knowledge. While these examples are not comprehensive of every crop grown in every location in western Washington, they do demonstrate two important points:

- Under reasonable assumptions, the value of compost appears likely to exceed cost for some crops grown in the region, especially for higher value crops in locations that minimize transport costs.
- Compost can have a wide range of values, depending on the cropping system to which it is applied.

Two important items should be noted in these scenarios: First, while the increase in net returns per acre was greater for the blueberry scenario than it was for the raspberry scenario, compost was more valuable in raspberries because only 7.5 dry tons was applied as opposed to 20 dry tons for blueberries. This difference highlights the importance of further field testing to refine application rate and yield increase assumptions. Second, while these scenarios assume transportation to locations within King and Snohomish Counties, most raspberry and blueberry production in western Washington actually occurs further away, in Skagit and Whatcom Counties. In these areas, municipal compost would likely be competing with compost derived from dairy manure. These complexities are not addressed in this report, but would be a relevant topic for future study.

Hopefully, this report stimulates a greater sharing of information on practices and benefits of compost on crop production within specific production systems. It also sheds light on the agronomic dynamics that most need to be clarified if funding for field trials were available. In particular, field trials could provide more information on the effect of soil properties and past management on yield response due to municipal compost application in western Washington soils and allow more finely tuned estimates of yield responses for particular crops based on site-specific conditions (i.e., prediction of whether the crop yield response will be low, medium or high based on site conditions and management history).

This report also provides insights that can be used to consider whether changes to the overall municipal waste and compost collection program would increase the use of compost on farms. These changes are considered in more depth in *Chapter 4, Lessons for Compost Policy: What Can Recycling Policy Tell Us?* These could include subsidies to farmers for purchasing compost, or subsidies of equipment used to spread compost (e.g., via equipment sharing programs) to name a few. In particular, funding provided to shared compost spreaders may be worth additional investigation as it would address high barriers to entry for using compost, given that custom spreading services are not available in all areas, and are quite expensive in others. It should be noted that programs exist in some conservation districts in western Washington to loan out manure/compost spreaders to help farms with manure management, but it is not clear whether they are being used by crop farmers for compost application.

Compost quality is an important driver of demand from end users. Concerns that farmers have over contamination of fields can outweigh the potential benefits to soil health or yield resulting from application of municipal compost. This has been perceived as a significant enough issue to motivate the creation of a working group to identify solutions (see the report *Washington State Organics Contamination Reduction Workgroup: Report and Toolkit*, 2017). Though this issue is not unique to agricultural end users, it may be especially problematic for them due to their price sensitivity (thus unwillingness to pay for further processing to remove contaminants), sensitivity to how their products are perceived, and preference for giving up modest gains to avoid risk of significant losses, even if that risk is small. On the last point, most farming operations are sole proprietorships with limited financial resources. It is therefore reasonable that they would give up a guaranteed modest improvement in profitability to avoid a calamitous outcome even if it is unlikely. This is probably how many farmers view the trade-off between using and not using compost, when contamination is a concern.

Additional important detail on this work is available in the technical report *Differentiating the Value and Cost of Compost Across Likely Farm Use Scenarios in Western Washington* on the [WTFT 2017-2019 webpage](#) of Washington State University's CSANR.

3.3 References

Washington State Organics Contamination Reduction Workgroup: Report and Toolkit. 2017. https://static1.squarespace.com/static/585c2db75016e175c9d685b7/t/5936deae1b631baae4303748/1496768177591/Washington+State+Organics+Contamination+Reduction+Workgroup_FINAL.pdf

Chapter 4: Lessons for Compost Policy: What Can Recycling Policy Tell Us?

Michael Brady

4.1 Introduction

Municipal compost collection programs across the US, including those in western Washington, have expanded significantly in recent years. This expansion has been driven by the willingness of many citizens to pay to reduce their environmental impact by having their compostable waste collected separately— and by policies, typically at the state-level, that require recycling or recovery of organic wastes. In contrast to a typical market, the amount of compost produced is determined by population, household food purchases, landscaping, and waste collection programs, which are *supply side* factors. It does not depend on *demand side* factors, such as how much compost farmers, gardeners, landscapers or other end users are willing to purchase at different price levels.

The objective of this report is to provide perspective on whether interacting with the waste and compost stream at some of the various points along the system (shown in Figure 4) may be more effective than others. The ideal way to study this would be to create a large data set on compost programs in many locations with information on outcomes (e.g., cost, quantity of organics sent to landfills, quantity composted) and perform a statistical analysis to see which program approaches were more effective. Unfortunately, this is not feasible with compost programs because they have only been implemented recently in the US. A next best approach is to review studies that inform compost policy even though they did not explicitly study compost. In this case, two major bodies of work were reviewed for their insights relevant to compost.

The first relevant area of study related to a particular challenge that has been identified by Washington organics management stakeholders relating to organics management – that of encouraging more compost use by agriculture (Yorgey et al., 2016). With respect to this challenge, this project focused on the body of knowledge relating to the adoption of alternative practices and technologies by farmers. There are important lessons from this literature on how farmers’ weigh potential gains relative to potential losses when there is uncertainty. Findings point towards a specific policy approach.

The second major area of study reviewed is the sizeable literature analyzing recycling programs. Compost is a type of recycling program that focuses on organic materials as opposed to glass, metal, or some other material. While the nature of the material being reused is an important consideration in how to design a program, there are significant similarities. Because general recycling programs have existed for much longer, there is a chance to learn from what has worked and not worked within these programs. The gold standard for informing compost policy would be to have data on compost program design and various outcomes. Hopefully, this type of

data set will be available in the future. In the meantime, a review of policies related to recycling provides some relevant insights.

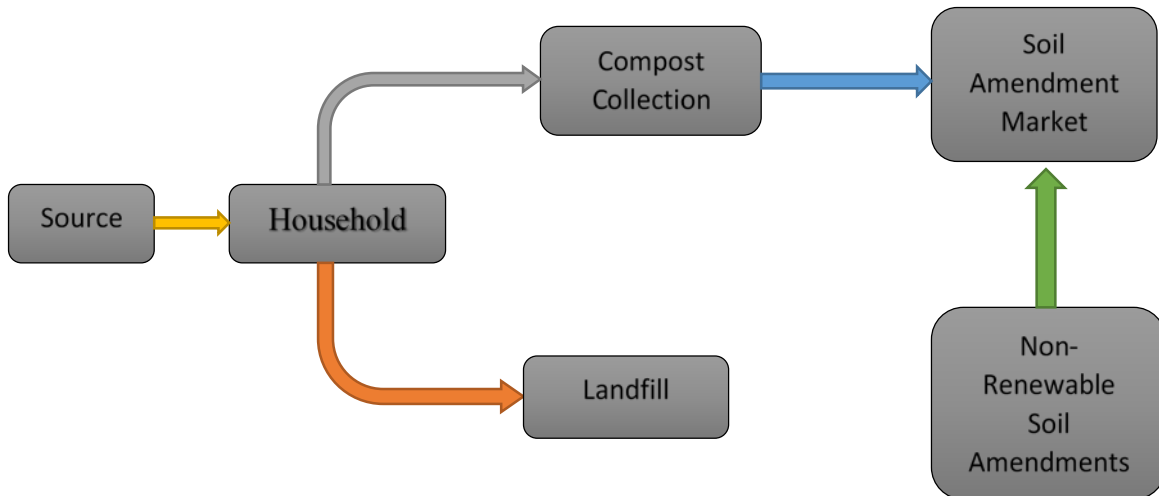


Figure 4: The household waste and compost generation pathway

4.2 Background relating to the approach used

Two possible different questions that can be asked include the “why” and the “how” of implementing organics diversion and recycling programs. This report pertains to the “how” question, which asks the question of how to design a recycling program to achieve a certain goal (such as a certain level of organics diversion) most efficiently (lowest cost).

In contrast, an economic approach to the “why” question would be to compare the benefits of the program relative to the costs. Within this approach, analysis should attempt to include all costs including externalities, not just those that are currently reflected in prices. For example, if compost used by farmers leads to a reduction in inorganic fertilizer use, and therefore a reduction in greenhouse gases, this should be quantified as a benefit even if there is not currently an active carbon market. Benefits and costs should also reflect impacts that occur in the future. Including all costs and benefits is, not surprisingly, difficult to do in practice, especially when considering the future.

The challenge of considering future repercussions of actions taken now underpins the concept of sustainable development. The idea was first formalized by the Brundtland Commission in the book *Our Common Future* (1987), and it was very successful in galvanizing the scientific community towards the goal of developing integrated models of biophysical and human systems that would inform society as to whether current paths of resource use were sustainable. One key

point made through this work was that existing patterns of industrial production, consumption, and technology were often developed without any regard for the finiteness of resources (McDonough and Braungart, 2000; Garcia-Perez, 2013). Another was that there was enough low hanging fruit (actions that could be relatively easily taken that would substantially improve sustainability) that sustainability could be achieved without a radical reduction in current living standards.

An alternative framework that leads to the same general policy prescription of acting now to more responsibly use natural resources was provided by economist Martin Weitzman (1974) fifteen years before the Brundtland Commission. Weitzman's probabilistic/risk management approach is more straightforward than sustainable development because it deals with uncertainty about the future head-on. Weitzman argued that catastrophic events, like a collapse in food production due to climate change, deserve action now even if the probability of catastrophe is low because societies voice through political action an aversion to risk.

Sustainability has served as a fundamental rationale for the adoption of recycling and organics diversion. However, it is also fair to say that the concept of sustainability has not proven to be *on its own*, enough to design policy, though it certainly informs it. There are a few reasons for this. First, sustainability must contend with the inherent uncertainty relating to impacts that occur in the future. Second, sustainability is usually not a binary choice (with one clearly "sustainable" choice and one entirely "unsustainable" choice) (Neumeyer, 2003). Instead the question is more likely one of degree: different policies may lead to slightly cleaner air and water, but are unlikely on their own to be the difference between humanity thriving and mass extinction.

Because the question is one of degree, there is a necessary and important *political* process by which societies make these types of decisions. When faced with such uncertainty over the future, it is perfectly reasonable for individuals to express through a political process a desire to err on the side of caution in terms of preserving the environment and natural resources, but this political process is distinct from economic analysis. An economist can give a menu of efficient policies, but only the political process can choose which option best meets priorities. This review was focused on just that: given organics diversion goals, what policies can be used to meet these goals at relatively low cost with relatively few unintended consequences.

4.3 Enhancing end use of municipal compost by agriculture

Before proceeding to consider studies that look at recycling program design broadly, there is an action more focused on compost that should be considered. Currently, the structure of composting is such that there is no direct subsidy to the purchaser of compost. There is an implicit subsidy in that composters receive a payment for collecting compost, which should lower its cost. If more public funding were available to subsidize compost purchases, one impactful way to utilize such resources is likely to invest in compost spreading equipment. The challenge for some farms is that compost requires a specific type of machinery that they do not already own. Even if compost was heavily subsidized, it would still be necessary for a farmer to

invest in new equipment. There are many research articles suggesting that this creates a huge barrier to enhanced use.

The basic idea behind these studies is that most farmers have been shown to be “loss averse,” meaning they are very hesitant to expose themselves to losses if the gains are uncertain (Bocqueho et al., 2013; Liu, 2013; Menapace et al., 2013). There are many aspects of compost that fit this situation. As discussed in *Chapter 3, Differentiating the Value and Cost of Compost Across Likely Farm Use Scenarios in Western Washington*, there is still significant uncertainty over how much compost will increase yields in different situations. Farmers may be willing to take this risk if compost is cheap enough. However, even if compost were really cheap, many would still have to go out and make a significant investment in new machinery. This introduces the potential for large losses if they don’t see a benefit. Given this, one strategy that would make a loss-averse farmer more willing to take a gamble is any program that eliminates the need for a capital investment in machinery. If spreading equipment were made available (such as through partnerships with conservation districts or other appropriate local entities), even loss averse farmers might not mind spending some money on compost to give it a try. If it does not work, they are only out the money they spent on that one application of compost. What a farmer wants to avoid is being stuck with a loan on a piece of machinery that turns out to not be useful. Loss aversion is also one likely explanation for why many farmers are so concerned about contamination risk from compost. They feel more strongly about avoiding losses than seeking out gains – and once imported onto the farm, contaminants may be difficult or impossible to remove. Thus, this framework also supports continued action to reduce the risk of contaminants such as plastic or glass in compost.

4.4 Economic perspective on municipal solid waste management

From the mid-1980s to mid-2000s, the number of households in the US with access to organized recycling services as part of their waste collection services went from almost none to 48% of the population, which was the outcome of more than 8,000 municipalities developing recycling programs (Kinnaman, 2006). The result has been consequential. Kaufman et al. (2004) estimates that diversion rates of the waste stream to recycling exceeded 30 percent by the year 2000. This diversion rate has fallen abruptly in the last few years, due to a series of decisions in late 2017 and 2018 that severely restrict China’s acceptance of recycled materials from other countries. This shift, driven in a large part by concerns relating to the quality and contamination of the recyclable materials, has sent shockwaves through recycling programs causing many municipalities to substantially alter the types of materials that are collected.

One challenge in designing a recycling program is that there are lot of different ways to create “carrots” (incentives) and “sticks” (punitive measures) to reduce waste. The substantial variation in recycling programs across locations makes it possible to empirically analyze the relationship between program design and outcomes. The objective of this report is to review the recycling

literature to identify key findings, discuss policy implications of those findings, and then consider to what degree these lessons and policy prescriptions also apply to compost program design. The aim is to complement earlier studies that have focused on technical aspects of compost production and use on farms, as well as those that have considered possible policy changes to composting programs.

Compost programs are at a similar stage as recycling programs were in the mid-1990's, in the sense that they have been implemented in many municipalities, but it is still relatively early in terms of assessing the effectiveness of different approaches. Thinking more broadly about approaches to solid waste and its recycling, a number of approaches to reduce the landfilled waste stream have been implemented, including:

- Fees: also called pricing or unit based pricing;
- Deposits/refunds;
- Advance disposal fees (fee added to consumer product to account for disposal costs);
- Taxes;
- Tax-subsidy combinations;
- Standards;
- Location of service provision (e.g., curbside versus drop-off); and
- Regulations with fines.

All of these policy levers that have been used for recycling could also be applied to composting, though there are also some important differences that should be kept in mind. For example, any recycling policy that relies on the material maintaining its physical structure will be less relevant to compost. Another thing to keep in mind is that the demand for compost is less well developed than some recycling materials. That said, it is also the case that there are substantial differences between industrial processes and markets for paper versus glass versus aluminum.

4.5 Learning from the economics literature on recycling

Recycling programs spread across much of the US, Europe, and parts of Asia in the 1990s. Seeing differences in approaches, this naturally led an academic literature in economics and public policy considering how to best design a waste program that incorporates recycling. Most of the published studies were conducted between 1996 and 2006. Since about 2006 until recently recycling programs have been relatively mature with less policy experimentation, and the academic literature has been less active as a result. However, there is likely to be a renewed interest in studies reassessing recycling programs in the next few years in response to the recent

shift in recycling programs, mentioned above, as the result of China's unwillingness to accept recycled materials.

Summary of Findings from the Recycling Literature

Taken as a body, the literature on recycling suggests several lessons relevant to composting programs:

- Empirical analysis is critical for projecting the effect of changes in policies. We are on the cusp of having enough experience with municipal composting programs that empirical studies could be performed to evaluate the relationship between approach and outcomes.
- Volume or quantity fees for waste disposal are the simplest to implement, but households are not that *responsive* to fees. This is particularly the case for the cross-price effect of garbage fees on recycling rates (i.e., increasing garbage fees in the hopes of enhancing recycling rates). What this means for compost is that increasing the price that households pay for disposing of waste to landfills may not have much of an effect on the amount of compostable waste diverted from the waste stream.
- Waste production increases with income, but at a fairly slow rate (i.e., income elasticity of demand for waste is positive but is highly inelastic). The implication for compost is that there is little reason for pessimism that higher income households will generate a lot more compostable waste than lower income households.
- While a significant departure from current policy that would need further study, a tax-based approach (i.e., a landfill tax) might provide a statewide context that would encourage some municipalities to adopt composting (those in which the benefits outweigh the costs), without requiring action for those municipalities where costs would outweigh the benefits. This finding is important to contrast with the second bullet point. While *households* are not that responsive to prices charged for putting waste in landfills, *waste management entities* (public or private), are very responsive. This makes sense given that households have lots of other day-to-day concerns, while waste management entities have most of their attention focused on these issues. Therefore, a landfill tax is more likely to increase the amount of compostable waste diverted from landfills than a higher waste fee charged to households.
- There are some interesting results showing that deposit-refund policies are effective, but they probably have less relevance for compostable materials.
- Though the literature on this topic did not provide specific evidence related to the effect of standards on contamination levels, there was also some indication that policies focusing on the upper end of the waste generation pathway are likely to be more effective than the current mid-stream focus on the household. It would stand to reason that a focus on standards may be helpful to reducing contamination, and thus increase the *value* of the compost product. There is increasing political momentum to outright ban waste that is especially problematic once it enters the waste stream. Plastic bags are the most well-known, but there is also movement on

plastic straws. Plastic utensils are another potential target. It is possible that the most impactful change for developing compost demand is the elimination of a few waste types that cause contamination if they enter the compost stream.

