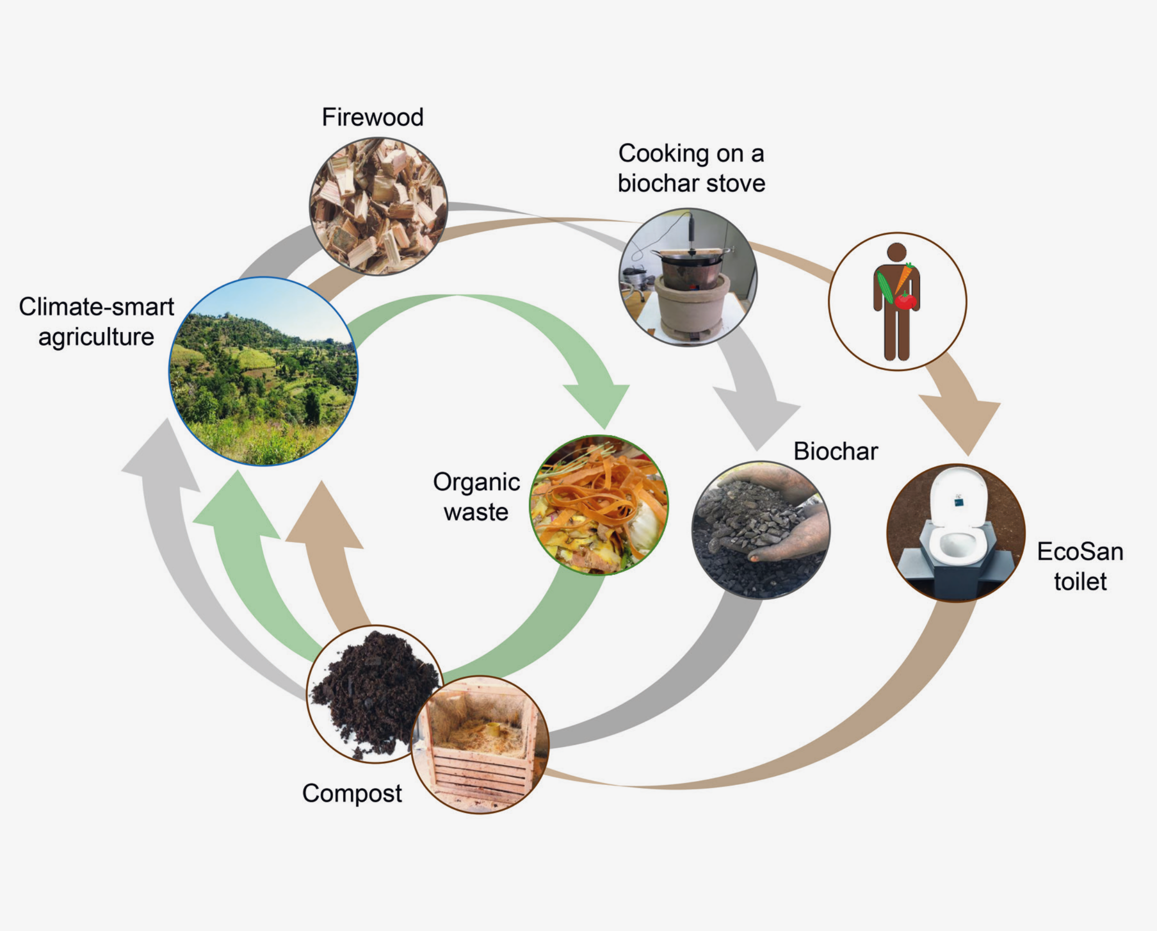


Ecological sanitation via thermophilic co-composting of humanure and biochar as an approach to climate-smart agriculture

Daniela Castro Herrera

Energie & Umwelt / Energy & Environment
Band / Volume 573
ISBN 978-3-95806-622-9

Mitglied der Helmholtz-Gemeinschaft

JÜLICH
Forschungszentrum

Schriften des Forschungszentrums Jülich
Reihe Energie & Umwelt / Energy & Environment Band / Volume 573

Forschungszentrum Jülich GmbH
Institut für Bio- und Geowissenschaften
Agrosphäre (IBG-3)

Ecological sanitation via thermophilic co-composting of humanure and biochar as an approach to climate-smart agriculture

Daniela Castro Herrera

Schriften des Forschungszentrums Jülich
Reihe Energie & Umwelt / Energy & Environment Band / Volume 573

ISSN 1866-1793 ISBN 978-3-95806-622-9

Bibliografische Information der Deutschen Nationalbibliothek.
Die Deutsche Nationalbibliothek verzeichnet diese Publikation in der
Deutschen Nationalbibliografie; detaillierte Bibliografische Daten
sind im Internet über http://dnb.d-nb.de abrufbar.

Herausgeber und Vertrieb: Forschungszentrum Jülich GmbH
Zentralbibliothek, Verlag
52425 Jülich
Tel.: +49 2461 61-5368
Fax: +49 2461 61-6103
zb-publikation@fz-juelich.de
www.fz-juelich.de/zb

Umschlaggestaltung: Grafische Medien, Forschungszentrum Jülich GmbH

Titelbild: Forschungszentrum Jülich GmbH

Organic waste:
“Compost can be beautiful” by Ruth Hartnup
https://www.flickr.com/photos/ruthanddave/33886991770/
License CC BY 2.0

Climate-smart agriculture:
“Agroforestry mountain landscape, Ethiopia”
by Stefan Hügel and Ruth Schaldach
https://www.ruvival.de/examples-agroforestry-systems-ethiopia/
License License CC BY-SA 4.0

Person icon:
https://commons.wikimedia.org/wiki/File:Person_icon_BLACK-01.svg
License CC BY-SA 4.0

Druck: Grafische Medien, Forschungszentrum Jülich GmbH

Schriften des Forschungszentrums Jülich
Reihe Energie & Umwelt / Energy & Environment, Band / Volume 573

D 5 (Diss. Bonn, Univ., 2021)

ISSN 1866-1793
ISBN 978-3-95806-622-9

Vollständig frei verfügbar über das Publikationsportal des Forschungszentrums Jülich (JuSER)
unter www.fz-juelich.de/zb/openaccess.

Referent: *Prof. Dr. Nicolas Brüggemann*
Korreferent: *Prof. Dr. Wulf Amelung*

Tag der mündlichen Prüfung: 8. November 2021

Angefertigt mit Genehmigung der Landwirtschaftlichen Fakultät der Universität Bonn

Falling asleep with the whooping of hyenas and the crickets' chirping as a lullaby and starting my days with the songs of countless birds, the smell of roasting coffee and the playful stare of Grivet monkeys right outside the kitchen window … that is the life at the Wondo Genet College campus. Wondo Genet means "paradise" in Amharic and it definitely feels like it. There is so much to learn from this culture-rich land and it is inspiring to see how, despite their struggles throughout history, people in Ethiopia embrace you with their kindness every day. Ethiopia gave me a beautiful gift that I will always treasure.

*To those **Ethiopian women** who taught me that standing **tall** and **strong** has a meaning beyond the one I knew*

ACKNOWLEDGEMENTS

These last four years have taken me through a very dynamic journey full not only of professional growth but also of valuable life experiences. During this ride, I was always accompanied by wonderful people that guided me, taught me, supported me and motivated me.

I wish to express my sincere appreciation to my supervisor, Professor Nicolas Brüggemann, whose expertise and support was always invaluable. I would like to also extend especial thanks to Dr. Katharina Prost for her academic guidance and priceless support. Nicolas and Katharina, the insightful feedback that you have given me elevated my work to a higher level.

My gratitude also goes to our Ethiopian project partners Dr. Menfese Tedese, Dr. Fantaw Yimer, Dr. Dong-Gill Kim, Dr. Mersha Gebrehiwot and Marius Bierig for their guidance, contribution and kindness. My field research and time in Ethiopia relied on the support of many people that through their engagement and love pushed this work forward. For this, I would like to say thank you from the bottom of my heart to Aynalem Korma (Mititi), Doktar Elias, Tsegaye Tabi, Takalech Chufamo, Tizazu Animaw, Edao Osu, Tsion Asfaw and Seble Metaferia. You all treated me as part of your family and that gave me strength.

Many thanks go to my Jülich colleagues and friends, Franz Leistner, Matthias Claß, Rüdiger Reichel and Holger Wissel. My dear office mate and friend, Paulina Deseano Diaz, I want to thank you for all those insightful conversations. Your friendship and advice are precious to me. To my Jülich friends Gabriela Perez Andrade, Lya Paas Oliveros, Amirpasha Mozaffari and Helena Horda thanks for making my time fun and meaningful.

Very importantly, I would like to thank my parents, Olga and Jorge Luis, my sister Cathy, my brother Coqui, and my brother-in-law Jorge for their unconditional love and for always believing in me. You have always been my motivation. In particular, my incredibly special gratitude goes to my mother, Olga, who gave me wings to fly and overcome my difficult moments. To my second and German family, Johannes, Maria and Josef, thanks for giving me so much love and taking this "tica" into your family. Johannes, your love was a light during this journey.

I do not want to end this aknowledgements section without thanking my thesis topic. Delving into this research topic kept me motivated every day of my PhD work and through it, I developed an especial and closer connection with nature. There are three thoughts that will always remain in my mind: 1) **poop** is **not** waste, 2) **soil** is **not** dirt and 3) both are **great**.

ABSTRACT

Lack of basic sanitation and appropriate waste management, limited access to mineral and organic fertilizers, and land degradation are major public health and food security challenges, particularly in developing regions with financial and infrastructural limitations such as in Sub-Saharan Africa. Developing integrated interventions that address these challenges is of great relevance and is becoming more urgent as the effects of climate change increase and as the global population continuously rises. Here, we developed an appropriate-technology ecological sanitation concept via thermophilic composting of human excreta and cattle manure as an approach for climate-smart agriculture. For this, inside traditional wooden compost boxes, we composted human excreta, and separately cattle manure, both with kitchen scraps and teff straw, sawdust and biochar as bulking agents, to produce a compost free of phytotoxicity and pathogens, but rich in nutrients that can be used to improve soil fertility. In order to maximize the benefits of this approach, we particularly aimed at:

(1) Investigating the dynamics of key nutrients and physical and chemical parameters of four composting treatments −human excreta or cattle manure, with and without biochar− to evaluate the feasibility of the appropriate-technology composting process, the type of manure used and the effect of biochar during composting.

(2) Quantifying CO_2, CH_4, N_2O, and NH_3 emissions of the different composting treatments to assess their environmental impact and the effect that biochar has on these gas emissions when used as amendment during composting.

(3) Exploring the nutrient dynamics and greenhouse gas emissions of these four types of compost when applied at two different rates (total compost N equaled 170 kg N ha^{-1}, and three times this amount) to a sandy soil at 25°C to evaluate their potential as fertilizers and the role of biochar in increasing C sequestration and reducing nutrient leaching in agricultural soils.

We found that our appropriate-technology thermophilic composting process enabled a well-running and hygienically safe composting not only of cattle manure, but also of human excreta as a hygienically critical waste, as demonstrated by the low nutrient losses, the temperature course, and the relatively low N_2O and CH_4 emissions. Phosphorus and K delivered by both compost forms showed that the plant demand for P and K based on maize at tropical temperature conditions can be fully covered through human excreta and cattle manure-derived compost application, even at the lower application rate, and could be especially suitable for highly weathered and depleted soils in the tropics with very low P, K and organic matter contents. In contrast, the N provided by all compost treatments was not enough to meet the crop N demand. Our research also demonstrated that compost, especially biochar-compost mixtures, may contribute to carbon sequestration and nutrient retention in agricultural soils and decrease the dependency on synthetic fertilizers, especially on mineral P and K. This work thus demonstrates that the ecological sanitation concept via thermophilic composting with biochar addition is a feasible and climate-smart approach with low requirement for investment and with high potential to increase access to sanitation, soil fertility and food security, and to contribute to climate change mitigation, ecological waste management and sustainable agricultural production.

ZUSAMMENFASSUNG

Fehlende sanitäre Grundversorgung, unangemessenes Abfallmanagement, begrenzter Zugang zu mineralischen und organischen Düngemitteln sowie Bodendegradation sind große Herausforderungen für die öffentliche Gesundheit und die Ernährungssicherheit, insbesondere in Entwicklungsregionen mit finanziellen und infrastrukturellen Einschränkungen wie in Afrika südlich der Sahara. Die Entwicklung integrierter Maßnahmen, die diese Herausforderungen angehen, ist von großer Bedeutung und wird immer dringlicher, da die Auswirkungen des Klimawandels zunehmen und die Weltbevölkerung kontinuierlich wächst. Im Rahmen dieser Arbeit wurde ein technologisch geeignetes, ökologisches Sanitärkonzept auf der Basis thermophiler Kompostierung von menschlichen Ausscheidungen und Rindermist als Ansatz für eine klimafreundliche Landwirtschaft entwickelt. Dazu kompostierten wir in traditionellen Holzkompostboxen menschliche Ausscheidungen und separat Rindermist mit Küchenabfällen und Teffstroh, Sägemehl und Biokohle als Füllstoffe, um einen Kompost zu erzeugen, der frei von Phytotoxizität und Krankheitserregern ist und reich an Nährstoffen, die zur Verbesserung der Bodenfruchtbarkeit verwendet werden können. Um die Vorteile dieses Ansatzes zu maximieren, hatten wir insbesondere folgende Punkte als Ziel:

(1) Untersuchung der Dynamik von Schlüsselnährstoffen und physikalischen und chemischen Parametern von vier Kompostierungsbehandlungen (menschliche Ausscheidungen oder Rindergülle, mit und ohne Biokohle), um die Machbarkeit des technologisch geeigneten Kompostierungsprozesses, die Art der verwendeten Gülle und den Effekt von Biokohle während der Kompostierung zu bewerten.

(2) Quantifizierung der CO_2-, CH_4-, N_2O- und NH_3-Emissionen dieser Kompostierungsprozesse, um ihre Umweltauswirkungen zu beurteilen und den Effekt zu evaluieren, den Biokohle auf diese Gasemissionen hat, wenn sie als Zusatz während der Kompostierung verwendet wird.

(3) Untersuchung der Nährstoffdynamik und der Treibhausgasemissionen dieser vier Komposttypen bei zwei Ausbringungsraten (gesamte Kompost-N-Menge äquivalent zu 170 kg N ha^{-1} sowie die dreifache Menge) in einem Sandboden bei 25°C, um ihr Potenzial als Dünger und die Rolle von Biokohle bei der Erhöhung der C-Sequestrierung und der Reduzierung der Nährstoffauswaschung in landwirtschaftlichen Böden zu bewerten.

Wir fanden heraus, dass der thermophile Kompostierungsprozess mit angepasster Technologie eine gut funktionierende und hygienisch sichere Behandlung nicht nur von Rindermist, sondern auch von menschlichen Ausscheidungen als hygienisch kritischem Abfall ermöglichte, was durch die geringen Nährstoffverluste, den Temperaturverlauf und die relativ geringen N_2O- und CH_4-Emissionen gezeigt wurde. Die P- und K-Nachlieferung bei beiden Aufbringungsraten von Kompost aus menschlichen Ausscheidungen und Rindermist war ausreichend, um den auf Basis von Mais berechneten P- und K-Bedarf der Pflanzen bei tropischen Temperaturen vollständig zu decken. Daher erscheint diese Art der organischen Düngung besonders geeignet für stark verwitterte und ausgelaugte Böden in den Tropen mit sehr geringen P-, K- und organischen Substanzgehalten. Im Gegensatz dazu war die von allen Kompostbehandlungen nachgelieferte N-Menge nicht ausreichend, um den N-Bedarf der Pflanzen zu decken. Die Forschungsergebnisse zeigten auch, dass Kompost, insbesondere Biokohle-Kompost-Mischungen, zur Kohlenstoffbindung und Nährstoffretention in landwirtschaftlichen Böden beitragen und die Abhängigkeit von synthetischen Düngemitteln, insbesondere von mineralischem P und K, verringern können. Die vorgelegte Arbeit zeigt somit, dass dieses ökologische Abfallentsorgungskonzept über thermophile Kompostierung mit Biokohle-Zugabe ein praktikabler und klimafreundlicher Ansatz mit geringem Investitionsbedarf, aber hohem Potenzial ist, um den Zugang zu sanitären Versorgung, die Bodenfruchtbarkeit und die Ernährungssicherheit zu verbessern und zum Klimaschutz, einer ökologischen Abfallwirtschaft und nachhaltiger Landwirtschaft beizutragen.

RESUMEN

La falta de saneamiento básico y de una gestión adecuada de residuos, el acceso limitado a los fertilizantes minerales y orgánicos y la degradación de suelos son grandes retos que amenazan la salud pública y la seguridad alimentaria, especialmente en las regiones en desarrollo con limitaciones financieras y de infraestructura, como en el África subsahariana. El desarrollo de intervenciones integradas que aborden estos retos es de gran relevancia y se hace más urgente a medida que aumentan los efectos del cambio climático y la población mundial. En este estudio desarrollamos un concepto de saneamiento ecológico como enfoque para agricultura climáticamente inteligente utilizando tecnología apropiada mediante el compostaje termofílico de excremento humano y estiércol de ganado. Para ello, dentro de una caja tradicional de madera, compostamos excrementos humanos, y por separado estiércol del ganado, junto con restos de cocina, y como materiales voluminosos utilizamos paja de tef, serrín y biocarbón para producir un compost libre de fitotoxicidad y patógenos, rico en nutrientes que puede utilizarse para mejorar la fertilidad del suelo. Con el fin de maximizar los beneficios de este enfoque, en particular:

(1) Investigamos la dinámica de los nutrientes clave y los parámetros físicos y químicos de cuatro tratamientos de compostaje (excremento humano o estiércol de ganado, con y sin biocarbón) para evaluar la viabilidad del proceso de compostaje con tecnología adecuada, el tipo de estiércol utilizado y el efecto del biocarbón durante el compostaje.

(2) Cuantificamos las emisiones de dióxido de carbono, metano, óxido nitroso y amoníaco de estos procesos de compostaje para evaluar su impacto ambiental y el efecto que tiene el biocarbón en estas emisiones de gases cuando se utiliza como enmienda durante el compostaje.

(3) Exploramos la dinámica de los nutrientes y las emisiones de gases de efecto invernadero de estos cuatro tipos de compost a dos tasas de aplicación (el total de nitrógeno del compost equivalía a 170 kg ha^{-1}, y tres veces esta cantidad) a un suelo arenoso bajo temperaturas tropicales para evaluar su potencial como fertilizantes y el papel que el biocarbón tiene en el aumento del secuestro de carbono y la reducción de la lixiviación de nutrientes en los suelos agrícolas.

Descubrimos que nuestro proceso de compostaje termofílico utilizando tecnología adecuada permitió un tratamiento seguro e higiénico no sólo del estiércol del ganado, sino también de residuos críticos, como los excrementos humanos, según demostrado con las bajas pérdidas de nutrientes, el perfil de temperatura y las emisiones relativamente bajas de óxido nitroso y metano. El fósforo y el potasio aportados por ambas tasas de aplicación de compost indicaron que la demanda de fósforo y potasio basados en maíz bajo temperaturas tropicales puede cubrirse por completo mediante la aplicación de compost derivado de excrementos humanos y de estiércol de ganado, y que podría ser especialmente adecuado para suelos muy erosionados en el trópico con contenidos muy bajos de fósforo, potasio y materia orgánica. En cambio, el nitrógeno aportado por todos los tratamientos de compost no fue suficiente para satisfacer la demanda de nitrógeno del cultivo. Nuestra investigación también demostró que el compost, especialmente las mezclas de biocarbón y compost, pueden contribuir al secuestro de carbono y a la retención de nutrientes en los suelos agrícolas y a la disminución de la dependencia de los fertilizantes sintéticos. Por lo tanto con este estudio demostramos que este concepto de saneamiento ecológico a través del compostaje termofílico con adición de biocarbón es un enfoque factible e inteligente desde el punto de vista climático, con un alto potencial para aumentar el acceso al saneamiento, la fertilidad del suelo y la seguridad alimentaria, y para contribuir a la mitigación del cambio climático en los sectores de saneamiento, gestión de residuos y agricultura.

CONTENTS

LIST OF FIGURES

LIST OF TABLES

LIST OF ABBREVIATIONS

C	Carbon
Ca	Calcium
CEC	Cation exchange capacity
CH_4	Methane
CM	Cattle manure control
CM1	Soil + cattle manure compost (low application rate)
CM2	Soil + cattle manure compost (high application rate)
CM+BC	Cattle manure amended with biochar
CM+BC1	Soil + cattle manure with biochar compost (low application rate)
CM+BC2	Soil + cattle manure with biochar compost (high application rate)
CO_2	Carbon dioxide
EBC	European Biochar Certificate
EC	Electrical conductivity
EcoSan	Ecological sanitation
GHG	Greenhouse gas
HM	Humanure control compost
HM1	Soil + humanure compost (low application rate)
HM2	Soil + humanure compost (high application rate)
HM+BC	Humanure amended with biochar compost
HM1+BC1	Soil + humanure with biochar compost (low application rate)
HM1+BC2	Soil + humanure with biochar compost (high application rate)
K	Potassium
K_{av}	Plant-available potassium
MF	Mineral fertilizer
Mg	Magnesium
MC	Moisture content
N	Nitrogen
NH_3	Ammonia
NH_4^+	Ammonium

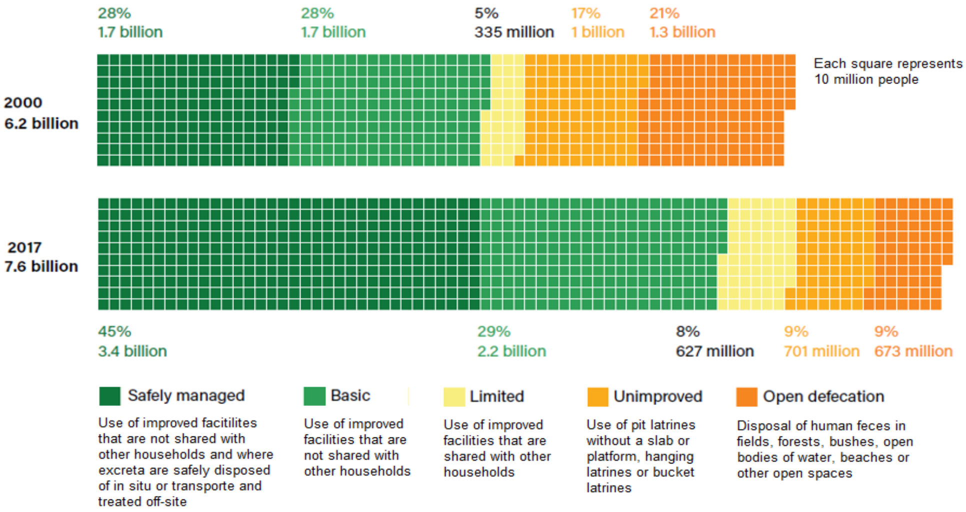
NO_3^-	Nitrate
N_2O	Nitrous oxide
OM	Organic matter
P	Phosphorus
PAH	Polycyclic aromatic hydrocarbons
P_{av}	Plant-available phosphorus
SE	Standard error
SOC	Soil Organic Carbon
SOM	Soil Organic Matter
TCa	Total calcium
TK	Total potassium
TLUD	Top-lit up-draft
TMg	Total magnesium
TN	Total nitrogen
TOC	Total organic carbon
TOM	Total organic matter
TP	Total phosphorus
WGCF-NR	Wondo Genet College for Forestry and Natural Resources
wt	Weight

1

INTRODUCTION

1.1 Motivation and background

The population using safely managed sanitation services increased from 28% in 2000 to 45% in 2017 (Figure 1.1; WHO and UNICEF 2020). Despite this progress, 4.2 billion people worldwide still lack safely managed sanitation, including 2 billion who still do not have access to basic sanitation facilities such as toilets or latrines, and of which, 673 million still defecate in the open, e.g. in street gutters, behind bushes or into open bodies of water (UN 2020b; WHO 2019a). In Sub-Saharan Africa, for instance, 18% of the population use a safely managed sanitation service, with rural areas suffering the most from a poor or inexistent sanitation (Figure 1.2; UN 2020a).



Note: Improved facilites include flush/pour to piped sewer systems, septic tanks or pit latrines, ventilated improved pot latrines, composting toilets or pit latrine with slabs

Figure 1.1. *Global sanitation coverage for years 2000 and 2017 (Modified after WHO and UNICEF 2020).*

This inadequate sanitation and lack of a proper management of this critical human waste reduce availability of a safe water supply, causing 432,000 diarrheal deaths annually due to the transmission of diseases such as cholera, diarrhea, dysentery, hepatitis A, typhoid and polio (WHO 2019a).

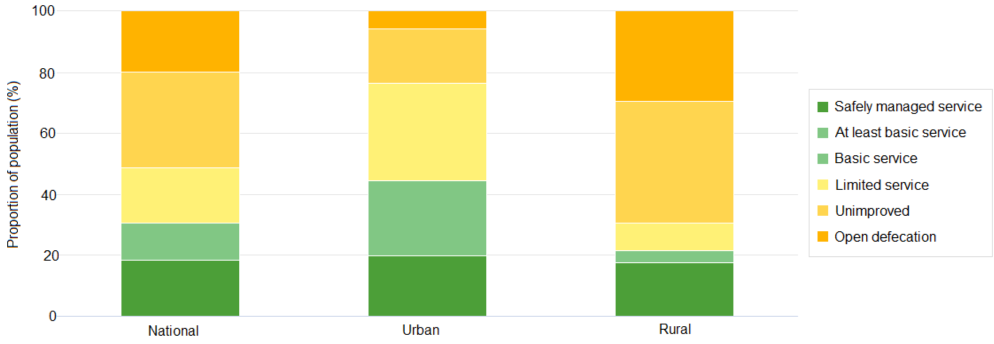


Figure 1.2. *Proportion of the population using sanitation services in Sub-Saharan Africa in 2017 by service level and by location (UN 2020a).*

Besides endangering public health, discharging untreated human excreta in the environment and the use of on-site technologies, such as septic systems and pit latrines, produce avoidable greenhouse gas (GHG) emissions like methane (CH_4) or nitrous oxide (N_2O; Reid et al. 2014; Strokal and Kroeze 2014) and break the human nutrient cycle, producing a permanent drain of nutrients from agricultural and food systems (Figure 1.3; Langergraber and Muellegger 2005; Ryals et al. 2019; Trimmer and Guest 2018). This loss of valuable soil nutrients makes our food system heavily dependent on non-renewable synthetic fertilizers (Akram et al. 2019; Elser and Bennett 2011; Trimmer and Guest 2018).

Since the 1960’s and 1970’s, mineral fertilizers have been used to dramatically increase food production to feed the “Green Revolution” (Maheshwari 2014). This new approach saved millions from starvation; however, this mineral fertilizer production depends on regionally concentrated and non-renewable supplies of phosphate rock and potash ores, and on an energy intensive process (Haber-Bosch) to convert atmospheric nitrogen (N) gas into ammonia-based fertilizer (Alewell et al. 2020; Cordell et al. 2009; Elser and Bennett 2011). Moreover, many agricultural soils in Europe and North America have surpassed critical nutrients levels and have led to water contamination, algal blooms and eutrophication (Cordell et al. 2009; Elser and Bennett 2011). While these nutrients get lost from agricultural soils and from the discharge of untreated human waste (Figure 1.3), regions with the greatest needs for sanitation development such as Sub-Saharan Africa and in particular Ethiopia, have also limited access to fertilizers (Reid 2020) and experience the most severe degradation in the world (FAO 2019; Nkonya et al. 2016). It is estimated that 33% of the Earth's soils are already degraded, with Sub-Saharan Africa having 65% of its agricultural soils degraded since the middle of the twentieth century, as well as 31% of permanent pastures, and 19% of woodlands and forests (FAO 2015, 2019).

