Life cycle energy and greenhouse gas assessment of the co-production of biosolids and biochar for land application

Leslie Miller-Robbie a, Bridget A. Ulrich a, Dotti F. Ramey a, Kathryn S. Spencer b, Skuyler P. Herzog a, Tzahi Y. Cath a, Jennifer R. Stokes c, Christopher P. Higgins a, *  

a NSF Engineering Research Center ReNUWIt, Department of Civil and Environmental and Engineering, University of California, Berkeley, Berkeley, CA 94720, USA  
b Clackamas County Water Environment Services, Oregon City, OR 97045, USA  
c NSF Engineering Research Center ReNUWIt, Department of Civil and Environmental and Engineering, University of California, Berkeley, Berkeley, CA 94720, USA

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A B S T R A C T  
The co-production and co-application to land of biochar and biosolids may make conventional wastewater treatment more sustainable; therefore, this study provides a first life cycle evaluation of the co-location of a pyrolysis (biochar production) plant with a municipal wastewater treatment plant. Life cycle energy use and greenhouse gas emissions are evaluated for the co-production of these two materials. Three US national scenarios are included: biosolids only produced and disposed of as is the current common practice in the US (landfilled, incinerated, and land applied); biosolids only produced but not land applied; and co-production of biosolids and biochar and their disposal as currently practiced. A case study is also presented with slightly different disposal practices. Large contributors to life cycle greenhouse gas emissions are from the disposal of biosolids in landfills and incinerators, making alternatives like land application attractive for mitigating greenhouse gas emissions. The addition of biochar production adds little to the overall energy use but provides substantial (26%) reduction in greenhouse gas emissions for the national case, largely due to the recalcitrant carbon storage in biochar. Because biochar production is an energy positive process, incorporation of the energy co-products into the plant is explored for producing high quality Class A biosolids. The addition of biochar to biosolids is shown to be more sustainable from a co-production standpoint. This study contributes a benchmark analysis in evaluating the co-production of biochar and biosolids for potential environmental, economic, and health benefits and the results are useful for utility planning.

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1. Introduction  
The long-term sustainability of conventional wastewater treatment may be threatened by potential environmental and human health risks associated with their end-of-life. Land application of biosolids is a widespread practice on a global scale (Chang et al., 2002). Currently in the United States, 55% of biosolids are land applied with the remaining 45% disposed through incineration and landfiling (North East Biosolids and Residuals Association (NEBRA), 2007). Land application allows for the nutrients in biosolids to be recycled back into the environment, displacing energy-intensive synthetic fertilizers that unnaturally fix high levels of nitrogen (N) (Vitousek et al., 1997) and increasing long-term reserves of carbon in soils. However, chemicals in biosolids may contribute to contamination of various environments.

Domestic wastewater contains thousands of synthetic organic and inorganic chemicals which remain in the biosolids following wastewater treatment. Levels of metals and pathogens in biosolids are regulated in the US (United States Environmental Protection Agency, 1994), but other pollutants, including synthetic organic chemicals, are not. As contaminants of emerging concern are being studied (Kinney et al., 2006; Sepulvado et al., 2011), further restrictions on biosolids land application may be implemented to protect surrounding water and soil quality from contamination (Camp Dresser and McKee Inc., 2011). In some European countries, agricultural land application of biosolids has been restricted due to concerns about related health risks (European Commission, 2002; Fytiti and Zabaniotou, 2008). To reap the benefits of biosolids...
land application in the future, one strategy is to co-apply a material that sorbs contaminants to mitigate their potential adverse environmental impacts.

A potential sorption material for many contaminants of emerging concern is biochar, a charcoal-like material made from pyrolysis of agricultural residues (e.g., corn stover, wood wastes, yard trimmings, etc.). Pyrolysis stabilizes the carbon taken up by plants and stores it in biochar, providing a carbon-sink (Meyer et al., 2011; Sparrevik et al., 2013; Woolf et al., 2010). Biochar has been used throughout the world to increase soil health and fertility (Glaser et al., 2001; Kimetu et al., 2008). It has been shown to increase soil nutrient retention and nutrient availability (Belanger et al., 2004; Major et al., 2010; Prendergast-Miller et al., 2014). There is also growing evidence of the sorption capacity of biochar for herbicides and metals (Park et al., 2011; Spokas et al., 2009; Uchimiya et al., 2010). The production of biochar is a net energy positive process, as syngas, bio-oil, and thermal heat are also produced during pyrolysis (Laird et al., 2009).