4.6 Conclusion

In this report, two demand side interventions relevant to increasing the use of compost by agriculture were considered: subsidies to farmers for purchasing compost, or subsidies of equipment used to spread compost (e.g., via equipment sharing programs). Based on the literature review, a subsidy to farms using compost is expected to have little effect on demand. However, subsidizing custom-hire spreaders or programs providing access to shared equipment may be effective for increasing compost use on farms. It should be noted that programs exist in some conservation districts in western Washington to loan out manure/compost spreaders to help farms with manure management, but it is not clear whether they are being used by crop farmers for compost application. Thus, in at least some areas, there may be an opportunity to build on these existing programs at relatively low cost.

In addition, broader policies relating to composting more generally were investigated, though limitations in the available literature led to an investigation of policies relating to recycling, of which organics recycling is one unique example. The economics literature on recycling arrived at a general consensus that policy intervention at the middle of the waste stream, or the household curbside, is an inefficient approach even if all social costs are properly priced in. The reasons for this include a lack of incentive to get final recyclable product to end users, and the fact that households are relatively non-responsive to changes. The suggestion to move policy intervention to the ends of the stream by focusing on contaminant reduction/product standards and landfill taxes would be a radical change to the dominant curbside tipping fee approach used now. It may be that the conclusion is that compost is substantially different enough from other recycling to nullify this policy recommendation. However, there is enough similarity to make it at least worth considering.

More information on the demand side of municipal compost is available in *Chapter 3: Differentiating the Value and Cost of Compost Across Likely Farm Use Scenarios in Western Washington*.

Additional important detail on this work is available in the technical report *Lessons for Compost Policy: What Can Recycling Policy Tell Us?* on the [WTFT 2017-2019 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

David Gang, Douglas Collins, Tom Jobson, Steven Seefeldt, Anna Berim, Nathan Stacey, Neda Khosravi, and Wendy Hoashi-Erhardt

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) is a promising efficient low-cost solution to that may reduce gas emission during and after the composting process (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 study in 2017 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, grown in soil with biochar added. This biennium’s work builds on these earlier results and seeks to reproduce and expand 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 WSU Compost Facility, Pullman, Washington. Gas emissions were sampled. Emissions were also sampled from laboratory co-composting experiments under more controlled conditions. The resulting co-compost, along with the compost, and biochar were used to amend soils for greenhouse and field trials. Additional detail including biochar and compost characteristics is provided in the technical report for this project (Gang et al., 2019).

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.

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 yd³) 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 yd³) 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 5).



Figure 5: 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 6). 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 ($\mu\text{g}\cdot\text{m}^{-2}\cdot\text{hr}^{-1}$) was calculated. Analysis of the field samples from Lenz and WSU revealed that emitted monoterpene compounds, principally α -pinene and limonene, were a large fraction of the total VOC emissions flux density. The WSU compost also had significant emissions of S-containing compounds, principally dimethyl disulfide and dimethyl sulfide, which in some samples was 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.

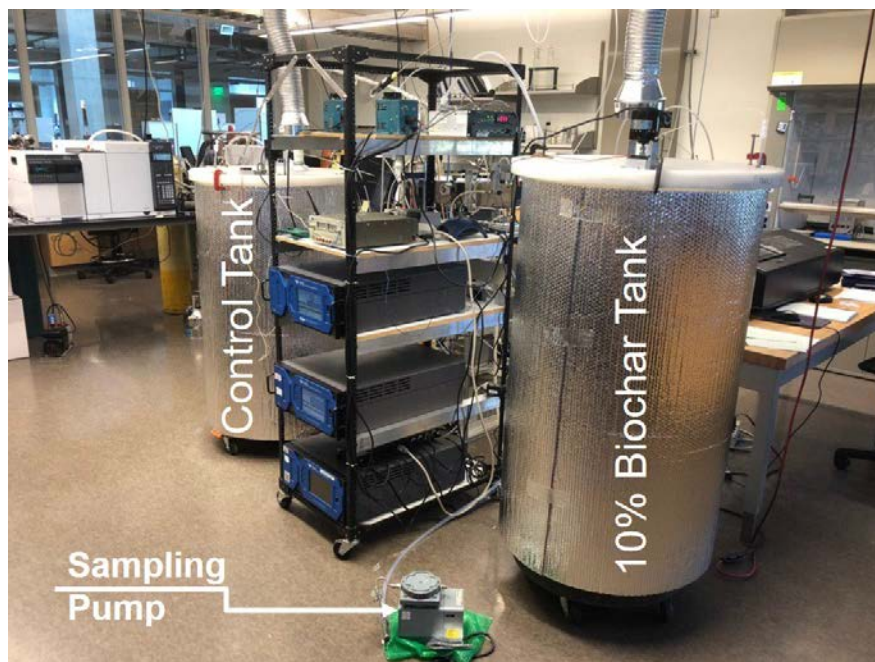


Figure 6: Photograph of compost lab test setup

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, dimethyl disulfide, and several other compounds that have not yet been identified (PTR-MS ion signals at m/z 69, 83, and 135) (Table 5). Biochar was not shown to be as effective at reducing emissions of hydrogen sulfide (H_2S) and dimethyl sulfide (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 5: Summary of VOC emissions from compost for both trials in control and biochar co-composted tanks

Compound	Sampled Tank	1 st Trial		2 nd Trial	
		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 ₂ S	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 7). 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.

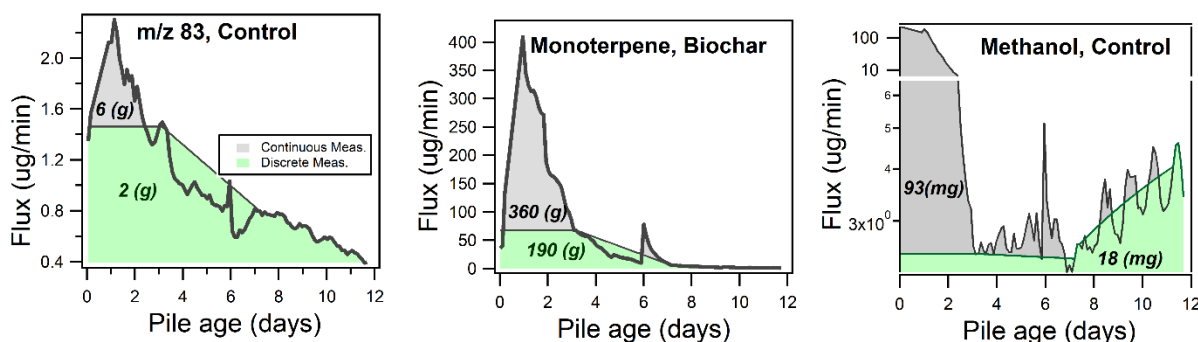


Figure 7: Comparison of emission profiles of m/z 83, monoterpene, and methanol estimated by continuous measurement and discrete measurements. Black trace shows measured fluxes. Area under this curve is the mass emitted. Green shading illustrates the area of the flux profile if discrete sampling at day 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, and Lenz composts. Biochar and compost sources are summarized in Table 6. Note that the biennium timeframe allowed for only an initial year of data collection, while normally two to three years of data collection are usually considered to be needed to capture variability due to weather conditions and other factors, as was alluded to above. Further data collection is anticipated for coming years.

Table 6: Summary of biochar and compost sources used in the experiments described in this report

Experiment	Biochar source	Compost source	Measurements
Emissions	Oregon Biochar Solutions	WSU & Lenz (on-site)	Air emissions
Basil – field (Colbert)	Amaron Energy	WSU 2017 Footehills	fresh plant mass phytochemical composition
Basil – greenhouse	Amaron Energy Oregon Biochar Solutions	WSU 2017 WSU 2018 Lenz	fresh plant mass phytochemical composition
Strawberry – greenhouse	Amaron Energy	WSU 2017	yield berry number single berry mass
Strawberry – field (Puyallup)	Oregon Biochar Solutions	Lenz	yield soil properties
Potato – field (Mount Vernon)	Oregon Biochar Solutions	Lenz	yield soil properties

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 8). 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 the technical report).

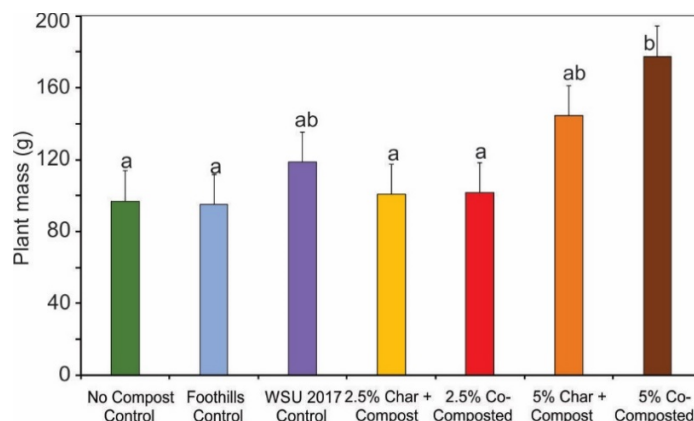


Figure 8: Effect of amendments on fresh mass of Genovese sweet basil grown at Foothills Farm (Colbert, Washington). Average plant mass \pm s.e.m. (n=10) are shown. Same lower case letters indicate there was no significant difference between treatments ($p < 0.05$). Note: “char + compost” indicates char added post-composting, while “co-composted” indicates char added prior to composting.

A greenhouse study with two basil cultivars and three biochar-amended composts produced different results for the two basil cultivars, but overall showed only moderate increases in biomass production. (Data is shown in Gang et al. [2019].) The vegetative yield increase observed in the greenhouse experiment in 2017 was not well reproduced with the same biochar-compost mixture in this study; however, the plant growth conditions were significantly different between the two years, with the latter plants grown much longer and past ideal fresh cut harvest date. 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.

Field trials involving strawberry and potatoes were conducted at Washington State University Research and Extension Centers in Puyallup and Mount Vernon, Washington, respectively. 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 (data shown in the technical report). 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 9).

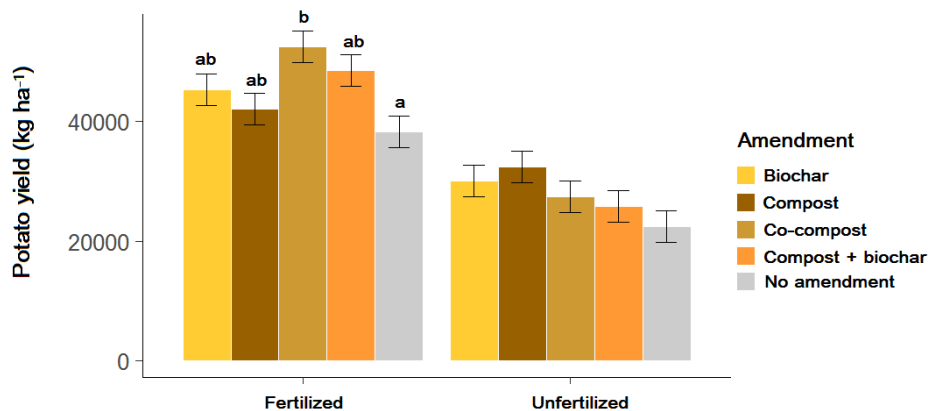


Figure 9: Potato yield at Mount Vernon as affected by fertilizer and amendment. Note: significant differences between treatments were observed only in fertilized plots

A greenhouse-based experiment with strawberries indicated productivity increases were observed in some of the biochar-compost treatments but were overall only moderate, and require further studies for reliable conclusions (Figure 10). In this experiment, additional supplementation with inorganic fertilizer interfered with the effects of biochar-compost application. Importantly, biochar addition, either co-composted or added at time of planting, had no negative impact on strawberry yield or quality.

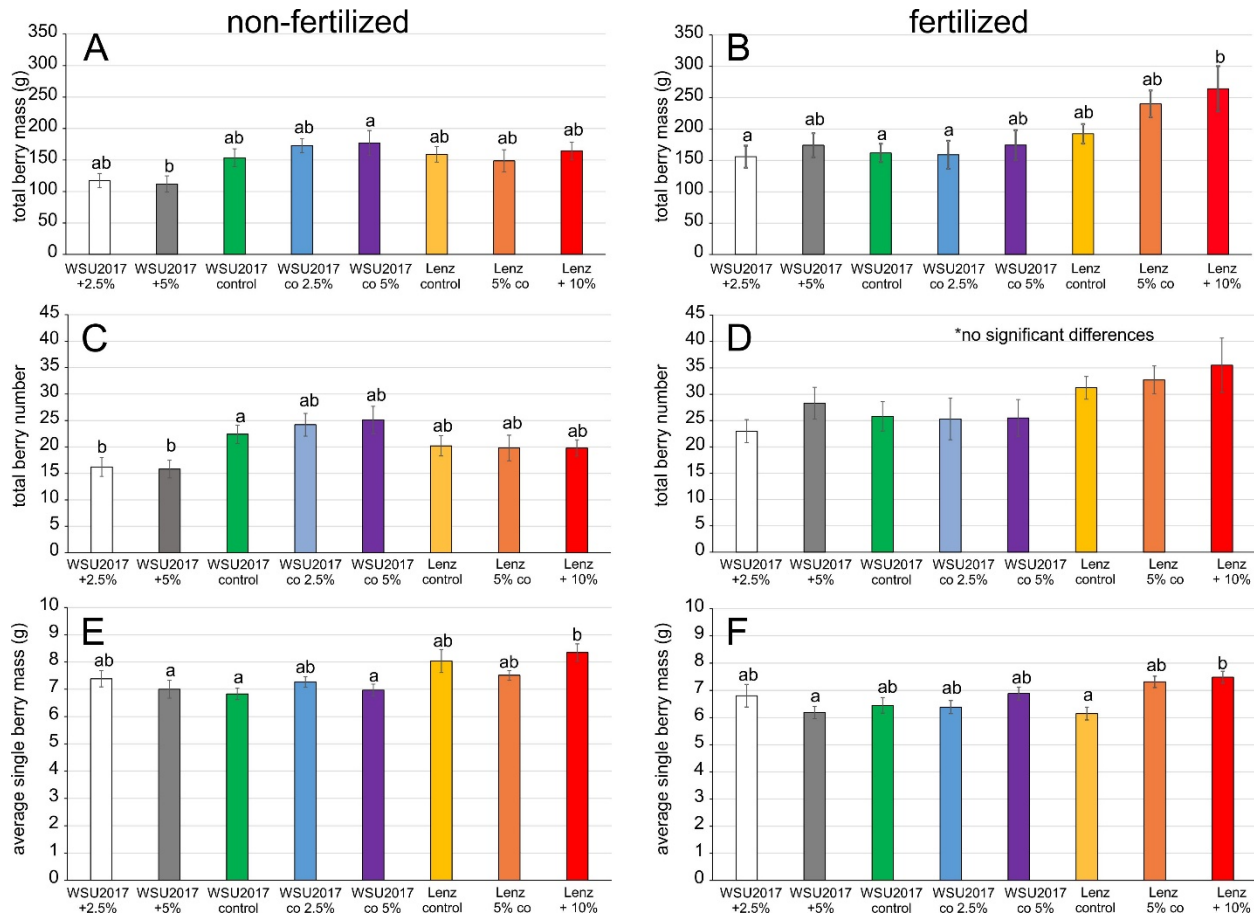


Figure 10: Effect of compost and biochar amendments on productivity of Albion strawberries. Two cohorts of plants (fertilized and non-fertilized) were subjected to 8 treatments. Fruit was harvested as it ripened over 22 harvests from each plant. Total berry mass (A and B) and berry number (C and D) per plant were recorded. Average single berry mass (E and F) was calculated from the totals. Results are means \pm s.e.m. ($n=10$). Different letters indicate significant difference between means, calculated by one-way ANOVA with Tukey post-hoc tests ($p<0.05$). Note: “char + compost” indicates char added post-composting, while “co-composted” indicates char added prior to composting.