Figure 1.3. *Broken human nutrient cycle. This broken cycle produces a permanent drain of nutrients from agriculture that makes our food system heavily depended on non-renewable synthetic fertilizers and in many cases leads to pollution of water bodies (Modified after Jenkins 2005).*

Realizing the human rights to clean water and sanitation, as well as the need to increase food security and sustainable nutrient management practices, recovering nutrients from human excreta could offset significant amounts of synthetic fertilizer use and thereby increase nutrient access and improve water quality in low-income countries like Ethiopia (Akram et al. 2019; Heinonen-Tanski and van Wijk-Sijbesma 2005; Langergraber and Muellegger 2005; Reid 2020; Trimmer and Guest 2018). In Ethiopia, agriculture is the backbone of the population and a key sector of the country's economy (FAO 2016). This sector is the main source of livelihoods for 85% of the population with over 95% of the annual gross total agricultural output of the country generated from smallholder farmers (FAO 2016). Yet, this sector is unable to feed the population due to low crop productivity caused by low nutrient inputs and soil erosion (Figure 1.4; FAO 2016). Moreover, unpredictable weather, droughts and heavy rainfalls that come with climate change and the increasing costs of mineral fertilizers will continuously worsen this situation (Cordell et al. 2009; FAO 2016; Manning 2010). Hence, there is an urgent need in Sub-Saharan Africa, especially in a region like Ethiopia for "climate-smart agriculture" to address food security in a changing climate. Climate-smart agriculture integrates three main pillars for sustainable development: i) sustainable increase of agricultural productivity and farmer's incomes, ii) build adaptation and resilience capacity to climate change, and iii) reduce GHG emissions (FAO 2016, 2021).

Offsetting unsustainable fertilizer use and increasing nutrient access will, however, also require an increased recycling of other nutrient-rich organic wastes such as animal manures. With the continuously increasing world population, it is estimated that by 2030, the global annual production of human feces will exceed 1×10^{12} kg of fresh fecal biomass per year, and this projection is about four times higher for animal feces, with an anticipated amount of 3.6×10^{12} kg (Berendes et al. 2018). Thus, to sustainably meet crop nutrient needs, the agricultural, sanitation and waste management sectors need to be integrated under a circular economy concept that recognizes waste streams as input streams (Simha and Ganesapillai 2017).

Figure 1.4. *Land degradation in Ethiopia (photos by Dong-Gill Kim).*

Ecological sanitation via thermophilic composting offers a hygienically safe solution to convert human fecal material and urine (humanure), and animal manure, together with other organic waste materials such as vegetable scraps, into a stabilized, pathogen-free and nutrient-rich fertilizer and soil amendment (Figure 1.5; more about EcoSan and thermophilic composting in sections 2.1 and 2.3, respectively; Huuhtanen and Laukkanen 2006; Jenkins 2005; Ryals et al. 2019). Thermophilic composting, in contrast to conventional wastewater treatment processes, has low capital and operating costs and does not require electricity and therefore offers the opportunity to establish more appropriate on-site sanitation and waste management services in regions, such those in Ethiopia, that suffer from economic and infrastructural limitations (Maheshwari 2014; Reid 2020).

Even though composting offers feasible ways to deal with organic waste, it presents some constraints, especially for developing countries, such as cost-intensive turning of the heaps (in industrialized countries often done with large-scale and expensive machines) and loss of N due to leaching as well as ammonia (NH_3) and N_2O emissions, and potentially also CH_4 emissions (Bernal et al. 2009; Dalzell et al. 1987; Martins and Dewes 1992; Shuval et al. 1981). These constraints can be reduced with the use of appropriate technology, i.e. technology that is compatible with social and economic local conditions, that utilizes locally available materials, resources and tools, and that is maintained and operated by the local population (Shuval et al. 1981; Sianipar et al. 2013). In the context of composting, this can be achieved by carrying out the process in a traditional wooden compost box and by combining the nutrient-rich materials with suitable substrates such as straw, sawdust, and biochar, that allow a continuous pile aeration and thereby reduce pile-turning workload (Dalzell et al. 1987; Jenkins 2005; Peigné and Girardin 2004). In this regard, due to the biochar's recalcitrance, high surface area, microporosity, and sorption capacity, co-composting organic material with biochar has also shown potential benefits for reducing nutrient and gaseous losses during composting (further discussed in section 2.4.4; Lehmann and Joseph 2009). In addition, in view of the accelerated loss of soil organic carbon (SOC) and nutrients and their severe implications on agricultural productivity and climate change, interest in biochar as a complementary amendment to compost for carbon (C) sequestration and nutrient retention in agricultural soil has increased considerably (Agegnehu et al. 2017; Lehmann and Joseph 2009; Lorenz and Lal 2018; Meena 2020). Local communities could also benefit from the use of appropriate technology to create suitable options for producing biochar such as cook stoves that, besides producing biochar, could reduce indoor air pollution and the amount wood needed for cooking (further discussed in section 2.4.2; Roth 2014; Scholz et al. 2014).

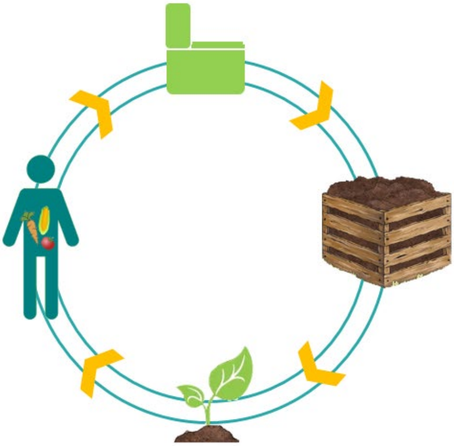


Figure 1.5. *Closing the human nutrient cycle with ecological sanitation via thermophilic composting.*

In order to maximize the benefits of this holistic approach, studies should monitor the composting process of human and animal excreta with biochar to evaluate its performance as management option for these critical forms of waste and its potential to mitigate GHG emissions during composting. Besides investigating the feasibility of methods for composting these excreta, studies should include their use as potential fertilizer and soil amendment in agriculture, especially the potential of compost-biochar mixtures in increasing C sequestration and reducing nutrient leaching in agriculture.

1.2 State of the art

1.2.1 Ecological sanitation

In most developed countries, sanitation is based on the premise that excreta are waste, repulsive and, therefore, need to be disposed of through “flush and forget” technologies (Simha and Ganesapillai 2017). These conventional forms of wastewater management and sanitation systems mix excreta with water –usually drinking water – and transport it to centralized water treatment plants for purification (Huuhtanen and Laukkanen 2006; Simha and Ganesapillai 2017). This disposal method, however, requires large investment costs and impedes the recovery of nutrients from human excreta (Hu et al. 2016; Simha and Ganesapillai 2017).

The concept of ecological sanitation (EcoSan) appeared in the in 1990s as an alternative way to tackle problems associated with the lack of sanitation worldwide and as sustainable approach to sanitation (Hu et al. 2016; Langergraber and Muellegger 2005). In this approach, human excreta are not considered as waste, but as a resource that is treated on site and the end product used as fertilizer in agriculture (Langergraber and Muellegger 2005; Simha and Ganesapillai 2017). Even though EcoSan is particularly suitable for regions with limited water access as well as financial and infrastructural limitations, it offers several appropriate solutions also for other regions depending on their resources availability (Hu et al. 2016).

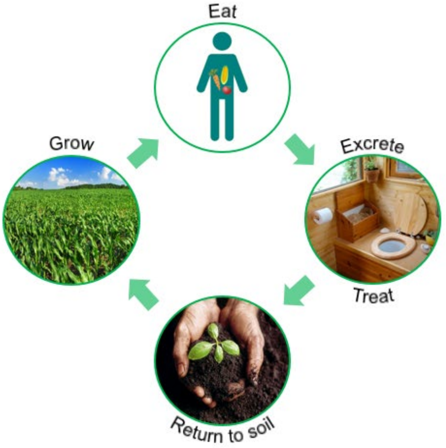


Figure 1.6. *Intact human nutrient cycle. In this cycle, the re-integration of nutrients from treated human feces back to the soil keeps the cycle intact, maintaining soil fertility and providing soils with sustainable nutrient sources (Modified after Jenkins 2005).*

EcoSan systems include different user interfaces (dry toilet, Urine Diversion Dry or Wet toilet, vacuum toilet), collection options (gravity, vacuum, motorized emptying sewage systems), storage schemes (septic tank, cesspit), and treatment technologies of either separated urine and solids or a mixture of both (struvite precipitation, composting, anaerobic digestion, filtration, pyrolysis, constructed wetlands), with scales ranging from simple household installations to complex decentralized systems (Hu et al. 2016; Somorin 2020; Werner et al. 2009). Regardless of the model, all EcoSan systems aim to close the "poop" loop, as it involves the re-integration of nutrients from treated human feces back to the soil (Figure 1.6; Huuhtanen and Laukkanen 2006; Kramer 2011; Langergraber and Muellegger 2005; Simha and Ganesapillai 2017). In particular, EcoSan offers potential solutions to: (1) providing safely managed sanitation, (2) reducing the health risks related to sanitation, waste and water pollution, (3) preventing the pollution of water bodies, (4) saving water, energy and resources,

(5) optimizing the management of nutrients, (6) preventing the degradation of soil fertility, by returning nutrients and organic matter (OM) to degraded agricultural soils.

1.2.1.1 Dry toilets

Here, in this research work, we focus on an EcoSan implementation that combines container-based toilets, also called dry toilets, for collecting human excreta (feces and urine) with an aerobic treatment via thermophilic composting to sanitize these excreta and produce a stabilized organic fertilizer and soil amendment.

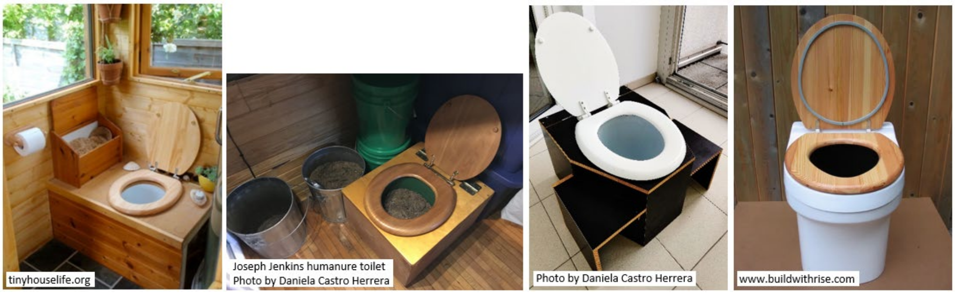


Figure 1.7. *Examples of container-based dry toilets.*

The dry toilet itself consists generally of a wooden or plastic box with a 20-L plastic receptacle below the toilet seat, which can be opened for the removal of the full container (Figure 1.7). For the operation of the toilet, it is necessary to place first a few centimeters of sawdust or other clean, fine and slightly moist C-based organic material into the bottom of the bucket such as leaves, rice hulls, sugar cane bagasse, or other similar materials (Jenkins, 2005). The toilet is operated like any other toilet, but instead of flushing with water, the contents (liquid or solid) will be covered with dry organic material, such as sawdust, after every use. Proper cover materials are absolutely essential for the successful operation of a dry toilet, as they prevent odor and flies, absorb moisture, and prepare the material for composting (Jenkins 2005). When the toilet receptacle is about 3/4 full, it is removed and closed, and a new receptacle is placed. The fecal material and the urine are collected together with adsorber materials, e.g., sawdust, and toilet paper in the buckets and can be stored until their use for composting. As these toilets do not separate feces and urine, this toilet design is an appropriate option when the nutrients from the urine are desired in the compost, provided that transportation to an offsite composting facility is not necessary or not far (Kramer 2011).

1.2.2 Fertilizer value of human excreta

Like animal manures, human urine and feces are also rich in plant macronutrients [N, phosphorus (P), potassium (K), calcium (Ca) and magnesium (Mg)] and micronutrients (zinc, copper, iron, manganese, boron and molybdenum) and therefore, can be also used as fertilizers of high quality (Table 1.1; Heinonen-Tanski and van Wijk-Sijbesma 2005; Jönsson et al. 2004). The amount and chemical composition of nutrients excreted by humans depend on age, gender, diet, calorie intake, geographical location, income levels and socio-cultural factors (Rose et al. 2015). The range of urine generation is 0.6−2.6 L cap^{-1} day^{-1} and fecal wet mass production 51−796 g cap^{-1} day^{-1} (Rose et al. 2015; Simha and Ganesapillai 2017), yielding per year an average total nutrient amount from excreta (urine + feces) of 5.7 kg N, 0.6 kg P and 1.2 kg K (Simha and Ganesapillai 2017). Of these nutrients, 90% of the total N, 50−65% of total P and 50−80% of total K are excreted in the urine (Simha and Ganesapillai 2017). Once adult, humans do not incorporate nutrients into new body tissue and, therefore, the amount of nutrients consumed and excreted is roughly equal (Jönsson et al. 2004). It is estimated that, if all the excreta of the whole world population were collected, one third of the world's mineral N use and 22% of the world's use of mined P could be replaced by N and P from excreta (WHO 2006).

Table 1.1. *Nutrient contents of human feces and different animal manures (Adekiya et al. 2020; Jenkins 2005; Moreno-Caselles et al. 2002; Turner and Leytem 2004).*

Manure	N (%)	P (%)	K (%)
Human	5.0−7.0	3.0−5.4	1.0−2.5
Cattle	0.9−2.4	0.1−0.8	1.5−2.1
Poultry	2.9−6.3	0.8−5.9	3.0−3.8
Pig	1.9−3.8	0.8−3.3	1.1−2.2
Horse	2.1−2.3	1.3−1.7	1.4−5.0
Sheep	1.9−3.8	1.3−1.9	1.3−4.1

In terms of chemical composition, human urine is composed of 91–96% water and the remainder of inorganic salts, urea, organic compounds, and organic ammonium salts, which are mainly plant available and composed of 14–19% N, 2.5–5% P, 3–4.5% K, 11–17% C, and 4.5–6% Ca (Jenkins 2005; Rose et al. 2015). Urea accounts for 75−90% of the total N in urine, and the remainder is mainly ammonium (NH_4^+), creatinine and very low concentrations of nitrate (NO_3^-; Jönsson et al. 2004; Rose et al. 2015). When applied to soil, urea is degraded to ammonium by urease, which is directly plant-available and an excellent N fertilizer, that can be transformed in soil to NO_3^-, which is also an important N source for many crops (Jönsson et al. 2004). The P and K in urine are mainly inorganic, plant available, and excreted in the form of phosphate and K ions, respectively (Simha and Ganesapillai 2017).

Feces are composed on average of 75% water, and 25% of a solid fraction composed of 5–7% N, 3–5.4% P, 1–2.5% K, 40–55% C, and 4–5% Ca contained in proteins, undigested fats, polysaccharides, bacterial biomass, ash, and undigested food residues (Table 1.1; Jenkins 2005; Rose et al. 2015). In contrast to urine, which has mainly water-soluble nutrients, feces contain both water-soluble nutrients, but also nutrients that are entrapped in larger particles of undigested matter that needs to be degraded to become available to plants (Jönsson et al. 2004). The solid part of the inorganic fraction is predominantly made up of calcium phosphate and iron phosphate particles that, depending on the soil conditions, can dissolve and become available for plants (Rose et al. 2015).

The main organisms posing threat to human health are pathogenic bacteria, viruses, parasitic protozoa and helminths, such as *Salmonella typhi*, *Salmonella paratyphi*, *Leptospira*, *Yersinia*, *Escherichia coli*, *Hepatovirus A*, *Ascaris* and others that can be found in large numbers in excreta of infected individuals (Huuhtanen and Laukkanen 2006; Jenkins 2005). In contrast to feces, urine presents less danger to human health as it contains few enteric microorganisms; however, some human pathogenic microorganisms as well as helminth eggs can be found also in urine (Rose et al. 2015). It is therefore crucial that adequate handling methods are applied to human excreta (collection, storing, and treatment procedure) to ensure safe and adequate sanitation and provide significant health benefits.

1.2.3 Thermophilic composting

Before use in cultivation, excreta should be handled and treated according to hygiene guidelines to ensure the inactivation of pathogens present in this critical material. Thermophilic composting provides the simplest and often most practical method for inactivating pathogens at temperatures of >55°C, >60°C or >65°C for 2 weeks, 6 days or 3 days, respectively (Berendes et al. 2015; Bundesministerium der Justiz und für Verbraucherschutz 2017; Preneta et al. 2013). Proper composting not only ensures the destruction of potential human or animal pathogens, but also converts human excreta or any other organic waste into a stabilized final product, free of phytotoxicity and with humic properties that can be used to improve soil fertility (Bernal et al. 2009; Jenkins 2005; Maheshwari 2014; Rynk et al. 1992). Any other kind of animal manure can also benefit from composting, as properly composted manures do not leach, like raw manures do and therefore, hold nutrients in soil, allowing for better nutrient management on farms and reducing nitrate contamination of water bodies (Jenkins 2005; Rynk et al. 1992).

Thermophilic composting is a decomposition process of organic materials by microorganisms in a predominantly aerobic environment that involves four different phases: mesophilic I (ambient temperature−45°C), thermophilic (>45°C), mesophilic II (45°C−ambient temperature), and a final maturation phase at ambient temperature (Figure 1.8; Bernal et al. 2009; Maheshwari 2014). These different phases can be easily identified by the temperature pattern over time, which reflects changes in the rate and type of decomposition processes taking place as composting proceeds (Rynk et al. 1992).

As soon as the feedstock materials are mixed together, the decomposition of OM by microorganisms starts immediately. During the first 1–3 days of this process, simple organic C compounds, such as sugars, amino acids, proteins, etc., are easily mineralized and metabolized by mesophilic bacteria and fungi, consuming O_2 and producing considerable heat and large quantities of CO_2, water vapor and other gases (Figure 1.8; Bernal et al. 2009; Maheshwari 2014; Rynk et al. 1992).

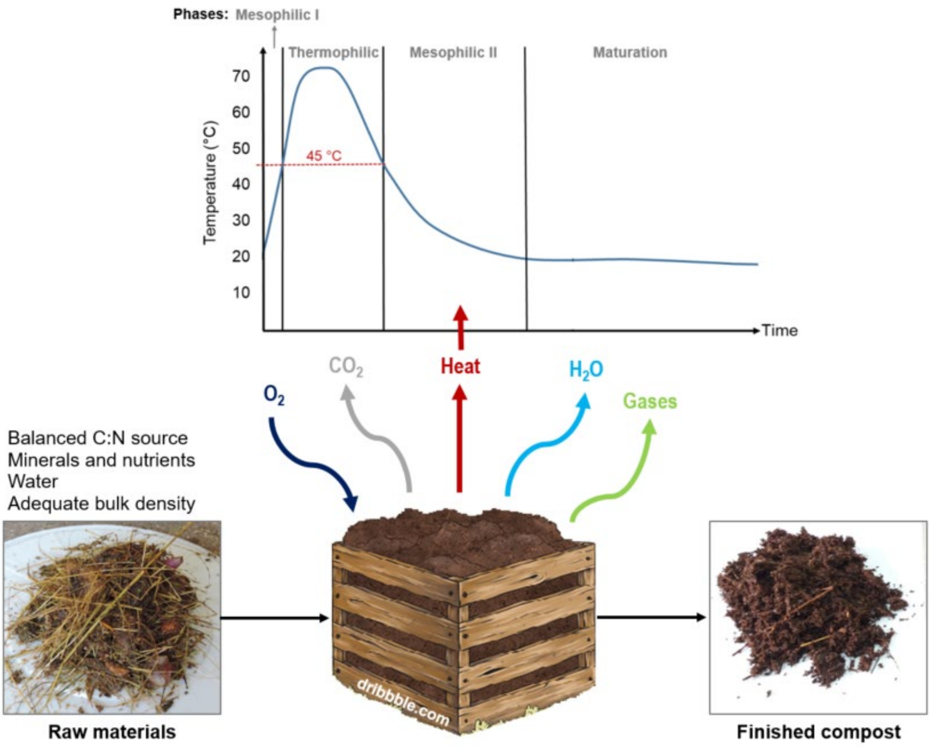


Figure 1.8. *Thermophilic composting process with a typical temperature curve of a compositing process without pile turning.*

The accumulation of this heat quickly raises the temperature of the pile up to 60−70°C, and thermophilic temperatures, i.e. above 45°C, are maintained for several days or weeks (Rynk et al. 1992). At these high temperatures, mesophilic bacteria become inhibited and are replaced by thermophilic microorganisms, and the high decomposition activity continues. This phase is also characterized by the destruction of human, animal and plant pathogens; however, it is not only the heat that allows hygienization, but also the competition for resources among the different communities of microorganisms present in the compost (Jenkins 2005). Once the easily degradable organic compounds are depleted, the microbial activity decreases, and as consequence, the temperature also gradually drops down (Maheshwari 2014). During the mesophilic phase II, or also called cooling phase, the composting pile is again colonized by mesophilic microorganisms that are able to degrade more complex organic materials such as the polysaccharides cellulose and hemicellulose, which are main components of plant cell walls (Bernal et al. 2009; Dalzell et al. 1987). During the cooling phase, lignin, also a main component of lignocellulosic materials, but which is resistant to bacterial degradation, is transformed by fungal species that produce lignolytic enzymes able to breakdown these more resistant C moieties (Maheshwari 2014). The degradation products of these lignocellulosic materials are also a main source of precursors of what later becomes chemically polymerized into humus, which can hold moisture, nutrients and minerals, and thus enhance soil quality (Maheshwari 2014). These processes occurr during the final stage of the composting process, called maturation, aging or curing phase, in which stabilization and humification of the OM occur, producing a mature and stable compost with humic characteristics (Bernal et al. 2009; Dalzell et al. 1987).

1.2.3.1 Compost parameters

For an efficient composting process to take place and to obtain an agricultural quality product, adequate initial substrate conditions need to be defined, controlled and maintained throughout the composting process (Bernal et al. 2009). Thus, the control of parameters, such as bulk density, C:N ratio, temperature, pH, moisture and oxygen supply, is crucial as they determine the optimal conditions for microbial development and OM degradation (Bernal et al. 2009; Dalzell et al. 1987; Rynk et al. 1992).

C:N ratio: Microorganisms require degradable organic C compounds as energy source and N for their development and activity, and therefore, a balanced supply of C and N is needed (Bernal et al. 2009). This can be obtained by defining a mixture of organic materials that yield an adequate initial C:N ratio. A proper initial C:N ratio for composting is in the range of 20:1–40:1 (Rynk et al. 1992). High C:N ratios make the process very slow, while low values lead to increased N losses via NH_3 volatilization or leaching from the composting pile (Bernal et al. 2009; Rynk et al. 1992). Since human excreta, animal manures and vegetable scraps have low C:N ratios, adding a bulking agent rich in C with little if any bound N, such as sawdust, straw, green cuttings, leaves, grass clippings or other C-rich substrate can increase and optimize this ratio (Jenkins 2005; Maheshwari 2014). Although the C:N ratio is a useful guide, the rate at which the C compounds of different lignocellulosic materials decompose must also be considered (Rynk et al. 1992). For instance, straw decomposes more easily than woody materials, such as sawdust (Rynk et al. 1992). During the course of composting, the decomposition of OM decreases the organic C content, which then reduces the weight of the pile and decreases the C:N ratio (Bernal et al. 2009). These C losses, however, are later on compensated by the stabilization processes occurring during the maturation phase (Bernal et al. 2009).

Bulk density and oxygen: The use of bulking agents not only balances the C:N ratio, but also adds structure to the composting mixture, decreasing the bulk density of the pile and thereby allowing adequate air distribution throughout the mixture (Bernal et al. 2009). As composting is an aerobic treatment, microorganisms rely on oxygen for the decomposition and transformation of OM (Rynk et al. 1992). The need for oxygen is greatest during the initial stages of composting, as readily degradable components of the raw materials are metabolized (Rynk et al. 1992). Proper aeration is therefore key for achieving hygienization temperatures and removing excess of water (Bernal et al. 2009; Rynk et al. 1992). If the O_2 supply is limited, the aerobic decomposition slows down and anaerobic conditions develop. Anaerobic decomposition involves different microorganisms and biochemical reactions that can lead to the undesirable production of gases such as CH_4 and N_2O (further discussed in chapter 3; Maheshwari 2014). Aeration can be provided by pile turning (manual or mechanical), forced aeration (fans or air blowers), and by natural air diffusion (e.g. with proper bulky materials; Rynk et al. 1992).

Temperature: Since the release of heat is directly related to the microbial activity, temperature is a good composting process indicator (Rynk et al. 1992). Heating is essential to kill pathogens and weed seeds and to enable the development of thermophilic microorganisms capable of degrading different compounds (Maheshwari 2014). Thus, it is crucial to provide proper composting conditions that stimulate microbial activity. The optimum temperature range for composting is 40–65°C, and above 55°C to kill pathogenic microorganisms (Bernal et al. 2009).

Moisture: Water provides a medium for chemical reactions, transport of nutrients and for mobility of microorganisms. A moisture content between 40−65% is generally recommended to support metabolic processes of the microbes; however, when highly porous substrates, such as straw, biochar and sawdust, are used, values above this range can also yield optimum results (Rynk et al. 1992). Low water contents can slow down microbial activity, while at high values, water can displace the air in the pore spaces of the composting mixtures, limiting air distribution throughout the pile and leading to anaerobic conditions (Dalzell et al. 1987; Maheshwari 2014). An excessive amount of water can also lead to NH_4^+ and NO_3^- losses via leachate (Bernal et al. 2009). During the course of composting, moisture levels decline as water evaporates due to the high temperatures during the thermophilic phase.

pH: Due to the broad spectrum of microorganisms and processes involved during composting, pH changes as composting proceeds; however, a pH of 5.5–9.0 is considered to support a good microbial activity (Rynk et al. 1992). Generally, pH declines during the initial stages of composting as the degradation of OM releases organic acids (Dalzell et al. 1987; Rynk et al. 1992). However, the pH rises quickly, as these carboxylic acids are rapidly broken down and as NH_4^+ or NO_3^- concentration increases (Maheshwari 2014).

1.2.3.2 Nitrogen losses during composting

Nitrogen losses during composting can occur by NH_3 volatilization, NH_4^+ and NO_3^- leaching, and by N_2O emissions through nitrification and denitrification (Bernal et al. 2009; Maheshwari 2014; Rynk et al. 1992). Besides decreasing compost quality, N losses generate environmental problems such as NO_3^- contamination of groundwater and climate change (Bernal et al. 2009).

Reducing NH_3 volatilization is very relevant for controlling N losses, as most N losses (up to 60% of initial N) during composting of animal manures have been found to be via NH_3 emissions (Bernal et al. 2009). The high initial NH_4^+ concentration and the presence of easily mineralizable compounds that form more NH_4^+ are main factors for NH_3 volatilization. The deprotonation of NH_4^+ leads to NH_3

formation, which increases the pH to values >7.5 and, together with the high temperature, favors the transfer of dissolved NH_3 to NH_3 gas (NH_3 emissions during composting are further discussed in chapter 3; Bernal et al. 2009).

To reduce N losses during the composting of organic waste rich in inorganic N, various solutions have been explored, e.g., the addition of biodegradable organic C sources to stimulate the partial incorporation of N into the organic matrix or its immobilization by the microbial biomass (Bernal et al. 2009; Cao et al. 2019). The use of porous additives, e.g., biochar, that absorb NH_3, NH_4^+ and NO_3^-, and that provide better aeration conditions, avoiding the development of anaerobic microsites that favor denitrification, has also been investigated (more about the use of biochar as compost amendment in 2.4.4; Cao et al. 2019).

1.2.3.3 Maturity and stability indicators

Compost quality is not only related to its fertilizer value, but also to its stability and maturity, which determine if the compost is ready and safe to be used in soil (Bernal et al. 2009; Maheshwari 2014). If unstable or immature compost is applied, it can induce anaerobic conditions as the soil microorganisms utilize oxygen to continue to break down the material, and can have a negative effect on plant growth due to the presence of phytotoxic substances such as an excess of NH_4^+(Bernal et al. 2009; Maheshwari 2014). Several physical, chemical and biological parameters have been proposed for evaluating compost maturity and stability, including color, odor, temperature, pH, electrical conductivity, cation exchange capacity, C:N ratio, NH_4^+-N, NH_4^+-N:NO_3^--N ratio, CO_2 production rate, seed germination index, pathogenic activity, among others (Table 1. 2; Bernal et al. 2009; Maheshwari 2014).