Although the life cycle environmental tradeoffs of biosolids and biochar have been studied separately (Peters and Rowley, 2009; Roberts et al., 2010), their combination has not been investigated prior to this study. In addition to the life cycle assessment (LCA) that includes three national-level scenarios and four scenarios applied to a real-world case study utility, the integration of energy between the pyrolysis plant and the wastewater treatment plant (WWTP) is explored.

2. Materials and methods

Process-based LCA methods were used for this LCA model. Material flow data were collected from both primary and secondary sources. Only energy use (EU) and greenhouse gas (GHG) emissions were included as impact categories. The impact assessment was carried out using global warming potential factors from IPCC’s 2007 Report (Intergovernmental Panel on Climate Change (IPCC), 2007). Global warming potential (in terms of carbon dioxide equivalents (CO2e)) was calculated for a 100-year time horizon. Carbon sequestration potential was only evaluated for the recalcitrant fractions of biochar and biosolids that have been shown to be stable for 100 years or longer (Recycled Organics Unit, 2007; Woolf et al., 2010). All infrastructure included in this study was assumed to have a 50-year lifetime (Horvath and Stokes, 2011). While all analysis is compliant with ISO 14040 standards (International Organization for Standardization, 2006), an extensive review by two LCA contributors was carried out in lieu of a review by an independent practitioner. Tables S1 and S2 in the supplementary information detail the data sources and emission factors employed throughout this study.

2.1. Goal and scope

This study uses LCA as a preliminary step to evaluate the EU and GHG emissions of a new approach in which yard trimmings are delivered to a biochar plant co-located with a WWTP. Upon completion, the two products are mixed and transported for land application in the model. This scenario is compared with other conventional and non-conventional scenarios. The study boundaries included all materials and energy inputs and outputs, as well as the energy to produce and transport all inputs and outputs, starting with waste biochar feedstock (domestic yard trimmings) and wastewater solids or biosolids input (Fig. 1). The WWTP prior to wastewater solids or biosolids input (including the anaerobic digester itself) was outside the study boundary. Building materials associated with the WWTP were not included because they are
necessary regardless of the addition of a pyrolysis plant. However, all construction materials associated with the pyrolysis plant were included. The pyrolysis plant was allocated solely to biochar and not to co-products because the potential uses for syngas and bio-oil were outside the scope of the LCA. This assumption is conservative because the inclusion of plant construction in the biochar analysis increases EU and GHG impacts. In reality, when co-products are included, the scenarios that include biochar will be even more beneficial. As shown in Fig. 1, utilizing waste heat is within the scope because it is used to dry feedstock; however, it cannot be transported to offset energy use in other systems, and related GHG emissions are assumed to be negligible. Energy to collect, transport, and shred biochar feedstock was included, but not the energy to cultivate the yard material because feedstock production is inevitable. Yard trimmings not used for biochar production in the national scenarios are either composted commercially (−60%) or landfilled (−40%) (United States Environmental Protection Agency, 2010). Yard trimmings in the case study are 100% composted commercially. Associated EU and GHG emissions were calculated for operations at each modeled facility. For land application, land-filling and waste-to-energy disposal practices, biosolids were produced using anaerobic digestion and dewatered prior to transportation. For incineration, wastewater solids from primary and secondary treatment were dewatered prior to incineration. Avoided processes as a result of this new approach were also inside the boundary, including avoided fertilizer, carbon dioxide (CO2) emissions sequestered as carbon in biosolids-amended soils, and carbon sequestered in biochar.

2.2. Study scenarios and case study

Three national-level scenarios for US biosolids handling operations and four case study scenarios for the Tri-City WWTP in Clackamas County, Oregon are evaluated. The national-level scenarios are:

N1) Current US biosolids disposal practice (55% land applied, 30% landfilled, and 15% incinerated) (North East Biosolids and Residuals Association (NEBRA), 2007);

N2) Biosolids land application is discontinued and all biosolids are landfilled or incinerated at rates consistent with current disposal practices (67% and 33%, respectively);

N3) Biochar (yard trimmings feedstock) and biosolids are co-produced, mixed at an approximate ratio of 1:3.5 biochar to biosolids (by weight) and land applied (only 55% of the total biosolids are used), and the remaining biosolids landfilled (30%) and incinerated (15%).

For the national-level study, the functional unit was one dry metric ton of biosolids generated at a WWTP, and the reference flow was 8.25 dry metric tons biosolids per day. A city of 300,000 people is expected to generate this reference flow (0.01 dry metric tons biosolids per person per year for 2009) (Metcalf and Eddy, 2003; Peters and Rowley, 2009), which is used to size the theoretical biochar and wastewater treatment plants. The amount of yard trimmings available for the theoretical city was conservatively estimated from global availability in urban areas at 0.005 dry metric tons yard trimmings per person per year (2009–2010 basis) (United States Environmental Protection Agency, 2010: Woolf et al., 2010). Biochar production from this feedstock was estimated at 1.3 dry metric tons of biochar per day. Unless specified otherwise, all units herein referred to as tons are dry metric tons.