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 available nitrogen, potassium, magnesium, sulfur, and zinc 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 one year of the 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 Mount Vernon. In unfertilized plots, treatments with compost increased available nitrogen (N), potassium (K), magnesium (Mg), sulfur (S), and zinc (Zn) more than biochar alone. (Data is shown in the technical report.) 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.

5.5 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, dimethyl disulfide, and other compounds that are not yet identified. Since monoterpenes were the most abundant VOCs 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 (hydrogen sulfide and dimethyl sulfide). 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, may provide a way to add value to compost as a soil amendment, though additional work, including through continuation of the current trials, is needed to provide more actionable insights. 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 crop yield increases, but this effect was only observed in fertilized split-plots.

Co-compost and the compost plus biochar were typically observed to affect soil physicochemical properties beneficially, especially by reducing bulk density. Our results suggest that blending compost with biochar may optimize the physical and chemical properties of each, but that this

effect is somewhat dependent upon the native soil and crop. Interestingly, in fertilized plots, amendment with biochar alone resulted in increased N availability, particularly in potato field trials in Mount Vernon.

For co-compost and compost plus biochar, as with biochar used alone, intent of use as a soil amendment needs to be carefully considered and clearly defined. The differences we observed between soil amendments considered in this study (i.e., co-compost, compost plus biochar, compost alone, or biochar alone) warrant this consideration, as growers using one or the other will likely see drastic differences in performance, and so, expectations for yield and soil responses should be specific to the product. For example, 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. Finally, it is important to note that data presented here is from one growing season, which makes it difficult to draw conclusions and make confident statements. Additional data collection from additional seasons would allow us to further evaluate the potential use of biochar and co-composted products as soil amendments, thereby improving recommendations to interested users. It should also be noted that because the characteristics of the biochar and compost impact chemical and biological processes, the use of different types of biochar or compost in these studies would be expected to yield different results.

Additional important detail on this work is available in the technical report *Integrating Compost and Biochar for Improved Air Quality, Crop Yield, and Soil Health* on the [WTFT 2017-2019 webpage](#) of Washington State University's CSANR.

5.6 References

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Chapter 6: Assessment of the Local Technical Potential for CO₂ Drawdown using Biochar from Forestry Residues and Waste Wood in 26 Counties of Washington State

James E. Amonette

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 (CO₂). 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 CO₂ 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 CO₂ drawdown technical potential of biochar for Spokane County with the aim to apply the method to the entire state in a separate, later effort. His 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 soil C and the exsolution of oceanic CO₂, were also considered.

In the present work, the approach of Amonette (2018) is strengthened in several ways. First, land capability classes and cropping systems are 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 soil organic C levels expected from additions of biochar) are updated to reflect recent literature suggesting a small enhancement of soil organic C by biochar amendments to agronomic soils. This effect is treated separately from the decreases in forest soil organic C levels expected from the removal of forestry residues to make biochar. Third, explicit time-dependent tracking of biochar production levels and biochar soil storage capacities is incorporated in order to account for the exports of biochar from counties that have exceeded their storage capacities to counties for which storage capacities in excess of their own biochar production capacity exist. This tracking allows the first assessment of the relative levels of production and consumption over time among the counties included in the study and sets the stage for a future economic assessment that includes transportation costs as a factor.

These improvements are discussed and the updated method is then applied to 26 counties (two-thirds of the counties in Washington State) to gain a more detailed and scientifically defensible

estimate of the technical potential of biochar technology to drawdown atmospheric CO₂ over a century.

6.1 Counties

Washington State has 39 counties, one of which (Spokane County) had been assessed by Amonette (2018). The goal of the current project was to bring assessment coverage to two-thirds of the counties (a total of 26 counties). Selection of the additional 25 counties was based on four equal criteria, 1) Municipal solid waste (MSW) production capacity, 2) timber harvest production capacity, 3) agricultural production, and 4) wildland-urban interface (WUI) fire risk. Counties were ranked by each criterion, and the top six in each category (seven for agricultural productivity) were selected for the study. Some counties appeared in the top six of more than one category, and so the category in which they scored the highest was the one for which they were selected, and they were removed from the rankings in the other categories.

The 26 counties selected for this study, and ranked by the above criteria for the category under which they were selected, are shown in Figure 11.

6.2 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 BGRAM program can be found in the online supplemental information file associated with the Woolf et al. (2010) publication.

MSW	Forest Biomass	WUI Fire Risk	Agric. Productivity
King	Grays Harbor	Yakima	Grant
Snohomish	Lewis	Stevens	Whitman
Clark	Skagit	Pend Oreille	Adams
Pierce	Mason	Okanogan	Benton
Thurston	Pacific	Chelan	Lincoln
Kitsap	Cowlitz	Klickitat	Franklin
			Walla Walla

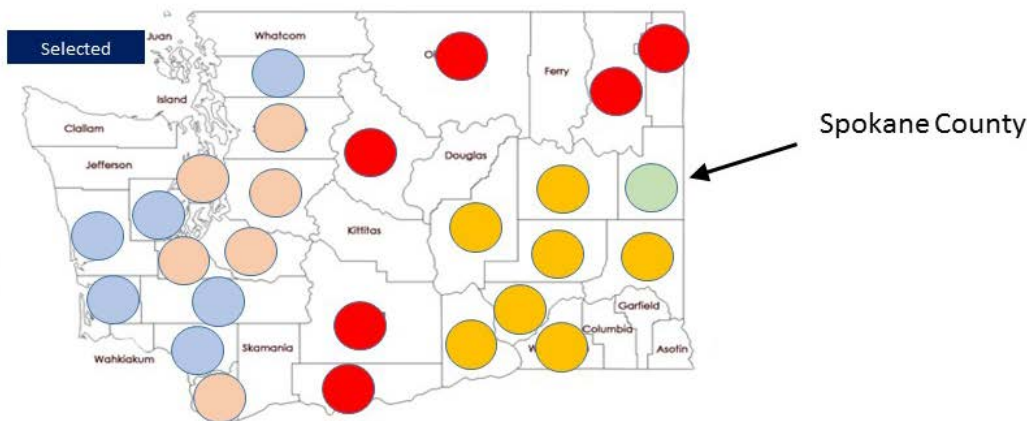


Figure 11: The four categories (MSW, Forest Biomass, WUI Fire Risk, and Agricultural Productivity) and top-ranking counties in each that were selected for this study. The 26 counties selected are shown by colored dots on the map, with the color corresponding to the category for which the county was selected. Spokane County (green dot) had already been selected based on previous work of Amonette (2018)

The BGRAM program 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, three primary feedstock streams were used: residual forest biomass from timber harvesting operations, wood reclaimed from 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 fourth, secondary feedstock stream, based on the additional drawdown stemming from biomass response to biochar amendment (i.e., enhanced yield), was considered in each scenario.

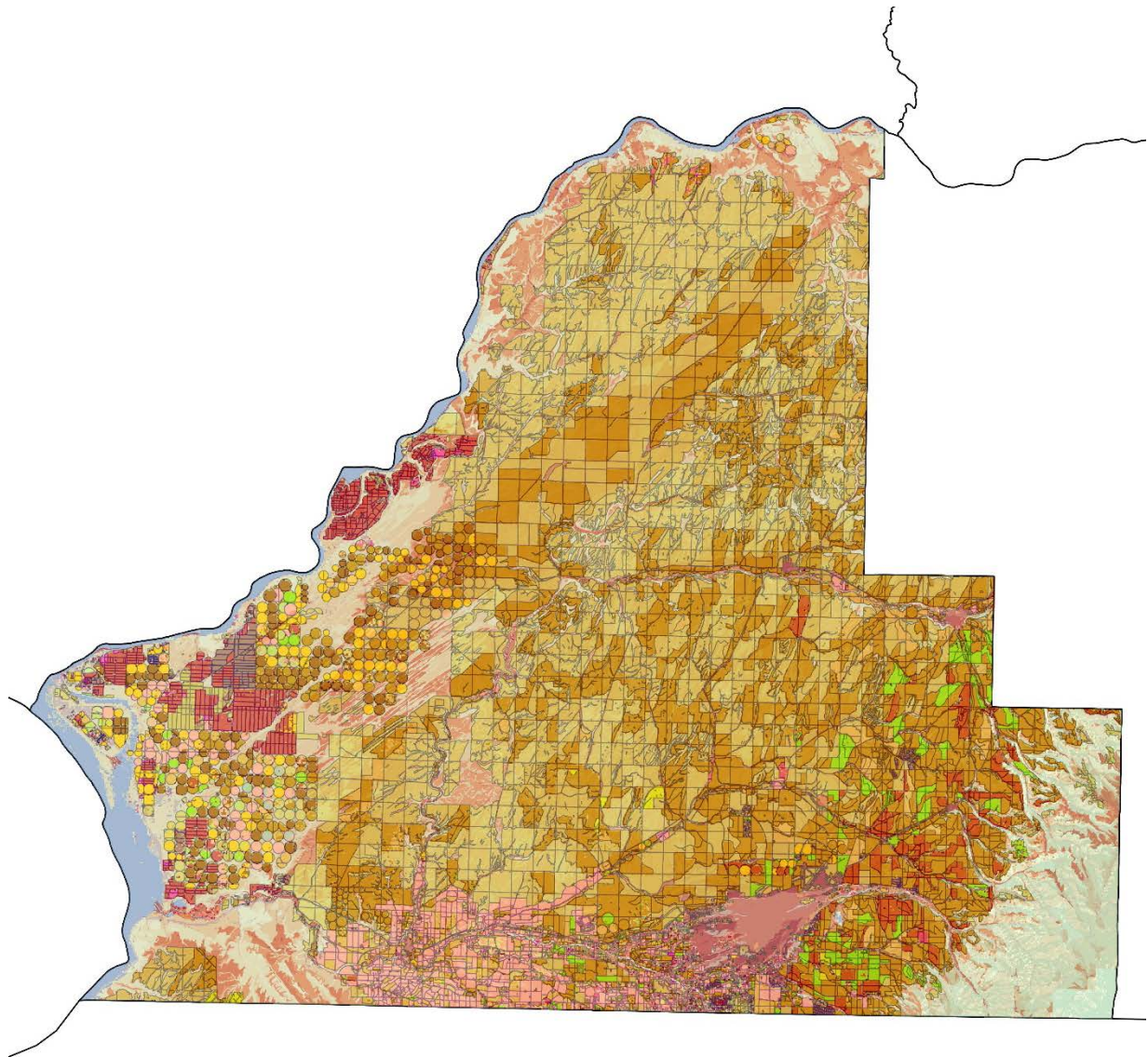
The enhanced-yield secondary feedstock stream in BGRAM required input data for the initial (i.e., pre-biochar) crop type and yield and for the soil productivity potential for each parcel of

cropped land in the county. Initial crop types and yields were obtained from the US Department of Agriculture's National Agricultural Statistics Service databases. For soil productivity potential, land capability classes developed as part of the US Department of Agriculture's Natural Resources Conservation Service soil survey database were used. A significant geographical information system (GIS)-assisted effort was required to integrate these data for each parcel of cropped land. The approach involved creation of two perfectly aligned raster datasets at 1-ha spatial resolution, one for crop type and the other for land capability class, and then interrogating them in a spreadsheet to assign the dominant crop type/land capability classification to each ha of cropped land in the county. These were then summed to yield the number of hectares in each category and normalized by crop to yield the fraction of each crop type grown in a particular land capability class. An example of the raster dataset images for crop type and land capability classification, which were developed for each county, is shown in Figure 12.

6.3 Biomass and processing scenarios

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 2018 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. The estimates assumed conservative, average, or aggressive timber harvest scenarios. Each of these timber harvest scenarios was further divided into two processing scenarios. One processing scenario involved only the fraction of the harvestable woody biomass brought to the landing that was subsequently transported to a central facility for processing (i.e., marketable biomass). The other processing scenario considered that all of the harvestable woody biomass brought to the landing was processed either at the landing using a mobile processing unit or at a central facility.

These estimates of MSW-recovered and timber-harvest residual biomass were combined into seven scenarios for each county. The first scenario consisted only of MSW-recovered biomass. The remaining six scenarios considered the MSW-recovered biomass in combination with one of the timber-harvest (conservative, average, aggressive) and biomass-processing (marketable biomass or all harvestable biomass) estimates just described.



0 4 8 12 16 20 miles



Walla Walla County
(WA071)

Figure 12: Map of Walla Walla County, Washington showing cropped agricultural lands (clearly delineated rectangular and circular polygons in a variety of colors), non-irrigated land capability classes (reddish, non-delineated zones), non-agricultural land (other non-delineated zones), and water (blue areas). Inset shows location of Walla Walla County in Washington State

For the 26 counties as a whole, the annual biomass inputs (reported as green weights) for the seven scenarios considered ranged from a total of 342 Mt for the MSW Only (Facility) scenario to 20,700 Mt for the Aggressive (Facility + Field) scenario, a factor of 60 (Table 7). On average, the proportion of the woody biomass coming from MSW was small, ranging from 1.6% to 5.8% of the total when grouped by scenario for the six scenarios that include timber-harvest residues. For individual counties, however, the MSW proportion ranged more widely. For example, the MSW proportion for the timber harvest residue scenarios in King County ranged from 12% to 29%, whereas the range for Grays Harbor County was 0.15% to 0.46%, reflecting the large differences in the types and quantities of biomass available in urban population centers and heavily timbered rural counties.

Table 7: Annual biomass inputs by harvest scenario summed for 26 counties in this study

Harvest scenario	Processing location		Biomass inputs			
	Facility	Field	Harvested biomass	MSW recovered wood	MSW green waste	Total biomass processed
			----- thousands of green metric tons -----			
MSW Only	X		n/a	303	39.4	342
Conservative	X		5,500	303	39.4	5,850
Average	X		7,870	303	39.4	8,210
Aggressive	X		9,330	303	39.4	9,670
Conservative	X	X	12,100	303	39.4	12,500
Average	X	X	17,100	303	39.4	17,500
Aggressive	X	X	20,300	303	39.4	20,700

6.4 Carbon drawdown potentials

To assess the climate impact of a given scenario, BGRAM calculates a variety of offsets for each feedstock stream, which are summed for the feedstock stream (Figure 13, left panel), and then the feedstock streams are summed to obtain a total offset (Figure 13, right panel). 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 offset, with biochar C added being most important for biochar and fossil-fuel offset being the most important for bioenergy.

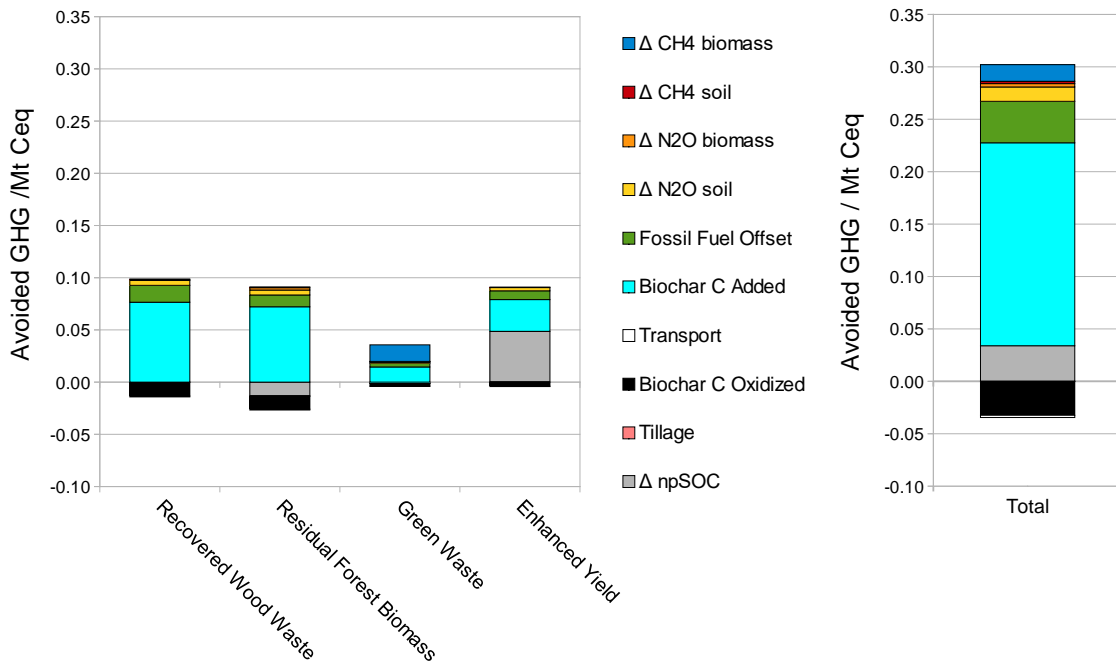


Figure 13: Contributions of feedstocks and offset mechanisms to the total offset for biochar (slow pyrolysis) under the Conservative (Facility) scenario for Walla Walla County. GHG = greenhouse gas, CH₄ = methane; N₂O = nitrous oxide; npSOC=non-pyrogenic soil organic carbon

The total 100-year offsets for biochar and bioenergy in the seven scenarios, summed for the 26 counties in this study, are listed in Table 8. Two offsets are reported: the *immediate* offset (Mt C_{eq}), which accounts for the initial C drawdown, and the *ultimate* offset [ppbv CO_{2(eq)}], which is expressed here in terms of atmospheric CO₂ levels and adds the long-term buffering response of the earth’s climate system to the initial C drawdown. In order to lower the ultimate (equilibrium) concentration of CO₂ in the atmosphere by 1 ppmv, 2.17 ppmv of CO₂ needs to be removed.