Table 1. 2. *Indicator parameters for compost stability and maturity (Bernal et al. 2009; Maheshwari 2014).*

Parameter	Value
Color	Dark brown or dark color
Odor	Like soil
Temperature	Ambient temperature and constant after turning
pH	Stable and around neutral
Electrical conductivity	Stable at the end (Maheshwari 2014) 2.0–3.5 mS cm^{-1} (Zaha et al. 2013)
Cation exchange capacity	> 0.6 meq g^{-1}
C:N ratio	<20
NH_4^+-N content	< 0.04 %
NH_4^+-N:NO_3^--N ratio	< 0.16
CO_2 production rate	< 120 mg CO_2 kg^{-1} h^{-1}
Seed germination index	> 80 %
Microorganism counts (total coliform, fecal coliforms and fecal enterococci)	< 500 MPN g^{-1}

1.2.3.4 Compost application in soil

Organic matter in soils represents residual compounds remaining after the decomposition of plant, fauna, and microbial inputs, with SOC being its main component and one of the key soil properties associated with many soil functions (FAO 2017; Paul 2015). Soil organic matter contributes to soil chemical fertility, as it constitutes a source of nutrients released with mineralization (N, P, and K) and as it retains nutrient cations (e.g., NH_4^+, K^+, Mg^{2+}, Ca^{2+}) with its usually negative surface charges (Paul 2015). It is also the basis of soil biological activity, being the source of C and energy for many microorganisms (Paul 2015). It also plays a major role in soil physical fertility, particularly by increasing water and nutrient holding capacity and soil structure stability, thus, promoting aggregate formation, which together with porosity, ensure sufficient aeration and water infiltration to support plant growth, and reduce soil erosion (FAO 2017; Paul 2015).

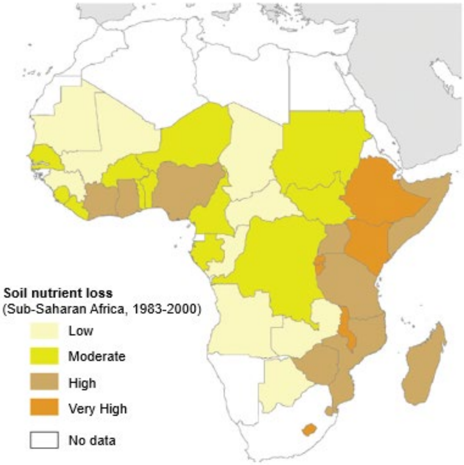


Figure 1.9. *Map of estimated nutrient loss from soil for Sub-Saharan Africa 1983−2000. Densely populated and hilly countries in the Rift Valley area show the highest losses owing to high levels of arable land, relatively high crop yields and significant erosion levels (European Union 2013).*

Soils are a major C reservoir containing more C than the atmosphere and terrestrial vegetation combined, with global SOC stocks estimated at an average of 1,500 Pg C in the first meter of soil, which is about 1.8 times more C than in the atmosphere and 2.3−3.3 times more than what held in the terrestrial vegetation of the world (FAO 2017). As the formation and decay of OM is an essential ecosystem process that contributes to the regulation of atmospheric trace gases, particularly CO_2, N_2O, and CH_4 (Paul 2015), the anthropogenic impacts on soil can turn it into either a net sink or a net source of GHGs (FAO 2017). In the presence of climate change, land degradation and unsustainable soil management, soils have lost substantial amounts of nutrients and SOC, and have also become a source of GHG emissions to the atmosphere (FAO 2017; Nkonya et al. 2016). According to the European Union (2013), nearly 500 million hectares in Sub-Saharan Africa, representing more than 16% of the total land area, were affected by some kind of degradation process, including the loss of nutrients and/or OM due to the practice of agriculture on poor or moderately fertile soils, without sufficient application of manure or fertilizer (Figure 1.9).

However, if managed wisely, soils have the potential to sequester large amounts of C, thus contributing to climate change mitigation and adaptation (Amelung et al. 2020). In addition, as SOC is an indicator of soil health, increasing soil C inputs offer opportunities to improve soil productivity and thereby ensure food security (FAO 2017). Given the role of soils to ensure crop productivity and to increase climate change mitigation and adaptation, the addition of organic amendments such as compost and biochar offer a sustainable soil management strategy that addresses the above challenges (Amelung et al. 2020; FAO 2017; Fisher and Glaser 2012; Lorenz and Lal 2018). Compost is an excellent soil conditioner that, besides being a source of macro- and micronutrients, provides a relatively stable form of OM to soil (Bernal et al. 2009). Thus, when applied to croplands, compost improves soil structure, aeration, drainage, water-holding capacity, nutrient sorption capacity, pH buffering capacity, and biological activity (Figure 1.10; Dalzell et al. 1987; Maheshwari 2014; Rynk et al. 1992).

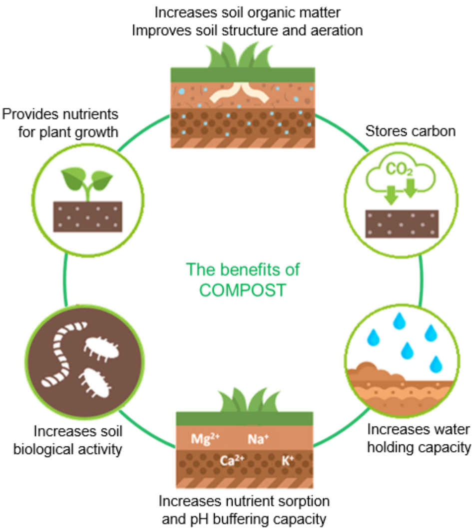


Figure 1.10. *Benefits of compost application in soils (modified after European Compost Network 2021).*

In contrast to chemical fertilizers, which supply nutrients in soluble form to plants, the nutrients in compost are mostly in a complex organic form and must be mineralized in the soil before they become available to plants and therefore, the level of plant-available nutrients in compost is lower than in mineral fertilizers (further discussed in chapter 4; Amlinger et al. 2003). For example, less than 15% of the total N in compost is typically available in the first cropping season, and only in the following years, the N from previous applications will gradually become available. Therefore, it is necessary to consider the benefits of compost as long-term fertilizer, requiring repeated applications over many years (Amlinger et al. 2003; Meena 2020). This can lead to a disadvantageous perception of compost compared to mineral fertilizers, with their higher concentrations of immediately plant-available nutrients (Meena 2020). However, it can be argued that too strict restrictions regarding the admissible nutrient loads limit compost applications in agricultural soils, and thereby not only deny the fertilizer potential of composting, but also the envisaged long-term soil improvement that is effected by regular compost application (further discussed in chapter 4).

A further problem is the unequal availability of the macronutrients. If the compost application rate covering the nutrient demand of the entire growing season was solely based on the amount of available N in compost, this could result in too high P or K inputs, especially in nutrient-rich soils (Maheshwari 2014). However, for highly nutrient-depleted soils with low OM content, such as sandy or highly weathered soils, large amounts of compost could help alleviate the low fertility of these soils (Kowaljow et al. 2017; Lorenz and Lal 2018; Steiner et al. 2007; Tsai and Chang 2020). An example of highly nutrient-depleted soils are Ferralsols, which occur mainly in tropical and subtropical regions and are widespread in Central, Eastern and Southern Africa and in Latin America (Figure 1.11; European Union 2013, 2015). These soils are common in areas of heavy rainfall and high temperature, and are deeply weathered soils that have lost nearly all of their weatherable minerals over time and have very low nutrient-holding capacity (European Union 2013). Therefore, more long-terms studies are required to develop compost application schemes suitable for different types of soil properties and climatic conditions (Amlinger et al. 2003; Maheshwari 2014).

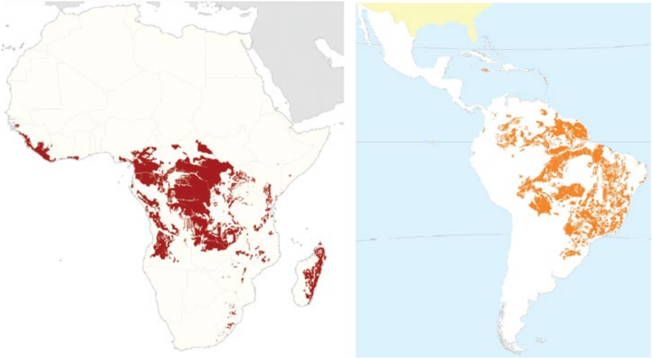


Figure 1.11. *The map shows where Ferralsols predominate. These soils cover about 10% of Africa and 17% of Latin America (European Union 2013, 2015).*

1.2.3.5 Composting human excreta and using human excreta-derived fertilizers

Composting of human excreta together with vegetable waste and animal manure has been a common practice in countries like China, Vietnam and Japan (also known there as night soil composting) for centuries and has been considered a vital aspect of maintaining the soil fertility of those countries

(Heinonen-Tanski and van Wijk-Sijbesma 2005; Shuval et al. 1981). The value of composting human excreta was also recognized in Europe; however, after the nineteenth century, European and North American farmers replaced the usage of excreta by chemically synthesized fertilizers (Maheshwari 2014; Shuval et al. 1981). Since the concept of EcoSan appeared in the in 1990s, interest has increased again, and several pilot projects using different EcoSan systems in diverse geographical settings have been implemented (Simha and Ganesapillai 2017). One example that shows increased engagement in sustainable sanitation is the SuSan network that came into existence in 2007, and which aim is to connect the work of people and organizations concerning the development of sustainable sanitation strategies and initiatives. Case studies presented by this network or from other sources that apply EcoSan via composting can be found across industrialized countries like Germany (Rauschning et al. 2009), Sweden (Coalition Clean Baltic 2009), the Netherlands (Bijleveld 2003), Denmark (Magid et al. 2006) and United States (Ersson and King 2019), emerging markets like India (Werner et al. 2009), and China (Zhou et al. 2010), and developing or least-developed countries such as Philippines (Bracken et al. 2009), Indonesia (Malisie et al. 2007), Vietnam (Jensen et al. 2008) and in many countries in Sub-Saharan Africa (Morgan 2007; von Münch and Ingle 2012).

One particular successful example is the container-based sanitation organization Sustainable Organic Integrated Livelihoods (SOIL) in Haiti, which successfully have adapted composting technologies to produce human excreta derived fertilizer and sell their product in the local market (Moya et al. 2019c). By mixing dry sugarcane bagasse with human feces to obtain an optimal C:N ratio and composting these materials in a 18 m^3 wooden compost bin (height 1−1.5 m), SOIL's system is able to reach temperatures above the 50°C threshold and remain there for more than two weeks, indicating efficient pathogen removal (Preneta et al. 2013). An additional top 5−10-cm layer of sugarcane husks is placed on top of the pile to maintain heat and protect it from wind. The composting process is static, i.e. not mixed, for six months, and during the first 2−3 months, approximately 90 L of urine collected from the urine-diverted toilets is added to the process over multiple watering sessions to increase moisture. In their study, Preneta et al. (2013) also discuss that despite measuring lower temperatures in the corners of their compost boxes, analysis showed that within eight weeks resistant pathogens like *Ascaris* are eliminated and the more general fecal contamination indicator *E. coli* is reduced to an acceptable level. This treatment process has also been described and investigated by Berendes et al. (2015) and Piceno et al. (2017), and their findings also indicated that SOIL's EcoSan composting process was effective in inactivating *Prevotella* and *Erysipelotrichaceae* (100% reduction of initial presence), *Ruminococcaceae* (98−99%), *Lachnospiraceae* (83−94%), *Escherichia* and *Shigella* (100%), and *Ascaris* spp. (100%). McNicol et al. (2020) also investigated the composting facility operated by SOIL by performing gas flux measurements to determine cumulative emissions of CO_2, CH_4 and N_2O over the composting life cycle and to estimate scalable emission factors. They found increased N_2O emissions, but these were outweighed by the one to two orders of magnitude smaller CH_4 emission factors than IPCC values for other excreta collection, treatment and disposal processes such as environmental discharge, dry and wet pit latrines, and septic systems. After accounting for GHG emissions throughout the sanitation cycle, including transport, urine and compost end use, at local scale, the climate change mitigation potential was 126 kg of CO_2-equivalent per capita per year for slum inhabitants. At the global scale for slum populations, they estimated that the implementation of human waste composting could mitigate 13−44% of CH_4 emissions from the sanitation sector.

Even though the feasibility of composting human excreta to obtain a safe product has been demonstrated, many organizations, projects and farmers have been affected by unclear regulations on the use of fertilizers derived from human excreta (Moya et al. 2019c) and by barriers due to the negative perceptions towards it (Gwara et al. 2021). For example, in the European Union, the application of sewage sludge on agricultural land is controlled in various ways in the different member states, with some countries imposing more stringent limits (Collivignarelli et al. 2019). In this regard, in their review paper, Gwara et al. (2021) found that horticultural exporters do not currently approve crops grown using human excreta-derived fertilizers for exporting to the European market. As a result of these perception issues, several countries such as United Kingdom, Sweden, United States, Australia and New Zealand have developed assurance schemes specific to biosolids to increase customers' confidence in their use in agriculture (Moya et al. 2019a). The main problems related with the reuse of biosolids concern the presence of heavy metals in the sludge (Collivignarelli et al. 2019). As human excreta collected from dry toilets has not been mixed with household and industrial wastewater streams, as it happens in the case of human excreta in sewage sludge, the amount for example of heavy metals is expected to be substantially lower (Jönsson et al. 2004; Rose et al. 2015). Thus, the use of human excreta from EcoSan models such as container-based systems should be evaluated separately; yet, schemes specific for the use of fertilizers derived directly from fresh human excreta from dry toilets are missing (Moya et al. 2019a). In Germany, for instance, the fertilizer regulation (Düngemittelverordnung; DüMV) lays down the requirements for placing fertilizers on the market, however, fresh separated human excreta are not listed as allowed main ingredient for producing fertilizers (Bundesministerium der Justiz und für Verbraucherschutz and Bundesamts für Justiz 2012). Governmental support could help to enhance research, knowledge, awareness, and social acceptance, and thereby mitigate perceived barriers on the use of nutrients from human excreta (Gwara et al. 2021). Therefore, there is a need for clear policies and the development of specific standards on this matter to ensure the safety and quality assurance for this type of fertilizers and soil amendments.

1.2.4 Biochar

1.2.4.1 What is biochar?

Biochar is a heterogeneous substance rich in aromatic C and minerals which is produced by pyrolysis, a process in which biomass, such as wood, leaves, or any other C-rich material, are broken down at temperatures ranging from 350−1000°C in a low-oxygen thermal process (EBC 2012). Charcoal is also produced by this thermochemical process, however, the difference between biochar and charcoal lies in that biochar is produced to be used as soil amendment for agronomic or/and environmental management (Lehmann and Joseph 2009).

During pyrolysis, mass is lost mostly in the form of volatile organics, producing a significant volume reduction. After this shrinkage, the remaining carbon skeleton consists of highly conjugated aromatic compounds characterized by rings of six C atoms linked together, aromatic-aliphatic organic compounds of complex structure (including residual volatiles), and a series of voids, formed as pores of various sizes and cracks, that may contain mineral compounds (inorganic ash; Figure 1.12; Lehmann and Joseph 2009).

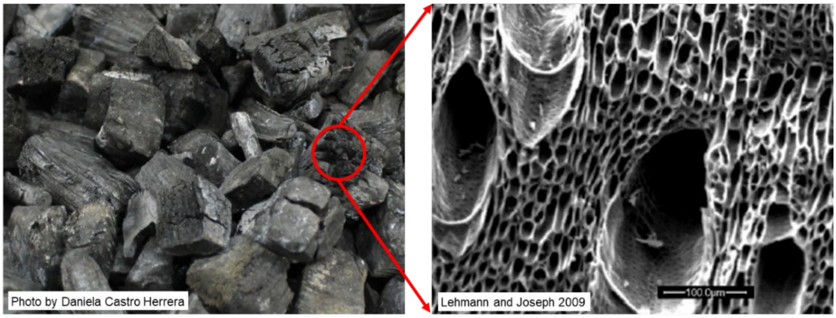


Figure 1.12. Wood-derived biochar (left). Scanning electron microscopy image showing the microporous structure of a wood-derived biochar produced by pyrolysis (right).

This complex C matrix has irregular arrangements that also contains heteroatoms such as H, O, minerals, and trace amounts of N and S, which are predominantly located on the edges of the aromatic compounds as components with various functional groups (Lehmann and Joseph 2009). The elemental composition of biochar varies according to the raw biomass material from which the biochar was produced and the characteristics of the carbonization process (Cha et al. 2016). This final structure is what gives biochar its unique characteristics, like high surface area, porosity, and surface charges that offer a range of potential agricultural applications (Hossain et al. 2020). Due to its polycyclic aromatic structure, biochar is chemically and microbially stable and can persist in the environment over decades or centuries (Glaser et al. 2001). Additionally, the high number of micropores contributes the most to the biochar's large surface area and is responsible for the high adsorptive capacities of different molecules such as gases, nutrients and solvents (discussed in more detail in sections 2.4.4 and chapter 2, 3, and 4; Lehmann and Joseph 2009).

1.2.4.2 Biochar production with pyrolytic cook stoves

There are several carbonization processes to produce biochar that range from traditional kilns such as in pits, mounds, and brick kilns, to more advanced and efficient technologies including drum pyrolyzers, rotary kilns, pyrolysis reactors, gasifiers, and wood-gas stoves, all of which produce varying quantities of gas and liquids along with biochar (Cha et al. 2016; Lehmann and Joseph 2009). This section will focus on the use of small pyrolytic top-lit updraft (TLUD) micro-gasifiers (Figure 1.13), here also called biochar cook stoves, for the production of biochar. These stoves, besides producing biochar, generate also combustible gases that create cooking heat (Roth 2014).

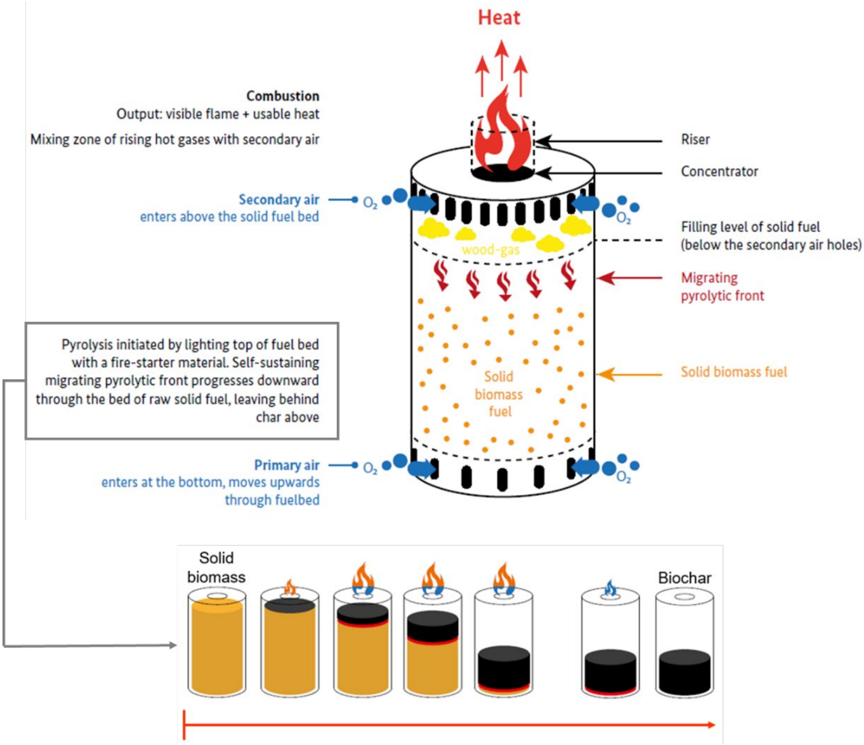


Figure 1.13. Basic design features of a char-making TLUD micro-gasifier (modified after Roth 2014).

A TLUD gasifier consists of a combustion unit (e.g., a tin can) with separate entry holes for primary and secondary air (Figure 1.13). For operation, the combustion unit is filled with dry solid biomass fuel, and the fire is lit at the top, producing a “pyrolysis flame front”. This flame front moves downward through the mass of solid fuel converting the biomass into biochar while the produced gases (wood gas or smoke) travel upwards. The progression of the pyrolysis front is controlled by regulating the primary airflow. During the combustion process a small amount of primary air moves upward from the bottom allowing only a partial combustion of the created wood gas, which then enters in contact with the oxygen coming from the secondary air holes and is burnt producing a very clean yellow-orange flame. Above the pyrolysis front, the created char accumulates without burning due to the lack of oxygen. When all the solid biomass is pyrolyzed, the flame size decreases and the color turns into a blue flame, indicating that the biochar is ready to be harvested, cooled down and stored (Roth 2014).

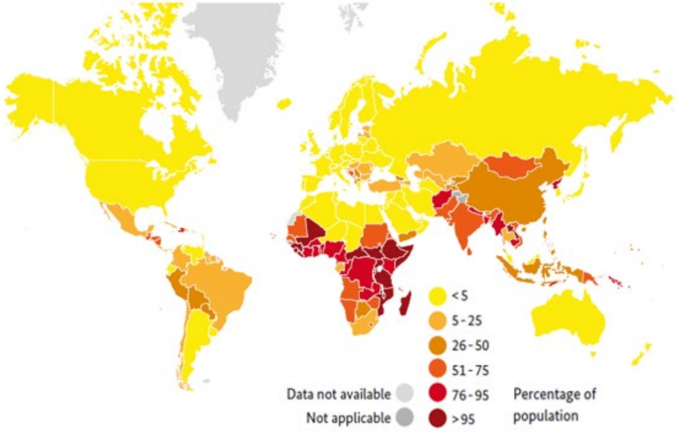


Figure 1.14. *Regional distribution of population cooking with solid fuels (Roth 2014).*

The development of these type of cook stoves arises from the need to improve traditional cooking performance and to address the negative health impacts associated with the toxic emissions from burning solid biomass (Roth 2014). It is estimated that three billion people worldwide cook their food with traditional stoves or open fires by burning solid biomass, such as firewood, dung, charcoal and agricultural waste (Figure 1.14; WHO 2018). These cooking practices produce harmful emissions that cause fatal illnesses and that claim the lives of 3.8 million people every year (WHO 2018). Moreover, the high demand and abuse of wood for firewood and the inefficient performance of this traditional way of cooking lead to land degradation and deforestation (Birzer et al. 2013). The use of TLUD micro-gasifiers offer advantages for cooking applications and for domestic heating, including reduction of smoke, steady hot flame, no need for a continuous wood feeding during the operation, higher heat efficiency compared to traditional cooking stoves and, therefore, less wood demand (Figure 1.13; Roth 2014).

Figure 1.15. *Examples of top-lit up-draft microgasifiers.*

1.2.4.3 The origin of interest in biochar

The increasing interest in biochar and its application in soils is related to the re-discovery of the highly fertile Amazonian Dark Earths known as Terra Preta (Glaser et al. 2001). After the characterization of these soils, the scientific community recognized that the enhanced fertility of Terra Preta soils resulted from their higher levels of soil organic matter, nutrient-holding capacity, and nutrients such as N, P, K and Ca, higher pH values and higher moisture-holding capacity (Glaser et al. 2001). However, they also observed that many of the surrounding soils were weathered, red, acidic and nutrient poor, typical of the humid tropics (Deutsche Bundesstiftung Umwelt 2015). These opposite soil properties clearly pointed to an anthropogenic origin of Terra Petra soils (Figure 1.16).

Figure 1.16. *Typical soil profile of Terra Petra sites (left) and surrounding Ferralsol regions (right; Glaser et al. 2001).*

Further studies revealed that Terra Preta soils are of pre-Columbian origin and that were formed over centuries by repeated input of nutrients in form of food waste and human excrements together with charred residues (biochar; Deutsche Bundesstiftung Umwelt 2015; Glaser 2007). The biochar's high recalcitrance plays a prominent role in Terra Preta genesis, as it allows biochar to act as a significant C sink, by stabilizing soil organic matter that otherwise would be difficult under humid tropical conditions (Glaser et al. 2001). In addition, surface oxidation of these charred residues during all these years produces carboxylic groups on the edges of the aromatic backbone, which increases its nutrient-holding capacity (Glaser et al. 2001). However, the high amounts of charcoal found in the Terra Preta soils do not primarily contribute to high nutrient contents and therefore, the addition of extra nutrient sources is also key in these soils' genesis. Some potential nutrient sources include human and animal excrements (rich in P and N), mammal and fish bones (rich in P and Ca), ash residues of incomplete combustions (rich in Ca, Mg, K, P and charcoal), terrestrial plant biomass (e.g., green manure, compost), and aquatic plant biomass (e.g., algae; Glaser 2007).

1.2.4.4 Biochar as compost amendment and co-composted biochar application in soil

As mentioned in section 2.3.1, for a successful composting process to take place and to reduce nutrient and GHG losses, suitable initial conditions such as aeration, feedstock formulation, bulking agents need to be defined, controlled and maintained throughout the composting process. Biochar addition at the beginning of composting can promote composting processes by providing those aforementioned suitable composting conditions. Furthermore, when co-composting biochar by mixing biochar with compost feedstocks before starting a composting process, its surfaces interact with microorganisms, minerals, dissolved organic and inorganic compounds, and gases (Hagemann et al. 2017). These interactions created during composting form nutrient-rich organo-mineral phases that are characterized by high concentrations of oxidized functional groups, redox-active mineral oxides and surface-nutrient loading that enhance the agronomic performance of biochar as a soil amendment (Joseph et al. 2010; Kammann et al. 2015). Moreover, the biochar's recalcitrance, high surface area, microporosity and sorption capacity are very important characteristics which influence all of the essential functions of soil fertility, including water and air supply, nutrient cycling and microbial activity (Lehmann and Joseph 2009).

Given these unique properties of biochar, a number of benefits have been documented when biochar is applied as compost amendment and to soil in combination with compost (some of these benefits of biochar as compost amendment are further discussed in chapters 2 and 3, and as soil amendment in chapter 4; Fisher and Glaser 2012; Godlewska et al. 2017; Guo et al. 2020; Wu et al. 2017a; Xiao et al. 2017). Biochar produced from different feedstocks and under different pyrolysis conditions exhibits a range of physical and chemical properties, and therefore the effects observed with biochar addition may vary (Lehmann and Joseph 2009). Some of the advantages offered by biochar addition in both composting and soil include:

Increased aeration: Given that biochar has a very porous nature, its application to compost and soils has been experimentally linked to improved aeration conditions (He et al. 2017; Lehmann and Joseph 2009; Liu et al. 2017b). Improved aeration will be partly due to higher air-filled porosity and improved supply of oxygen throughout the compost and soil matrix (Lehmann and Joseph 2009). Where sufficient O_2 is available, aerobic respiration will be the dominant metabolic pathway for energy generation, resulting in water and CO_2 as the primary metabolic end products and thereby reducing the incidence of anaerobic conditions required for CH_4 and N_2O production (further discussed in chapter 3; Chen et al. 2017; Jia et al. 2016; Li et al. 2016; Sonoki et al. 2013; Wang et al. 2013; Wang et al. 2018).

Improved water holding capacity: The high porosity of biochar may also allow it to retain more moisture, which may result in an overall increase in the water-holding capacity of compost and soils (Fisher and Glaser 2012). For example, the limited capacity of sandy soils to store water and plant nutrients is partly related to the relatively small surface area of these soil particles and therefore, the addition of compost with co-composted biochar can increase the water-holding capacity of these type of soils (Lehmann and Joseph 2009).

Liming effect: It has been well documented that biochar application can significantly alter compost and soil pH due the alkaline nature of the ash contained in biochar, which is rich in Ca, Mg, K and Na oxides, hydroxides and carbonates (further discussed in chapters 2, 3 and 4; (Agyarko-Mintah et al. 2017; Jia et al. 2016; Lehmann and Joseph 2009). The liming effect of biochar not only ameliorates soil acidity, but can influence the activity of bacterial populations, e.g., that of the enzyme N_2O reductase of denitrifying microorganisms, resulting in less N_2O emission via denitrification (Lehmann and Joseph 2009).

Increased cation exchange capacity: Interactions between biochar, compost and soil particles, dissolved organic matter, gases, microorganisms and water increase the concentration of oxygenated functional groups, such as phenolic, carboxylic, hydroxyl, carbonyl and quinone groups, on the surface of biochar (Fisher and Glaser 2012; Lehmann and Joseph 2009). In the case of composting, the high temperatures during the thermophilic stage play also an important role in driving the oxidation of fresh biochar surfaces and creating negative charges in deeper layers of biochar particles (Fisher and Glaser 2012; Wu et al. 2017b). These negatively charged particles increase the charge density on the surfaces of biochar and are therefore responsible for the increased cation exchange capacity in compost and soil (Zhang and Sun 2014).

Provision of extra habitat for microorganisms: The porous structure of biochar, its high internal surface area and its ability to retain water and to adsorb soluble organic matter, gases and inorganic nutrients are likely to provide a highly suitable habitat for microorganisms particularly for bacteria, actinomycetes and arbuscular mycorrhizal fungi (Figure 1.17; Fisher and Glaser 2012; Lehmann and Joseph 2009; Steiner et al. 2010).