To account for the specific disposal practices and local conditions for the case study WWTP, specific local scenarios (similar to the national scenarios) were included for the Tri-City WWTP case study. For example, the biosolids land application rate for this WWTP (84%) is higher than the national average (55%). This is expected as the rate for beneficial reuse of biosolids is high in the state of Oregon, estimated at approximately 98% of those produced (Oregon Association of Clean Water Agencies (2012)). Also, the availability of yard waste was higher than global averages, which allowed for a higher ratio of biochar to biosolids produced. The four case study scenarios include:

C1) Current Tri-City WWTP biosolids disposal practice (83.7% land applied, 14.6% landfill, and 1.6% combusted in a waste-to-energy facility);

C2) Biosolids land application is discontinued with all biosolids landfilled or combusted at ratios consistent with current disposal practices (90% and 10%, respectively);

C3a) Biochar (yard trimmings feedstock) and biosolids are co-produced at rates consistent with material availability and mixed at an approximate ratio of 1:1 biochar to biosolids (by weight) and land applied (only 83.7% of the total biosolids are used). The remaining biosolids are landfilled (14.6%) or combusted (1.6%);

C3b) Same as C3a, but biochar production is based on the amount of energy from excess heat generated during the pyrolysis process required to produce Class A biosolids and therefore mixed at an approximate ratio of 1:3.8 biochar to biosolids (by weight) and land applied.

Due to the large amount of yard trimmings available in the case study, slightly more biochar than biosolids can be produced for land application, and therefore, a large amount of excess energy is available for the model. Scenario C3b is included to limit biochar production based on the amount of energy that is needed to produce higher quality, Class A biosolids.

For the Tri-City WWTP case study (2013 average flow of 9 million gallons per day (MGD)), the functional unit remained one ton of biosolids, and the reference flow was 3 tons biosolids per day, generated by the 64,000 people that this plant serves (0.02 tons biosolids per person for the year of 2013). The amount of yard trimmings recovered and commercially composted from these 64,000 residents was estimated using data collected by the Oregon Department of Environmental Quality (personal communication with Peter Spendelow) for the Portland metropolitan area and was 10.8 tons per day (0.06 tons of yard trimmings per person for the year 2012; 55% moisture content assumed (Cooger et al., 2002)). From this feedstock, 2.6 tons of biochar per day could be produced at the pyrolysis plant.

2.3. Wastewater solids and biosolids systems

The system boundaries included the dewatering process of the wastewater solids or biosolids with the addition of polymer and high-speed centrifugation, followed by transportation to final deposition, and end-of-life of the biosolids (Fig. 1). An emulsion polymer is used (national case: 27 kg neat polymer per ton biosolids and 13 kg neat polymer per ton wastewater solids (United States Environmental Protection Agency, 2000); case study: 44 kg neat polymer per ton biosolids; assumed 41% active). Methods considered for managing wastewater biosolids at end-of-life include beneficial reuse by land application of Class B biosolids, disposal of dewatered biosolids in a landfill, and disposal of incinerated solids ash. The case study included combustion of biosolids at a waste-to-energy facility but not incineration.
2.3.1. Land application

The modeled process for biosolids land application included dewatering biosolids from an anaerobic digester, transporting to the land application site, and spreading the biosolids across agricultural fields. Dewatering was modeled with the addition of polymer to the biosolids prior to centrifugation to achieve 78% moisture content (Metcalf and Eddy, 2003; National Biosolids Partnership, 2005; Peters and Rowley, 2009).

For both the national and case study scenarios, dewatered biosolids (78% moisture content) were transported to the land application site (412 km round trip). Biosolids were transported twice per week with a model GMC c8500 Dump Truck (cargo weight 12 tons per load assumed) (Horvath and Stokes, 2011) and were transported three times per week by a 5-axle semitrailer combination truck (cargo weight limit 24 tons per load assumed) (Bureau of Transportation Statistics, 2012; Federal Highway Administration, 2000; Parker Ag Services LLC).