The 100-year climate offsets generally follow the expected trend established by the size of the biomass inputs (Table 8). Thus, addition of biochar and bioenergy production in the field (i.e., Facility + Field scenarios) roughly doubles the climate offsets over those obtained when only centralized facilities (Facility scenarios) are used for processing. For biochar, the immediate offset ranges from 11 Mt C_{eq} for the MSW Only (Facility) scenario to 354 Mt C_{eq} for the Aggressive (Facility + Field) scenario. The corresponding ultimate offset for biochar ranges from 2 ppbv CO_{2(eq)} to 77 ppbv CO_{2(eq)}. The offsets from bioenergy in Washington State are roughly half of those estimated for biochar (Table 8). This is largely due to the low C intensity of the primary energy supply, but also to the large degree of enhanced yield obtained when biochar is

applied to soils. For C drawdown purposes, then, biochar is twice as effective as bioenergy in Washington State.

Table 8: 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 the 26 counties in this study

Harvest scenario	Processing location		Total 100-year offsets				
	Facility	Field	Biochar	Bioenergy	Biochar	Bioenergy	Bioenergy / Biochar
			Mt C _{eq} (immediate)		ppbv CO _{2(eq)} (ultimate)		
MSW Only	X		11	5	2	1	0.50
Conservative	X		113	54	24	12	0.48
Average	X		154	75	33	16	0.49
Aggressive	X		178	88	38	19	0.50
Conservative	X	X	222	114	48	25	0.51
Average	X	X	303	158	66	34	0.52
Aggressive	X	X	354	186	77	40	0.53

6.5 Reconciling production and storage capacities

The results for the 26 counties differ substantially within a given scenario as well as across scenarios. To offer an idea of the major trends in the county-level results within a scenario, the cumulative 100-year biochar-C gross production levels for the Aggressive (Facility + Field) scenario with the counties ranked from highest to lowest are plotted (left axis) in Figure 14. In this scenario, largely rural counties with ample timber-harvest activity, led by Grays Harbor and Lewis Counties, tend to dominate production. With the exception of Yakima County, which is large and diverse, the counties with significant agricultural sectors contribute little to the overall production of biochar C. However, in part this is due to the present study’s focus on biochar derived from woody biomass. Biochar can be made from a variety of feedstocks, including agricultural residues as well as timber-harvest residues, and a full analysis of the contribution of these counties would include both categories of feedstocks.

Producing biochar, however, is only half of the solution. A place to store it is needed, and currently the most favorable storage option is to incorporate biochar into agricultural soils. The biochar-C storage capacities of the 26 counties are also plotted (right axis) in Figure 14. Counties with small woody-biomass biochar production capacities generally have large biochar-C storage

capacities. 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 study period. Large-scale adoption of biochar technology, therefore, will require a substantial effort to

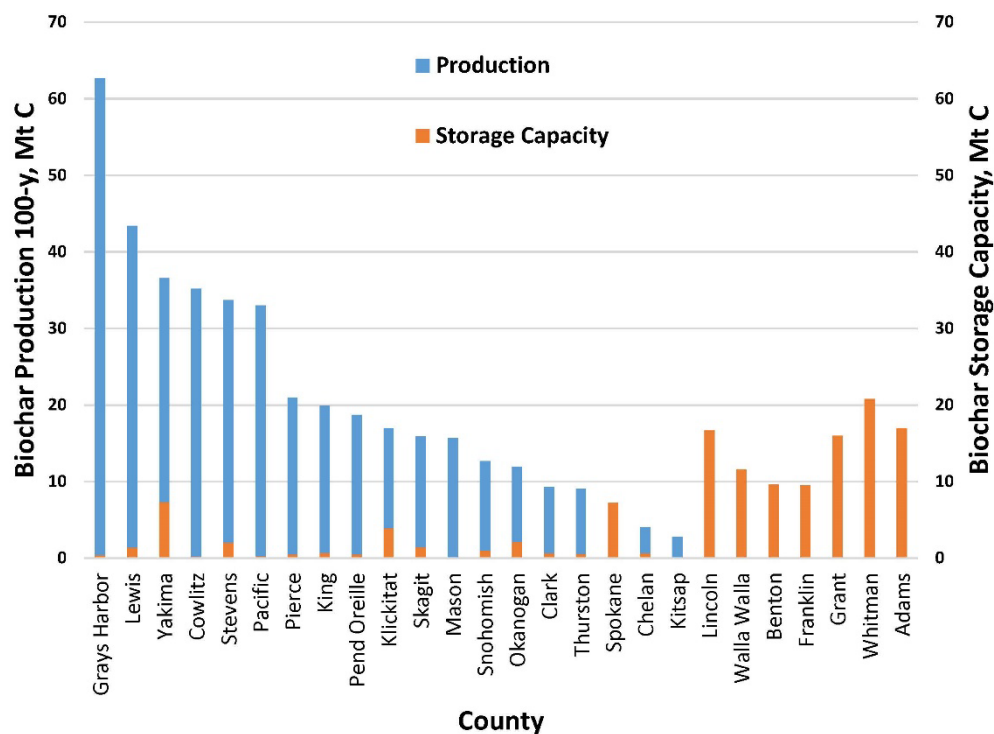


Figure 14: Cumulative 100-year biochar production in the Aggressive (Facility + Field) scenario and the initial biochar storage capacity in agricultural soils for each of the 26 counties in this study

transport not only biomass to processing facilities, but also biochar to storage sites that may be 100-200 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. Some relevant work in this regard is described in *Chapter 10, Biochar Production in Biomass Power Plants: Techno-Economic and Supply Chain Analyses*.

Another, somewhat more tractable issue, relates to the overall 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 seven scenarios considered fully saturate the available storage capacity during the first 100 years (Figure 15), and the sixth timber harvest residue scenario would reach saturation within 109

years. Addition of biochar produced from agricultural residues will shorten the time to saturation even further. 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. These concepts need research to identify which are viable solutions to the biochar storage problem. Fortunately, the current results suggest that we will have several decades at least to develop alternative storage options.

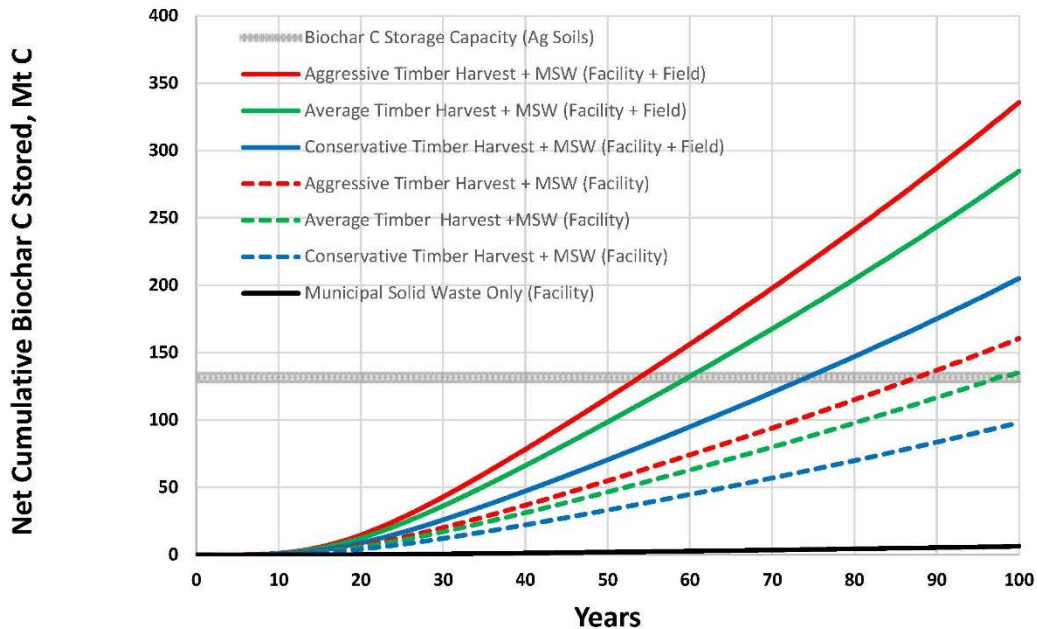


Figure 15: Cumulative net biochar C stored during the first 100 years of production in each of the seven scenarios summed for the 26 counties in this study, and the sum of the initial biochar storage capacity in agricultural soils for the 26 counties

6.6 Conclusion

This assessment of the C-drawdown potential of biochar technology when implemented in 26 selected counties of 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 11 Mt C_{eq}.
- Addition of timber-harvest residual biomass to the MSW biomass results in 113 to 354 Mt C_{eq} depending on the harvest and processing scenario.
- Addition of field processing of biomass to that done in centralized facilities roughly doubles the available biomass and, consequently, the C drawdown potential.

- When equilibrium with the climate system reservoirs is considered, an ultimate greenhouse gas offset can be calculated in terms of decreases in atmospheric CO₂ levels. This metric yields a drawdown potential range from 2 to 77 ppbv CO_{2(eq)}.
- Because this study focuses on woody biomass as feedstock, it underestimates the total drawdown potentials of biochar, particularly in the southeastern portion of the state, by not considering agricultural residues as feedstocks.
- In Washington State, use of the same woody biomass to generate bioenergy instead of biochar yields half of the C 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 26-county biochar-C soil storage capacity will be saturated in 54 to 109 years for the scenarios that include 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.

Additional important detail on this work is available in the technical report *Assessment of the Local Technical Potential for CO₂ Drawdown using Biochar from Forestry Residues and Waste Wood in 26 Counties of Washington State* on the [WTFT 2017-2019 webpage](#) of Washington State University's CSANR.

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Chapter 7: Production of Engineered Biochars for Phosphate Removal from Waste Lignocellulosic Materials: First, Second, and Third Generation Engineered Products

Michael Ayiania, Sohrab Haghighi Mood, Yaime Jefferson Milan, and Manuel Garcia-Perez

Authors' note: This study's results have been published as

Ayiania M, Carbajal-Gamarra FM, Garcia-Perez T, Frear C, Suliman W, Garcia-Perez M: Production and characterization of H₂S and PO₄³⁻ carbonaceous adsorbents from anaerobic digested fibers. *Biomass and Bioenergy*, January 2019, 339-349.

Haghighi-Mood S, Ayiania M, Jefferson-Milan Y, Garcia-Perez M: Nitrogen Doped Char from Anaerobically Digested Fiber for Phosphate Removal in Aqueous Solutions, Paper submitted to *Chemosphere*, 2019

Portions of this chapter, including figures, were taken directly from these publications.

7.1 Introduction

This project examines several different strategies for creating engineered biochars with enhanced performance characteristics from waste lignocellulosic materials, as part of a broader approach for improving the economics of biochar production. Specifically, the goal was to improve capacity for adsorption of phosphates and hydrogen sulfide by chars derived from several lignocellulosic materials including fiber from anaerobically digested dairy manure (AD fiber), urban wood residuals, and wheat straw. The impact on water holding capacity was also examined as this is an important function for biochar incorporated into soils. The biochar feedstocks and performance specifications were chosen specifically for their potential to integrate several potential municipal biorefinery scenarios.

Within this project, the objective was to evaluate and compare the phosphate adsorption capacity resulting from three strategies for production of biochars: carbon dioxide (CO₂)-activation, nitrogen doping either through a one-step (pyrolysis under NH₃, activation under NH₃) or two-step (pyrolysis under N₂, activation under NH₃) process, and metal-N-doping.

7.2 First generation biochars

The *first generation* of engineered biochars were made with a pyrolysis step followed by an activation step with CO₂ (Figure 16). Performance thus relied on physical activation due to enhanced generation of micropores. Previous work involving biochar from AD fiber, produced without an activation step, resulted in negligible phosphate adsorption. In this study, char produced from AD fiber was activated with CO₂, resulting in an improved phosphate adsorption capacity of 32.4 mg g⁻¹ biochar. The hydrogen sulfide (H₂S) adsorption capacity of AD fiber-derived chars was 51.2 mg g⁻¹. The breakthrough time for adsorption of H₂S for AD fiber-derived char produced at 750°C was 11.0 hours, which compared favorably to commercial activated carbon (Darco; 3.25 hours). The production temperature of the biochar had a significant influence on the capacity of the resulting biochar to adsorb and retain H₂S and phosphate. The ash content of biochar (particularly Mg, Ca, and Fe) and micropore volume appear to be very important for the adsorption of phosphate and H₂S, making CO₂-activated char from AD fiber an effective char for these purposes.

7.3 Second generation biochars

The *second generation* engineered biochars involved two types of “nitrogen doping” (the process of introducing nitrogen functional groups into a carbonaceous material), used to introduce nitrogen functional groups (e.g., amides, aromatic amines, and pyridinic groups), which are good adsorbents for negatively charged ions such as phosphate (Yin et al., 2017). Second generation biochars were made through one of two processes (Figure 16):

- A two-step process of pyrolysis followed by activation with ammonia (NH₃) gas (instead of the CO₂ used for activation of first generation biochars), or
- A one-step process involving pyrolysis under NH₃ gas.

High surface area and pore volume are key biochar properties that relate to water and nutrient cycling, microbial activity as well as sorption potential for organic and inorganic compounds as well as gaseous pollutants. Thus, within these experiments, we measured the impacts of the two N-doping processes at different temperatures on the amount of N incorporated, the surface area of the resulting char, and the phosphate adsorption capacity.

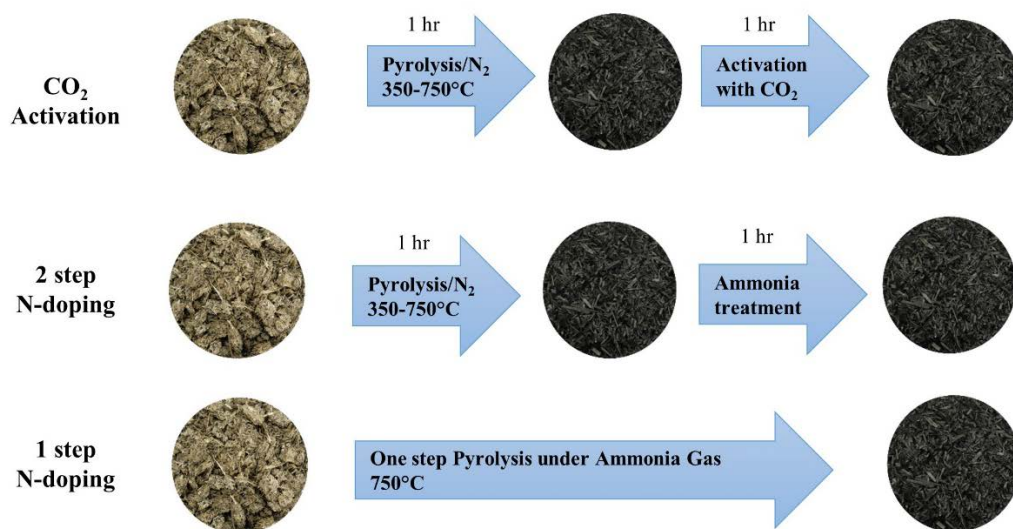


Figure 16: Summary of CO₂ activation (first generation) and nitrogen doping (N-doping; second generation) processes used in this report

Since nitrogen functional groups are important for phosphate adsorption, it was useful to know which nitrogen functional groups were likely to be formed on biochar under specific process conditions. For this purpose, density functional theory was used to study the thermodynamic stability of nitrogen functionalities in three graphene structures as a function of temperature and pressure, providing insight into the most favorable nitrogen functionalities present in N-doped biochar. A phase diagram confirmed that pyridinic groups are most stable functional groups present for the conditions studied. This information was used to optimize conditions (i.e., temperature and pressure) for production of N-doped biochar for phosphate removal.