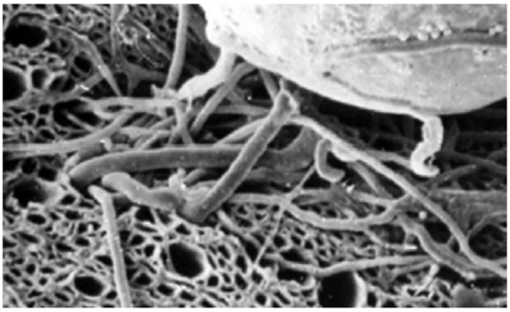


Figure 1.17. *Mycorrhiza fungal hyphae growing into biochar pores (Lehmann and Joseph 2009).*

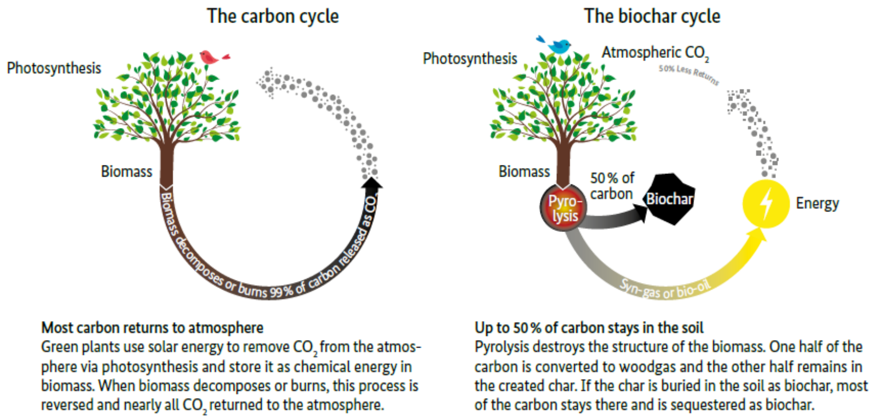
Reduced nutrient losses and GHG emissions: The large number of micropores in biochar is responsible for the high adsorptive capacities for gases such as N_2O, CO_2, CH_4, and NH_3 (Hestrin et al. 2020; Kamman et al. 2017; Steiner et al. 2010), as well as dissolved organic matter and nutrients, including NO_3^- and NH_4^+ (Kammann et al. 2015). This adsorption can take place either through charge or covalent interaction or through interactions with oxidized functional groups present on biochar surfaces (Hagemann et al. 2017). This can thus reduce leaching losses nutrient, such as NO_3^-, NH_4^+, PO_4^{3-}, and K^+ during composting (further discussed in chapter 2; Hagemann et al. 2018; Joseph et al. 2017; Steiner et al. 2010; Wang et al. 2017), and in agricultural soils (further discussed in chapter 4; Agegnehu et al. 2017; Glaser and Lehr 2019; Hagemann et al. 2017; Kammann et al. 2015; Tsai and Chang 2020). In addition, since biochar application alters the supply of electron acceptors such as $O_2 > NO_3^- > SO_4^{2-} > PO_4^{3-} > CO_2$, and that of electron donors as NH_4^+, as well as the redox potential (through improved aeration), adding biochar to compost and soil can also control the emissions of gases such as N_2O, NH_3, and CH_4 (Lehmann and Joseph 2009). For example, during the heterotrophic denitrification process, denitrifiers use NO_3^- as the electron acceptor and prefer low to zero dissolved O_2 (low redox potential) and therefore, by adsorbing NO_3^- and providing more O_2, biochar may reduce N_2O production occurring via nitrification and denitrification (further discussed in chapter 3; Lehmann and Joseph 2009).

A particular benefit when applying co-composted biochar to soil is:

Carbon sequestration: The macro-molecular structure of biochar dominated by aromatic C makes biochar more recalcitrant to microbial decomposition, with a turnover time of hundreds to thousands of years and thereby, improving C sequestration in soils (Fisher and Glaser 2012; Lehmann and Joseph 2009). As already mentioned in section 2.3.4 (Compost application in soil), it has been widely recognized that SOC sequestration can be of great importance as a climate change mitigation and adaptation measure. Thus, the addition of a stable C input as biochar, and even better of co-composted biochar offers a potential sustainable soil carbon sequestration practice via SOM sorption and organo-mineral interaction (further discussed in chapter 4; Agegnehu et al. 2017; Amelung et al. 2020; Fisher and Glaser 2012; Lorenz and Lal 2018). Soil aggregation and formation of organo-mineral complexes may be the most important stabilization process in topsoil of agroecosystems (Lorenz and Lal 2018). Additionally, biochar application to soil transfers C originating from atmospheric CO_2 to the soil, protecting it from combustion and maintaining it in a relatively stable form for a long period in comparison to the parent organic materials (Figure 1.18; Fisher and Glaser 2012; Roth 2014).

And a benefit observed specifically for composting is:

Increasing thermophilic temperatures and accelerating the decomposition of OM: The additional habitats and suitable conditions for microorganisms, and the improved aeration conditions caused by biochar addition to composting mixtures has been linked to enhanced microbial activity and, therefore, to the resulting higher temperatures during the thermophilic phase and to the faster decomposition of the organic matter (Awasthi et al. 2017; Chen et al. 2017; Zhang and Sun 2014).



Most carbon returns to atmosphere
Green plants use solar energy to remove CO_2 from the atmosphere via photosynthesis and store it as chemical energy in biomass. When biomass decomposes or burns, this process is reversed and nearly all CO_2 returned to the atmosphere.

Up to 50 % of carbon stays in the soil
Pyrolysis destroys the structure of the biomass. One half of the carbon is converted to woodgas and the other half remains in the created char. If the char is buried in the soil as biochar, most of the carbon stays there and is sequestered as biochar.

Figure 1.18. *The carbon cycle versus the biochar cycle (Roth 2014).*

1.3 Objectives and outline of this work

Ecological sanitation via thermophilic composting of human excreta by using appropriate technology could be a feasible option for implementing a climate-smart agriculture concept that increases access to safely managed sanitation, improves waste management, soil health and food security, and opens ways for climate change mitigation in developing countries. Realizing the potential that this holistic approach offers, with the present work, we aimed to evaluate the feasibility of an appropriate-technology EcoSan concept as an approach to climate-smart agriculture in safely and properly treating human and animal excreta with other organic waste materials and biochar via thermophilic composting, and in producing compost for fertilization and soil amendment purposes. In order to maximize the benefits of this approach, we focused our research on these three main research objectives, each of which is addressed in different chapters:

1. Investigate an appropriate-technology thermophilic composting process of human excreta, and separately of cattle manure, both with kitchen scraps, sawdust, teff straw and biochar.

In *Chapter 2*, we examine how key nutrients and physical and chemical parameters of four composting treatments (human excreta or cattle manure, with and without biochar) behave during appropriate-technology thermophilic composting, during which different degradation and transformation processes take place. By following the dynamics of different nutrients and parameters, we aimed particularly to:

i) Evaluate if using a traditional compost box and bulking materials such as sawdust, straw and biochar as appropriate technology options enable a well-running and hygienically safe treatment of human excreta and cattle manure via thermophilic composting to produce a mature, nutrient-rich fertilizer, free of pathogens and phytotoxic substances, and with the least pile turning workload possible.

ii) Compare the composting process of human excreta with that of cattle manure. We hypothesized that under the same and suitable composting conditions humanure composting would behave differently than cattle manure, as the higher nutrient contents in human excreta compared to cattle manure could provide better and more balanced conditions for microbial activity, thereby yielding higher temperatures and accelerating the degradation process. Additionally, we expected that this higher nutrient load in human excreta compared to cattle manure would be reflected in the final composts.

iii) Given the benefits associated with biochar, we were particularly interested in studying the effect that biochar has during composting. We hypothesized that the use of biochar would reduce the losses of organic matter, organic C, and nutrients, and increase thermophilic temperatures and aeration conditions during composting as well as the pH and cation exchange capacity of the final composts.

2. Quantify CO_2, CH_4, N_2O, and NH_3 emissions of the appropriate-technology thermophilic composting process of the four composting treatments.

To further evaluate our thermophilic composting process, in *Chapter 3*, we analyzed the GHG and NH_3 emissions occurring during thermophilic composting of human excreta and cattle manure with and without biochar addition. By quantifying and following these gas emissions, we aimed particularly to:

i) Assess the impact of our appropriate-technology composting process of human excreta on the atmosphere, and compare it with that of cattle manure composting. We hypothesized that humanure composting would result in higher NH_3 emissions and in a faster decrease of CO_2 emissions compared to cattle manure due to the higher thermophilic temperatures and faster degradation process as consequence of the better conditions for microbial activity provided by the higher nutrient contents in human excreta. In addition, due to the high presence of methanogens in the digestive tract of ruminant animal, we hypothesized that cattle manure would yield higher emissions of CH_4.

ii) Evaluate the biochar's climate change mitigation potential during the treatment of these excreta. In this regard, we hypothesized that biochar would improve aeration conditions inside the composting pile, and reduce the emissions of CH_4, N_2O and NH_3 due to better oxygen supply and/or other biochar-mediated effects such as adsorption effects.

3. Investigate the use of human excreta and cattle manure compost, with and without biochar, and especially of large application rates, as fertilizers and soil amendments.

Besides investigating the composting process of human excreta and cattle manure, we also studied the use of the four types of compost produced (human excreta or cattle manure compost, with and without co-composted biochar) as soil fertilizer. For this, in *Chapter 4*, we explored in a 180-day incubation experiment the nutrient dynamics and GHG emissions of these four types of compost at two application rates (total compost N equaled 170 kg N ha^{-1}, and three times this amount) to a sandy soil at 25°C, which was intended to resemble tropical temperatures of the Ethiopian highland. To elucidate the role of these composts as fertilizers and soil amendments, we specifically aimed to:

i) Evaluate the mineralization dynamics during the application of human excreta and cattle manure-derived compost in soil. In this regard, we hypothesized that under tropical conditions (in this study at an average soil temperature of 25°C) complete mineralization of the non-biochar-amended composts added would take place during the 180 days of incubation.

ii) Evaluate the potential that biochar-compost mixtures have in increasing C sequestration and reducing nutrient leaching in agricultural soils. Here, we wanted to test the hypotheses that the addition of relatively small amounts of co-composted biochar has the potential to stabilize soil organic matter and decrease C mineralization, reduce CH_4 and N_2O emissions and the risk of nutrient leaching from soil, mainly as nitrate and phosphate.

iii) Assess the application of large amounts of compost. We hypothesized that applying the maximum amount of compost allowed in Germany would not increase the risk of nutrient leaching from agricultural soils.

Following these three chapters, *Chapter 5* seeks to connect their findings and discuss the implication that these have for the future of sustainable sanitation and soil nutrient management. In this work, Ethiopia is chosen as a model region; however, this integrated approach could be easily applicable to other regions worldwide.

2

NUTRIENT DYNAMICS DURING COMPOSTING OF HUMAN EXCRETA, CATTLE MANURE AND ORGANIC WASTE AFFECTED BY BIOCHAR

Based on:

Castro-Herrera, D., Prost, K., Schäfer, Y., Kim, D-G., Yimer, F., Tadesse, M., Gebrehiwot, M., Brüggemann, N. Nutrient dynamics during composting of human excreta, cattle manure and organic waste affected by biochar. *Manuscript submitted to Journal of Environmental Quality*

Abstract

Lack of sanitation and limited access to fertilizers are global challenges, particularly in developing countries. Using compost made via thermophilic composting from human excreta could represent a strategy for increasing agricultural productivity and public health. We developed an appropriate-technology ecological sanitation concept by composting human excreta, and separately cattle manure, with kitchen scraps, teff straw, sawdust and biochar to produce a pathogen-free and nutrient-rich fertilizer and soil amendment. We followed the dynamics of the most important nutrients (N, P, K), as well as physical (temperature, moisture content), and chemical parameters (pH, electrical conductivity, cation exchange capacity, total organic matter, total organic C, Ca, Mg, and micronutrients) throughout the process. We implemented a well-running and hygienically safe thermophilic composting process, as assessed by the low N, P, K, Ca, and Mg losses, and the temperature profile. Average temperatures in the compost reached values above 60°C for 7, 6, 5, and 8 consecutive days for treatments containing human excreta, human excreta amended with biochar, cattle manure, and cattle manure amended with biochar, respectively. In the cattle manure with biochar treatment, biochar led to a significant temperature increase with a maximum value of 65.9°C reached at day 6. Biochar reduced losses of organic matter (18−23%), C (33−42%), N (49−100%), and decreased the extractable NO_3^--N (32−36%) in the final compost. The tested ecological sanitation concept may thus represent a strategy to increase access to sanitation, food security, waste management and sustainable agricultural production, and to decrease N losses due to nitrate leaching.

2.1 Introduction

About 2 billion people do not have access to basic sanitation, and of these, 673 million practice open defecation (WHO 2019b). This combined with poor or inexistent waste management systems, make dumping of human excreta and other organic waste in streets, drains, and riverbanks a common practice (Langergraber and Muellegger 2005; Orner and Mihelcic 2018; Somorin 2020). This not only pollutes water, thereby endangering public health and killing 432,000 people every year (WHO 2019b) but it also produces a drain of nutrients from agricultural and food systems (Langergraber and Muellegger 2005; Ryals et al. 2019; Trimmer and Guest 2018). Instead of recycling nutrients from organic waste and sanitation, atmospheric N is converted by a high-energy intensive process (Haber-Bosch), and P mined from finite phosphate rock deposits to produce mineral fertilizers, making our food system heavily dependent on non-renewable synthetic nutrients (Akram et al. 2019; Alewell et al. 2020; Elser and Bennett 2011; Trimmer and Guest 2018). In addition to the limited access to fertilizers, those regions with the lowest sanitation coverage, such as Sub-Saharan Africa, experience the most severe land degradation in the world (Nkonya et al. 2016; Reid 2020).

Nitrogen, P, K and other macro- and micronutrients in human and animal excreta could replace a substantial part of synthetic fertilizers and increase nutrient access in low-income countries (Akram et al. 2019; Berendes et al. 2018; Trimmer and Guest 2018). The major proportion of plant-available nutrients excreted by humans is found in urine (15–19% N, 2.5–5% P, and 3–4.5% K, 4.5–6% Ca), while part of the nutrients in feces (5–7% N, 3–5.4% P, and 1–2.5% K, 4–5% Ca) are not soluble but locked in aggregated larger particles of indigestible fat, protein and vegetable fibers (Jenkins 2005; Jönsson et al. 2004).

Every year, on average, each human excretes 50 L of feces (Ecodesign: The bottom line 2012). With the continuously increasing world population, it is estimated that by 2030, the global annual production of human feces will exceed 1×10^{12} kg of fresh fecal biomass per year (Berendes et al. 2018). For animal manures, this projection is about four times this amount (Berendes et al. 2018). Thus, the management of human and animal excreta is a crucial factor both for fertilizer production and for addressing public health and environmental pollution issues. Ecological sanitation (EcoSan) offers potential solutions to address these challenges and represents a sustainable nutrient management strategy (Akram et al. 2019; Ryals et al. 2019). Thus, authors like Jenkins (2005) call human excreta also “humanure” in order to stress its fertilizing potential.

EcoSan via thermophilic composting, i.e., reaching temperatures >45°C and, for hygienization, >55°C, >60°C or >65°C for 2 weeks, 6 days or 3 days, respectively (Bundesministerium der Justiz und für Verbraucherschutz 2017; Piceno et al. 2017; Preneta et al. 2013) is a feasible way to transform excreta, together with other organic waste, into compost. The end product is free of phytotoxicity and pathogens, rich in nutrients and humus, and can be used to improve soil fertility (Jenkins 2005; Maheshwari 2014; McNicol et al. 2020; Ryals et al. 2019; Somorin 2020). Nevertheless, constraints associated with composting are loss of nutrients due to leaching and/or gaseous emissions (Bernal et al. 2009; Martins and Dewes 1992; Steiner et al. 2010), and, particularly for developing countries, limited financial resources and access to infrastructure and technology, e.g., for turning compost piles (Dalzell et al. 1987; Shuval et al. 1981). Thus, for developing communities, the use of appropriate technology, i.e. technology suitable for prevailing social and economic conditions, is crucial, as it provides the chance to be self-sufficient (Sianipar et al. 2013). One approach is developing a composting process in wooden compost boxes with a volume >1 m^3 together with bulking materials, such as straw, sawdust, and biochar that allow pile aeration, thereby reducing the need for a constant heap turning (Dalzell et al. 1987; Jenkins 2005; Peigné and Girardin 2004).

Biochar is a carbon-rich material produced by thermochemical conversion of biomass under limited supply of O_2 (Lehmann and Joseph 2009). Due to its unique properties, such as recalcitrance, high surface area, microporosity, and sorption capacity, biochar shows great potential as amendment for composting of various types of waste (Chen et al. 2017; Lehmann and Joseph 2009; Prost et al. 2013). Numerous studies have observed that composting organic materials with biochar improves aeration conditions (Lehmann and Joseph 2009; Liu et al. 2017b); reduces nutrient losses, especially of N (Joseph et al. 2017; Steiner et al. 2010; Wang et al. 2017); leads to higher temperatures during the thermophilic phase (Awasthi et al. 2017; Chen et al. 2017) and to faster organic matter (OM) degradation (Zhang and Sun 2014). It additionally increases cation exchange capacity (CEC) of the final compost (Zhang and Sun 2014); and provides additional microhabitats for microorganisms, enhancing their activity (Lehmann and Joseph 2009).

To our knowledge, no studies combining thermophilic composting of human excreta from EcoSan and biochar exist to date. We aimed to implement an appropriate-technology thermophilic composting process of human excreta to produce a pathogen-free and nutrient-rich organic fertilizer and soil amendment with the least pile turning workload possible. With respect to this main objective, we evaluated the effect of biochar addition to compost, and the type of manure (humanure and cattle manure), by following the dynamics of key nutrients and of chemical and physical parameters.

2.2 Materials and methods

2.2.1 Biochar production

We produced biochar from Eucalyptus wood (*Eucalyptus camaldulensis*) with a top-lit up-draft (TLUD) micro-gasifier at pyrolysis temperatures between 500−600°C and residence time of 40−50 min. The TLUD micro-gasifier (Noah stove, patent pending, appendix A: Figure A.1, Figure A.2) was designed and developed by the clay construction company PRO LEHM (Langballig, Germany) and built from local materials (mix of clay and sand soil fractions with teff straw and used tin cans) in Wondo Genet College of Forestry and Natural Resources (WGCF-NR), southern Ethiopia. We used wood pieces with average dimensions of 3.2 x 2.8 x 1.8 cm and moisture content (MC) of 12–16% to produce biochar.

Before adding to the compost mixture, we crushed the biochar pieces and sieved them to <14 mm, with 85% of the particles in the range of 2−12 mm, 9% between 1−2 mm, 3% between 0.5−1 mm, and 3% below 0.5 mm.

Biochar was analyzed according to the requirements of the European Biochar Certificate (EBC 2012) by Ruhr Lab GmbH (Gelsenkirchen, Germany). Physical and chemical properties show that the biochar (Table A.1) fulfills the "premium" requirements of the European Biochar Certificate. The polycyclic aromatic hydrocarbons concentration exceeds the premium threshold value but complies with the "basic" criteria. Hence, the Noah cookstove is suitable for cooking and for producing biochar for compost and soil amendment.

2.2.2 Thermophilic composting and experimental design

We conducted the composting from March to November 2019 in a roofed facility inside WGCF-NR. Ambient temperatures ranged from 15.0−27.0°C. Compost treatments consisted of human excreta (from now on called "humanure") or cattle manure, vegetable scraps (mainly cabbage plus peels of onions, potatoes and carrots), teff straw, and sawdust, with and without biochar. For collecting humanure, we built an EcoSan toilet house inside the college's campus, consisting of bucket-based dry toilets (Figure A.3). Human fecal material and urine were collected together with sawdust and toilet paper (10:1 w/w fresh sawdust:humanure). Cattle manure was collected from the WGCF-NR's dairy facility together with sawdust (here referred to as cattle manure, 4:1 w/w fresh sawdust:cattle manure), which had been used as bedding material. These two different sawdust:manure ratios (4:1 for cattle manure and 10:1 for humanure) arose from the different collection practices. Physical and chemical properties of these materials are listed in Table A.2 (Appendix A).

Trials comprised four treatments, whereof two were controls and two biochar (BC)-amended treatments, each with four replicates. One of the controls (HM) consisted of a mixture of fresh humanure (50% dry wt), vegetable scraps (32% dry wt), and teff (*Eragrostis tef*) straw (18% dry wt). The second control (CM) was composed of fresh cattle manure (55% dry wt), and the same amount of the other organic materials (dry wt: 29% of vegetable scraps and 16% of teff straw). The other treatments (HM+BC and CM+BC) were prepared with the same mixtures mentioned before, but with biochar addition (19% dry wt; Appendix A: Table A.3). The compost substrates were manually mixed and filled into wooden boxes with dimensions of 1.5 x 1.5 x 1.4 m (width x depth x height) until a height of 1−1.1 m (Appendix A: Figure A.4, Figure A.5). To provide aeration, compost boxes were built with gaps between the boards and with bamboo poles arranged in the bottom (Appendix A: Figure A.6). For insulation purposes, a layer of 10–15 cm of teff straw was placed on the bottom, back, front, left, and right side of the box, and after placing the composting materials, the top of the pile was also covered with teff straw (Appendix A: Figure A.6). Initial MC of the composts ranged between 78.6−81.4%

(fresh wt), and bulk density between 344−377 kg m^{-3}. No further MC adjustment was made throughout the composting period.

A randomized complete block design of four blocks was applied. Each block started 15−18 days after the other and contained one replicate of each treatment (Appendix A: Figure A.7). After 101 days of composting, germination index tests revealed insufficient maturity levels and therefore, to stimulate the ongoing process, replicates of the same treatment (with different composting times) were mixed and turned. Thereafter, composting continued for another 72 days. Since each block started 15−18 days after the other, the composting time, during which replicates were combined, was 165, 148, 129, and 113 days for replicates from blocks 1, 2, 3, and 4, respectively.

2.2.3 Compost sampling and sample preparation

Two samples were collected on day 0, 7, 14, 21, 28, 56, 87, and 185 from each pile. Each sample consisted of six subsamples taken from the center, edge and corner of both top and middle horizontal plane of the pile (Figure A.8). Fresh compost samples were ground to <10 mm with a hi-speed multifunctional grinder (RRH-100, Ririhong, China) and used for determining MC, pH, electrical conductivity (EC), CEC, NH_4^+, NO_3^-, total organic matter (TOM), and germination index (GI). Total organic C (TOC), total N (TN), total P (TP), total K (TK), available P (P_{av}), available K (K_{av}), total Ca (TCa), total Mg (TMg), and micronutrients (Zn, Cu, Fe, Mn, B, and Mo) were determined from dry (at 105°C) and ball-mill ground samples. We analyzed two laboratory replicates for each parameter, and one replicate for TP, TK, P_{av}, and K_{av}.

2.2.4 Control and indicator parameters of the composting process

2.2.4.1 Temperature

Compost temperature was measured at 18 different points (Figure A.9) every day the first 28 days, thereafter every two days, and after day 87 every two weeks until day 101. Thereafter, temperature was measured on day 129 and 143.

2.2.4.2 Moisture content, pH, electrical conductivity, and cation exchange capacity

Moisture content was determined by drying the samples at 105°C for 24 h. pH was measured with a glass electrode (WTW 3310 SET2, SenTix41, Xylem Analytics, Weilheim, Germany) in a water extract (1:10, w/v) according to FAO (2008). The EC was determined with a conductivity meter (WTW Cond 3110 SET1, TetraCon325, Xylem Analytics) after 1 h of shaking in deionized water (1:5, w/v) and filtering (filter paper MN615, Macherey & Nagel, Düren, Germany) according to FAO (2008). For CEC, we followed the method described by FAO (2008), mixing fresh compost samples with 1.0 mol L^{-1} sodium acetate solution (1:5, w/v) for 5 min to saturate the exchangeable sites with Na^+. Thereafter, exchangeable Na^+ was replaced with 1.0 mol L^{-1} ammonium acetate solution (1:5, w/v) for 5 min, and the extract filtered (filter paper MN640m, Macherey & Nage, Düren, Germany). We measured Na^+ concentration of the filtrate with a flame photometer (BWB-XP, England). Final compost samples (day 185) were not analyzed for CEC due to university closure caused by COVID-19.

2.2.4.3 Germination index

We determined the GI to evaluate phytotoxicity and maturity of the compost (adapted from Li et al. 2015), by measuring germination and root elongation of 10 cress seeds (*Lepidium sativum*) after incubating them at 25°C for 48 h in petri dishes containing filter paper and 5 mL of filtered water-compost extracts (1:10 w/v, shaking for 1 h at 250 rpm, filter paper MN615, Macherey & Nagel) or 5 mL distilled water as control.

2.2.5 Total organic matter, total organic carbon and nutrients analysis

Total OM was measured through loss of weight on ignition with a muffle furnace at 400°C for 4 h. Total organic C was measured with an elemental analyzer (Thermo Scientific Flash 2000 coupled with a Thermo Scientific IRMS Delta V Plus, USA).

We extracted NH_4^+ and NO_3^- from fresh compost samples with 0.01 mol L^{-1} $CaCl_2$ (1:10 w/v for 2 h). After filtering (filte paper MN640d, Macherey & Nagel), we analyzed NH_4^+-N with a spectrophotometer (WTW photoLab 7100VIS, Xylem Analytics) with the indophenol blue method (FAO 2008; VDLUFA 2014). Nitrate-N was analyzed with ion chromatography (Dionex ICS-3000, USA). We determined TN by summing up NH_4^+-N, NO_3^--N, and N contents of the solid residue remaining after extraction. This residue was dried at 105°C for 24 h, ground with a ball mill and measured for N with the same elemental analyzer used for TOC analysis. Plant-available P and K were determined by inductively coupled plasma optical emission spectroscopy (ICP-OES, iCAP6500, Thermo Fisher Scientific, Oberhausen, Germany), after extracting with 0.05 mol L^{-1} calcium-acetate-lactate solution (CAL extraction, 1:20 w/v, shaking for 90 min at 200 rpm). Total P, K, Ca, Mg, and micronutrients were also determined by ICP-OES, after an aqua regia microwave extraction/digestion.

2.2.6 Determination of losses of organic matter, C and nutrients

To calculate losses of OM, TOC, TN, TP, TK, TCa and TMg during composting, we considered the final and initial pile weights (Appendix A: Table A.4) by using equation 2.1.

$$\text{Loss}(\%) = \left(\frac{W_0P_0 - W_{185}P_{185}}{W_0P_0}\right) \times 100 \qquad \text{(Equation 2.1)}$$

where W_0 is the pile weight at day 0 (kg dry wt.); P_0 the value for the chemical parameter at day 0 (mg kg^{-1} dry wt.); W_{180} the pile weight at day 185 (kg dry wt.); P_{180} the value for the chemical parameter at day 185 (mg kg^{-1} dry wt.).

2.2.7 Statistical analysis

Since we combined all replicates of the same treatment during the maturation phase, we divided our statistical analysis accordingly. For day 0 and 185, we conducted a one-way ANOVA followed by the general linear model procedure for least squares means with the Tukey method for the adjustment for multiple comparisons, when the data was balanced, and the Tukey-Kramer approximation, if the data was unbalanced. From day 0 to 87, we conducted a one-way repeated-measures ANOVA by using a mixed model with the Kenward-Roger method.

Data are reported as mean values ±SE and we considered a statistical significance level of $\alpha=0.05$. From day 0 to 87, parameters are reported as mean values of four replicates (n=4), and for day 185, as mean values of samples collected from three different points of the compost pile (n=3). Statistical analyses were conducted with the SAS software University Edition (SAS Studio, version 5.1).

2.3 Results and Discussion

2.3.1 Ecological sanitation

The dry toilets and the collection and storing of excreta, toilet paper and sawdust in one bucket showed to be a good alternative, with regard to cleanliness, odor, and utilization, showing the suitability of sawdust as an adsorber material (Obsa 2019).

2.3.2 Control and indicator parameters

2.3.2.1 Temperature

In all treatments, we observed a rapid increase in average temperatures above 45°C on day one, indicating the beginning of the thermophilic phase (Figure 2.1; Appendix A: Table A.5). The continuously increasing microbial activity increased the temperature further, reaching peak values of 65.1°C on day 4 for HM, of 64.2°C on day 3 for HM+BC, of 62.1°C on day 5 for CM, and of 65.9°C on day 6 for CM+BC. Thermophilic temperatures were maintained until day 18 for HM and CM+BC, day 16 for HM+BC, and day 17 for CM. As consequence of the high temperatures and the competition among microorganisms, a destruction of pathogens can be assumed (Senesi 1989). We measured average temperatures above 60°C during 7, 6, 5, and 8 consecutive days for HM, HM+BC, CM, and CM+BC, respectively. For CM+BC and the critical treatments containing human excreta, the duration of these hygienization temperatures complied with the sanitation hygienic treatment requirements for organic waste according to the German Biowaste Ordinance (Bundesministerium der Justiz und für Verbraucherschutz 2017). This ordinance demands for sanitation purposes, a temperature ≥55°C for an uninterrupted period of 2 weeks, ≥60°C over 6 days, or ≥65°C over 3 days has to be maintained.