Biosolids or biosolids/biochar mix were modeled as being spread onto land using a tractor and a manure spreader (personal communication with Mike Scharp of Parker Ag Services) fueled by off-road diesel (Argonne National Laboratory, 2010). The total nitrogen content of the biosolids was used to calculate the amount of urea N fertilizer avoided, equating to 41.8 kg energy avoided (Sawyer et al., 2010) and 4000 kg CO2e avoided per ton N in biosolids (Sylvis Environmental, 2011). An estimated first-year, plant-available nitrogen factor for anaerobically digested biosolids of 35% was included to calculate avoided fertilizer (Pacific Northwest Extension, 2007). Emission factors for CO2, methane (CH4), and nitrous oxide (N2O) associated with land application were taken from the BEAM 1.1 model (Sylvis Environmental, 2009, 2011) (Table 1).

2.3.2. Landfilling

To dispose of biosolids at a landfill, it was assumed that stabilization would be required to reduce vector attraction. Anaerobic digestion produced biosolids destined for the landfill, and dewatering was modeled in the same manner as for land application (United States Environmental Protection Agency, 1999). Nationally, about 4% of biosolids are disposed of in dedicated surface disposal units, or monofills (North East Biosolids and Residuals Association (NEBRA), 2007). In this study, it was assumed that processes associated with monofills are similar to landfills.

For both the national and case study scenarios, dewatered biosolids were transported to a landfill (226 km round-trip) using a dump truck and a combination truck (described above), each for half of the trips and stockpiled at the landfill. Standard landfill/earth moving equipment (Caterpillar D8N Dozer) was used to spread and compact the biosolids (Horvath and Stokes, 2011). Table 1 provides emission factors attributed to landfilling biosolids.

2.3.3. Incineration

The third solids-management method considered was incineration of wastewater solids and landfilling of the ash residue. In this study, 1 ton of biosolids (digested solids) were produced for every 1.7 tons of wastewater solids (undigested solids) due to the loss of volatile solids during anaerobic digestion, for a total of 13.75 tons wastewater solids per day (Metcalf and Eddy, 2003). Kim et al. states that the minimum gross calorific content of solids required to sustain autothermal combustion is 3.35 MJ/kg (Kim et al., 2005). Using higher heat values for primary and secondary solids (25.6 MJ/kg and 18.6 MJ/kg, respectively) (Metcalf and Eddy, 2003), the minimum solids content required to sustain autothermal combustion is 15% solids, which is achieved by centrifugation.

For the national scenarios only, incineration was modeled to take place on site so transportation of wastewater solids would not be required. The incinerator was specified to be a continuously operated fluidized bed incinerator with electrostatic precipitation and wet scrubbing for the air emissions control equipment. Direct emissions were based on CH4 and N2O released during high-temperature incineration (48.5 g-CH4 and 4304 g-N2O per ton undigested solids) (Sylvis Environmental, 2011), and the energy use was based on the net consumption of electricity for operation and effluent gas treatment (1753 MJ per ton undigested solids) (Sub and Rousseaux, 2002).

The modeling assumed that ash and solid waste from the electrostatic precipitator would require transport to a landfill (226 km round-trip). The ash content of wastewater solids was assumed to be 25% (Metcalf and Eddy, 2003) and combustion was assumed to be complete, generating 3.4 tons of ash per day with a density of 500 kg/m3 (Kosior-Kazberuk, 2011). It was assumed that the solid waste generated was not hazardous based on the industrial pre-treatment regulations that require wastewater utilities to develop and implement a program to protect plant operations (including solids management).

2.3.4. Combustion at waste-to-energy facility

For the case study only, a small amount of biosolids (2%) are sent to a mass-burn waterwall resource recovery facility which uses dry flue gas scrubbers and fabric filter baghouses to control emissions of acid gases and particulates. The net energy production for these biosolids is estimated at 15 MJ per ton biosolids based on a dewatered biosolids energy content of 82 MJ per ton biosolids and a plant energy production of 19% net/total. Direct emissions are based on CH4 and N2O released during high-temperature incineration (48.5 g-CH4 and 4304 g-N2O per ton digested solids) (Sylvis Environmental, 2011). Transport modes to the facility are modeled the same as for landfill (above) and the distance assumed is 115 km per load. Ash transport attributed to biosolids is assumed to be negligible and is not modeled in this study.