Nitrogen content and surface area of biochars were measured because they are associated with greater capacity for phosphate adsorption. It was possible to obtain char derived from AD fiber (using the two-step N-doping method - pyrolysis under nitrogen gas, followed by activation under NH₃) containing 8.7% N by weight, respectively. Using one-step N-doping method (pyrolysis under NH₃-activation under NH₃), it was possible to increase nitrogen content of chars derived from AD fiber and cellulose up to 16.0 and 12.5% by weight, respectively. Two-step N-doping resulted in chars with slightly increased surface area and nearly doubled the phosphate adsorption capacity (32.4 to 63.1 mg g⁻¹) compared to chars produced from the same AD fiber with CO₂ activation. When AD fiber-derived char was produced with NH₃ in a single step compared to two-step N-doping, phosphate adsorption capacity nearly doubled again (63.1 to 110.3 mg g⁻¹; green rows in Table 9).

Table 9: Summary of phosphate adsorption capacity of chars from feedstocks and processes used in this study. First generation (blue rows) were obtained from CO₂-activation (pyrolysis then activation under CO₂; blue rows). Second generation (green rows) were obtained using two-step N-doping (pyrolysis under nitrogen gas and activation under ammonia) under a range of temperatures, or one-step N-doping (pyrolysis under ammonia) at 750°C. Third generation (orange) studies used cellulose and wheat straw as feedstocks for metal-N doping.

Feedstock	Process	T(°C)	Langmuir-Freundlich phosphate adsorption capacity (mg g ⁻¹)
AD Fiber	CO ₂ -activation	350	3.4
		450	4.9
		550	7.2
		650	7.3
		750	32.4
AD Fiber	2 step N-doping	350	6.3
		450	7.8
		550	12.3
		650	43.6
		750	63.1
	1 step N-doping	750	110.3
Cellulose	1 step N-doping	800	21.4
	Mg + standard pyrolysis	800	7.8
	Mg + 1 step N-doping	800	335
	Ca + 1 step N-doping	800	178
	Fe + 1 step N-doping	800	11.7
Wheat Straw	Mg + standard pyrolysis	400	18.8
		500	30.8
		600	60.1
	Mg + 1 step N-doping	400	136.2
		500	194.8
		600	288.4

In order to better understand the parameters in biochar responsible for phosphate adsorption, the research team made biochar from pure cellulose at various production temperatures (500-900°C). In this set of experiments, while the greatest N content occurred at 800°C, biochar produced at 900°C resulted in the greatest surface area and pore volume (data not shown).

Our team also conducted analysis of water holding capacity with N-doped biochars produced from urban wood residuals (particle board and compost overs). When raw (non N-doped) char from particle board was blended with Quincy sand soil at a rate of 10% by weight, water holding

capacity more than doubled compared to no biochar, from 29.9 to 69.6% by weight. However, N-doping provided little benefit compared to untreated (raw) biochar, and actually *reduced* the water holding capacity compared to raw biochar at higher application rates (Figure 17).

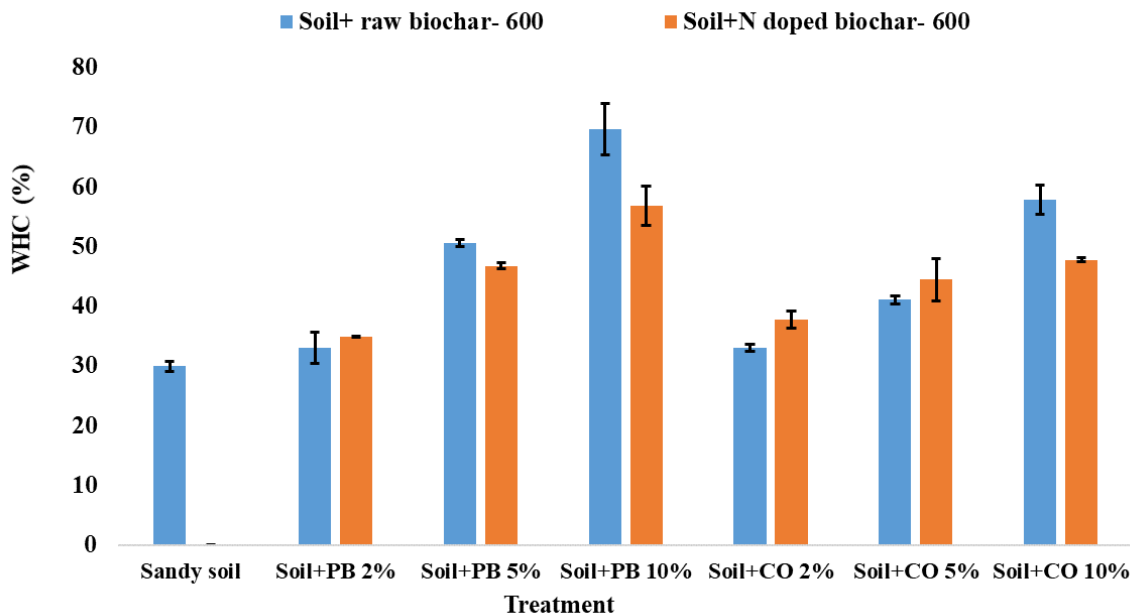


Figure 17: Effect of addition of raw and N-doped biochar produced at 600 °C at 2, 5, and 10% by weight on water holding capacity (WHC) of sandy soil. PB = particle board feedstock, CO = compost overs feedstock

7.4 Third generation biochars

In order to further increase phosphate adsorption capacity beyond what was achieved with N-doping, third generation engineered biochars were prepared from cellulose by impregnating feedstock with metals and then using the one-step N-doping process shown in Table 9 to create a metal-N-doped biochar. Since the presence of three metals – magnesium (Mg), calcium (Ca), and iron (Fe) – in biochar is known to improve phosphate adsorption capacity, these were the metals used here. Cellulose was used as a feedstock in order to isolate the effects of the other components of lignocellulosic materials. X-ray photoelectron spectroscopy (XPS) studies confirmed preferential bonding of Mg with N. Chars doped with Mg or N alone had phosphate adsorption capacities of 7.8 mg g⁻¹ or 21.4 mg g⁻¹, respectively. However, doping the char with both Mg and N increased the adsorption capacity to 335 mg g⁻¹, more than 15-fold more than materials doped with N alone, and more than 40-fold more than materials doped with Mg alone (Table 9).

Similar materials were produced from wheat straw. Untreated wheat straw has a very low phosphate adsorption capacity (5 mg g⁻¹) due to lack of metal content in the ash. When treated

with Mg prior to pyrolysis, wheat straw char produced at 600°C had a phosphate adsorption capacity of 60 mg g⁻¹. When char from wheat straw was produced by treatment with Mg followed by one-step-N-doping, it adsorbed 288 mg g⁻¹, an almost five-fold increase in adsorption capacity compared to the wheat straw treated with Mg only (Table 9; Figure 18). There were synergistic effects between Mg or Ca with N-doped char, with the Mg-N- and Ca-N-doped chars providing substantially improved capacity to remove phosphate ions. Fe-N-doped biochar, however, had a poor capacity for phosphate adsorption (Table 9).

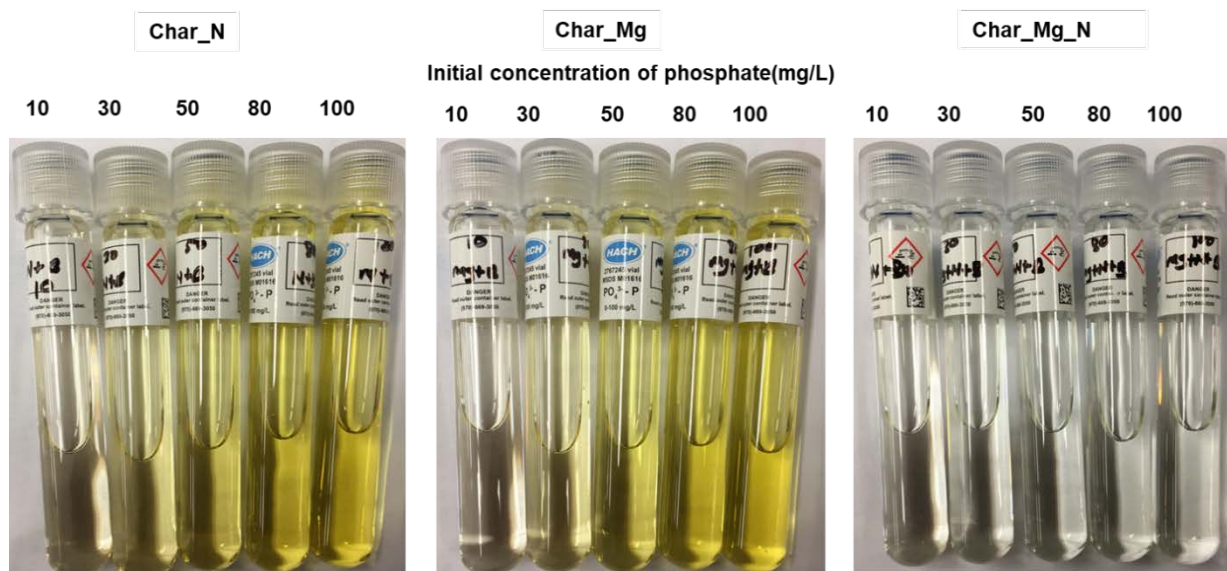


Figure 18: Phosphate adsorption by metal and N-doped biochar derived from cellulose (pyrolyzed at 800°C) from solutions of varying initial concentrations. The color intensity is directly proportional to the remaining concentration of phosphate after treatment with equal amounts of biochar for 24 hours, with more intense yellow indicating higher concentrations of phosphate. The photo on the left shows phosphate concentration after being in contact with N-doped biochar (Char_N). The photo in the middle shows solutions that have been in contact with Mg-doped biochar (Char_Mg). The photo on the right shows solutions after contact with Mg-N-doped biochar (Char_Mg_N). The fact that all test tubes are colorless on the right indicates that this biochar has been very effective at removing phosphate ions

7.5 Conclusion

As a result of this project, our team developed a method to produce carbonaceous adsorbents from waste lignocellulosic materials – AD fiber, urban wood residuals, and wheat straw – with great capacity to adsorb H₂S, and phosphate from aqueous effluents. With further development, products such as these have could eventually be integrated within a municipal biorefinery. For example, activated biochar derived from AD fiber could be used for H₂S removal from AD biogas and phosphate removal from AD effluent, as well as reducing emissions of H₂S from compost. Alternatively, biochar could be used to remove phosphate from other wastewaters in a variety of situations. Within either of these scenarios, the resulting phosphate-charged biochar could perhaps be sold as a nutrient-rich soil amendment. Future work will involve the study of

the desorption process to get a mechanistic understanding of how strongly the nutrients are bonded to the biochar, affecting nutrient bioavailability to plants. Further research is also required to standardize the design rules of biochar, which govern feedstock selection and carbonization conditions, leading to desired characteristics for specific nutrient- or pollutant-removal capabilities.

Additional important detail on this work is available from the technical report *Production of Engineered Biochars for Phosphate Removal from Waste Lignocellulosic Materials: First, Second, and Third Generation Engineered Products* on the [WTFT 2017-2019 webpage](#) of Washington State University's CSANR.

7.6 Reference

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Chapter 8: A Rapid Test for Plant-Available Water-Holding Capacity in Soil-Biochar Mixtures

James E. Amonette, Markus Flury, and Jun Zhang

As described in Chapter 6, production of biochar from waste woody biomass and forestry residues followed by its addition to agricultural soils can provide significant climate benefits and enhance agricultural productivity while adding value to significant components of the organic waste stream. For this approach to be economically practical, however, demand for biochar in the agricultural sector needs to increase. One potential benefit of biochar application is the enhancement of plant-available water holding capacity (PAWC) with potential increases in productivity if the crop is water-limited. In this chapter, we describe the development and application of a rapid test to assess the impact of biochar addition on PAWC levels in soils.

The PAWC is a measurement of how much water can be stored in a particular soil for use by plants, and is an important indicator of how crops will fare in droughty or water-limited conditions. However, because measurement of PAWC by standard methods can take days, if not weeks, to complete, it is both difficult and expensive to estimate the changes in PAWC expected when soil is amended with biochar. A more cost-effective measurement could have widespread impacts, both enhancing scientists' ability to test large numbers of soil-biochar combinations, facilitating the demonstration of biochar's benefits to agriculture, and, ultimately, the growth in agricultural demand for biochar and enhancing development of mechanistic insight into the impact of biochar amendments to soils on PAWC.

The relatively recent commercial development of disposable centrifuge filter units and highly accurate electronic analytical balances, coupled with the ready availability of laboratory centrifuges capable of handling dozens of samples simultaneously, suggests that a centrifuge-based method patterned on earlier work (Briggs and McLane, 1907; Russell and Richards, 1938) relating to estimating PAWC of soils may be feasible. In Part 1 of this project, we refined and calibrated an inexpensive, rapid method for measuring PAWC of soil-biochar mixtures. In part 2 of this project, we applied this method to a suite of 72 binary soil-biochar mixtures.

8.1 Biochars and soils

Both parts of this project utilized four types of biochar, representing the types of feedstocks and biochar manufacturing processes most likely to be encountered for large-scale application to agricultural soils (Table 11). We selected for use nine soils from Washington State having textures that encompass the range of textures typically found in the top horizon of Washington soils (Figure 19). Biochar-soil mixtures involved two different rates of biochar applied on the basis of biochar carbon content (0.5 and 2.0% by weight biochar carbon). Part 1 utilized one randomly selected biochar source-application rate combination for each soil, while Part 2 utilized all combinations, for a total of 72 soil-biochar mixtures.

Table 10: Provenance and selected physical properties of the biochars used in the project

Biochar	Feedstock	Manufacturing Process	Supplier	Contact Angle¹	Bulk Density	Organic C Content
				(degrees)	(g cm ⁻³)	(wt. %) ²
CP-Wd	Wood (pine)	Engineered	Cool Planet, CO	95 ± 5	0.24	82.2
SP-Wd	Wood (pine)	Slow pyrolysis	Biochar Now, CO	93 ± 1	0.17	81.3
G-Wd	Forest wood residuals (Douglas fir, pine)	Gasification	Oregon Biochar Solutions, OR	< 10	0.09	85.0
G-Ws	Straw (wheat)	Gasification	Ag Energy Solutions, WA	< 10	0.19	66.9

¹ The angle formed in the water phase between the water-solid surface and the water-air surface when water contacts a solid in the presence of air. Contact angle is measured to determine the hydrophobicity of a surface. A high contact angle indicates greater hydrophobicity (less interaction of water with the surface).

² Reported on an oven-dry basis

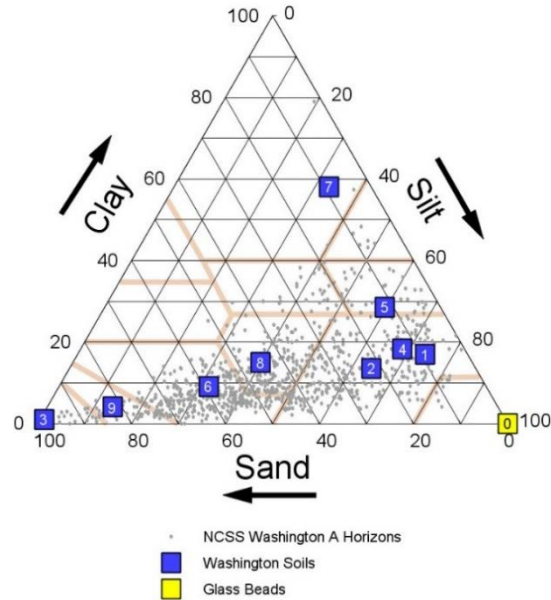


Figure 19: Soil textural triangle showing textural distribution of Washington A horizons in the USDA National Cooperative Soil Survey database, and the nine natural Washington soils and one synthetic soil (borosilicate glass beads) used in this work

Table 11: Names, typical crop, and selected physical and chemical properties of soils used in test

Soil	Typical Crop	Series Name ¹	Textural Class ²
S0 ³	-- ⁴	-- ⁴	silt
S1	potato	Skagit	silt loam
S2	strawberry	Sultan	silt loam
S3	potato	Quincy	sand
S4	wheat	Palouse	silt loam
S5	forest	Salkum	silty clay loam
S6	-- ⁵	Kapowsin	sandy loam
S7	forest/ pasture	Salkum B/C-Horizon	clay
S8	-- ⁵	Briscot/Kitsap	loam
S9	forest/ pasture	Harstine	loamy sand

¹NRCS-USDA National Cooperative Soil Survey, Official Soil Series Description

²USDA Soil Classification

³Solid borosilicate glass beads

⁴Not applicable

⁵No typical crop listed

8.2 Method development and calibration

The centrifuge method developed in this project is based on applying a specific level of water potential to a sample using a centrifuge. The measurable quantity in the centrifuge method is the moisture equivalent (i.e., the quantity of water retained per unit mass of oven-dry soil after centrifugation at a relative centrifugal force of 1000 g.) Calibration of this result with PAWC results obtained by the standard (reference) method using a pressure-plate (Klute, 1986) and a dew-point-psychrometer (Campbell, 2012) allows a value for a centrifuge PAWC value to be calculated from the moisture equivalent. The centrifuge values were plotted against the reference values to determine correlation for a pooled sample of soils and the nine randomly selected soil-biochar mixes (Figure 20). Note that PAWC values are most easily expressed in terms of weight of water per weight of oven-dry soil, thus, as weight percent.