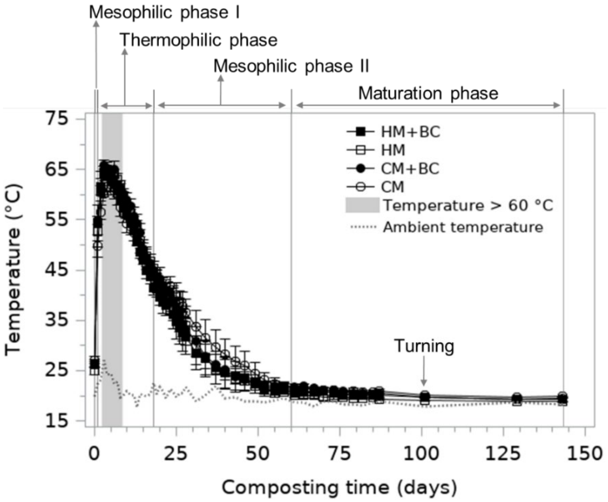


Figure 2.1. ***Temperature evolution during the course of composting.*** *Vertical lines mark the different phases during composting. Values are mean ± SE (n=4). Since replicates of the same treatment were combined during the maturation phase, values reported on day 129 and 143 are mean ± SE of three and two replicates, respectively. HM, humanure control; CM, cattle manure control; HM+BC, humanure amended with biochar; CM+BC, cattle manure amended with biochar.*

When summing up temperatures for days with temperatures >55°C, we observed a significant increase by 24% for CM+BC compared to CM (p=0.0358). Awasthi et al. (2017) and Chen et al. (2017) also detected significantly longer thermophilic phases when co-composting biochar. These findings suggest biochar may stimulate decomposition processes during composting due to its high porosity and surface area that provide better O_2 availability and distribution through the composting mixture (Lehmann and Joseph 2009). Additionally, biochar's complex porous structure holds water, soluble OM, gases, nutrients, and serves as microhabitat for microorganisms, providing better conditions for their metabolic processes (Lehmann and Joseph 2009). In contrast to cattle manure treatments, we did not detect a significant difference in the temperature sum >55°C between HM and HM+BC. After easily degradable organic substrates started to become depleted, the pile temperature continuously decreased until ambient temperature was reached. Ambient temperature was maintained during the maturation phase, during which OM stabilization and humification take place (Bernal et al. 2009).

2.3.2.2 Moisture content, pH, electrical conductivity, and cation exchange capacity

Moisture continuously decreased for all treatments throughout composting, from initial values of 78.6−81.4% to final values of 59.8−63.3% (Figure 2.2 A, Table 2.1). We explain this significant decline with the loss of water via leachate (not analyzed), and through evaporation, caused by the high thermophilic temperatures. The initial MC was above the normally recommended moisture conditions of 40−65% (Rynk et al. 1992; USDA 2000), but in the maximum range for highly porous materials, such as straw and sawdust (Golueke, 1972 [as cited by Gottschall (1992)]). Biochar-amended treatments had significantly lower MC than the controls (Table 2.1), indicating that adding a dry and porous material, such as biochar, can improve air diffusion inside the compost mixture and thereby, allowing higher evaporation (Liu et al. 2017b).

Changes in pH and EC followed the same pattern in all treatments throughout composting (Figure 2.2 B, C). During the first three weeks, pH continuously increased, with highest values reached on day 21 (8.5−9.0). This initial pH increase occurred as hydrolysis reactions, that consume a large quantity of H^+, predominated during the thermophilic phase (López-Zavala 2019). The initial decrease in EC confirms this hydrolysis activity, as the consumption of protons reduces their concentration in the compost's aqueous solution. Afterwards, EC increased and stabilized to 3.11−4.49 mS cm^{-1} after day 56, likely to be induced by an increase in nitrification activity, triggered by the increase in NH_4^+ availability and polymerization reactions of the remaining OM (Maheshwari 2014).

Biochar-amended composts exhibited significantly higher final pH values than the controls (Table 2.1), most probably due to the addition of alkalinity via the ash content in biochar (Lehmann and Joseph 2009; Xingyong et al. 2016), caused by the presence of Ca, Mg, K, and Na oxides, hydroxides and carbonates in the ash (Lehmann and Joseph 2009). Cattle manure composts showed significantly higher final pH values than the humanure counterparts (Table 2.1). We associate this with the 2.2−2.3 times larger final NO_3^- contents in the humanure treatments, suggesting larger nitrification activity in these composts compared to the cattle manure ones. Electrical conductivity exhibited higher final values measured for HM and HM+BC than for CM and CM+BC, but no clear biochar effect (Table 2.1).

After 87 days of composting, we measured a 1.1−2-fold increase in CEC in all treatments (Figure 2.2 D), caused by a higher number of functional groups, result from the OM oxidation during humification (Bernal et al. 2009). We did not observe any significant difference among treatments on day 0, but on day 87, we measured significantly lower CEC values in the biochar-amended treatments compared to the controls (Table 2.1). Conversely, Zhang and Sun (2014) reported higher final CEC values for their biochar treatments. However, we cannot evaluate the effect of biochar addition on CEC, as we did not obtain values for the final samples from day 185.

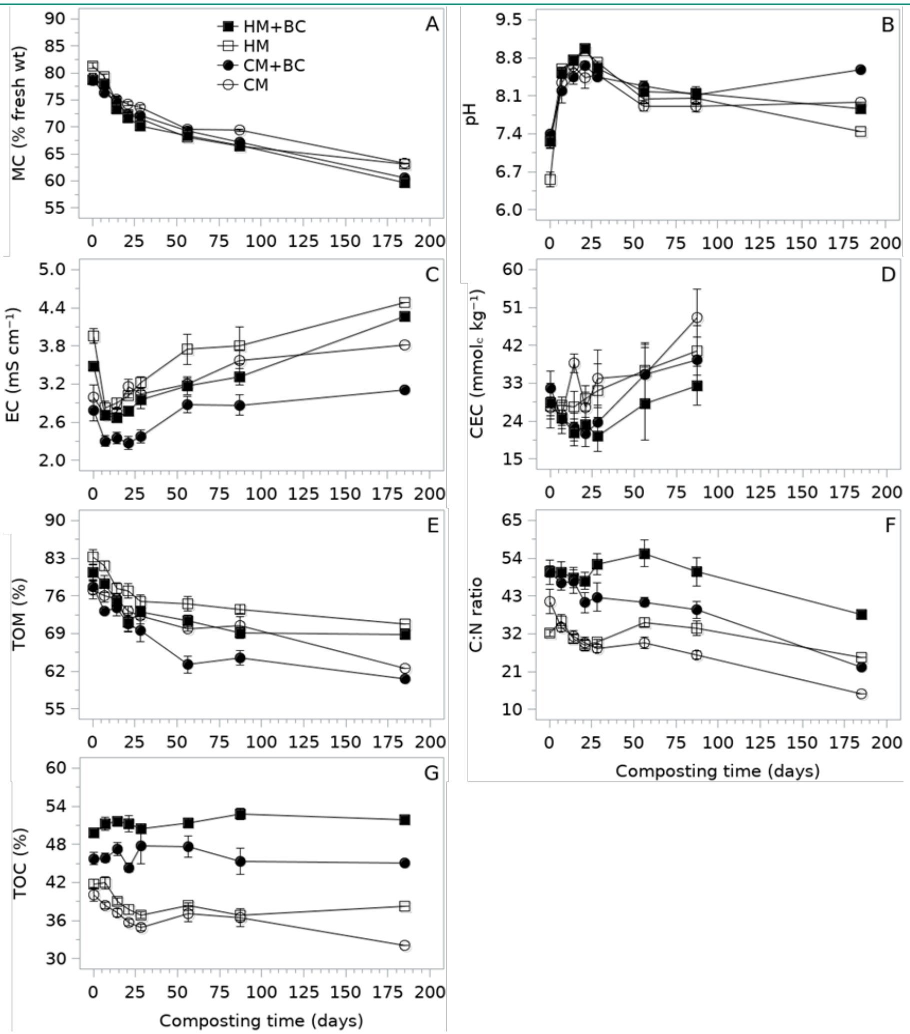


Figure 2.2. ***Chemical and physical parameters during the composting process.*** *(A) Moisture content (MC), (B) pH, (C) electrical conductivity (EC), (D) cation exchange capacity (CEC), (E) total organic matter (TOM), (F) carbon to nitrogen ratio (C:N), (G) total organic carbon (TOC). Values are mean ± SE (n=4 for days 0−87). At day 185, SE for mean values (n=3) are not shown, since another experimental and sampling design was used for this sampling period. HM, humanure control; CM, cattle manure control; HM+BC, humanure amended with biochar; CM+BC, cattle manure amended with biochar*

2.3.2.3 C:N ratio

Initial C:N ratios ranged from 49.8−50.2 and 32.3−41.4 for compost treatments with and without biochar, respectively. All C:N ratios continuously decreased until day 28. Among the recommended conditions for rapid composting, a C:N range of 20:1−40:1 is considered reasonable to ensure an appropriate nutrient balance, however, values slightly outside this range can also yield successful results (Rynk et al. 1992). After day 28, we observed an increase in C:N ratios in humanure treatments, while those in cattle manure treatments continued decreasing (Table 2.1, Figure 2.2 F), probably due to a faster C degradation rate in these last ones. We attribute this to larger amounts of sawdust in the humanure treatments. Sawdust is a complex C-rich organic material, high in lignin, making it recalcitrant to biological breakdown (Maheshwari 2014). Thus, after day 28, when easily degradable C compounds started to become depleted, the remaining C was the recalcitrant one of the sawdust and therefore, the degradation rate decreased. Final C:N ratios significantly decreased to 14.6−25.1 and 22.2−37.5 for treatments with and without biochar, respectively, but with significantly lower values in cattle manure treatments.

Biochar-amended composts had significantly higher C:N ratios throughout composting, which we attribute to the addition of biochar (Hagemann et al. 2018), as it has a high content of stable C compounds, which make it recalcitrant to degradation (Lehmann and Joseph 2009).

2.3.2.4 Germination index

We started assessing compost's phytotoxicity from day 87 on, assuming the absence of phytotoxic compounds (especially NH_4^+) and the maturity of the compost from this day on. However, after 101 days of composting, GIs <80% revealed insufficient maturity levels. In order to stimulate the ongoing process, replicates of the same treatment were combined, mixed, and turned. Afterwards the compost was left for another 72 days to mature. While turning, we observed that piles had compacted, which we assume may have reduced O_2 availability, thereby hampering the conversion of NH_4^+ to NO_3^- (Tiquia et al. 1996). Thus, teff straw did not prove to be a suitable bulking agent, if no or limited turning is planned during composting. A grass with stronger stems, e.g. wheat straw, that could resist compaction, should be preferred for this purpose, whereas teff straw can be used if regular compost turning is envisaged. We did not observe any significant differences between treatments regarding GI (Table 2.1). Final GI values of all treatments >100% classified the composts as mature, indicating the absence or only negligible concentrations of water-soluble phytotoxic compounds that may inhibit seed germination and root development (Bernal et al. 2009; California Compost Quality Council 2001).

We did not observe any significant differences between treatments regarding GI (Table 2.1). Final GI values of all treatments >100% classified the composts as mature, indicating the absence or only negligible concentrations of water-soluble phytotoxic compounds that may inhibit seed germination and root development (Bernal et al. 2009; California Compost Quality Council 2001).

2.3.3 Total organic matter, organic C and N dynamics

During the active phase of composting (i.e. thermophilic and mesophilic phase II), TOM contents continuously decreased (Figure 2.2 E), with a faster reduction during the first three weeks, likely due to higher availability of easily degradable compounds at the beginning of the process. Thereafter, OM degradation rate gradually declined as labile C sources decreased, and polymerization and humification reactions predominate during mesophilic II and maturation phases (Bernal et al. 2009). We did not observe any effect of biochar addition or type of manure on the initial and final TOM values, while HM+BC and CM+BC treatments showed significantly higher TOC contents than the controls (Table 2.1). However, as already explained for the C:N ratio, increased TOC contents can be attributed to the recalcitrant nature of biochar (Hagemann et al. 2018). Considering the mass loss of the piles, losses of TOM (50.1−68.6%) and TOC (39.3−69.1%; Table 2.1) were within the expected range for proper thermophilic composting (Bernal et al. 2009), and the addition of biochar led to a relative reduction of TOM losses by 18−23% and of TOC by 33−42% compared with the controls. We attribute this higher amount of remaining TOM and TOC in the biochar-amended composts mainly to biochar's stability against the high microbial activity occurring during composting (Hagemann et al. 2018).

We did not observe a significant difference among treatments regarding initial TN, while cattle manure treatments showed significantly higher final TN contents compared to humanure treatments (Table 2.1). Contents of N, P, and K in human excreta (5−7% N, 3−5.4% P, and 1.0−2.5% K; Jenkins 2005) are larger than in cattle manure (0.9−2.4% N, 0.1−0.8% P, and 1.5−2.1% K; Adekiya et al. 2020; Moreno-Caselles et al. 2002). Nevertheless, this difference was not reflected in our initial and final composts, as it was diminished by the larger sawdust amount in the humanure (1:10 human excreta:sawdust) compared to cattle manure treatments (1:4 cattle manure:sawdust). Nitrogen losses during composting ranged from -26.9% for CM+BC to 46.6% for HM (Table 2.1). While TN losses in HM were in the expected range for proper thermophilic composting (Bernal et al. 2009) and in HM+BC and CM even lower, the negative values for CM+BC would represent a net N gain, which is not possible. We assume this result was caused by imprecisions during sampling of the heterogeneous composting material on day 0. Nevertheless, as TN losses in the other treatments were either low or within the expected range, and as the TOM and TOC losses were consistent, we feel comfortable to state that the composting process, especially of HM+BC, CM, and CM+BC, conserved most of the N contained in the original feedstocks. Lower TN losses for cattle manure treatments suggest a higher amount of readily available C compounds in these treatments. This might have provided a more balanced supply of C and N for microbial activity and growth (Figure 2.2 F; Figure 2. 3 B), resulting in a better incorporation of N into the organic fractions or its immobilization by microbial biomass (Bernal et al. 2009). We thus recommend partly substituting sawdust with another, more easily degradable dry-toilet adsorber material (e.g. coffee husks) to reduce N losses during composting of human excreta. We found that amending humanure with biochar reduced N losses by 49%, and apparently 100% in CM+BC as compared to their controls (Table 2.1), which we mainly attribute to better aeration and absorption of available N compounds. Similarly, Steiner et al. (2010) and Wang et al. (2017) report a reduction of N losses during composting with biochar addition. However, here, except for the HM treatment, N losses were lower than the ones reported by these authors (50−61% for control and 31−44% for biochar treatments).

2.3.4 NH_4^+ and NO_3^- dynamics

We measured highest NH_4^+-N concentrations (1329−4695 mg kg^{-1}) on day 0. Afterwards, NH_4^+-N continuously declined, either due to nitrification, microbial immobilization, and/or losses via leaching and NH_3 volatilization (Bernal et al. 2009). On the final day of composting, we measured low NH_4^+-N values in all treatments (49−78 mg kg^{-1}), with no significant differences among them (Figure 2. 3 C, D; Table 2.1). After temperatures fell below thermophilic values (<45°C), nitrification started, reflected by the NO_3^--N increase from day 28 until the end of composting to 907−3123 mg kg^{-1} (Figure 2. 3 E, F; Table 2.1). In a likely chain of processes, urea, the predominant organic N form, was first hydrolyzed by urease to NH_4^+, and then converted into NO_3^- by ammonia-oxidizing and nitrite-oxidizing bacteria (Jönsson et al. 2004).

Biochar had a significant impact on mineral N dynamics, demonstrated by the significant reduction of extractable NO_3^--N in final biochar composts by 32−36% compared to the controls. Joseph et al. (2017) found that nitrate and phosphate are not only retained by biochar through conventional anion exchange capacity, but also by its layer of organic substances derived from the compost material, combined with organomineral coatings and precipitates. This sorption of NO_3^- on biochar's surface suggests that co-composted biochar could decrease NO_3^- leaching during field application (Kammann et al. 2015). In terms of nitrification, the low NH_4^+-N (<400 mg kg^{-1}) values and NH_4^+-N:NO_3^--N ratios (<0.16) characterize our composts as mature and ready to be used as organic fertilizer (Bernal et al. 2009).

2.3.5 Phosphorus and K dynamics

On day 0, P_{av} and K_{av} fractions were the predominant P and K forms, with 68−92% relative to their total amounts (Table 2.1). This is expected since P and K in urine, feces and animal manure are mainly inorganic and excreted in the form of plant-available phosphate and K ions (Jönsson et al. 2004). The amount of P_{av} and K_{av} remained constant until day 87, but after 185 days, we detected on average a 5-fold decrease in P_{av} and K_{av} (Figure 2. 3 E, G; Table 2.1). At the end of composting, P_{av} and K_{av} amounted to 11−15% of the respective total P and K. This decline in P_{av} and K_{av}, and low total P and K losses (Table 2.1) reveal a strong shift to more stable P and K forms. During composting, labile organic P compounds, such as phospholipids, DNA and simple phosphate monoesters are mineralized (Turner and Leytem 2004; Wang et al. 2019). Once as phosphate, P can be associated with Ca (apatite), Mg (struvite) or with oxides and hydroxides of Al, Fe and Mn, and precipitate in insoluble form (Turner and Leytem 2004; Vandecasteele et al. 2017). The mobility and water solubility of these P forms depend on different compost characteristics, such as pH, Ca, Mg, Al, Fe and Mn contents, which could strongly reduce P solubility at a pH>8.5 (Vandecasteele et al. 2017). In the case of K, this easily exchangeable cation can be reversibly bound by the continuously increasing number of functional groups that result from the oxidation of OM during humification, as confirmed by the significant increase in CEC of the final composts.

We found that the presence of biochar neither affected final plant-available nor total P and K contents. However, the negative values of total P and K losses in the biochar-amended treatments show that biochar introduced additional P and K (Table 2.1). Since P and K losses were already low in the control composts, the presence of P and K in biochar, which typically accumulate during pyrolysis of biomass (Lehmann and Joseph 2009), becomes more evident, as proportionally more biochar is sampled at the end of the composting process due to mass reduction. These low P and K losses indicate that a well-running composting process can reduce P and K loss risk in runoff from agricultural soils and, therefore, decrease the risk of water pollution when compost is used as fertilizer.

2.3.6 Calcium, Mg and micronutrients

We did not detect a significant difference of Ca and Mg contents between biochar-amended composts and controls (Table 2.1). This indicates that the presence of these cations in the biochar's ash did not add a significant amount to the composting mixture. We measured low Ca and Mg losses ranging from -4.2−21.3%, with no clear biochar effect (Table 2.1). At an alkaline pH, these cations precipitate as carbonates (Joseph et al. 2017), making them less prone to leaching. Moreover, the strong influence of Ca and Mg on P-binding and availability, and the low solubility of their phosphates at high pH values (Jönsson et al. 2004; Vandecasteele et al. 2017) could have also reduced their loss.

Biochar addition did not have a significant effect on Zn, Cu and Fe, and did not show a clear trend for B and Mo (Table 2.1). Initial and final Mn contents were 2−2.5 times larger in composts containing biochar compared to the controls (Table 2.1). This indicates that the addition of biochar introduced a significant amount of this micronutrient (Barker and Pilbeam 2015) to the composting mixture, also confirmed by the high concentration of this trace element measured in the pure biochar (Table A.1).

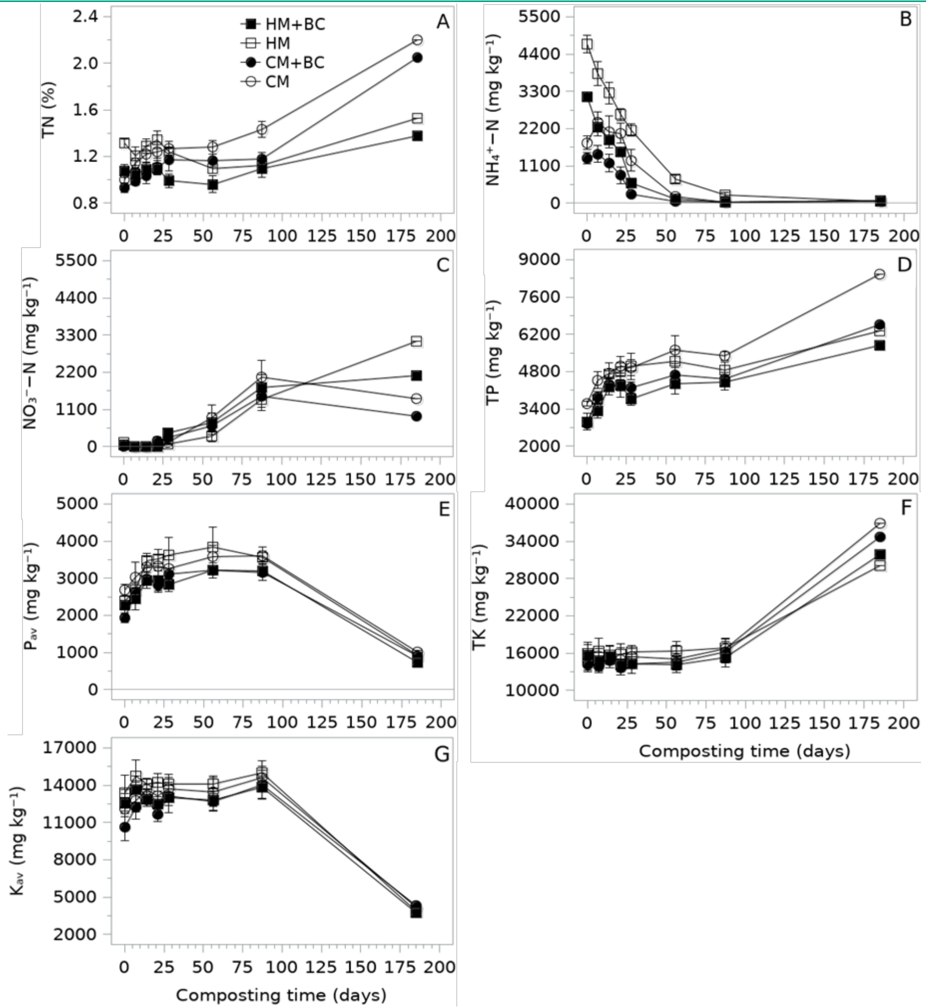


Figure 2. 3. Nutrient dynamics during the composting process. *(A) total nitrogen (TN), (B) NH_4^+-N, (C) NO_3^--N, (D) total phosphorus (TP), (E) plant-available phosphorus (P_{av}), (F) total potassium (TK), (G) plant-available potassium (K_{av}). Values are mean ± SE (n=4 for days 0−87). At day 185, SE for mean values (n=3) are not shown, since another experimental design was used for this sampling period. HM, humanure control; CM, cattle manure control; HM+BC, humanure amended with biochar; CM+BC, cattle manure amended with biochar.*

Table 2.1. *Physical and chemical parameters of the initial and final compost. Values are means (± SE) with n=4 for day 0 and n=3 for day 185. Different superscript letters indicate significant differences at $p < 0.05$ among treatments (across columns) for each day separately.*

Parameter	Treatment							
	Day 0				Day 185			
	HM	HM+BC	CM	CM+BC	HM	HM+BC	CM	CM+BC
MC (% fresh wt.)	81.4 ± 0.6 ^{b}	78.7 ± 0.7 ^{a}	79.1 ± 0.8 ^{ab}	78.6 ± 0.7 ^{a}	63.1 ± 0.2 ^{c}	59.8 ± 0.3 ^{a}	63.3 ± 0.1 ^{c}	60.6 ± 0.1 ^{b}
pH ¶	6.6 ± 0.1 ^{a}	7.3 ± 0.1 ^{b}	7.3 ± 0.1 ^{b}	7.4 ± 0.1 ^{b}	7.4 ± 0.1 ^{a}	7.9 ± 0.1 ^{b}	8.0 ± 0.1 ^{b}	8.6 ± 0.1 ^{c}
EC (mS cm^{-1})	3.95 ± 0.11 ^{c}	3.48 ± 0.05 ^{b}	3.00 ± 0.19 ^{a}	2.78 ± 0.16 ^{a}	4.49 ± 0.05 ^{c}	4.27 ± 0.07 ^{c}	3.82 ± 0.05 ^{b}	3.11 ± 0.07 ^{a}
CEC † ($mmol_c$ kg^{-1})	27.6 ± 5.3 ^{a}	28.4 ± 3.1 ^{a}	27.2 ± 2.6 ^{a}	31.9 ± 3.9 ^{a}	40.8 ± 5.9 ^{b}	32.4 ± 4.6 ^{a}	48.6 ± 6.7 ^{c}	38.5 ± 5.5 ^{ab}
Germination index (%)	–	–	–	–	127 ± 20 ^{a}	117 ± 16 ^{a}	172 ± 23 ^{a}	160 ± 30 ^{a}
TOM (%)	83.2 ± 1.4 ^{b}	80.4 ± 1.5 ^{ab}	77.0 ± 1.6 ^{a}	77.6 ± 1.2 ^{a}	70.9 ± 2.2 ^{b}	68.8 ± 2.4 ^{ab}	62.6 ± 1.3 ^{ab}	60.6 ± 1.2 ^{a}
TOC (%)	41.8 ± 0.7 ^{a}	49.8 ± 0.5 ^{c}	40.1 ± 1.1 ^{a}	45.8 ± 1.0 ^{b}	38.3 ± 2.0 ^{a}	51.9 ± 1.7 ^{c}	32.1 ± 1.1 ^{a}	45.0 ± 0.7 ^{b}
TN (%)	1.3 ± 0.1 ^{b}	1.1 ± 0.1 ^{a}	1.0 ± 0.1 ^{a}	1.0 ± 0.1 ^{a}	1.5 ± 0.1 ^{a}	1.4 ± 0.1 ^{a}	2.2 ± 0.1 ^{b}	2.1 ± 0.2 ^{b}
C:N ratio	32.3 ± 0.9 ^{a}	50.2 ± 1.6 ^{b}	41.4 ± 3.5 ^{ab}	49.8 ± 3.5 ^{b}	25.1 ± 1.8 ^{b}	37.5 ± 1.6 ^{c}	14.6 ± 1.0 ^{a}	22.2 ± 1.9 ^{b}
NH_4^+-N (mg kg^{-1})	4,695 ± 264 ^{c}	3,144 ± 143 ^{b}	1,782 ± 198 ^{a}	1,329 ± 168 ^{a}	52 ± 8 ^{a}	75 ± 16 ^{a}	78 ± 14 ^{a}	49 ± 4 ^{a}
NO_3^--N (mg kg^{-1})	117 ± 78 ^{b}	46 ± 29 ^{ab}	7 ± 4 ^{ab}	41 ± 29 ^{a}	3,123 ± 34 ^{d}	2,112 ± 50 ^{c}	1,418 ± 96 ^{b}	907 ± 140 ^{a}
P_{av} (mg kg^{-1})	2,675 ± 113 ^{a}	2,277 ± 177 ^{a}	2,675 ± 108 ^{a}	1,938 ± 90 ^{a}	925 ± 77 ^{ab}	732 ± 56 ^{a}	1,009 ± 46 ^{b}	896 ± 14 ^{ab}
K_{av} (mg kg^{-1})	13,365 ± 1,852 ^{a}	12,553 ± 1,737 ^{a}	12,123 ± 1,846 ^{a}	10,638 ± 2,418 ^{a}	3,857 ± 259 ^{a}	3,780 ± 251 ^{a}	4,177 ± 37 ^{a}	4,350 ± 255 ^{a}
TP (mg kg^{-1})	2,920 ± 146 ^{a}	2,920 ± 308 ^{a}	3,600 ± 82 ^{a}	2,855 ± 90 ^{a}	6,333 ± 33 ^{a}	5,800 ± 351 ^{a}	8,467 ± 289 ^{b}	6,567 ± 67 ^{a}
TK (mg kg^{-1})	15,922 ± 1,852 ^{a}	15,650 ± 1,732 ^{a}	14,575 ± 1,360 ^{a}	14,150 ± 1,100 ^{a}	30,033 ± 1,317 ^{a}	31,900 ± 1,453 ^{ab}	36,967 ± 579 ^{b}	34,667 ± 1,202 ^{ab}
TCa (mg kg^{-1})	11,800 ± 750 ^{a}	14,600 ± 1,307 ^{a}	16,000 ± 1,732 ^{a}	21,100 ± 1,437 ^{b}	21,400 ± 933 ^{a}	21,500 ± 1,234 ^{a}	36,570 ± 536 ^{b}	33,200 ± 690 ^{b}
TMg (mg kg^{-1})	1,820 ± 94 ^{a}	2,160 ± 88 ^{ab}	2,330 ± 191 ^{b}	2,540 ± 89 ^{b}	3,280 ± 110 ^{a}	3,220 ± 146 ^{a}	4,603 ± 18 ^{b}	4,347 ± 35 ^{b}
TOM loss ‡ (%)	–	–	–	–	61.0 ± 0.8 ^{b}	50.1 ± 1.3 ^{a}	68.6 ± 1.0 ^{c}	52.8 ± 1.0 ^{a}
TOC loss ‡ (%)	–	–	–	–	58.2 ± 0.8 ^{b}	39.3 ± 0.5 ^{a}	69.1 ± 1.0 ^{c}	40.4 ± 1.7 ^{a}
TN loss ‡ (%)	–	–	–	–	46.6 ± 1.4 ^{c}	23.9 ± 5.2 ^{b}	14.1 ± 5.7 ^{b}	-26.9 ± 5.4 ^{a}