2.4. Biochar system

Biochar is produced by thermal decomposition of biomass under oxygen-limited conditions. While several technologies exist for biochar production, the main body of literature focuses on pyrolysis and gasification (Meyer et al., 2011). Pyrolysis occurs under oxygen-free conditions, while gasification occurs under low-oxygen conditions. Gasification processes are typically optimized for syngas production, having lower biochar yields than pyrolysis processes. Moreover, research suggests that slow pyrolysis (longer residence times and lower temperatures than fast pyrolysis) has a greater potential for carbon sequestration, retaining up to 50% of the...
feedstock carbon as stable carbon in the biochar (Gaunt and Lehmann, 2008; Ibarrola et al., 2012). Therefore, a slow pyrolysis process with high biochar yields was modeled for this study to evaluate sequestration benefits. For all appropriate scenarios, the biochar production system was based on an improved-kiln slow pyrolysis reactor operated by Biochar Solutions Inc. (Carbondale, CO) (Biochar Solutions, 2013). While the materials requirements and production capacity specific to this system were considered, the material and energy balance parameters (i.e., biochar and co-product yield, biochar composition) were based on a broader compilation of data from the literature to make these results more broadly applicable. However, the mass and energy balance parameters considered in this analysis were generally within ~30% of those communicated by Biochar Solutions (Biochar Solutions, 2013).

This system included four phases: pre-processing, start-up, pyrolysis reaction, and post-processing. During pre-processing, the feedstock is homogenized in an electric hopper and then dried. Once the feedstock is fed to the reactor, a small amount of natural gas is used for start-up, or ignition of the feedstock. During slow pyrolysis, the feedstock is thermally degraded under oxygen-free conditions at peak temperatures of 300–600 °C and residence times of 8–24 h (Mayja, 2012). Longer residence times are required for lower temperature processes, and vice versa. The peak process temperature presents a trade-off in char properties — lower temperatures mean higher biochar yields but higher temperatures mean more of the carbon in the biochar is recalcitrant (i.e., fixed carbon that will not enter the atmosphere as CO2). Therefore, a moderate peak temperature of 450–500 °C and residence time of 4–6 h was chosen to balance these properties (personal communication with Jonah Levine of Biochar Solutions, Inc.). Once the pyrolysis reaction is complete (during post-processing), the char is air-dried, sieved, and packaged in plastic drums.

The three products of this process are biochar (primary product), syngas, and bio-oil (co-products). Biochar yields for slow pyrolysis are generally between 20% and 50% by dry weight for a wide range of feedstocks, including nut shells, corncobs, husks, stalks, and various types of wood (Mayja, 2012); a biochar yield of 30% was assumed here to be representative of the chosen pyrolysis conditions (Mayhead, 2010). Furthermore, it was assumed that the process uses yard trimmings as the input to achieve high recalcitrant carbon contents (e.g., 80% of carbon in biochar is fixed) (Roberts et al., 2010; personal communication with Jonah Levine of Biochar Solutions, Inc.). Biochar produced from feedstocks with high nutrient contents (e.g., food waste and manure) tend to have less recalcitrant carbon and more amorphous labile carbon (no defined structure and unstable) than those produced from plant feedstocks (e.g., wood, yard trimmings). Therefore, only plant-based feedstocks were considered for this study to increase the carbon sequestration potential of the biochar (Harvey et al., 2012; Singh et al., 2012).

Syngas and bio-oil are generally produced in similar quantities during slow pyrolysis, thus it was assumed that the yield for each co-product is 35% by weight (Mayhead, 2010). The syngas is primarily composed of carbon monoxide, hydrogen gas, and CH4, while bio-oil is composed of larger oxygen-containing organic molecules. Bio-oil has a relatively high energy content; when compared to the heating value of petroleum fuels, bio-oil from fast pyrolysis has a relative heating value of approximately 40% (on a mass basis), while syngas has a relative heating value of approximately 15% (Czernik and Bridgewater, 2004; Laird et al., 2009). Both co-products may be used in conventional boilers and consequently in on-site applications. Additionally, it was assumed that 90% of the heat lost through the reactor walls can be recovered and used for heating applications such as drying feedstock (Roberts et al., 2010). In the model used in this study, ~35% of the heat from pyrolysis was used for drying feedstock and the remaining was used elsewhere.

For this study, CO2 emissions resulting from the combustion or decay of syngas, bio-oil, and the labile portion of biochar were not counted. The major reasons are: 1) the carbon in the plant tissue was captured from the atmosphere during the plant’s lifetime (biogenic), 2) the biomass source is a waste byproduct of other human activities, and 3) no land use change occurs to produce this biomass (Searchinger et al., 2009). Likewise, in scenarios where the yard trimmings are not used to make biochar, they are composted and landfilled, and resulting biogenic CO2 emissions are not counted though CH4 and N2O emissions are included as appropriate. However, the recalcitrant fraction of carbon in biochar has been shown to have a half-life of greater than 100 years, even millennia (Woolf et al., 2010). Therefore, this carbon was assumed to be sequestered in biochar and credits were allocated herein.