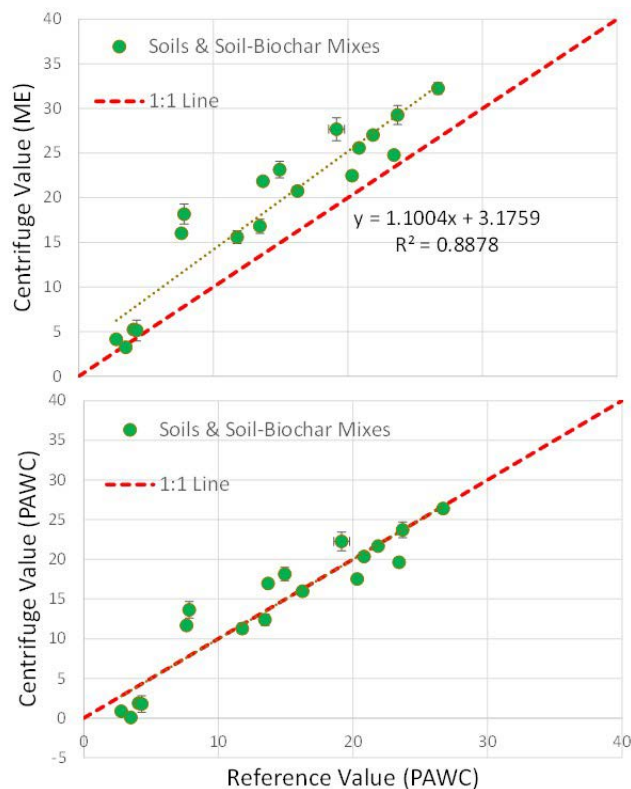


Figure 20: Pooled correlation between centrifuge-measured moisture equivalent (ME) and conventionally measured reference value for plant-available water-holding capacity (PAWC) of nine natural soils and nine mixtures of these soils with biochars (top); Pooled correlation between centrifuge- and conventionally measured reference PAWC values for soils and soil-biochar mixes (bottom). Error bars are one standard deviation

Further, we looked at the change in PAWC (Δ PAWC) predicted for biochar amendments to natural soil for the nine samples analyzed by both the centrifuge and reference methods. This Δ PAWC (Figure 21) is the difference between the water holding capacity of the natural soil and

that of the natural soil after amendment with biochar. In general, the results obtained by the two methods agree very well and for eight of the nine soils are not significantly different ($P = 0.05$) from one another. In the case of one soil amended with a hydrophobic biochar (SP-Wd), a decrease in Δ PAWC was measured by the reference method, while an increase in Δ PAWC was measured by the centrifuge method.

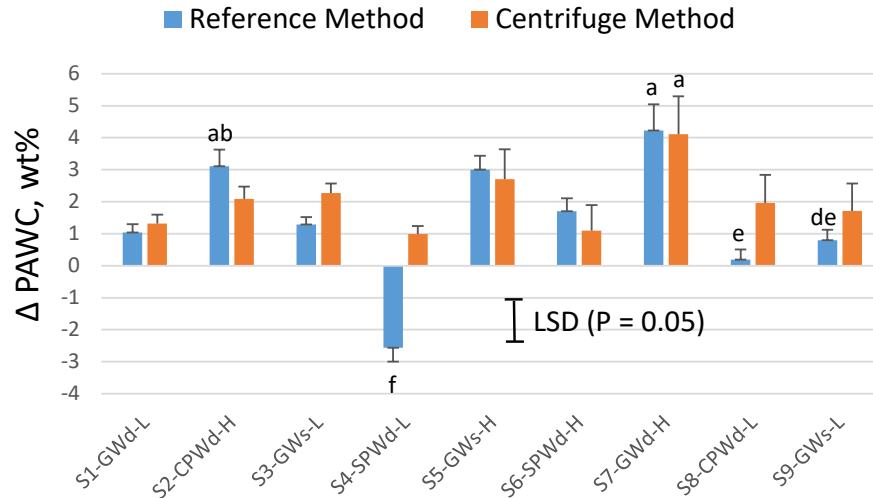


Figure 21: Comparison of the changes in plant-available water-holding capacity (PAWC) for selected biochar-soil mixtures measured by the reference and centrifuge methods. Error bars indicate one standard deviation. Different letters above error bars indicate significant ($P=0.05$) differences among means. Means having error bars without letters are similar to 3 or more other means. LSD (least significant difference) = 1.20 weight %. Soil and biochar sample codes are defined in 10 and 11; L= biochar added at 0.5% by weight carbon, H= biochar added at 2.0% by weight carbon

The calibration dataset using soils of different textures and different types of biochars against measurements made with standard pressure-plate and dew-point-psychrometer methods showed high linearity and a good coefficient of correlation between centrifuge and standard methods (Figure 19; $R^2 = 0.8878$). Change in PAWC was also reasonably well captured, with no significant ($P = 0.05$) statistical difference between the centrifuge and reference approaches for eight of the nine soils (Figure 21).

8.3 Application of the centrifuge method

The second part of this project focused on demonstrating the utility of the centrifuge method to obtain PAWC data for 72 biochar rate-soil-biochar type combinations and, as a possible

consequence, developing further mechanistic insight into the impact of biochar amendments to soils on PAWC.

When grouped in terms of soil type alone, the measured Δ PAWC means (averaged across both application rates) ranged from a low of 2.0 wt. % for the S3 (Quincy sand) to a high of 3.7 wt. % for S9 (Harstine loamy sand). The ranking of the two coarsest soils (S3 and S9) at opposite ends of the Δ PAWC response spectrum (Figure 22) is curious and suggests that, in this instance at least, mineralogy might be more important than texture. The Quincy sand (S3) is of eolian origin and largely derived from basalt, whereas the Harstine loamy sand (S9) is derived from glacial drift with influences from volcanic ash.

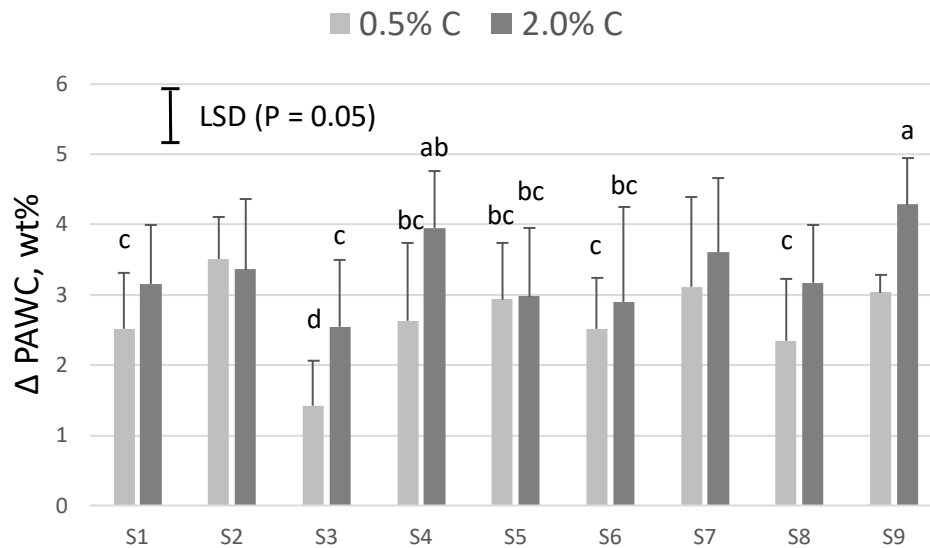


Figure 22: Mean changes in plant-available water-holding capacity (PAWC) measured by the centrifuge method as a function of soil type and biochar amendment rate. Error bars represent 1 standard deviation. Different letters above error bars indicate significant ($P=0.05$) differences among means. Means having error bars without letters are similar to 3 or more other means. LSD (least significant difference) = 0.78 weight %.

When grouped by biochar type only, the two hydrophobic biochars (SP-Wd and CP-Wd) give the lowest mean Δ PAWC values (2.5% and 2.8% by weight, respectively), whereas the two hydrophilic biochars (G-Wd and G-Ws) yield similar and significantly higher ($P = 0.05$) mean Δ PAWC values (3.3% by weight and 3.4% by weight, respectively). We note that hydrophobic properties are common with fresh biochars and tend to vanish as the biochars age, either in soil or as amendments to compost. These results, therefore, indicate the initial PAWC of the biochar-soil mixtures and underscore the need to be able to measure the PAWC repeatedly and inexpensively over time.

Overall, the Δ PAWC was 2.7% by weight and 3.5% by weight for the 0.5% C and 2.0% C application rates, respectively, indicating that biochar increases the PAWC of soils, but the

contribution of biochar is not linearly proportional to the amount of biochar added. If confirmed by additional study, such insights could be helpful to determining the most advantageous biochar application rates, considering both economics and impact on PAWC.

The analysis of soils alone, biochars alone, and soil-biochar mixtures allowed for a determination of the relative contributions the individual components to the increase of PAWC in soil-biochar combinations. The fraction of the increase in PAWC that is not explained by data from unmixed soils and biochars, was attributed to inter-particle effects (e.g., the creation of new pore spaces between biochar and soil particles). In this experiment, inter-particle effects were clearly associated with different biochar types and ranged from 56% (G-Wd) to 96% (SP-Wd) of Δ PAWC. These results suggest that especially for hydrophobic biochar (e.g., SP-Wd) Δ PAWC results primarily from inter-particle effects.

8.4 Conclusion

The centrifuge method refined and calibrated in this project is a useful technique to determine PAWC of soils and allows rapid screening of different soils and biochar mixtures. While it is unlikely to replace existing standard (reference) methods for determining PAWC using a pressure-plate and a dew-point-psychrometer, it provides good correlation to the reference method, and thus will likely be used as a complementary measurement. In Part 2 of this study, roughly five days of effort yielded a dataset that would have taken several months to collect by the standard method.

The results we obtained by application of the calibrated centrifuge method to a large set of samples lead to the following conclusions regarding the effects of biochar amendments on the PAWC of soils: (1) biochar increases the PAWC of soils, but the contribution of biochar is not linearly proportional to the amount of biochar added; (2) soil texture, and possibly soil mineralogy, in some instances, seem to have a large impact on the degree to which biochar increases PAWC; and (3) inter-particle effects are the largest contributor to the overall impact of biochar on PAWC.

Additional important detail on this work is available in the technical report *A Rapid Test for Plant-Available Water-Holding Capacity in Soil-Biochar Mixtures* on the [WTFT 2017-2019 webpage](#) of Washington State University's CSANR.

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Chapter 9: Using CropSyst to Evaluate Biochar as a Soil Amendment for Crops

Claudio O. Stöckle, Nigel Pickering, and Roger Nelson

9.1 Introduction

Biochar has been documented to affect crop production in terms of water holding capacity and adsorption of positively charged soil chemicals and nutrients such as ammonium. This study explored biochar effects on yields under irrigation, using potatoes grown in a loamy sand soil in central Washington as a model scenario, a soil condition under which the addition of biochar might offer some advantage. We used a crop growth simulation model (CropSyst) to evaluate the long-term effect of various kinds of biochar, different application rates, and under varying water supply regimes. Crop simulation is a useful tool because the model can be run over a period of many years, looking at average results and distributions, thus eliminating the possibility of a dry or a wet year influencing the outcome. [CropSyst](#) is a multi-crop simulation model developed at Washington State University. Model descriptions are available in Stöckle et al. (1994) and Stöckle et al. (2003), and some examples of applications are presented in Stöckle et al. (2014). The model uses a daily time step to evaluate simulation results over multiple years. The model simulates the soil water budget, soil-plant nitrogen budget, crop canopy and root growth, dry matter production, yield, residue production and decomposition, and erosion. Management options include cultivar selection, crop rotation, irrigation, nitrogen fertilization, tillage operations, and residue management.

9.2 Methods and model parameters

For this study, addition of two biochars (high and low ammonium adsorption capacity) was simulated, added at rates of 96 and 288 tons per hectare and mixed to a soil depth of 0.3 m (resulting in much larger application rates than usually reported). Four combinations of biochar type and application rate, along with a control treatment (no biochar), were simulated under three different irrigation schemes (full, deficit, and no irrigation) to give a total of 15 runs (Table 12). Thirty years of daily weather data (1981-2010) provided weather input to the 15 simulated scenarios. We evaluated the effect of biochar incorporated into the topsoil on dry yield, net irrigation (irrigation application losses not included), deep percolation, nitrogen applied, nitrogen leached, and nitrous oxide emissions. Deep percolation, nitrogen leached and nitrous oxide emissions are of interest as they relate to nitrogen losses and the potential for negative impacts on water quality and increased greenhouse gas emissions.

Table 12: Summary of CropSyst model scenarios

Model Run	Irrigation	Soil	% Biochar (w/w)	Biochar Applied (t/ha)	Type of Biochar
FI-LS	Full	Loamy Sand	0	0	None
FI-BL2	Full	Loamy Sand	2	96	Low adsorption
FI-BL6	Full	Loamy Sand	6	288	Low adsorption
FI-BH2	Full	Loamy Sand	2	96	High adsorption
FI-BH6	Full	Loamy Sand	6	288	High adsorption
DI-LS	Deficit	Loamy Sand	0	0	None
DI-BL2	Deficit	Loamy Sand	2	96	Low adsorption
DI-BL6	Deficit	Loamy Sand	6	288	Low adsorption
DI-BH2	Deficit	Loamy Sand	2	96	High adsorption
DI-BH6	Deficit	Loamy Sand	6	288	High adsorption
NI-LS	None	Loamy Sand	0	0	None
NI-BL2	None	Loamy Sand	2	96	Low adsorption
NI-BL6	None	Loamy Sand	6	288	Low adsorption
NI-BH2	None	Loamy Sand	2	96	High adsorption
NI-BH6	None	Loamy Sand	6	288	High adsorption

Modeling captured both the physical and chemical impacts of biochar addition to soils. Physical impacts documented in the literature included a decrease in bulk density, and an increase in porosity, saturation and available water, and a mixed effect on saturated and unsaturated hydraulic conductivity. Hydraulic conductivity measures the ease with which water moves through pore spaces or fractures. Documented chemical impacts include an increase in carbon content, cation exchange capacity (CEC) (Tian et al., 2016), soil pH (Liu et al., 2013), and soil sorption of chemicals and nutrients (Tian et al., 2016). Sorption is the removal of a chemical from aqueous solution by partitioning a fraction of the chemical mass from the soil solution onto the surface of soil solid particles, while desorption is the release of a sorbed chemical back into the soil solution. By maintaining an equilibrium between the chemical mass sorbed and in solution phases, the soil regulates the supply of ammonium and other positively-charged chemicals to the crop. The increase in CEC allows for a higher degree of adsorption that augments adsorption of Ca^{2+} , Mg^{2+} , Na^+ , NO_3^- , NH_4^+ , K^+ , and SO_4^{2-} (Strawn et al., 2015).

Soil parameters utilized in the model were based on a review of the literature, and are summarized in Table 13.