Table 2.1. *Continuation*

Parameter	Treatment							
	Day 0				Day 185			
	HM	HM+BC	CM	CM+BC	HM	HM+BC	CM	CM+BC
TP loss ‡ (%)	–	–	–	–	0.1 ± 5.2 ^{b}	-19.7 ± 12.6 ^{ab}	9.2 ± 2.1 ^{b}	-39.5 ± 4.7 ^{a}
TK loss ‡ (%)	–	–	–	–	10.6 ± 9.2 ^{b}	-23.1 ± 13.1 ^{ab}	-0.5 ± 9.9 ^{b}	-51.1 ± 13.4 ^{a}
TCa loss ‡ (%)	–	–	–	–	18.3 ± 5.4 ^{a}	10.1 ± 11.5 ^{a}	6.5 ± 12.3 ^{a}	2.0 ± 10.4 ^{a}
TMg loss ‡ (%)	–	–	–	–	17.8 ± 4.1 ^{ab}	12.3 ± 4.8 ^{ab}	21.3 ± 9.0 ^{b}	-4.2 ± 5.1 ^{a}
Zn (mg kg^{-1})	59.9 ± 4.0 ^{ab}	52.5 ± 1.8 ^{a}	63.0 ± 4.5 ^{ab}	65.6 ± 1.9 ^{b}	94.9 ± 3.1 ^{a}	90.0 ± 5.8 ^{a}	123.0 ± 0.6 ^{a}	154.7 ± 29.8 ^{a}
Cu (mg kg^{-1})	9.4 ± 0.7 ^{a}	9.5 ± 0.4 ^{a}	11.2 ± 0.8 ^{a}	11.9 ± 1.1 ^{a}	12.7 ± 0.9 ^{ab}	12.0 ± 0.6 ^{a}	15.3 ± 0.3 ^{bc}	16.7 ± 0.7 ^{c}
Fe (mg kg^{-1})	2,260 ± 205 ^{ab}	1,700 ± 168 ^{a}	2,390 ± 215 ^{ab}	2,900 ± 317 ^{b}	5,250 ± 293 ^{a}	4,030 ± 209 ^{a}	5,980 ± 135 ^{a}	8,900 ± 2292 ^{a}
Mn (mg kg^{-1})	144 ± 22 ^{a}	360 ± 36 ^{b}	224 ± 30 ^{a}	426 ± 49 ^{b}	236 ± 12 ^{a}	621 ± 29 ^{c}	357 ± 3 ^{b}	800 ± 8 ^{d}
B (mg kg^{-1})	9.8 ± 1.0 ^{a}	11.7 ± 1.3 ^{a}	10.3 ± 2.3 ^{a}	11.7 ± 0.8 ^{a}	17.5 ± 1.0 ^{a}	15.3 ± 0.8 ^{a}	29.0 ± 0.4 ^{c}	21.3 ± 0.7 ^{b}
Mo (mg kg^{-1})	2.7 ± 0.3 ^{ab}	2.2 ± 0.1 ^{a}	3.3 ± 0.6 ^{b}	2.5 ± 0.3 ^{ab}	3.1 ± 0.3 ^{a}	2.2 ± 0.4 ^{a}	4.4 ± 0.2 ^{b}	< 2.0 ^{a}

HM, humanure control; CM, cattle manure control; HM+BC, humanure amended with biochar; CM+BC, cattle manure amended with biochar; EC, electrical conductivity; CEC, cation exchange capacity; –, not analyzed; TOM, total organic matter; TOC, total organic carbon; TN, total nitrogen; P_{av}, plant-available phosphorus; K_{av}, plant-available potassium; TP, total phosphorus; TK, total potassium; TCa, total calcium; TMg, total magnesium.

[¶] Measured in distilled water.

[‡] CEC final values correspond to day 87 and with n=4

2.4 Conclusions

With appropriate technology, using sawdust as adsorber material for dry toilets, compost boxes and a suitable composting mixture, a thermophilic composting process with low nutrient losses could be implemented, and a pathogen-free, mature and nutrient-rich fertilizer free of phytotoxic substances could be produced. We report the first thermophilic composting of human excreta with biochar addition and its nutrient dynamics. We found that using biochar as compost amendment decreased extractable NO_3^--N contents by 32−36% in the final composts. That biochar can capture nitrate when co-composted with nutrient-rich organic materials has important implications for decreasing N losses via leaching from agricultural soil after field application. Our findings show that EcoSan via thermophilic composting of humanure represents a promising way for recycling nutrients and eliminating pathogens from human excreta. It could therefore be a crucial component for improving public health, food security and water quality, particularly in areas suffering from financial and infrastructure limitations. Moreover, the increased amount of relatively recalcitrant C in compost and especially in biochar-compost has the potential of increasing soil C storage, with promising implications for climate change mitigation and adaptation. However, the long-term stability of soil organic carbon from fields amended with biochar-compost requires further investigation.

3

BIOCHAR ADDITION REDUCES NON-CO_2 GREENHOUSE GAS EMISSIONS DURING COMPOSTING OF HUMAN EXCRETA AND CATTLE MANURE

Based on:

Castro-Herrera, D., Prost, K., Kim, D-G., Yimer, F., Tadesse, M., Gebrehiwot, M., Brüggemann, N. Biochar addition reduces non-CO_2 greenhouse gas emissions during composting of human excreta and cattle manure. *Manuscript in preparation.*

Abstract

Sustainable management of human excreta with appropriate technology could be a promising approach for addressing public health, environmental pollution and climate change. This could be even more relevant in regions, where dumping of organic waste and excreta is common practice. Instead, ecological sanitation combined with thermophilic composting could represent a feasible way to transform human excreta into a stabilized, pathogen-free and nutrient-rich fertilizer. Additionally, if combined with suitable bulking materials, such as sawdust and straw, and additives, such as biochar, it could be a suitable waste management strategy for reducing greenhouse gas (GHG) emissions. During a 143-day thermophilic composting, we measured GHG (CO_2, CH_4, N_2O) emissions with a portable FTIR gas analyzer and sampled NH_3 as ammonium in a H_2SO_4 trap. Thermophilic composting of human excreta and cattle manure together with teff straw, organic waste and biochar was conducted in wooden boxes. In this study, we investigated the effect of biochar as compost amendment on GHG and NH_3 emissions. We found that, in the course of composting, biochar addition significantly reduced N_2O emissions by 56−57% and CH_4 emissions by 65−91%. Overall, non-CO_2 GHG emissions were reduced by 51−71%. In contrast, we did not observe a significant effect of biochar addition on CO_2 and NH_3 emissions. Previous data already showed that, when applying this composting method, it is possible to hygienize human fecal material. Our data, therefore suggest that thermophilic composting with biochar addition and appropriate technology represents a safe and cost-effective waste management practice for producing a humus-rich fertilizer and reducing GHG emissions.

3.1 Introduction

Every day, humans excrete tons of feces and urine, and it is estimated that the global annual production of human feces will exceed 1×10^{12} kg of fresh fecal biomass per year by the year 2030 (Berendes et al. 2018). A significant part of these excreta ends up in streets, drains, and waterbodies (Somorin 2020), as an estimated amount of 4.5 billion people worldwide live without access to safe sanitation facilities, and either defecate in the open or in toilets or pit latrines where excreta are disposed of in-situ (WHO 2019a, 2019b). The discharge of this untreated organic material into the environment poses a serious public health risk that results in the death of an estimated 432,000 people every year (WHO 2019b). Moreover, discharging untreated human excreta in the environment and the use of on-site technologies, such as septic tanks and pit latrines, are a significant source of greenhouse gas (GHG) emissions. Uncollected human waste accounts for approximately 80% of the global N_2O emissions originating from human waste (Strokal and Kroeze 2014), and pit latrines alone for approximately 1–2% of global anthropogenic CH_4 emissions (Reid et al. 2014). Meanwhile, as nutrients and gases from this organic material accumulate in our ecosystems, regions with the lowest sanitation coverage, such as Sub-Saharan Africa, experience limited access to fertilizers (Reid, 2020) and the most severe land degradation in the world (Nkonya, Mirzabaev, & Braun, 2016).

Human excreta are an organic material rich in plant macro- and micronutrients that could help millions of farmers to restore humus content and soil fertility of degraded soils and thereby boost agricultural productivity. By this means, additionally, the dependence on non-renewable synthetic fertilizers could be reduced (Akram et al. 2019; Berendes et al. 2018; Jenkins 2005; Trimmer and Guest 2018). Ecological sanitation via thermophilic composting, i.e., reaching temperatures >45 °C for a time that allows hygienization, may be a feasible way to transform human excreta into a hygienized and stabilized nutrient-rich compost and soil amendment (Bernal et al. 2009; Jenkins 2005; McNicol et al. 2020; Ryals et al. 2019; Somorin 2020). Furthermore, this integrated sanitation approach may also build resilience towards climate change through soil humus and thus carbon sequestration in agricultural ecosystems and, at the same time, mitigate climate change by reducing GHG emissions in sanitation and waste management.

In contrast to conventional wastewater treatment and waste management processes, composting can be conducted with low operation costs and does not necessarily depend on an external supply of water, energy, infrastructure and technology (Reid 2020; Somorin 2020). It could therefore be a feasible and environmentally friendly sanitation and waste management approach, especially for developing countries. Since composting is an aerobic degradation process, it provides a potential way for treating organic waste with significantly lower GHG emissions than existing treatment and disposal methods (McNicol et al. 2020; Ryals et al. 2019). McNicol et al. (2020) estimated that at a global scale, the implementation of human waste composting for the one billion people, who live in slums, could mitigate 13–44% of CH_4 emissions from the sanitation sector. Nevertheless, anaerobic sites inside a compost pile created by inappropriate composting management conditions can lead to substantial CH_4 and N_2O emissions (Brown et al. 2008; Rynk et al. 1992). Moreover, without a sufficient supply of easily degradable carbon at the beginning of composting, a significant fraction of the initial N contained in N-rich organic materials can be lost as NH_3 gas, thereby reducing the agronomic value of the final compost (Bernal et al. 2009; Eghball et al. 1997).

It is thus that the use of organic waste rich in easily degradable carbon, like e.g. vegetable scraps, as an additional feedstock for composting could reduce NH_3 emissions (Gottschall, 1992). Moreover, the application of appropriate bulking materials, such as sawdust and straw could enhance pile aeration and thereby reduce emissions of CH_4 and N_2O, which evolve due to anaerobic conditions (Bernal et al. 2009; Chowdhury et al. 2014; Czekała et al. 2015; Sommer and Møller 2000). Biochar, a carbon-rich material produced by thermochemical conversion of biomass under limited supply of O_2 and with unique properties, such as recalcitrance, high surface area, microporosity, and sorption capacity, has also been reported to improve aeration conditions (He et al. 2017; Lehmann and Joseph 2009; Liu et al. 2017b), and thereby to reduce CH_4 (Chen et al., 2017; Jia, Wang, Wenqiao, Xiaotang, & Baozhu, 2016; Sonoki et al., 2013), N_2O (Li et al., 2016; Wang et al., 2013; Wang et al., 2018), and NH_3 gas emissions (Agyarko-Mintah et al. 2017; Chen et al. 2017; Janczak et al. 2017; Steiner et al. 2010).

To our knowledge, no studies that measure NH_3 and GHG emissions during thermophilic composting of human excreta together with biochar exist to date. Our aim was to quantify CO_2, CH_4, N_2O, and NH_3 emissions from the thermophilic composting of human excreta together with sawdust, straw and organic waste, and for comparison that of cattle manure, to evaluate the effect of biochar on these gas emissions when used as an amendment during composting.

3.2 Materials and methods

3.2.1 Biochar production

As described in chapter 2 (section 2.2.1), we produced biochar from Eucalyptus wood (*Eucalyptus camaldulensis*) with a top-lit up-draft micro-gasifier to produce biochar. Before adding to the compost mixture, we crushed the biochar pieces and sieved them to <14 mm, with 85% of the particles in the range of 2−12 mm, 9% between 1−2 mm, 3% between 0.5−1 mm, and 3% below 0.5 mm.

The biochar was analyzed according to the requirements of the European Biochar Certificate (EBC 2012) by Ruhr Lab GmbH (Gelsenkirchen, Germany). Physical and chemical properties of the biochar show that it fulfills the "premium" requirements of the European Biochar Certificate, except for the polycyclic aromatic hydrocarbons concentration, which exceeds the premium threshold value, but complies with the "basic" criteria (chapter 2, section 2.2.1).

3.2.2 Thermophilic composting and experimental design

Described in detail in chapter 2, section 2.2.2.

3.2.3 Greenhouse gases and ammonia flux measurements

We measured fluxes of CO_2, CH_4, N_2O, and NH_3 with a cylindrical static chamber. The chambers consisted of a collar constructed from a 25-cm diameter and 30-cm length PVC pipe, and as chamber top, a PVC lid attached to a flexible rubber coupling with a 25-cm diameter and a 16.5-cm height was used (Appendix B: Figure B.1 A). The chamber lid was equipped with two stainless steel tube fittings (6 mm) used as sampling ports. A one-meter long flexible inert tubing with an inner diameter of 4 mm was connected to a gas-tight stainless steel tube fitting (¼ in) as vent tube to minimize pressure perturbations during chamber deployment. An additional hole was drilled to fit a plastic cable gland with an adjustable diameter for measuring temperature during gas flux measurements. On day 0, collars were inserted 12 cm deep in the center of each pile, right after the compost box was filled with fresh compost material. For gas sampling, the lid was placed on the collar by tightening the bottom stainless steel band of the flexible rubber coupling to the collar. The internal part of the chamber had a height of 27 cm, an area of 0.05 m^2 and a volume of 0.0135 m^3.

We carried out GHG flux measurements on days 0, 1, 2, 4, 6, 9, 13, 20, 27, 34, 41, 48, 55, 62, 69, 76, 82, 87, 101, 115, 129 and 143 with a portable FTIR gas analyzer ((Appendix B: Figure B.1 B); DX4015, Gasmet Technologies Oy, Vantaa, Finland) with a 10-min deployment time. We measured the initial and final internal chamber temperatures with a digital thermometer (GMG 285, Pt 1000, Greisinger, Germany).

We expressed the GHG fluxes as grams of CO_2-C, milligrams of CH_4-C, or milligrams of N_2O-N per square meter per day and calculated them by using: Equation 3.1.

$$\mathrm{Flux}_{\mathrm{GHG}} = \frac{S}{1000000} \times \frac{P}{R\,T} \times \frac{V}{A} \times M \times 60 \times 24 \qquad \text{(Equation 3.1)}$$

where S is the slope of the linear equation fitted to the change of the gas concentrations in the chamber during the measurements (ppm min^{-1}); 1,000,000 is used to convert ppm ($\mu L\ L^{-1}$) into L L^{-1}; P is the air pressure (atm) at 1,700 m; R the universal ideal gas constant (L atm mol^{-1} K^{-1}); T the average temperature (K) of the chamber during deployment time; M the molar mass of C or N (g mol^{-1}); and 60 and 24 the values used to convert time from min to day. Concentration values obtained during the first 2 min were not used for calculating the slope due to fluctuations produced by pressure disturbances during chamber closure. Fluxes with linear fits of $R^2 < 0.81$ were considered below the detection limit and therefore, reported as zero.

We estimated cumulative emissions by calculating the emission average rate of two consecutive measurement days and multiplying it by the time between these two measurements. Thereafter, we summed up all the resulting values and expressed the cumulative fluxes as g CO_2-C m^{-2}, mg CH_4-C m^{-2}, and mg N_2O-N m^{-2}. We also expressed cumulative GHG emissions as percent of initial C and N to compare our results with other studies.

We estimated total GHG emissions in terms of g CO_2-C equivalents (CO_2-C_{eq}) m^{-2} by using a 100-year global warming potential time horizon with climate-carbon feedback. We used conversion factors of 12.36 kg CO_2-C_{eq} per kg CH_4-C, and 127.71 kg CO_2-C_{eq} per kg N_2O-N according to (IPCC 2013)
For NH_3, we carried out gas sampling on days 1, 2, 3, 5, 7, 10, 14, 20, 27, 34, 41, 48, 55, 62, 69, 76, 82, 87, 101 and 115 for one hour, and we measured the initial and final internal chamber temperatures. The NH_3 trapping system consisted of the same static chamber used for GHG measurements, a bubbler containing 150 mL of 0.01 mol L^{-1} H_2SO_4 solution to trap the emitted NH_3, a micro-pump (KNF, NMP 03 KP DC-L, Germany) to pull the air inside the chamber through the system, and a rotameter (Brooks Instruments, model 1250, USA) to regulate the flow rate at 150 mL min^{-1} (Appendix B: Figure B.1 C, D). The air was directed into the acid trap with PTFE tubes with 6 mm diameter to trap NH_3 as ammonium in the acid solution. Thereafter, we determined ammonium concentration with the indophenol blue method and a spectrophotometer (WTW photoLab 7100 VIS, Germany) according to FAO (2008) and to VDLUFA (2014).

We expressed NH_3 fluxes as milligrams of NH_3-N per square meter per day and calculated them by using Equation 3.2:

$$\text{Flux}_{NH_3\text{-}N} = \frac{C \times 150}{R} \times \frac{1}{A} \times 24 \qquad \text{(Equation 3.2)}$$

where C is the concentration of NH_4^+-N measured in the acid trap solution with the spectrometer (mg L^{-1}); 150 the volume of the 0.01 mol L^{-1} H_2SO_4 solution in the trap (mL); R the recovery (divided by 100) of the trapping system at the average ambient temperature during the collection of the sample (%); A the area covered by the chamber (m^2); and 24 the value used to obtain the emissions per day.

For calculating the percentage of NH_4^+-N recovered with this trapping system, we flushed, for one hour, ammonia coming from a cylinder (215.9 ppm ± 2%) through the static chamber and bubbled it subsequently into the trap containing 150 mL of the 0.01 mol L^{-1} H_2SO_4 solution at a flow rate of 150 mL min^{-1}. We repeated this three times at four temperatures of the acid solution of 15, 20, 25, and 30°C. According to the amount, in mg, of expected NH_4^+-N to obtain after one hour of flushing at a flow of 150 mL min^{-1}, we calculated the percentages of recovery at each temperature and plotted them against temperature. The function obtained, allowed us to calculate a recovery for each ammonia-sampling event, specific for the ambient temperature during the hour of collection. When ammonium concentrations were below the detection limit, a correction factor according to the recovery was not possible, and therefore, fluxes were set to zero. We calculated cumulative NH_3-N fluxes in the same way as for GHG and expressed them as mg m^{-2} and % of initial N.

3.2.4 Statistical analysis

Data are reported as mean values ±SE and by considering a statistical significance level of $\alpha=0.05$. For GHG and NH_3 fluxes, we conducted a one-way repeated-measures ANOVA by using a mixed model with the Kenward-Roger method. Gas fluxes are represented as mean values of four replicates (n=4). Statistical analyses were conducted with the SAS software University Edition (SAS Studio, version 5.1).

3.3 Results and Discussion

Results of temperature measurements (together with pH, moisture, C/N ratio, cation exchange capacity, and germination index), reported in chapter 2, showed a well-functioning composting process with distinct temperature phases typical for thermophilic composting. In detail, mesophilic phase I lasted from day 0–1, the thermophilic phase from day 1–18, mesophilic phase II from day 19–60, and the maturation phase from day 60−143. During the thermophilic phase, maximum average temperatures of 65.1°C, 64.2°C, 62.1°C, and 65.1°C, were reached for HM, HM+BC, CM, and CM+BC, respectively (chapter 2: section 2.3.2.1).

3.3.1 Gas fluxes during composting

3.3.1.1 CO2 emissions

During the mesophilic I (<45°C) and the thermophilic (≥45°C) phase, CO_2 emissions sharply increased to maximum values ranging from 326−542 g CO_2-C m^{-2} d^{-1} (Figure 3.1 A) due to the rapid degradation of easily decomposable organic compounds (Keener et al. 2000). After the peak in respiration on day 1 and 2, CO_2 fluxes declined exponentially, while temperatures continuously increased until reaching peak values of 62.1−65.9°C on day 4 for HM, day 3 for HM+BC, day 5 for CM, and day 6 for CM+BC (chapter 2: section 2.3.2.1). During the last part of the mesophilic II (45°C−ambient temperature) and entire maturation phase (ambient temperature), CO_2 fluxes ranged from 4.1−13.3 g CO_2-C m^{-2} d^{-1}, indicating low microbial respirations and thus a depletion of easily degradable organic substrates together with that OM stabilization and humification (Bernal et al. 2009).

3.3.1.2 CH_4 emissions

Similar to CO_2 emissions, CH_4 production from all treatments also rapidly peaked at day 1 to 420−978 mg CH_4-C m^{-2} d^{-1} (Figure 3.1 B). At the beginning of composting, maximum degradation of OM with high O_2 consumption by microorganisms occurs, decreasing O_2 availability inside the pile

(Rynk et al. 1992). Low O_2 concentrations create anoxic conditions that favor anaerobic degradation of organic compounds, such as acetate and CO_2 by methanogenic organisms, that lead to CH_4 production (Lehmann and Joseph 2009; Sonoki et al. 2013). This initial CH_4 peak was immediately followed by a rapid decrease of these emissions for HM, HM+BC, and CM+BC treatments, while for CM treatments, CH_4 emissions larger than 400 mg CH_4-C m^{-2} d^{-1} were measured for three consecutive weeks. Methane emissions decreased to non-detectable fluxes from day 41−48 until the end of composting, as O_2 consumption rate and pile moisture declined, favoring an aerobic environment and CH_4 oxidation by methanotrophs (Rynk et al. 1992; Sonoki et al. 2013).

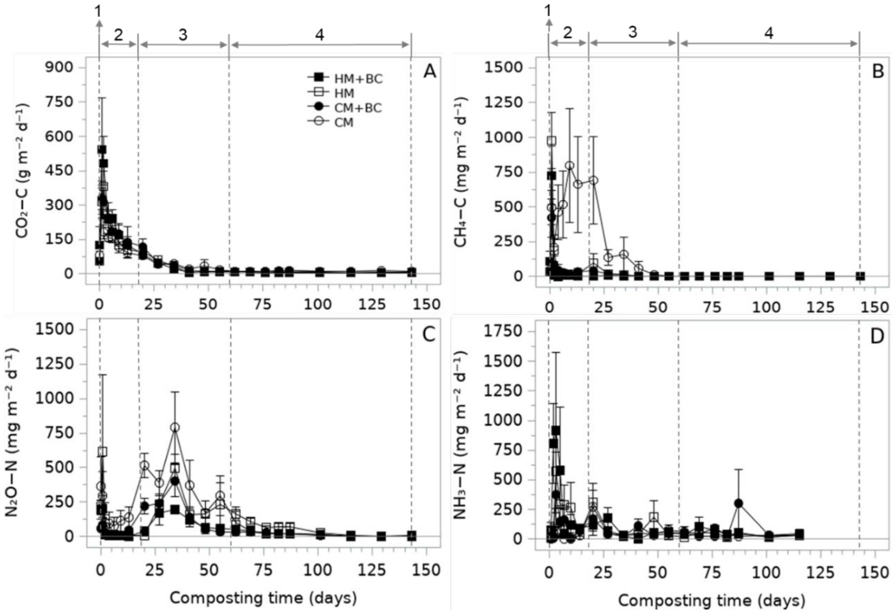


Figure 3.1. ***GHG and NH₃ emission rate dynamics during the composting process.*** *(A) CO_2-C, (B) CH_4-C, (C) N_2O-N, (D) NH_3-N. Values are mean ± SE (n=4). HM, humanure control; CM, cattle manure control; HM+BC humanure amended with biochar; CM+BC, cattle manure amended with biochar. 1, mesophilic I phase; 2, thermophilic phase; 3, mesophilic II phase; 4, maturation phase.*

3.3.1.3 N_2O emissions

Nitrous oxide emissions exhibited a bimodal pattern (Figure 3.1 C) with an initial increase on day 1, followed by a decline as pile temperatures continuously increased. After temperatures fell below thermophilic values on day 34, the largest N_2O emissions of 198−789 mg N_2O-N m^{-2} d^{-1} were measured. This phenomenon is also described by Hellmann et al. (1997). They attribute the finding to the observation that nitrifying and denitrifying microorganisms are generally not thermophilic and that there is thus only minor production of N_2O during the thermophilic phase, which mainly evolves from cooler parts of the compost pile.

We explain the enhanced N_2O production right after the thermophilic phase with the lower O_2 availability inside the compost pile, which could have favored bacterial denitrification (Strokal and Kroeze 2014). This biological process is performed either by nitrifying bacteria, which use NO_2^- as an alternative electron acceptor when O_2 is limited or/and by heterotrophic denitrifying aerobic bacteria, that reduce NO_3^- or NO_2^- to N_2, producing N_2O as an intermediate product (Lehmann and Joseph 2009). In our study, the occurrence of nitrification was suggested by a constant NH_4^+ decline and NO_3^- increase (chapter 2: Figure 2. 3). The assumption of a sequential occurrence of denitrification and then of nitrification activity was supported by the increase in pH from 6.6−7.4 to maximum values of 8.5−9.0, and a subsequent decrease after day 28 to pH values below 8.3 (chapter 2: Figure 2.2), as denitrification consumes protons, while nitrification produces them. During the maturation phase, N_2O emissions remained below 100 mg N_2O-N m^{-2} d^{-1} after day 48, 69 and 101, from both biochar-amended treatments, CM, and HM, respectively.

3.3.1.4 NH_3 emissions

We measured the highest NH_3 emissions during the first five days of composting, with mean values of 143−918 mg NH_3-N m^{-2} d^{-1} (Figure 3.1D). The main factor promoting NH_3 emissions is a combination of high NH_4^+ concentrations, elevated temperatures, and pH >7.5 (Bernal et al. 2009; Rynk et al. 1992). The observed NH_3 peak can be thus explained by both the deprotonation of NH_4^+ to NH_3 at high NH_4^+ content and enhanced volatilization of NH_3 at elevated temperature and pH >7.5 during the thermophilic phase (chapter 2: Figure 2.2). After the passing peak temperatures on days 3−6, NH_3 emission rates declined below 100 mg NH_3-N m^{-2} d^{-1} and remained relatively stable.

3.3.2 Cumulative gas emissions

3.3.2.1 Cumulative CO_2 emissions

Cumulative CO_2 emissions from all treatments ranged from 4916−5736 g CO_2-C m^{-2}, with no significant difference among them (Table 3.1). Nonetheless, we observed a tendency towards higher values for composts containing biochar, which suggests that biochar addition may stimulate OM decomposition during composting. We attribute this to biochar's high porosity and surface area potential to provide better O_2 availability and distribution through the compost mixture (He et al. 2017; Lehmann and Joseph 2009; Liu et al. 2017b). In addition, biochar adsorbs water, soluble organic compounds, gases and nutrients, and serves as microhabitat for microorganisms inside its complex porous structure, thereby providing better conditions for their metabolic processes (Lehmann and Joseph 2009).