2.5. Plant energy integration

Co-location of a pyrolysis plant with a WWTP facilitates integration of the excess energy products from biochar production into the WWTP. The syngas and bio-oil produced by pyrolysis can be combusted to power a boiler. The heat energy from the reactor can be recovered directly by pumping a heat transfer fluid through a jacket around the reactor. Therefore, the co-products of biochar could be used for electrical generation, to heat buildings, and to heat biological processes in cold weather seasons. The excess heat from the reactor may be best used within the WWTP to produce a higher quality biosolids product for land application as it can be used for drying processes without consuming additional fossil fuels.

Most health concerns associated with biosolids application can be reduced by meeting the strict requirements for producing Class A biosolids. Class A biosolids are considered of exceptional quality and can be used for broader agricultural applications when compared to Class B biosolids; according to the US EPA’s Part 503 Rule, Class A biosolids “can generally be applied as freely as any other fertilizer or soil amendment to any type of land” (United States Environmental Protection Agency, 1994). Generating Class A biosolids increases the amount of biosolids that can be land-applied and decreases transport distances associated with biosolids application because more options are available. Unfortunately, Class A criteria do not account for potential health risks associated with contaminants of emerging concern in biosolids, but the co-application of biochar and Class A biosolids could alleviate some of these concerns as discussed in Section 3.4.

To achieve Class A biosolids, US EPA regulations must be met for vector attraction, pathogen content, and metals. To sufficiently reduce vector attraction and pathogen content to required levels, biosolids must be dried in 10% water content at 80 °C or higher for 20 min or longer (United States Environmental Protection Agency, 1994). Wastewater influent levels for metals were assumed to be low enough that the biosolids meet the metals content requirements following treatment at the WWTP, which is often the case with domestic wastewater. The amount of energy needed to produce Class A biosolids is equivalent to that required to dewater biosolids to 10% content, as shown in Equation (1):

\[
E_D = (M_{WC} - M_{WD}) \times E_{WR}
\]

where \(E_D\) is the energy required to dry biosolids (MJ/day), \(M_{WC}\) is the mass of water in centrifuged solids (tons water/day) [tons biosolids/day * % water in centrifuged solids], \(M_{WD}\) is the mass of water in dried solids (tons water/day) [tons biosolids/day * % water in target solids], and \(E_{WR}\) is the dryer energy requirements (MJ/tons.)
water removed). This energy requirement was compared to the amount of available energy from biochar production in Section 3.3. The production of Class A biosolids using a Komline-Sanderson paddle dryer to achieve this drying requirement is examined in the results for scenario C3b.

3. Results and discussion

The life cycle EU and GHG emissions results are presented followed by an analysis of energy integration between the WWTP and pyrolysis plant and a discussion of biochar, biosolids, and contaminants of emerging concern for co-application.

3.1. Energy use and GHG emissions from co-production of biosolids and biochar

For all scenarios, EU and GHG emissions by life cycle stage (i.e., inputs to biochar production as well as inputs to biosolids processing, transportation, and end-of-life) are summarized in Figs. 2 and 3. Results show that biosolids processing is the most energy-intensive life cycle stage for all scenarios, largely due to electricity use for dewatering biosolids in a centrifuge and offsite production of the polymer used for biosolids dewatering (Fig. 2). The higher energy use for transportation in scenarios N3 and C3a can be attributed to the transport of materials for biochar production (primarily the transport of natural gas needed for start-up of the pyrolysis process). Energy use at end-of-life is dependent on the type of disposal and how much it is used for each scenario. Higher energy use at end-of-life is due to electricity use for incineration, and smaller amounts are due to fuel for spreading and compacting biosolids into a landfill and for spreading biosolids at land application sites.

In Fig. 3, the GHG emissions do not follow the pattern of energy use (Fig. 2) because not all GHGs are energy related. For example, in the national scenarios (Fig. 3A), the end-of-life stage contributes the highest GHG emissions primarily due to direct emissions during landfiling and incineration. For the case study scenarios (Fig. 3B), impacts from large amounts of yard trimmings and biochar are reflected in the high emissions from composting and landfilling yard trimmings for scenarios C1 and C2, and a high amount of negative emissions due to recalcitrant carbon in the biochar at end-of-life.

Overall, for the national case (Fig 2A), the addition of biochar production adds 5% to EU between scenarios N1 and N3 but results in a 26% reduction in GHG emission (Fig. 3A). When compared to discontinued land application (scenario N2), the addition of biochar production is even more sustainable, with the N3 scenario yielding a 58% reduction in GHG emissions. Co-application of biochar with biosolids may also allow for continued land application and nutrient recycling.