Table 13: Soil parameters used in CropSyst model scenarios; BC = biochar

Parameter	Material	Loamy Sand	Loamy Sand + 2% biochar (w/w)	Loamy Sand + 6% biochar (w/w)	Comment
Bulk density (g/cm ³)	Soil mix	1.60	1.40	1.20	Imhoff and Nakhli (2017)
Wilting point (cm ³ /cm ³)	Soil mix	0.04	0.04	0.04	Imhoff and Nakhli (2017)
Field capacity (cm ³ /cm ³)	Soil mix	0.13	0.15	0.18	Imhoff and Nakhli (2017)
Saturation (cm ³ /cm ³)	Soil mix	0.43	0.48	0.52	Imhoff and Nakhli (2017)
Campbell β (-) ¹	Soil mix	2.90	2.84	2.53	Derived
Campbell Ψ_e (kPa) ²	Soil mix	1.00	1.32	2.27	Derived
NH ₄ K (L/mg)	Soil mix with low BC	0.001	0.003	0.006	Interpolated
NH ₄ Qmax (mg/g)	Soil mix with low BC	3.30	3.26	3.18	Interpolated
NH ₄ K (L/mg)	Soil mix with high BC	0.001	0.004	0.011	Interpolated
NH ₄ Qmax (mg/g)	Soil mix with high BC	3.30	3.28	3.23	Interpolated

¹ β = coefficient in the Campbell (1985) equation for soil water retention

² Ψ_e = air entry potential in the Campbell (1985) equation for soil water retention

The kind of biochar and application rate changed the soil moisture and ammonium adsorption parameters of the top 30 cm of soil. The water supply regime changed the irrigation amount. Nitrogen application was adjusted for each run to avoid crop nitrogen deficiency (as reported by the model).

9.3 Results

The effect of biochar on potato yields was negligible under full irrigation, but with small gains of about 4% when water was limiting (deficit and no irrigation treatments) (Figure 23; Table 14).

Full irrigation assumes that plants receive irrigation sufficient to meet evapotranspiration demand and achieve maximum yields, so it would not incorporate the benefits of additional water holding capacity when water supply is limiting yields. This result is consistent with dryland cereal crop experiments reported in the literature, when conducted in agricultural soils not limited by low pH or low fertility.

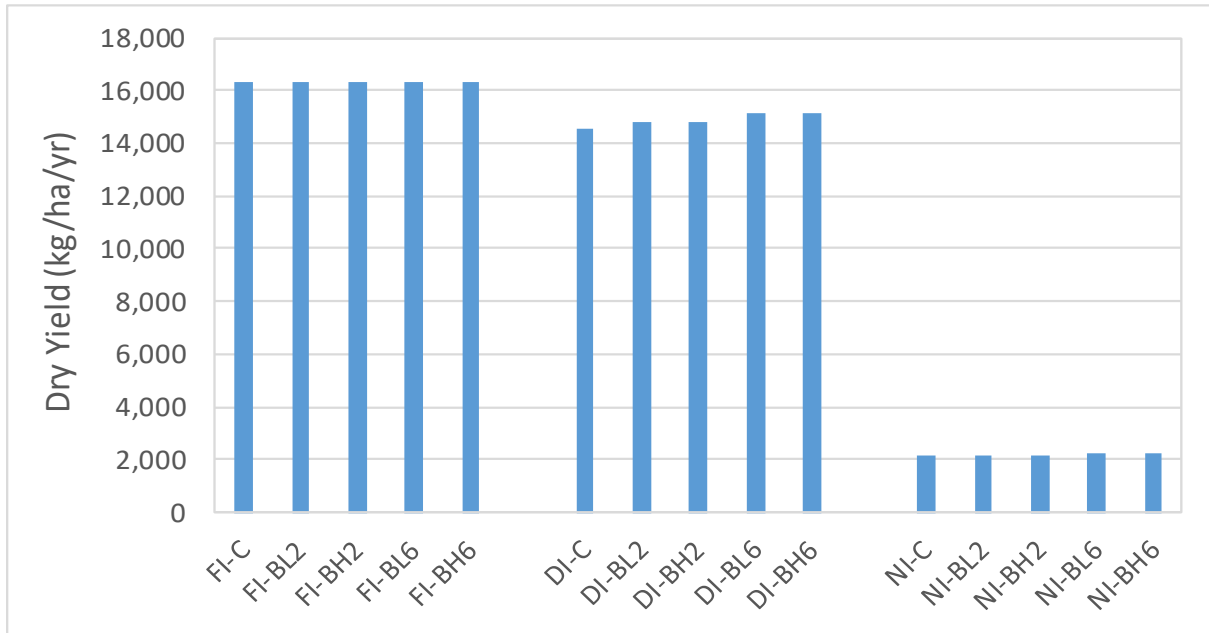


Figure 23: Potato dry yield for different CropSyst scenarios. See Table 12 for description of model runs, and Table 13 for soil parameters used

Table 14: Scenario results. See Table 12 for description of model runs, and Table 13 for soil parameters used

Scenario	Dry Yield (kg/ha)	Net Irrigation ¹ (mm)	Deep Percolation (mm)	N Applied (kg/ha)	N Leached (kg/ha)	N ₂ O-N Emission (kg/ha)
FI-LS	16,302	676	10.4	309	6.7	1.4
FI-BL2	16,305	683	9.9	308	6.9	1.6
FI-BH2	16,305	683	9.9	308	6.9	1.6
FI-BL6	16,308	694	10.7	308	7.8	1.6
FI-BH6	16,308	694	10.7	308	7.8	1.7
DI-LS	14,524	524	1.2	290	1.7	1.2
DI-BL2	14,815	530	0.7	293	0.9	1.3
DI-BH2	14,815	530	0.7	293	0.9	1.3
DI-BL6	15,145	531	0.4	296	0.6	1.4
DI-BH6	15,145	531	0.4	296	0.6	1.4
NI-LS	2,123	0	0.7	117	2.1	0.7
NI-BL2	2,171	0	0.4	118	1.1	0.7
NI-BH2	2,171	0	0.4	119	1.1	0.7
NI-BL6	2,208	0	0.3	117	0.9	0.8
NI-BH6	2,209	0	0.3	117	0.9	0.8

¹ Seasonal precipitation average was 64 mm

On average, the net amount of irrigation water applied was essentially not affected by biochar addition, with small increases in amounts of irrigation water applied linked to the small yield gains (Table 14). Net irrigation demand is almost exclusively determined by weather and crop growth. Similarly, biochar addition did not substantively affect water drainage below the root zone depth (deep percolation), though deep percolation clearly decreased with deficit and no irrigation.

Nitrogen leaching, as expected, decreased with reductions in irrigation water applied and deep percolation (Table 14). However, biochar addition caused only insignificant differences and are likely due to small yield differences and concurrent nitrogen uptake changes, as well as interactions between irrigation and fertilization timing calculated by the model. It seems that the relatively fast conversion of ammonium to nitrate (nitrification) neutralized the effect of the additional ammonium sorption capabilities provided by biochars, providing no advantage in terms of leaching. It is unclear if biochar can provide a degree of physical protection against the microbial activity responsible for nitrification, thus delaying the conversion process.

The emissions of nitrous oxide, a powerful greenhouse gas affecting climate change, were proportional to the amount of ammonium fertilizer applied, decreasing in the order full irrigation

> deficit irrigation > no irrigation (Table 14). The loamy sand soil considered in this study drains fast and does not provide opportunities for soil saturation leading to denitrification (a major process enhancing nitrous oxide emissions). For this reason, the bulk of the emissions in this study came from nitrification, the conversion of ammonium to nitrate, and thus were related to the amount of ammonium fertilizer applied. The addition of biochar slightly increased nitrous oxide emissions in all irrigation treatments, more so when biochar application rates were increased from 96 to 288 tons per hectare. Because this effect was not related to the biochar ammonium adsorption capabilities, these results suggest that an increase in water holding capacity may have created slightly better conditions for microbial activity.

9.4 Conclusion

Based on the physical and chemical mechanisms modeled in this study, the addition of biochar to fully irrigated crops is unlikely to improve yields unless acidic conditions are improved by biochar additions. However, yields of dryland crops grown in the Inland Pacific Northwest might benefit, particularly if biochar is applied at high rates, as water is more often limiting in these systems. However, the economic benefit of biochar addition will require evaluation, as dryland crops have a lower value per acre than irrigated crops.

Additional important detail on this work is available from the technical report *Using CropSyst to Evaluate Biochar as a Soil Amendment for Crops* on the [WTFT 2017-2019 webpage](#) of Washington State University's CSANR.

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Chapter 10: Biochar Production in Biomass Power Plants: Techno-Economic and Supply Chain Analyses

Manuel Garcia-Perez, Michael Brady, and Abid H. Tanzil

10.1 Introduction

The adoption of biochar soil amendments has the potential to reduce and recycle woody biomass waste streams in Washington State, draw down atmospheric carbon dioxide, and improve crop productivity. The substantial acreage of high value crops in Central Washington located in relatively close proximity to woody debris, suggests potential for the use of biochar in these agricultural systems. Potential exists to produce biochar from modification of existing large biomass boilers. However, in recent years, biomass boilers in Washington State have been shuttered, with the low selling price of electricity in Washington as one primary contributing factor. While electricity prices in the US range from \$0.06 to \$0.35 per kilowatt hour (kWh), Washington's average electricity price is \$0.07 per kWh, which is 23% less than the national average (Electricity Local, n.d.).

An improved understanding of biochar techno-economics and potential use values in our region is needed to help target future research work to identify ways that the nascent regional biochar sector can be supported. (A view of the current sector, and the various entities that operate within that sector, can be seen in map form at <https://www.pnwbiochar.org/producers/>.) The goal of this report is to assess the potential market for biochar in the Pacific Northwest by comparing production costs relative the value of biochar for agricultural uses. The assessment consists of three parts. First, a techno-economic analysis is used to estimate production costs, and thus, the minimum selling price at which biochar can be produced. Second, a supply analysis is used to evaluate the potential regional supply of feedstocks at different price levels. Finally, a demand analysis estimates the value of biochar, which can be thought of as the maximum purchase price, based on value due to crop yield improvements and carbon sequestration.

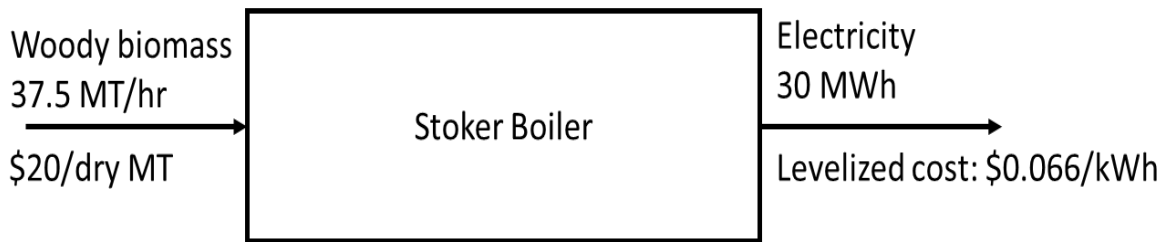
10.2 Techno-economic analysis

A baseline techno-economic analysis for a 30 MW power plant producing only electricity was conducted and biochar minimum selling price was calculated as a function of feedstock cost and production capacity (Figure 24, Case 1). In our second study we conduct a techno-economic analysis for a modified power plant producing biochar (Figure 24, Case 2) and calculated biochar minimum selling price as a function of electricity wholesale price, feedstock cost, biochar yield and production capacity. This task was conducted using information from the literature (California Biomass Collaborative; Tiangco et al., 2005) and interviews with the industry representatives. A few key technical assumptions are as follows: capacity was assumed to be 37.5 dry metric ton (MT) per hour, ash production of 5%, feedstock price of \$20 per dry MT,

biochar yield of 15%, electricity price (levelized) of \$0.066 per kWh. More details on technical and financial assumptions, capital costs and operational and maintenance cost assumptions are provided in the technical report associated with this chapter.

The mass and energy balances of the two cases studied are summarized in Figure 24. In Case 1, which represents the base case, 37.5 MT of biomass are processed per hour to produce 30 MWh of electricity (no biochar). In Case 2 study, the grade velocity was assumed to be accelerated to limit the combustion of fixed carbon. Thus, the same unit is used to produce a combination of electricity and biochar, with the electricity production reduced to 25.5 MWh and 5.62 MT per hour of biochar. The minimum selling price (MSP) of biochar was estimated to be \$151.50 per MT.

Case 1



Case 2

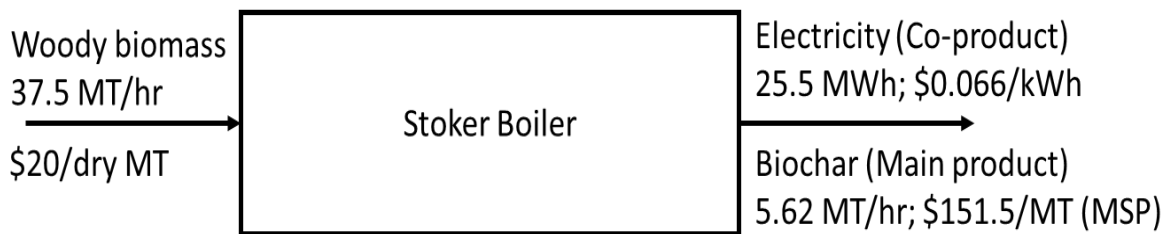


Figure 24: Overall mass balance of the two cases studied for biochar production (MT=metric ton)

The effect of feedstock cost and electricity price in the estimation of MSP of the char produced in the boilers is shown in Figure 25. Note that the feedstock cost can be positive or negative depending on the type of biomass used. (Negative cost means that the biochar maker is paid for accepting the feedstocks.) This figure clearly shows the potential for economic conditions in which biochar production cost could be low enough to justify its use in large-scale agriculture. For example, if the biochar unit is able to buy feedstock for \$20 per dry MT and is able to sell the electricity for \$ 0.1 per kWh, then the unit is able to give the biochar away for free (MSP is zero).

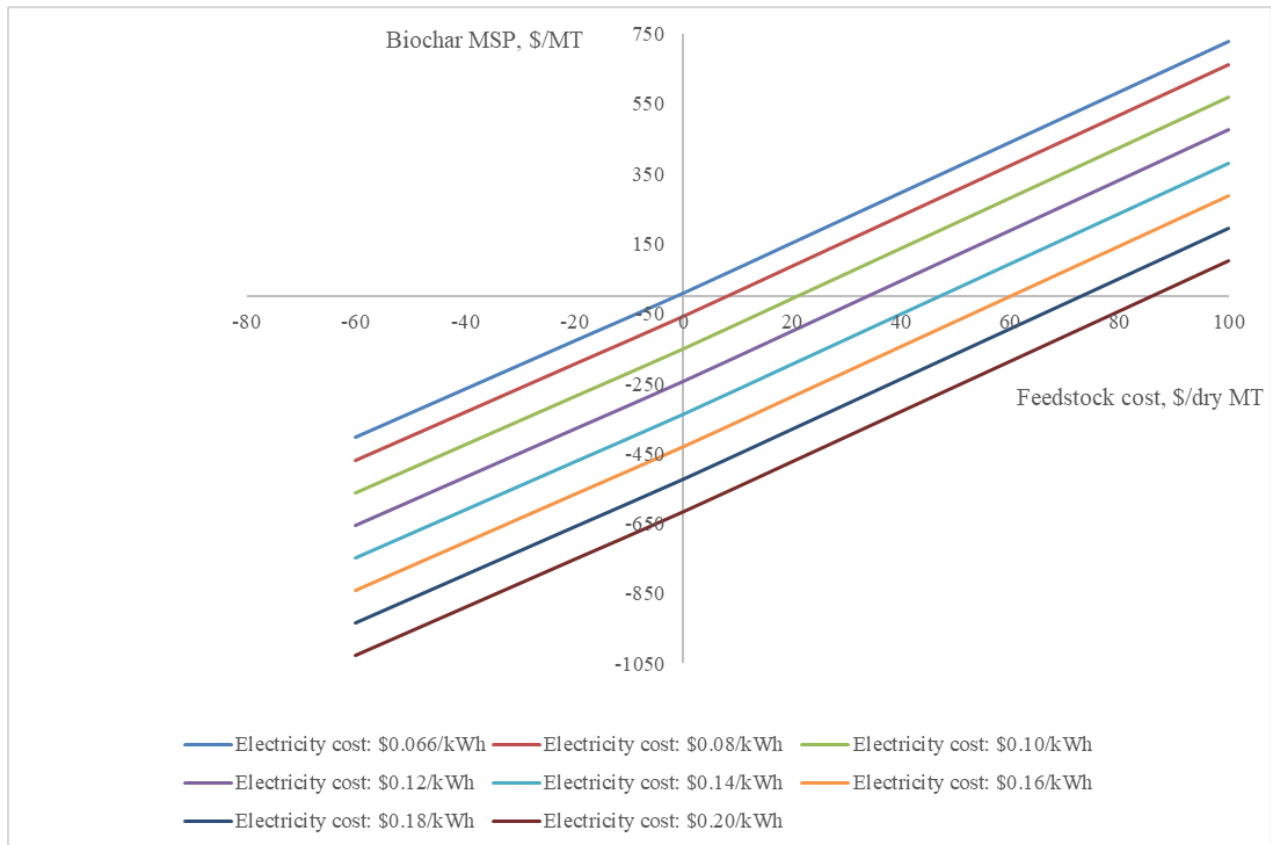


Figure 25: Biochar minimum selling price (MSP) as a function of feedstock cost and electricity price

Our team also calculated biochar price needed to compensate for the losses of energy revenue because of net conversion of biochar. At an electricity price of \$0.066 per kWh, the electricity revenue lost annually by changing to biochar production would be \$92.1 million. In order to compensate for this loss, the resulting biochar would need to be sold at a price of \$583/MT. The reader should note that the main concepts and premises used for this calculation are very different to those used to calculate Figure 25. In the case of Figure 25, electricity production and commercialization is subsidizing the production of biochar. In the calculation detailed above, biochar needs to be sold at a price that generates the same income as electricity. Correspondingly, its price is much higher.