In our study, 18−25% of the total initial C of the compost was lost as CO_2 in humanure treatments (Table 3.1). McNicol et al. (2020) report a similar loss of 21−24% of the initial C as CO_2 during 6−8 months of composting of human fecal material from urine-diverting dry toilets together with bagasse

(bagasse:feces ratio of 2:1), with a start of turning after 8 weeks (Table 3.2). These results are slightly larger than the ones for our HM+BC treatments and slightly lower than for our HM treatments, which can be explained by the recalcitrant nature of the biochar, resulting in larger carbon content of the final compost with biochar (chapter 2: Figure 2.2, Table 2.1).

Cumulative CO_2-C emissions from cattle manure treatments from our study comprising 20−23% of initial C (Table 3.1) are within the range of results reported by different studies for static, i.e. without turning, (22−29%) and active composting, i.e. with turning, (17−49%) of cattle manure (Table 3.2).

Table 3.1. *Cumulative emissions of GHG (143 days) and NH_3 (115 days) expressed as mass per m^2, and % of initial C or N. Global warming potential of combined GHG emissions, expressed as CO_2-C_{eq} over a 100-year time horizon with climate-carbon feedback (calculation see section 2.3). Different superscript letters indicate significant differences at $p < 0.05$ among treatments.*

Parameter	Treatment			
	HM	HM+BC	CM	CM+BC
Cumulative emissions [†]				
CO_2-C (g m^{-2})	4,916 ± 709 ^{a}	5,407 ± 1,068 ^{a}	4,972 ± 640 ^{a}	5,736 ± 1,089 ^{a}
CH_4-C (mg m^{-2})	2,434 ± 546 ^{a}	858 ± 317 ^{a}	16,968 ± 5,680 ^{b}	1,500 ± 253 ^{a}
N_2O-N (mg m^{-2})	15,466 ± 4,026 ^{b}	6,757 ± 1,370 ^{a}	22,664 ± 5,761 ^{c}	9,719 ± 3,036 ^{a}
NH_3-N (mg m^{-2})	9,651 ± 1,844 ^{a}	9,881 ± 1,648 ^{a}	6,309 ± 1,525 ^{a}	8,646 ± 2,643 ^{a}
% of initial C				
CO_2-C	25	18	23	20
CH_4-C	0.012	0.003	0.080	0.005
% of initial N				
N_2O-N	2.3	1.1	4.7	1.3
NH_3-N	1.5	1.6	1.2	1.4
CO_2-C_{eq} (g m^{-2})				
CH_4-C	30	11	210	19
N_2O-N	1865	907	3231	971
Total [†]	1,895 ± 518 ^{b}	918 ± 174 ^{a}	3,441 ± 711 ^{c}	990 ± 389 ^{a}
Total + CO_2-C [†]	6,921 ± 1,089 ^{a}	6,280 ± 1,039 ^{a}	8,076 ± 778 ^{a}	6,995 ± 1,112 ^{a}
% of total GHG emissions				
CO_2-C	71.0	86.1	61.6	82.0
CH_4-C	0.4	0.2	2.6	0.3
N_2O-N	28.6	13.7	35.8	17.7

HM, humanure control; CM, cattle manure control; HM+BC, humanure amended with biochar; CM+BC, cattle manure amended with biochar.

[†] Values provided as means (± SE) with n=4.

A comparison from our findings with those from Hao et al. (2001) and Ahn et al. (2011) for active composting (Table 3.2) confirms former observations that turning is rebuilding the compost pile's porosity, which usually decreases during the composting process due to decomposition and settling of the pile. Therefore, it improves passive air exchange. Additionally, turning exchanges material at the pile's surface with material from the interior thereby exposing all material equally to the different decomposition conditions in the inside and surface of the pile (Rynk et al. 1992). By this means turning accelerates the composting process, which is reflected by higher microbial activity and CO_2 emissions (Peigné and Girardin 2004; Tiquia et al. 1996). Hence, frequently turned piles showed higher emissions than the ones from our study, which were only turned once after 101 days. In addition to more frequent turning, Hao et al. (2001) used wheat straw as a bulking agent, which consists of strong stems with a larger diameter compared to teff straw (used in our study), thereby ensuring aeration of the compost piles along the whole composting process.

3.3.2.2 Cumulative CH_4 emissions

Methane emissions predominantly occurred during the thermophilic phase of composting and shortly afterwards (Figure 3.1). The most important factor determining CH_4 production is the intensity of anaerobic conditions within the compost pile. The evolvement of anaerobic sites in turn is influenced by height and shape of the pile, moisture content, turning frequency and bulking material (Peigné and Girardin, 2003). Biochar addition significantly reduced cumulative CH_4 emissions by 91% in cattle manure treatments. A similar, though not statistically significant, trend could be observed for humanure treatments, with HM+BC showing 65% reduced cumulative CH_4 emissions compared to its control (HM; Table 3.1). We attribute this finding to improved aeration conditions in the biochar amended composts, caused by the biochar's highly porous structure, which could have favored aerobic conditions and increased CH_4 diffusion and availability for methanotrophs (Lehmann and Joseph 2009). This influence of biochar on CH_4 metabolism was also observed by Sonoki et al. (2013) during composting of cattle manure. They quantified the level of *mcrA* encoding methyl coenzyme M reductase and *pmoA* encoding methane monooxygenase and found a 2-fold decrease of methanogens (*mcrA*) and a 3-fold increase of methanotrophs (*pmoA*) in the mixture containing biochar during the thermophilic phase of composting. We measured higher CH_4 emissions from CM when compared to HM, which we attribute to the larger amount of methanogens in the digestive tract of ruminant animals, and therefore, in their manure (Kim et al. 2014).

We found that C lost as CH_4 from the humanure treatments was 0.003−0.012% of the total initial C in the compost piles (Table 3.1). These CH_4 emissions were 67 times lower than the mean cumulative CH_4-C emissions (0.5% of initial C) estimated by McNicol et al. (2020) during composting of human feces and bagasse. The first phase of their composting process consisted of an 8-week composting phase without turning inside compost boxes of wooden boards with spaces in between. These boxes were similar to the ones used in our study but with a significantly larger in size and volume (Table 3.2). Our smaller initial pile volume of 1.5 m x 1.5 m x 1.1 m, i.e. 2.5 m^3, may have thus allowed a better air diffusion into the composting substrate, as usually, a distance of more than 1 m from the pile surface and sides to the center of the pile will hinder air diffusion inside to the center (Dunst 2015).

Cumulative CH_4-C emissions from the cattle manure control treatment in our study comprising 0.08% of initial C (Table 3.1) are in the lower range of results reported by different studies for static (0.03−1.8%) and active (0.1−2.4%) composting of cattle manure (Table 3.2). Lower CH_4-C emissions than those in the cattle manure control treatment in our study were reported by Sommer (2001), who measured a 0.03% loss of CH_4-C of initial C for compacted as well as for cut and mixed static windrows (Table 3.2). Even though a larger mass of cattle manure was piled up for composting in the study from Sommer (2001) compared to the one in our study, we attribute their lower CH_4 emissions to the lower initial moisture content of 62−64% of their composting piles and their use of wheat straw as a bulking agent. Their composting process therefore should have ensured a better aeration compared to the one in our study (79−81% and 60−63% moisture content at the beginning and end of composting, respectively [chapter 2: Figure 2.2, Table 2.1], and teff used as bulking agent). In contrast, CH_4-C emissions from biochar-amended cattle manure piles in our study (Table 3.1) were six times lower than the ones reported by Sommer (2001), supporting our assumptions of insufficient oxygen supply in the treatments without biochar in our study and the one from Sommer (2001). However, except for the study from Sommer (2001), all composting studies with cattle manure cited in Table 3.2 have in common that the composting piles or windrows comprised a larger volume and longer distances for air diffusion from the surface to the inside of the piles compared to ours. We thus conclude that the better aeration of our composting piles during the thermophilic phase, due to their smaller volume of 2.5 m^3 led to reduced CH_4 emissions. This effect seemed to have outweighed the impact of moisture and turning, since i) the piles in our study had high initial moisture contents of 78.6−81.4%, and since ii) we had only turned the compost once after 101 days (chapter 2: section 2.2.2).

3.3.2.3 Cumulative N_2O emissions

The addition of biochar reduced N_2O emissions by 56 and 57% for humanure and cattle manure treatments, respectively (Table 3.1). Some possible effects of biochar addition include improved O_2 availability and distribution through the compost mixture due to the biochar's highly porous surface (Lehmann and Joseph 2009). This porous matrix has also been shown to capture NO_3^- through conventional anion exchange with functional groups and organo-mineral complexes, and by unconventional H-bonds created through electron donation of the π-orbitals of the polyaromatic systems to the electron-deficient hydrogen in water of the hydrated NO_3^- (Joseph et al. 2017; Kammann et al. 2015). We detected a significant reduction of extractable NO_3^- by 32−36% in treatments amended with biochar compared to non-biochar-amended treatments, which we attributed to strong NO_3^- adsorption by biochar (chapter 2: Table 2.1). Reduced availability of NO_3^- as an electron acceptor has been linked with the suppression of the *nirK* gene abundance of denitrifying bacteria, which encodes the enzyme responsible for the reduction of NO_2^- to NO (Kammann et al. 2015; Lehmann and Joseph 2009; Li et al. 2016). Consequently, the shift of the denitrifying gene expression favors the N_2O-reducing enzymatic activity encoded by the *nosZ* gene (Kammann et al. 2015; Lehmann and Joseph 2009; Li et al. 2016), finally leading to a reduction in N_2O emissions. The increase in pH due to the alkalinity provided by the presence of ash in biochar and/or the adsorption of N_2O in biochar have also been reported to lower the N_2O:N_2 ratio, thereby stimulating the activity of N_2O reductase in denitrifying microorganisms (Lehmann and Joseph 2009; Yanai et al. 2007). Other abiotic mechanisms investigated are, e.g., the role of metals present in biochar, such as TiO_2 and Fe cations, that can act as catalysts for the reduction of N_2O (Lehmann and Joseph 2009). These different findings therefore suggest that a combination of a range of both biotic and abiotic mechanisms are involved in the reduction of N_2O emissions due to biochar addition (Cao et al. 2019).

Consequently, N lost as N_2O from the treatments with biochar in our study was 1.1−1.3% of initial N, whereas treatments without biochar comprised N_2O-N losses of 2.3−4.7% of initial N in the compost (Table 3.1). Still, this N fraction lost as N_2O from the humanure composts without biochar is comparable with the range of 0.8−1.9% measured by McNicol et al. (2020). In contrast, N_2O-N emissions from cattle manure treatments without biochar (4.7% of initial N; Table 3.1) were significantly higher than those reported by the studies cited in Table 3.2, showing N losses in the range of 0.1−2.3% of initial N. We attribute these higher N_2O-N emissions, which evolved before and after the thermophilic phase, to the higher moisture content of the cattle manure piles of 75−79% after the thermophilic phase in our study to a range of 64−77% initial moisture contents from the other studies (Table 3.2). This surplus of water may have hindered air diffusion in the already compacting piles, thereby increasing the number of anaerobic sites within the piles. This assumption is supported by the fact that only cumulative N_2O-N emissions of 1.3% of initial N from cattle manure treatments with biochar (Table 3.1) were inside the range of emissions reported by other studies for static (0.1−2.3%) and active (0.2−1.9%) composting of cattle manure (Table 3.2).

When comparing CH_4 and N_2O emissions from our composting trials with other studies (Ahn et al. 2011; Chadwick 2005; El Kader et al. 2007; Hao et al. 2001; Maeda et al. 2013; McNicol et al. 2020; Sommer 2001), our data suggest that the combination of biochar with sawdust, straw plus a compost box design that allows aeration throughout the whole compost, leads to lower CH_4 and N_2O emissions. However, higher N_2O emissions of our cattle manure control treatments suggest that further improvements of the approach are needed, like a reduction of the initial moisture content and/or the use of another bulking material than teff straw and sawdust, in order to even further increase aeration inside the pile. We thus recommend using grass with stronger stems, e.g., wheat straw, that could resist compaction longer and thereby reduce the need for pile turning, and not adding any additional water when mixing the composting substrate.

3.3.2.4 Cumulative NH_3 emissions

Cumulative NH_3 emissions from all treatments ranged from 6309−9881 mg NH_3-N m^{-2}, which corresponded to 1.2−1.6% of the initial TN (Table 3.1). The share of N lost as NH_3 emissions in our study was in the lower range of those emissions reported by other studies for static (0.26−9%) and active (3−13%) composting of cattle manure (Table 3.2).

All studies cited in Table 3.2 for cattle manure have in common that their composting mixtures had lower initial C:N ratios of 17−24, while the initial compost mixtures in our study had initial C:N ratios of 32.3 (HM), 50.2 (HM+BC), 41.4 (CM), and 49.8 (CM+BC; chapter 2: Table 2.1). Not only the C:N value differed between our compost piles and the ones in the above cited studies, but also the accessibility of the carbon for the microbial biomass. Besides the source of carbon provided from human or animal feces, we used a mixture of teff straw, sawdust and cabbage waste, with the last one providing easily degradable and thereby for microorganisms easy accessible carbon (Peigné and Girardin 2004), while El Kader et al. (2007), Maeda et al. (2013) and Sommer (2001) used straw and dried grass as extra carbon source. Moreover, Maeda et al. (2013) measured a higher initial pH value of 7.9 and average compost temperatures of > 70°C for several days, while our pH ranged between 6.6−7.4 and maximum average temperatures were 65.1°C, 64.2°C, 62.1°C, and 65.9°C, for HM, HM+BC, CM, and CM+BC, respectively (chapter 2: section 2.3.2.1). We thus conclude that the higher C:N ratios of our composting piles and the use of cabbage waste as an easily available carbon source, led to a better N assimilation by the microbial biomass and therefore to lower NH_3 emissions (Bernal et al. 2009). In addition, as elevated temperatures and pH >7.5 are some of the main factors promoting NH_3 emissions (Bernal et al., 2009; Rynk et al., 1992), the higher pH and temperatures reported by Maeda et al. (2013) can also explain the enhanced volatilization of NH_3 from their composting process.

We did not observe any significant differences in cumulative NH_3 emissions between treatments of our study, but a tendency to lower values for treatments containing cattle manure. Since all treatments reached similarly high temperatures and pH values >8.0 during the thermophilic phase, time in which ammonia emissions peaked, we attribute the difference between both types of manure to the 2.4−2.6 times higher initial NH_4^+ contents in the humanure treatments (chapter 2: Table 2.1).

In other studies, the addition of biochar was associated with decreased NH_3 emissions caused by the absorption of NH_3 and precursors, such as NH_4^+, urea and uric acid, by biochar (Agyarko-Mintah et al. 2017; Chen et al. 2017; Zhou et al. 2021). After oxidation of biochar with H_2O_2, Hestrin et al. (2020) found that the oxidized biochar increased microbial activity, doubled the composting rate, and reduced NH_3 loss. Oxidation or functionalization of the biochar's surface usually occurs in the course of composting (Hua et al. 2009; Prost et al. 2013; Wiedner et al. 2015). However, as most of the NH_3 in our study was emitted at the very beginning of the composting process, we assume that oxidation of the biochar's surface, which could have decreased NH_3 emissions, had not occurred at that point in time. In their global meta-analysis, Cao et al. (2019) quantified the impact of different additives on NH_3 and

GHG emissions and found that under alkaline conditions (pH>7.5) physical additives, such as biochar, resulted in lower reduction in NH_3 emissions compared to chemical additives, especially PO_4^{3-} and Mg^{2+} salts. They attributed this to the alkaline minerals present in biochar that shift the equilibrium from NH_4^+ to NH_3 volatilization at pH above 7.5. We thus attribute the non-significant effect of biochar addition on NH_3 emissions and the tendency to higher cumulative NH_3 emissions in biochar composts in our study to i) the fact that most of the biochar's surface oxidation had not occurred at the time of highest NH_3 emissions, and ii) the significantly higher pH values in the biochar compost treatments, which were in the range of 7.3−9.0 compared to 6.6–8.9 in our biochar-free compost treatments (chapter 2: Figure 2.2, Table 2.1).

3.3.3 Total greenhouse gas emissions

After converting cumulative CH_4-C and N_2O-N emissions to CO_2-C_{eq} by using a 100-year global warming potential of 12.36 and 127.71, respectively (IPCC 2013), and summing them up with the CO_2-C emissions, we obtained total GHG emissions over the 143 days of composting of 6921−8076 g CO_2-C_{eq} m^{-2}. We did not observe a significant effect of biochar addition to compost, neither of the type of manure used, on total GHG emissions (Table 3.1). This indicates that the lower CH_4 and N_2O emissions we observed for the biochar-amended treatments were compensated by their higher, although not significant, CO_2 emissions of those treatments. These results were not reflected in the significantly lower TOM and TOC losses of the biochar-amended composts, which we mainly attribute to the biochar's recalcitrance against decomposition (Hagemann et al. 2018). Since CO_2 is produced from the decomposition of any kind of organic waste, i.e., organic material that is derived directly or indirectly from recent plant material, it is considered to be climate-neutral. Thus, from a climate change perspective, we did not account for CO_2 emissions when estimating the GHG mitigation potential of biochar addition during composting (IPCC 2006). We found that the addition of biochar reduced non-CO_2 GHG emissions on average per pile by 2.2 kg CO_2-C_{eq} for HM+BC and of 5.5 kg CO_2-C_{eq} for CM+BC during the composting process, which corresponds to a non-CO_2 GHG emission mitigation of 51−71% (Table 3.1).

Carbon dioxide was the major GHG emitted during the composting process, accounting for 82.0−86.1% of the total CO_2-C_{eq} emissions from treatments containing biochar and 61.6−71.0% in the case of the controls. Methane accounted for 0.3−0.4% of the total CO_2-C_{eq} fluxes for HM, HM+BC and CM+BC, and 2.6% for CM. In contrast to CH_4, a higher share of the total flux was contributed by N_2O, with 13.7−17.7% for biochar-amended treatments, and 28.6−35.8% for controls (Table 3.1). Hence, co-composting of human and cattle manure with biochar can be an efficient strategy in reducing GHG emissions from composting. In addition, it will contribute to sequestering carbon when using the compost as a fertilizer and soil amendment due to its recalcitrant nature and by stabilizing non-biochar organic matter (Schwarzer, 2019).

Table 3.2. *Literature review of studies reporting cumulative emissions of GHG and NH_3 for human excreta and cattle manure composting with addition of a bulking agent* †

Feedstock	Bulking agent	System		Duration	CO_2-C % of initial C	CH_4-C % of initial C	N_2O-N % of initial N	NH_3-N % of initial N	Reference
Studies with GHG emissions measurements over a full composting process ‡									
Human fecal material from urine-diverting dry toilets	Bagasse (2:1, bagasse:feces)	Static in compost bins (16m^3) for 8–10 weeks. Afterwards turned every two weeks for eight weeks. Followed by a maturation phase of 2–4 months in static windrows		6–8 months	21–24	0.2–0.7	0.8–1.9	Not measured	McNicol et al. (2020)
Dairy manure	Bedding material (not specified)	Cylindrical piles 1200 kg mass, 1.9 m^3 volume 1.2 m height 1.4 m diameter C:N 27 76% moisture	Static	80 days	28.9	0.7	0.1	Not measured	Ahn et al. (2011) ¶
			Active (turned four times)		31.9	1.0	0.4		
Beef cattle farmyard manure	Bedding material (not specified)	Concrete block walls 3 m x 3 m 1.5 m high 14 m^3 C:N 17–24 77% moisture	Static and uncovered	90–109 days	Not measured	Period 1: 1.8% Period 2: 1.6%	Period 1: 2.3% Period 2: 0.1%	Period 1: 4.5% Period 2: 0.26%	Chadwick (2005)
Dairy cattle deep litter	Straw (bedding)	Static windrows 1.1–1.7 t weight 3.7 m long 1.9 m wide 1.3 m high C:N 18–21 62–64% moisture	Compacted	132 days	Not reported	0.03	0.3	12.2	Sommer (2001)
			Cut and mixed			0.03	0.1	5.9	

Table 3.2. *Continuation*

Feedstock	Bulking agent	System		Duration	CO_2-C % of initial C	CH_4-C % of initial C	N_2O-N % of initial N	NH_3-N % of initial N	Reference
Studies with GHG emissions measured only until the end of the thermophilic phase[§]									
Dairy cattle manure	Dried grass	Piles of 4 t of manure and 400 kg of dried grass C:N 24 72% moisture	Active (turned three times)	8 weeks	17.0	0.4	0.2	3.0	Maeda et al. (2013)[¶]
Dairy cattle manure	Straw (bedding)	Windrows 548−555 kg weight 1.98 m length 1.35 m width 0.78−0.78 high C:N 23−24 77% moisture	Static	46 days	Not reported	Not reported	1.2	9	El Kader et al.g (2007)
			Active (turned two times)				1.9	13	
Feedlot cattle manure	Straw (bedding)	Windrows 6.7−9.5 Mg weight 3.6 m x 14 m 1.4−1.6 m high C:N 19 71% moisture	Static	99 days	21.6	1.8	0.6	Not measured	Hao et al. (2001)[¶]
			Active (turned six times)		49.0	2.4	1.1		

† We only considered composting studies that i) used bulking materials, ii) reported compost temperature data, iii) showed a composting process reaching thermophilic temperatures, and iv) used compost volume and weights comparable or larger than the ones used in this study.

‡ We consider a full composting process as a process that covers all four composting phases, i.e. mesophilic I (ambient temperature−45°C), thermophilic (>45°C), mesophilic II (45°C−ambient temperature), and a final maturation phase at ambient temperature (compare Bernal et al., 2009).

§ These studies did not cover a full composting process, as they stopped the experiment at the end of the thermophillic phase or before the compost material reached ambient temperature.

¶ Cumulative GHG emissions were calculated by us as % of initial C or N using the data presented in the cited studies in order to make them comparable with the results from our study (for calculations see Appendx B: Table B.1).

3.4. Conclusions

We evaluated the effect of biochar addition on GHG and NH_3 emissions during thermophilic composting of human excreta and cattle manure. We found that composting of these organic materials with biochar was effective in minimizing CH_4 and N_2O emissions. However, we did not observe a significant effect on CO_2 and NH_3 emissions.

Ecological sanitation via thermophilic composting with biochar addition therefore represents an integrated sanitation strategy that can contribute to climate change mitigation by reducing GHG emissions in sanitation and waste management. Field studies with biochar-compost amendments have to prove if this ecological sanitation approach may even have a positive effect on food security and climate change mitigation by increasing soil fertility and soil carbon sequestration.

4

NUTRIENT AND CO_2 DYNAMICS AFTER APPLICATION OF BIOCHAR-AMENDED HUMAN EXCRETA COMPOST AND CATTLE MANURE COMPOST TO SANDY SOIL UNDER TROPICAL CONDITIONS

Based on:

Castro-Herrera, D., Prost, K., Kim, D-G., Yimer, F., Tadesse, M., Gebrehiwot, M., Asaye, Z., Brüggemann, N. Nutrient and CO_2 dynamics after application of biochar-amended human excreta and cattle manure compost to sandy soil under tropical conditions. *Manuscript in preparation.*

Abstract

The use of co-composted biochar in agriculture has the potential to counteract soil degradation, which is a particular problem in the tropics, contribute to climate change mitigation, increase soil nutrient retention and reduce the use of synthetic fertilizers. Here, we conducted a 6-month incubation experiment to investigate the mineralization dynamics of four types of compost (human excreta or cattle manure compost, with and without co-composted biochar) at two application rates (total compost N equaled 170 kg N ha^{-1}, and three times this amount) to a highly depleted sandy soil. Overall, we found degradation rates lower than 40% for all compost types over the 6-month incubation period at 25°C. Nitrogen availability in the high-rate compost treatments was comparable with a common-practice mineral fertilizer (MF) treatment, whereas P and K availability exceeded the MF treatment. In contrast, in the low-rate compost treatments, P and K availability was comparable to the MF treatment, whereas N availability was significantly below the MF level. While N_2O and CH_4 emission rates were generally very low, CO_2 emissions of the soil amended with the high rate of human excreta compost plus biochar were 36% lower than for the human excreta compost without biochar. In the same treatment, biochar decreased the extractable NO_3^- by 16% and available P by 7%, thereby reducing the risk of N and P leaching losses. Biochar addition also decreased available P in the biochar-cattle manure compost treatments by 22−24%. This is the first study that follows the greenhouse

gas emissions and nutrient dynamics after applying human excreta compost with and without biochar to soil. Our findings have important implications for the future of a sustainable return of nutrients and organic matter from human excreta to agricultural soils, with opportunities for C sequestration and nutrient retention.

4.1 Introduction

It is estimated that 33% of the Earth's soils are already degraded, and that over 90% could become degraded by 2050, especially in agricultural lands and in highly weathered and strongly leached soils of tropical regions (FAO 2019). The depletion of soil organic matter (SOM) and nutrients not only limits agricultural productivity and poses a threat risk to global food security, but also has severe implications for our environment, including water quality and climate change through increased CO_2 emissions (FAO 2019; Lorenz and Lal 2018). Soil carbon sequestration practices, as well as improved and sustainable nutrient management strategies in agricultural systems need to be implemented to counteract soil degradation, contribute to climate change mitigation and adaptation and to increase productivity of soils (Amelung et al. 2020; Maheshwari 2014). Sequestering carbon in arable soils may improve soil nutrient supply, water-holding capacity, and soil structure, and thereby contribute to closing agricultural yield gaps and reducing fertilizer needs (Amelung et al. 2020). To gain and maintain soil organic carbon (SOC) stocks, organic inputs into soil need to be increased (Amelung et al. 2020; Meena 2020).

Compost application in agriculture has been identified as a source of stable organic C and nutrients that can be used to improve soil fertility, soil structural stability, SOC content, cation exchange capacity, pH, and microbial biomass (Maheshwari 2014). Furthermore, the plant macro- and micronutrients present in the organic materials that can be used to produce compost, such as human excreta or animal manures, could help millions of farmers restore their degraded soils and boost agricultural productivity, instead of heavily depending on non-renewable synthetic fertilizers (Akram et al. 2019; Maheshwari 2014; Meena 2020). Additionally, the management of human and animal excreta is crucial for issues related to public health and environmental pollution, especially in regions that suffer from a deficiency or lack of adequate sanitation and waste management, such as Sub-Saharan Africa (WHO 2019b). Developing interventions that combine ecological sanitation, thermophilic composting, and climate-smart agriculture offer feasible solutions for treating these critical organic materials in a resource-efficient way, for increasing food security and for mitigating climate change through soil carbon sequestration in agricultural ecosystems (Akram et al. 2019; Maheshwari 2014; Meena 2020).

The slow release of organically bound nutrients from organic fertilizers into soil makes the use of compost less attractive as it might take several years until all the nutrients contained in these stable organic materials are provided in their plant-available form (Meena 2020). These limitations at the farmer's level due to the slower nutrient release of organic fertilizers compared to that of mineral fertilizers is also aggravated by the legal restrictions of their application to arable or grassland soils in amounts that do not consider this slow release (Meena 2020). In Germany, these legal limitations can be found in the German Fertilizer Ordinance ("Düngeverordnung") from 26th of May 2017 under Section 6 "Additional requirements for the use of specific fertilizers (Zusätzliche Vorgaben für die Anwendung von bestimmten Düngemitteln)", No. 4 (Bundesministerium der Justiz und für Verbraucherschutz and Bundesamts für Justiz 2017). To avoid nutrient leaching, e.g. specially of nitrate and phosphate, to the environment, the application of organic and organic-mineral fertilizers, including farm manures (also in mixture) is limited to a maximum annual amount of 170 kg of total N per hectare. In the case of compost, the total amount of N applied must not exceed 510 kg per hectare over a period of three years.

Despite of the many benefits that compost application has in enhancing soil fertility, the organic matter from compost is typically mineralized to CO_2 within a time frame of months to years, due to rapid decomposition rates under tropical conditions and thus, only a small portion of the applied organic compounds will be stabilized in the soil in the long term (Agegnehu et al. 2017). In view of this rapid mineralization of SOM and accelerated loss of SOC, interest in biochar as a complementary amendment to compost in agricultural soils has increased considerably (Agegnehu et al. 2017; Lehmann and Joseph 2009; Lorenz and Lal 2018; Meena 2020). Biochar, a carbon-rich solid product of thermochemical conversion of biomass residues with limited oxygen supply, has the potential to offer a variety of benefits, including carbon sequestration, improved soil properties and fertility, climate change mitigation (Glaser et al. 2001; Lehmann and Joseph 2009), and reduction of nutrient leaching (Hagemann et al. 2017; Kammann et al. 2015; Lehmann and Joseph 2009; Tsai and Chang 2020).