For the case study (Fig. 2B), overall EU is increased slightly with the addition of the pyrolysis plant (when compared to C1, a 9% increase for C2 and a 3% increase for C3a). Large reductions in GHG emissions for C3a and C3b result from fewer emissions from the biochar pyrolysis process when compared to alternative uses for yard trimmings with even greater reductions from the recalcitrant fraction of carbon in the biochar itself. Overall, the results show that small investments in materials and energy to add biochar production are effective in decreasing total system life cycle GHG emissions at various scales. Importantly, the results displayed in Figs. 2 and 3 do not account for energy generated from the pyrolysis plant.

A further breakdown of the largest GHG contributors to each life cycle stage for co-production (scenarios N3 and C3a) is illustrated in Fig. 4. For the national scenario (Fig. 4A), the largest GHG emissions contributor is direct biosolids disposal (landfill and incineration). There are also high energy-related GHG emissions due to the electricity use in dewatering biosolids and polymer production. For the national case, biochar production itself adds very little to overall GHG emissions (less than 1%).

For the case study (Fig. 4B), polymer production is a large contributor to GHG emissions due to a higher amount of polymer added to dewater biosolids when compared to the national case. A greater amount of biochar produced for this scenario also results in higher GHG emissions in biochar production and transport stages. A larger proportion of biosolids are land applied in scenario C3a and therefore disposal impacts are lower when compared to scenario N3, but it remains a significant contributor.

3.2. End-of-life life cycle stages for biosolids and biochar

A direct comparison of the end-of-life options is illustrated in Fig. 5 to better understand the GHG emissions per ton of biosolids (i.e., the percentages of biosolids destined for each option are not taken into account here). Incineration produces the highest total GHG emissions in CO2 equivalents (Nakakubo et al., 2012; Sylvan Environmental, 2009), followed by waste-to-energy (based on the same direct emissions from incineration) and landfiling for this model (Peters and Rowley, 2009). Land application has less impact due to lower direct emissions on site and offsets from avoided...
fertilizer use and carbon sequestration observed in biosolids-amended soils. Therefore, increasing the amount of land–applied biosolids provides GHG emissions benefits. These results are consistent with earlier work (Murray et al., 2008), which estimate that biosolids disposal (landfilling and combustion) are more problematic than beneficial reuse (land application) in terms of EU and GHG emissions.

Because biochar production is an energy positive process, the resulting energy co-products (syngas, bio-oil, and thermal heat from pyrolysis) were also quantified. This excess energy is not accounted for in the overall EU comparisons above, but is shown for co-production of biosolids and biochar in Fig. 6A and B. The EU values of the co-products are on the right. The results show that when co-products are included in the LCA, approximately 69% of the total energy use is produced in the co-products (Fig. 6A and C). Utilizing the energy-rich co-products could offset other energy sources and associated GHG emissions within the WWTP (see Section 3.3); however, such avoided GHG emissions are not considered in this model.

Fig. 6C shows the extent to which recalcitrant carbon in biochar contributes to reductions in system GHG emissions for the national case. For the case study (Fig. 6D), the large amount of biochar

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### Fig. 3.
Greenhouse gas emissions for all A) national and B) case study scenarios, displayed by life cycle stage. LA = land application. For scenarios without biochar production (N1, N2, C1, and C2), GHG emissions attributed to inputs for biochar production are the impacts from composting and landfilling of yard trimmings that would occur in lieu of biochar production.

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### Fig. 4.
Greenhouse gas emissions for co-production of biosolids and biochar for both the national scenario (N3, panel A) and case study scenario (C3a, panel B) by life cycle stage. Units are kg CO₂e per dry metric ton biosolids. Avoided/negative impacts and carbon stabilized in biochar are not shown.

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### Fig. 5.
GHG emissions for biosolids end-of-life options. Current percentages for disposal practices are not applied and data are shown per ton of biosolids disposed for each option. Equipment fuel combustion for spreading and compacting at the landfill is included, but is 0.4% of the total for GHG emissions at the landfill. Likewise, electricity generated at the waste-to-energy facility is ~0.4% of the waste-to-energy total and equipment fuel combustion for spreading biosolids on land is 3% of the landfill total. For the avoided fertilizer shown in this figure, 3.3% total nitrogen biosolids content (dry weight) is assumed (Metcalf and Eddy, 2003). Dotted lines represent the total for each option after including any negative emissions.