10.3 Regional feedstock availability

To gain a regional perspective on biochar production, we consider the amount of biochar feedstock available at different price levels. According to the US Department of Energy (2011), Washington is estimated to have just under 1 million dry MT of feedstocks available annually at \$20 or less per MT. Just under 1.1 million dry tons of feedstocks are thought to be \$40 or less per MT. Then, there are about 1.5 million dry MT at \$200 or less per MT. Given the fact that biochar

is still a nascent market, it makes sense to simplify the analysis by assuming that only the 1 million dry tons of low cost feedstock may be used. The cost of each unit of feedstock will vary along with transportation costs even if it is the exact same “roadside” cost. Bringing all this information together, we construct a regional supply curve for biochar as shown in Figure 26. The results from the previous section simply shift this curve upward to produce a (marginal) cost curve, which is simply the cost of producing each ton of biochar.

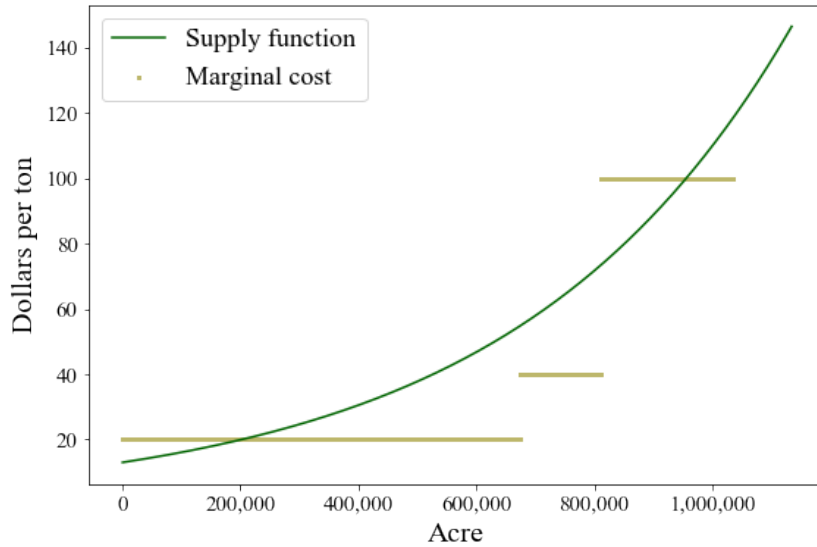


Figure 26: Regional supply curve for biochar for Washington State

10.4 Biochar value for agricultural uses in Washington State

To calculate the public and private value of utilizing biochar for agricultural uses, we consider its value due to carbon sequestration and crop yield improvement. A biochar application rate of 10 MT per acre is assumed throughout this study. Note that in previous studies, including Galinato et al. (2011), an avoided emissions value is included because using biochar affects soil acidity in a way that offsets the need to apply lime, and thus the greenhouse gas emissions associated with lime use are avoided. We did not consider avoided emissions in our value calculations for these reasons: soil acidity is less of an issue in Washington State as whole as it is in the Palouse (the focus of Galinato et al.) and the value derived from avoided emissions is likely to be a much smaller component than carbon sequestration and yield improvements.

Carbon Sequestration

Biochar reduces carbon in the atmosphere by sequestering carbon that otherwise would have been released (Laird, 2008). Currently, there is no direct financial incentive for a farmer in the US to use biochar for carbon sequestration. However, it is important to be prepared to understand the role that biochar could play if a climate policy like a carbon tax or cap and trade policy were adopted.

Galinato et al. (2011) estimated biochar values with carbon prices ranging from \$1 to \$31 per MT of carbon dioxide equivalent, based on trading on the Chicago Climate Exchange in 2008. This corresponds to a biochar value of \$2.93-\$90.83 per MT. More recently, carbon prices have ranged from \$1 to \$125 per MT of carbon dioxide equivalent (World Bank Group, 2019), with most of the observed prices are at the lower end of this range in the vicinity of \$10 per MT, which points towards exploring a price in the lower end of the range. However, the pricing of carbon is meant to provide incentives to reduce emissions so that the worst-case scenarios of global climate changes are not realized. Stiglitz and Stern (2017) have argued that the price required to alter behavior enough to avoid the worst climate change impacts is \$40-\$80 per MT of carbon dioxide equivalent.

Given that the motivation of this report is to provide perspective on potential outcomes, we calculate biochar values assuming carbon dioxide equivalent prices of \$10, \$40, and \$80 per MT. This corresponds to values of \$29.36, \$117.44, and \$234.88 per MT of biochar resulting in carbon sequestration. These values are added to the yield value described below.

Yield improvement

Several studies have reported yield increases from biochar application at rates between 2 and 20 tons per acre if appropriate nutrient management is followed (Galinato et al., 2011; Filiberto and Gaunt, 2013; Hussain et al., 2017). Although there is a great deal of uncertainty over the magnitude of yield improvements from biochar application, two recent meta-studies both arrive at 10% as a reasonable starting point (Jeffrey et al., 2011; Liu et al., 2013). Information on profit per acre is taken from recent crop enterprise budgets. We use the same set of values assumed in recent benefit-cost analyses on water storage projects (Yoder et al., 2014).

Biochar creates value by increasing agricultural productivity per acre. That additional production has a value that is equal to the change in yield due to biochar multiplied by the price per unit of output (i.e., crop price), assuming that costs per acre stay relatively constant. To convert from a value per acre to value per unit (ton) of biochar, the additional value (yield change times crop price) is divided by the biochar application rate. In economics, this value is referred to as the marginal value product (MVP).

Table 15 reports the estimated value for biochar for each crop group under three different yield improvement assumptions, both the one most supported by the literature, and two more optimistic yield assumptions. Values vary widely from nearly \$0 to \$163 per MT of biochar. Since a 10% increase in yields is the value most supported by the existing literature, as discussed previously, Table 16 incorporates both the yield improvement value, and the carbon

sequestration value, reporting biochar values by crop assuming a 10% increase in yield under all three carbon prices (\$10, \$40, and \$80 per ton of carbon dioxide equivalent). The take-home message comparing Table 15 and Table 16 is that the value of biochar depends significantly on whether there is a climate market for carbon sequestration. As the yield benefit varies from 10% to 30%, the value of biochar is increased somewhat, but an equivalent proportional change in carbon price has a much more significant effect. This is important to consider given the number of studies that question whether these higher yield increases are possible in non-tropical soils.

Table 15: Profit per acre, total acres, and estimated value of biochar (in dollars per metric ton [MT]) from three different levels (10%, 20%, and 30%) of yield improvement for crop groups in the Columbia River Basin of Washington State

Crop Group	Profit per acre ¹	Total acres in region ²	Estimated value of biochar (\$/MT)		
			Yield improvement		
			10%	20%	30%
Alfalfa	678	410,155	6.78	13.56	20.34
Apples	2,248	180,868	22.48	44.96	67.44
Asparagus	238	4,870	2.38	4.76	7.14
Concord	1,509	21,466	15.09	30.18	45.27
Hops	3,481	35,988	34.81	69.62	104.43
Mint	804	27,697	8.04	16.08	24.12
Miscellaneous	785	16,091	7.85	15.7	23.55
Other Grain	3	1,696,983	0.03	0.06	0.09
Other Hay	240	344,253	2.4	4.8	7.2
Other Tree	833	73,332	8.33	16.66	24.99
Other Veg	5,422	480,315	54.22	108.44	162.66
Pasture	479	311,193	4.79	9.58	14.37
Potatoes	1,155	180,254	11.55	23.1	34.65
Sweet Corn	436	65,643	4.36	8.72	13.08
Timothy	701	101,990	7.01	14.02	21.03
Wheat	40	2,309,819	0.4	0.8	1.2
Wine	2,630	56,969	26.3	52.6	78.9

¹ from Yoder et al. (2014)

² acres in Columbia River Basin of Washington State (WSDA, 2016)

Table 16: Estimated value of biochar (in dollars per metric ton [MT] of biochar) assuming a 10% yield increase and three different prices for carbon dioxide equivalents (CO_{2eq})

Crop Group	CO _{2eq} price		
	\$10/MT	\$40/MT	\$80/MT
Alfalfa	36.14	124.22	241.66
Apples	51.84	139.92	257.36
Asparagus	31.74	119.82	237.26
Concord	44.45	132.53	249.97
Hops	64.17	152.25	269.69
Mint	37.4	125.48	242.92
Miscellaneous	37.21	125.29	242.73
Other Grain	29.39	117.47	234.91
Other Hay	31.76	119.84	237.28
Other Tree	37.69	125.77	243.21
Other Veg	83.58	171.66	289.1
Pasture	34.15	122.23	239.67
Potatoes	40.91	128.99	246.43
Sweet Corn	33.72	121.8	239.24
Timothy	36.37	124.45	241.89
Wheat	29.76	117.84	235.28
Wine	55.66	143.74	261.18

10.5 Demand curve

One way to conceptualize the potential regional demand for biochar is with a demand curve - a plotted relationship that shows the value of each unit of a good used. In this case, the good is biochar and its value is the additional profit generated by increasing crop production. In the current absence of climate policy, the demand curve is constructed by assuming that the first unit of biochar is used on the field where it would create the greatest value. Referring to values in Table 15, the first application of biochar would go to the crop group “Other Veg.” Additional biochar would continue to go to this crop group until all of its acres have received biochar. Then, biochar would move onto the next highest valued crop group, which is hops. If one plots out the value of every unit of biochar potentially used in the region in this way, the result is the series of horizontal red lines as shown in Figure 27. An approximation to these series of “steps” is a smooth curve drawn through them (blue line).

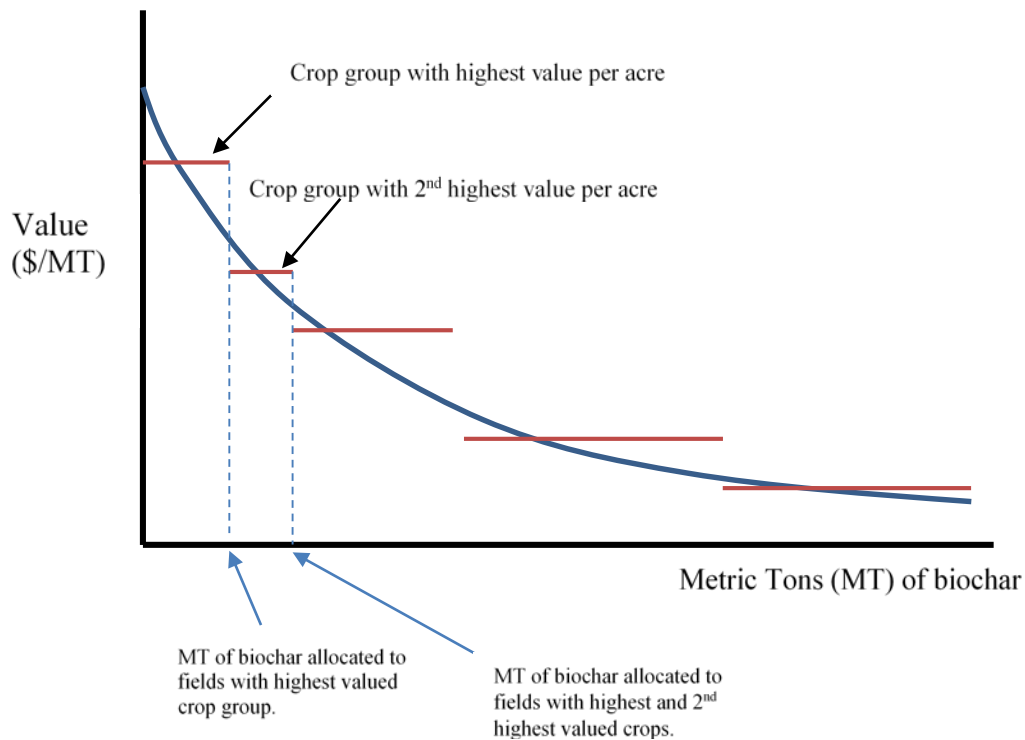


Figure 27: Example of a demand curve for biochar with five crop groups

A large number of potential scenarios were considered in this study, and a demand curve could be drawn for each one of them. However, the curve only has much of a downslope if yield increase is the primary source of value. If carbon prices are over \$10 per MT of carbon dioxide equivalent, then the demand curve becomes fairly flat with a low value of \$117 per MT and a high value of \$170 per MT. Figure 28 puts these values in perspective by showing approximations to biochar demand curves with a 10% yield increase only, and a 10% yield increase with a carbon price of \$40 per MT. There are about 6.5 million acres of cropland agriculture where biochar could be applied in the Columbia River Basin of Washington, which is used to specify the x-axis.

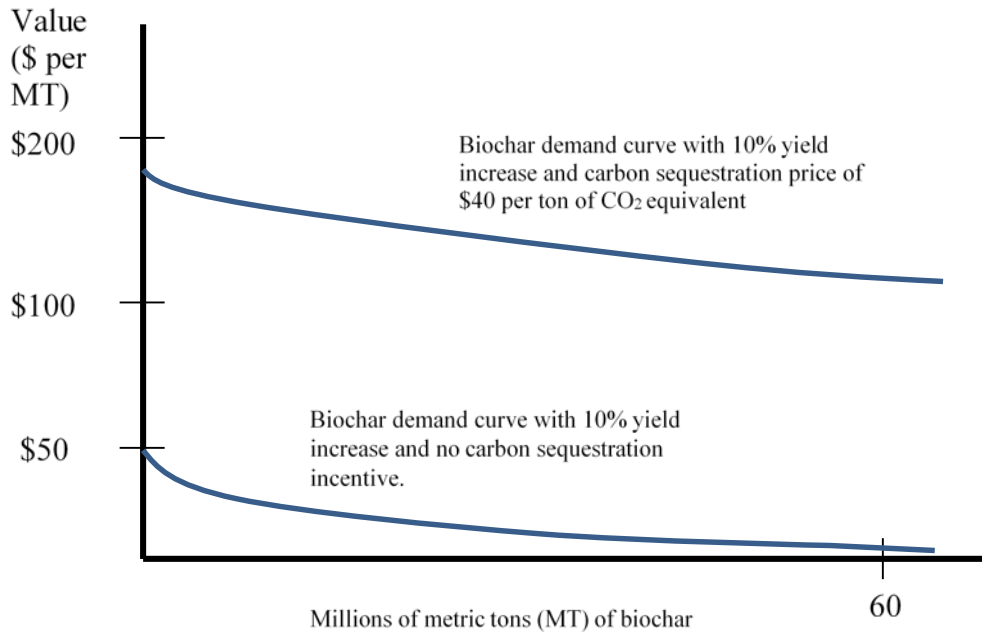


Figure 28: Approximations to biochar demand curves assuming a 10% yield increase, with and without a carbon market

10.6 Conclusion

There is clearly a potential market for biochar in the Pacific Northwest considering production cost and agricultural use values estimated in this report. However, a number of conditions need to be met for the value of biochar to exceed production costs. As discussed in the techno-economic analysis, there is a scenario where the minimum viable selling price for biochar is in the vicinity of \$150 per MT (Figure 24). In the agricultural use value section, there are a number of fields and scenarios where the value of biochar to agriculture exceeds this value. However, there is only one type of crop (mixed vegetables) that could justify the use of biochar without a climate policy that compensates farmers for sequestering carbon, and this requires a fairly optimistic yield improvement assumption of 30%. Biochar use in agriculture becomes much more feasible if there is a carbon market with prices nearing \$40 per MT of carbon dioxide equivalent. It should also be noted that there are a number of assumptions in this study that are uncertain. Our hope is that this report provides a better perspective on the conditions required for an active biochar market to develop. Most importantly, it provides a reference point for others to identify the potential market impacts if costs are lowered or values are increased.

Additional important detail on this work is available in the technical report *Biochar Production in Biomass Power Plants: Techno-Economic and Supply Chain Analyses* on the [WTFT 2017-2019 webpage](#) of Washington State University's CSANR.

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