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produced for scenario C3a contributes to large reductions in GHG emissions from recalcitrant carbon in biochar, but also at the land application site due to soil carbon sequestration and avoided fertilizer needs. GHG emissions from the labile carbon in biochar and any subsequent combustion-related emissions from biochar co-products (syngas and bio-oil) are not included due to their biogenic nature (see Section 2.4).

3.3. Energy integration at the WWTP

Co-production of biochar and biosolids is attractive from an energy standpoint because the energy-rich co-products generated from the pyrolysis process can offset energy use in the WWTP. Based on the model for biochar production at the Tri-City WWTP (9 MGD), there is an estimated 14,801 kWh per day available from excess heat, syngas, and bio-oil (scenario C3a). According to a report by the Electric Power Research Institute, electricity use for a 10 MGD advanced WWTP with nitrification is 17,912 kWh per day total of which 800 kWh per day is used for lighting and buildings (Burton, 1996).

Regarding the production of Class A biosolids, Scenario C3b for the case study models the system impacts with a sufficient (but not excess) amount of feedstock needed to produce enough excess useable heat from the pyrolysis reactor alone to generate Class A biosolids: 2.8 tons yard trimmings per day resulting in 0.7 tons biochar per day and 4670 MJ in excess heat per day after using a portion to dry feedstock (1297 kWh per day). The excess heat can be used to dry 2.5 tons of land-applied biosolids per day to 10% water content to achieve Class A biosolids quality.

For the national case, a total of 8533 MJ per day (2370 kWh per day) is needed to dry the 4.5 tons of land-applied biosolids per day. The excess heat amounts to 9065 MJ/day (2518 kWh/day). Therefore, based on the modeled national average, more than 100% of the energy needs for producing Class A biosolids are available and can be used in a dryer to dry biosolids.

3.4. Biochar, biosolids, and contaminants of emerging concern

Despite the sustainability benefits of biosolids use for land application, the practice involves the risk of releasing contaminants of emerging concern into the environment. Current regulations for biosolids focus on heavy metals only (United States Environmental Protection Agency, 1994). Other contaminants of concern include nutrients, biocides, flame retardants, steroids, hormones, and pharmaceuticals (United States Environmental Protection Agency, 2009). Some studies have shown that the use of biochar can sorb metals (Mohan et al., 2007; Uchimiya et al., 2010), decrease nitrate leaching from biosolids-amended soils (Knowles et al., 2011), decrease metal leaching and plant uptake from contaminated soils (Park et al., 2011), and reduce N2O and ammonia (NH3) emissions without the reduction of plant-available nitrogen (Taghizadeh-Toosi et al., 2011). Another study found that biochar amendment can either enhance or suppress GHG emissions relative to control soils depending on the soil and biochar properties (Spokas and Reicosky, 2009). The addition of biochar to biosolids may sorb contaminants of emerging concern (Koelmans et al., 2006); however, more experimental studies are needed to investigate potential reduction in contaminant transport. Literature gaps remain and include a general lack of studies evaluating impacts associated with the co-application of biochar and biosolids to agricultural lands (e.g., long-term carbon sequestration, soil fertility, crop yields, etc.), discrepancies in the application rates of investigated biosolids and

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biochar (Park et al., 2011; Spokas and Reicosky, 2009), variability in the properties of investigated biochar and biosolids (Mohan et al., 2007), and differences in soil types and climates between investigated sites. Due to the complexity of the biochar-biosolids-field system, future studies must be rigorous in isolating variables during experiments.

4. Conclusions

Biochar provides many benefits to biosolids management in terms of energy integration and reduction of GHG emissions at the plant level. This study shows that the small amount of energy and related GHG emissions invested in a pyrolysis plant can provide a large amount of usable energy and life cycle GHG reductions—scenario N3 GHG emissions are 26% less than scenario N1. Increased land application and decreased disposal of biosolids in landfills and incinerators can significantly reduce GHG emissions, as disposal contributes high GHG impacts. The case study results show that there may be an optimal point where both EU and GHG impacts are reduced and Class A biosolids are produced.

For all co-production scenarios (N3, C3a, and C3b), 100% of the energy needs to produce Class A biosolids are available directly from the pyrolysis plant in the form of waste heat. Class A biosolids are superior product that can be applied to various types of land. With increasing concern for continued release of contaminants of emerging concern to the environment through biosolids land application, solutions that simultaneously address global concerns as well as local human and ecological health are needed. This study contributes a first attempt in evaluating the co-production of biochar and biosolids for potential environmental, economic, and health benefits. The results can be used by WWTP utilities in planning for more sustainable approaches to energy use within the plant and land application of biosolids.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.jclepro.2014.12.050.

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