In order to expand the understanding of compost mineralization dynamics, especially at large application rates, under tropical conditions in SOC- and nutrient-depleted soils, we conducted a 6-months incubation experiment. For this, we measured greenhouse gas emissions and followed the nutrient dynamics after application of four different compost types, i.e. human excreta and cattle manure compost with and without biochar at two different application rates, and compared these dynamics with an unfertilized control and with a common-practice mineral fertilizer treatment. The amount of compost for the lower application rate was calculated, based on the total N content, to match the N application rate of the mineral fertilizer treatment (170 kg N ha^{-1}), while the higher compost application rate was three times higher than that of the lower compost application rate and therefore, at the upper limit of the regulations of the German Fertilizer Ordinance. With this study, we specifically aimed to test the hypotheses that i) under tropical conditions (in this study at an average soil temperature of 25 °C) complete mineralization of the compost added would take place during the 180 days of incubation, ii) the addition of relatively small amounts of co-composted biochar has the potential to stabilize soil organic matter and decrease C mineralization, iii) composts containing biochar addition would reduce CH_4 and N_2O emissions and the risk of nutrient leaching, mainly as nitrate and phosphate, from soil, and iv) applying the maximum amount of compost allowed in Germany would not increase the risk of nutrient leaching from agricultural soils.

To our knowledge, this is the first study that measures greenhouse gas emissions and follows the nutrient dynamics after the application of human excreta compost with and without biochar addition in soil.

4.2 Materials and Methods

4.2.1 Incubation experiment and experimental design

For the laboratory incubation experiment, ten different soil mixtures with four replicates each were placed in PVC tubes (21.5 cm long and 4.5 cm diameter) inside an incubator at temperatures ranging from 23.5−26.5 °C for six months (Appendix C: Figure C.1). The treatments consisted of a control containing 350 g (dry weight) of sandy soil, and nine more soil mixtures containing the same amount of soil with an addition of a mineral fertilizer or four different types of compost with two different application rates (Table 4.1; Appendix C: Table C.1). The height of the different soil mixtures filled inside the PVC tubes ranged from 13.7−14.9 cm. The sandy soil was prepared by mixing a standard silty sand agricultural soil (uS, No. 2.1, LUFA Speyer, Germany) and quartz sand (Quarzwerke, Germany) on a 1:5 ratio (dry basis). The mixture had a moisture content of 9.9 %, water holding capacity of 19.3 %, bulk density of 1.3 g cm^{-3}, and classified as class A soil with an available fraction of P and K of <15 mg kg^{-1} and <30 mg kg^{-1} for sandy soil, respectively (Landwirtschaftskammer Niedersachsen 2018; VDLUFA 1999, 2018). The mineral fertilizer used was a NPK 15-15-15 fertilizer with 6.3 % NO_3^--N, 8.7 % NH_4^+-N, 15 % P_2O_5, 15 % K_2O and 3.7 % SO_3 (Sulfan Mila Universal, YARA GmbH & Co. KG, Dülmen, Germany). The four types of compost used were human excreta (from now on called “humanure”), cattle manure compost, and each of these composts with biochar addition. We produced these organic fertilizers by thermophilic composting of humanure or cattle manure, vegetable scraps, teff straw, sawdust with and without biochar (18.5 % of compost dry weight) as described in chapter 2 (section 2.2.2). The main physical and chemical properties of the soil and compost treatments are listed in Table 4.1. The biochar in the amended composts was produced from Eucalyptus wood (*Eucalyptus camaldulensis*) with a top-lit up-draft micro-gasifier at pyrolysis temperatures between 500−600 °C and a residence time of 40−50 min. The biochar fulfilled the “premium” requirements of the European Biochar Certificate, except for the polycyclic aromatic hydrocarbons (PAHs) concentration, which exceeded the premium threshold value, but complied with the “basic” criteria (EBC 2012).

The amount of mineral fertilizer and compost for the lower application rate (Table 4.1) followed the regulations of the German Fertilizer Ordinance ("Düngeverordnung"; Bundesministerium der Justiz und für

Verbraucherschutz and Bundesamts für Justiz 2017) and of the Chamber of Agriculture of Nordrhein-Westfalen (Landwirtschaftskammer Nordrhein-Westfalen 2015) for N application for maize. According to these regulations, the N requirement of maize is 180−200 kg ha^{-1} and the application of compost in agricultural fields must not exceed an average amount of total N of 510 kg ha^{-1} over a period of three years. To calculate the amount of mineral fertilizer and compost for the lower application rate, we considered a N requirement of maize of 200 kg ha^{-1} and from this, we subtracted: 1) the amount of available N (sum of NH_4^+-N and NO_3^--N) in the soil (14 kg ha^{-1} in 13.7 cm), and 2) a soil N mineralization potential during the first two months of the cultivation period of 10 kg ha^{-1} (Landwirtschaftskammer Nordrhein-Westfalen 2015). The higher compost application rate was set three times higher than that of the lower compost application rate (Table 4.1).

The amount of water in the soil mixtures was adjusted to 60% of the water holding capacity of the control treatment and was gravimetrically maintained during the incubation period by irrigating each tube once a week according to the evaporative weight loss.

Table 4.1. *Description and nutrient application rates per treatment and amount of available nutrients and dissolved organic C and N on day 0. Values provided as means with $n=4$ and in dry weight basis. Different superscript letters indicate significant differences at $p<0.05$ among treatments.*

Treatment	Description	Amount of fertilizer (kg ha^{-1})			Amount extracted from soil mixtures on day 0 (mg kg^{-1})					pH on day 0
		Total			Available			Dissolved		
		N	P	K	N	P	K	organic C	N	
Control	Soil	-	-	-	6.2^{a}	6.0^{a}	10.0^{a}	3.4^{a}	2.2^{a}	4.9^{a}
Mineral fertilizer and lower compost application rate (according to the German Fertilizer Ordinance)										
MF	Soil + mineral fertilizer (NPK 15-15-15)	176	76	146	81.6^{e}	44.7^{bcd}	63.0^{ab}	4.9^{ab}	18.7^{c}	4.9^{a}
HM1	Soil + humanure compost	177	73	347	20.3^{abc}	35.3^{bc}	106.7^{b}	10.7^{bcd}	6.8^{ab}	6.1^{b}
HM+BC1	Soil + humanure with biochar compost	175	73	404	20.8^{abc}	25.3^{abc}	113.3^{b}	10.5^{abc}	5.0^{a}	6.4^{b}
CM1	Soil + cattle manure compost	176	68	295	13.5^{abc}	33.3^{bc}	101.3^{b}	8.4^{abc}	3.3^{a}	6.3^{b}
CM+BC1	Soil + cattle manure with biochar compost	176	56	296	11.8^{ab}	23.0^{ab}	80.7^{b}	13.0^{cd}	5.2^{a}	6.4^{b}
Higher compost application rate (three times more)										
HM2	Soil + humanure compost	531	220	1041	56.0^{d}	70.0^{de}	314.0^{de}	22.3^{de}	17.1^{c}	6.9^{c}
HM+BC2	Soil + humanure with biochar compost	526	220	1212	22.5^{bc}	68.3^{de}	324.0^{e}	23.9^{e}	12.9^{bc}	7.0^{c}
CM2	Soil + cattle manure compost	527	203	886	26.7^{c}	75.3^{e}	256.0^{cd}	17.8^{de}	5.9^{a}	7.0^{c}
CM+BC2	Soil + cattle manure with biochar compost	527	168	889	42.0^{d}	50.7^{cde}	228.0^{c}	20.9^{de}	7.5^{b}	6.9^{c}

4.2.2 Greenhouse gases flux measurements

We measured fluxes of CO_2, CH_4, and N_2O from the PVC tubes containing the incubated soil mixtures with the dynamic chamber method. The chamber consisted of a PVC lid with an inner diameter of 4.5 cm and a height of 10.6 cm, and was equipped with a rubber seal which fit gas-tight on the PVC soil columns. The chamber lid was equipped with two stainless steel tube fittings (¼ in) as sampling ports. The sampling ports were connected with 2.37 m long tubing in closed-loop mode to an infrared laser absorption analyzer (G2508, Picarro, Inc., Santa Clara, CA, USA; Appendix C: Figure C.2). A 1-m long open tubing with an inner diameter of 4 mm was connected to a third gas-tight stainless steel tube fitting (¼ in) as vent tube to minimize pressure perturbations during chamber deployment. The headspace comprised a soil surface area of 0.0016 m^2, and, depending on the treatment. a height of 13.7−14.9 cm and a volume of 0.00019−0.00021 m^3.

We carried out GHG flux measurements on days 0, 1, 2, 3, 5, 8, 15, 21, 29, 36, 44, 50, 57, 67, 71, 78, 86, 99, 113, 127, 141, 155, 169, and 180 with a deployment time of the analyzer of 10 min.

We calculated the GHG fluxes and expressed them as mg of CO_2-C m^{-2} d^{-1}, µg of CH_4-C m^{-2} d^{-1}, or µg of N_2O-N m^{-2} d^{-1} by using Eq. 4.1.

$$\mathrm{Flux}_{\mathrm{GHG}} = \frac{S}{1000000} \times \frac{P}{R\,T} \times \frac{V}{A} \times M \times 60 \qquad \text{(Equation 4.1)}$$

where S is the slope of the linear equation fitted to the change of the gas concentrations in the chamber during the measurements (ppm min^{-1}); 1,000,000 is used to convert ppm (µL L^{-1}) into the unit L L^{-1}; P is the air pressure (atm) at 80 m a.s.l.; R is the universal ideal gas constant (m^3 atm mol^{-1} K^{-1}); T is the average temperature (K) of the chamber during the deployment time; M is the molar mass of C or N (g mol^{-1}); and 60 the value used to convert the unit time from min to hours. The concentration values obtained during the first 2 min were not used for the calculation of the slope due to the fluctuations produced by pressure disturbances during the closure of the chamber. Fluxes with linear fits of $R^2 < 0.81$ were considered below the detection limit and therefore, we reported them as zero. We estimated the cumulative emissions by calculating the emission average rate of two consecutive measurement days and multiplying it by the time between these two measurements. Thereafter, we summed up all the resulting values and expressed the cumulative fluxes as g C m^{-2} for CO_2 and CH_4, and mg N m^{-2} for N_2O, and also as % of initial C or N.

Compost degradation rates after 180 days of incubation were expressed as % of organic C added with the compost and calculated based on the cumulative CO_2-C emissions attributed only to compost decomposition, i.e. by subtracting the cumulative CO_2-C emissions of the control from the compost treatments, by using Eq. 4.2.

$$\text{Compost decomposition rate (\%)} = \frac{\text{TOC - (CO}_2\text{-C} \times \text{A)}}{\text{TOC}} \times 100 \qquad \text{(Equation 4.2)}$$

where TOC is the total organic C added through compost (mg); CO_2-C the cumulative CO_2-C emissions attributed only to compost decomposition (mg m^{-2}); and A the area of the PVC tubes where the soil mixtures were placed (0.0016 m^2).

To determine expected percentages of compost degradation after one year of compost application, we fitted logistic decay functions to the degree of decomposition at each measuring day with time for each treatment.

4.2.3 Soil sampling and sample preparation

Soil samples were collected on days 0, 30, 60, 90, 120, 150, and 180 by emptying the soil contained in each PVC tube and mixing it thoroughly. At each sampling day, we destructively sampled 40 samples (four replicates of each of the 10 treatments) to follow the nutrient dynamics throughout the incubation process. Fresh soil samples were used for determining NH_4^+, NO_3^-, and plant-available P and K. Dry soil samples (at 105 °C) were used for measuring pH. Total C, total N, total P, and total K were determined from dried (at 105 °C) and ball-milled samples.

4.2.4 Chemical analysis

Ammonium and nitrate were determined by ion chromatography (Dionex ICS-3000, USA) after extracting fresh soil samples with 0.01 mol L^{-1} $CaCl_2$ (1:10 w/v ratio for 2 h; VDLUFA 2014). Available N was obtained by summing up NH_4^+-N and NO_3^--N. Plant-available P and K were determined by inductively coupled plasma optical emission spectroscopy (ICP-OES, iCAP 6500, Thermo Fisher Scientific, Oberhausen, Germany) after extracting fresh soil samples with 0.05 mol L^{-1} calcium-acetate-lactate solution (CAL extraction, 1:20 w/v, shaking for 90 min at 200 rpm; VDLUFA 2014). Total organic C and total N were measured with a CHN elemental analyzer (vario EL cube, Elementar). Total contents of P and K were determined by inductively coupled plasma optical emission spectroscopy (ICP-OES, iCAP 6500, Thermo Fisher Scientific) after an aqua regia (nitric and hydrochloric acid) microwave extraction/digestion. The pH was measured with a glass electrode (WTW multi 340i, Xylem Analytics, Weilheim, Germany) in a $CaCl_2$ 0.01 mol L^{-1} extract (1:2.5, w/v) according to ISO 10390 (ISO 2005). Dissolved organic C and dissolved N were measured with a Shimadzu TOC/TN analyzer in a $CaCl_2$ 0.01 mol L^{-1} extract (1:4, w/v ratio for 0.5 h, centrifugation at 3500 rpm for 10 min, and filtrating with a 0.45 µm cellulose membrane filter (PORAFIL MV, Macherey-Nagel, Düren, Germany).

4.2.5 Statistical analysis

All the statistical analyses were conducted with the SAS software University Edition (SAS Studio, version 5.1). Data are reported as mean values ± standard error (SE) of each treatment (n=4) and we considered a statistical significance level of $\alpha = 0.05$.

A one-way repeated-measures ANOVA by using a mixed model with the Kenward-Roger method was conducted to test differences in nutrient, total C, and greenhouse gas dynamics among the soil treatments over the 180 days of incubation. In this mixed model, we considered treatment, time, and time–treatment interaction effects, and we accounted for the correlation among observations on the same treatment by adding two random effects, replicate–treatment, and replicate–treatment–time effects.

To compare chemical parameters measured on day 0 and cumulative GHG emissions among the soil treatments, we conducted a one-way ANOVA followed by the general linear model procedure for least squares means with the Tukey method for the adjustment for multiple comparisons.

Cumulative CO_2-C emissions from the compost treatments were linearly correlated with the corresponding dissolved N values and dissolved organic C to dissolved N ratios to obtain Pearson correlation coefficients.

A one-way ANOVA by using the general linear model procedure for least squares means with the Tukey method was used to analyze if NH_4^+, NO_3^- and available N, P and K final values were significantly increased or decreased at the end of the incubation period.

4.3 Results and discussion

4.3.1 Greenhouse gas fluxes and cumulative emissions

4.3.1.1 CO_2

On the first day of the incubation experiment, CO_2-C emissions were above 100 mg m^{-2} h^{-1} from the soil mixtures containing compost, while the CO_2-C emissions from the control and MF treatments were below 20 mg m^{-2} h^{-1} (Figure 4.1). These higher respiration rates at the beginning of the incubation experiment reflect the higher microbial activity introduced through the incorporation of an organic carbon rich material into the soil, such as compost (Maheshwari 2014). These initially high respiration rates of the soil treatments amended with compost were immediately followed by a rapid decrease and thereafter by an increase again after day five, partially to values higher than the values of day 1 of the experiment. CO_2-C emissions were larger than 50 mg m^{-2} h^{-1} in HM+BC1, CM2 and CM+BC2 until day 57, in HM1 and HM+BC2 until day 78, and from HM2 until day 99. During this prolonged peak of soil respiration, we measured the largest CO_2-C emissions of 366 and 217 mg m^{-2} h^{-1} for soil mixtures with the higher application rate of humanure compost with and without biochar, respectively. In contrast, treatments with the lower humanure application rate as well as the cattle manure treatments with both application rates exhibited lower CO_2-C emissions with average values below 100 mg m^{-2} h^{-1}. Even lower CO_2-C emissions were measured from the control and MF treatments, which remained rather constant, with average CO_2-C emissions of 17 mg m^{-2} h^{-1} during the 180 days of the incubation period. During the last 81−123 days of incubation, all treatments remained stable and below 50 mg m^{-2} h^{-1}, indicating that the easily degradable organic substrates contained in the composts had been degraded, and stabilization and humification of the organic matter took place (Maheshwari 2014).

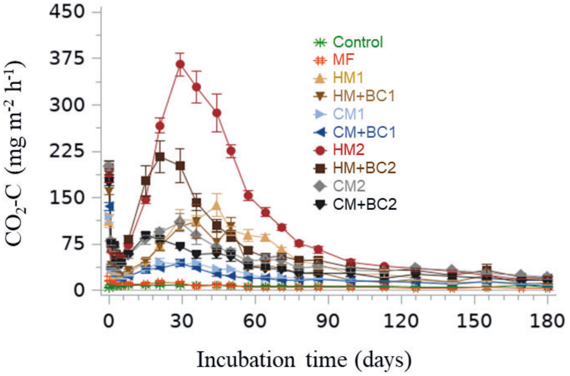


Figure 4.1. ***CO_2 emission rate dynamics.*** *Soil incubated at 25 °C, without any nutrient addition (control, green asterisk), with mineral fertilizer (MF, orange hash), with a lower compost application rate of: humanure (HM1, yellow triangle), humanure amended with biochar (HM+BC1, light brown triangle down), cattle manure (CM1, light blue triangle right), cattle manure amended with biochar (CM+BC1, blue triangle left), and with a higher compost application rate (three times more than the lower one) of: humanure (HM2, red circle), humanure amended with biochar (HM+BC2, brown square), cattle manure (CM2, gray diamond), cattle manure amended with biochar (CM+BC2, black home down). Values provided as means (± SE) with n=4.*

The cumulative CO_2-C emissions for all treatments ranged between 26−455 g m^{-2}, with the lowest emissions observed for the control and MF treatments, and the highest emissions from the higher compost application rate, especially from the humanure compost treatments (Table 4.2). As shown by Maheshwari (2014), other field compost application studies also observed elevated soil respiration when applying compost, which they attributed to the substantial increase in microbial activity in the soil as result of the mineralization of available carbon compounds contained in the organic amendments. Besides an enhanced microbial activity, these studies also observed significant increases in soil pH to optimum ranges for microbial growth. The addition of compost to soil in our study significantly increased the pH of the soil from 4.9 (control and MF) to 6.1−6.4 with the lower compost application rate and further to 6.9−7.0 with the higher application rate, with no effect of type of compost or biochar addition (Table 4.1). These results indicate that organic amendments have great potential of raising soil pH and thereby, ameliorating acid soils (Maheshwari 2014; Meena 2020).

It is worth noting that the application of three times the amount of compost in our treatments did not translate into three times the CO_2 cumulative emissions from the lower application rate. We attribute this either to an O_2 limitation inside the soil matrix or/and to higher respiratory C losses per unit of microbial biomass in the soil treatments from the low application rate. Even under well-aerated conditions, O_2 can be depleted inside microsites within soil aggregates when it is consumed faster than it can diffuse into such aggregates (Paul 2015). Thus, the O_2 consumption rate by microbial activity after the addition of a large amount of compost was faster than its diffusion rate, causing a saturation of aerobic CO_2 production. In regard to the higher respiratory C losses per unit of microbial biomass, Clayton et al. (2021) observed that microbial fractions of soil samples containing $< 1\%$ SOC were out of stoichiometric equilibrium and therefore, were inefficient at immobilizing C due to high microbial investments for resources acquisition in the resource-poor soils, whereas increasing SOC, especially above 1% SOC lead to a more efficient C acquisition.

The soil CO_2 emissions from the compost treatments observed in the current study were in agreement with other incubations studies conducted at 25°C with different organic amendments applied to sandy soils (Nguyen et al. 2014; Tsai and Chang 2019) and with field experiments conducted in Sub-Saharan Africa with different agricultural management practices, including application of animal manure (Kim et al. 2015; Musafiri et al. 2020). Nevertheless, CO_2 emissions from the control and mineral fertilizer treatments of these field experiments were substantially higher than the ones in our study, which we attribute to the lower total C content of the sandy soil used in our experiment.

For both compost application rates, soil mixtures containing humanure with and without biochar exhibited 1.8−2.2 times larger cumulative CO_2-C emissions than the corresponding cattle manure treatments (Table 4.2) Since microbial communities utilize dissolved organic substrates, i.e. available C and N, their concentration, composition and supply rate in soil have been recognized as one important driving factor regulating soil microbial biomass growth and thereby soil respiration (Paul 2015). These labile C and N sources, such as cellulose, hemicellulose, proteins, nucleic acids, and other C and N-containing compounds, provide the initial nutrients and energy source to boost microbial biomass growth

(Paul 2015). Soil cumulative CO_2-C emissions of the compost treatments were positively correlated with the amount of dissolved N measured on day 0 (correlation=0.807; Figure 4.2), while we observed a negative correlation between dissolved organic C to dissolved N ratios and cumulative CO_2-C emissions (correlation= -0.638; Figure 4.2). This indicates that both dissolved N and the ratio of dissolved organic C to dissolved N might have influenced microbial biomass growth and respiration. Therefore, we attribute the larger cumulative CO_2-C emissions from the humanure treatments to their higher dissolved N contents and their lower dissolved organic C to dissolved N ratios measured at the beginning of the incubation experiment (Table 4.1; Figure 4.2), which might have stimulated microbial growth and activity particularly strongly.

Table 4.2. *Cumulative emissions of greenhouse gases (expressed as mass per m^2, and % of initial C or N) and calculated decomposition rate (with respect to C) of the compost added after 1 year of application. Soil incubated at 25 °C, without any nutrient addition (control), with mineral fertilizer (MF), with a lower compost application rate of: humanure (HM1), humanure amended with biochar (HM+BC1), cattle manure (CM1), cattle manure amended with biochar (CM+BC1), and with a higher compost application rate (three times more than the lower one) of: humanure (HM2), humanure amended with biochar (HM+BC2), cattle manure (CM2), cattle manure amended with biochar (CM+BC2). Values provided as means (± SE) with n=4. Different superscript letters indicate significant differences at p < 0.0001 among treatments.*

Treatment	CO_2-C ($g\ m^{-2}$)	CH_4-C ($mg\ m^{-2}$)	N_2O-N ($mg\ m^{-2}$)	CO_2-C (% initial C)	CH_4-C (% initial C)	N_2O-N (% initial N)	Compost decomposition after one year (%)
Control	27 ± 2^a	1.1 ± 0.2^a	0^a	6.2	<0.0004	0	-
MF	26 ± 2^a	1.6 ± 0.4^a	7.7 ± 4.9^a	6.7	<0.0004	0.02	-
HM1	203 ± 13^c	0.9 ± 0.1^a	6.3 ± 6.2^a	34.6	<0.0004	0.01	56
HM+BC1	163 ± 13^c	1.3 ± 0.2^a	0^a	15.6	<0.0004	0	27
CM1	93 ± 3^b	1.5 ± 0.1^a	0^a	19.0	<0.0004	0	34
CM+BC1	86 ± 1^b	1.0 ± 0.2^a	0^a	12.0	<0.0004	0	19
HM2	455 ± 11^e	1.0 ± 0.3^a	0.3 ± 0.2^a	44.5	<0.0004	0	43
HM+BC2	290 ± 20^d	1.4 ± 0.2^a	2.7 ± 2.5^a	17.0	<0.0004	<0.005	17
CM2	206 ± 12^c	0.7 ± 0.3^a	3.1 ± 1.9^a	21.0	<0.0004	<0.005	30
CM+BC2	163 ± 9^c	1.5 ± 0.2^a	1.1 ± 0.9^a	12.5	<0.0004	<0.005	15

For the lower compost application rate, biochar addition did not have a significant effect on the CO_2-C cumulative emissions, although we observed a tendency towards lower values for treatments containing biochar (Table 4.2). For the higher compost application rate, we observed the same trend for the cattle manure treatments, while we found that amending soil with humanure compost with biochar reduced CO_2 emissions on average by 36% compared to HM2. In this incubation study, we hypothesized that the addition of relatively small amounts of co-composted biochar has the potential to stabilize organic matter and decrease C mineralization. This significant decrease in CO_2 emissions and the tendency towards lower values confirm our assumption that biochar plays a stabilizing role within the soil matrix. Glaser et al. (2001) showed that findings of charcoal and highly aromatic humic substances in anthropogenic soils of the Brazilian Amazon basin, known as "Terra Petra" soils, suggest that residues of incomplete combustion of organic materials are responsible for maintaining high levels of SOM in theses soils. Thus, Glaser et al. (2001) concluded that black carbon can act as a significant carbon sink and is a key factor for sustainable and fertile soils, especially in the humid tropics. Some recent studies have also indicated that the simultaneous application of biochar and compost resulted in enhanced C sequestration benefit (Agegnehu et al. 2017; Lorenz and Lal 2018; Tsai and Chang 2019). Biochar delivers a pool of C that is relatively resistant to mineralization, and due to the structural and chemical changes during composting and later in soil, the formation of surface functional groups such as carboxyl, phenolic, and carbonyl groups, provide interaction spots that may adsorb and stabilize SOC and nutrients (Lehmann and Joseph 2009; Lorenz and Lal 2018).

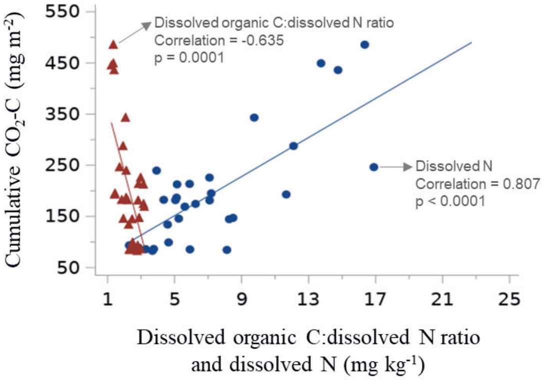


Figure 4.2. *Scatter plot with Pearson correlation coefficients between cumulative CO_2-C emissions and dissolved organic C to dissolved N ratio (red triangle) and between cumulative CO_2-C emissions and dissolved N (blue circle).*

In this study, another of our hypotheses was that under tropical conditions (in this study at 25 °C) complete mineralization of the compost added would take place during the 180 days of incubation. However, after fitting logistic decay functions to the cumulative CO_2-C emissions attributed to compost decomposition during the 180 days of incubation, we determined expected percentages of compost degradation after one year of application ranging from 15−27% for biochar composts and 30−56% for composts without biochar addition (Table 4.2). These values, besides disproving our hypothesis, also show that after one year compost degradation rates did not exceed 56% even for the high compost application rate, and therefore, besides building more C, compost provides a source of nutrients for two to three years. Furthermore, the lower degradation values for soil amended with biochar compost further confirm that applying biochar compost to soil could stabilize SOC by reducing C mineralization compared to application of compost alone. However, as different components of compost can have different decomposition rates, CO_2 emissions provide only a first proxy for total organic matter mineralization; but in this study, we assumed that nutrients, such as N, P and K are also released in their mineral form at the same rate.

4.3.1.2 CH_4 and N_2O

In contrast to CO_2 emissions, CH_4 and N_2O production from all treatments remained low and did not play an important role during the incubation period (Table 4.2; Appendix C: Figure C.3). We did not detect any significant differences across treatments, showing no effect of type of fertilizer and compost, application rate and addition of biochar on these non-CO_2 emissions throughout the study period.

We measured low CH_4 emissions below 1.6 mg m^{-2} and we did not observe a substantial CH_4 uptake (no negative fluxes). These results are not consistent with field studies conducted in Sub-Saharan Africa on maize fields in which the investigated soils predominantly acted as sinks of CH_4 (Musafiri et al. 2020). This indicates a well-aerated soil matrix in the different treatments in our study but also a low O_2 supply to oxidize CH_4 to CO_2. Even though we observed a saturation effect of CO_2 production for the higher compost application rate, the limited O_2 supply that restricted the further oxidation of organic matter, and thereby the production of CO_2 did not result in increased CH_4 emissions. Methanogenic activity develops optimally at very low redox potential, and as long as NO_3^- is present in the soil solution, the soil redox potential will remain at a level too high for CH_4 production. Thus, low CH_4 fluxes under these conditions are common (Paul 2015).

Nitrous oxide emissions were minimal and only in a few occasions significantly above the detection limit, showing insignificant N_2O production during microbial N transformations via nitrification or denitrification (Paul 2015). The high content of organic material obviously inhibited N_2O formation during nitrification (Liu et al. 2017a), while the more or less well-aerated status of our soil mixtures inhibited denitrification (Paul 2015). Cumulative N_2O emissions did not exceed 7.7 mg m^{-2} (77 g of N_2O-N ha^{-1} in six months) and were below the lower end compared with studies conducted in tropical croplands in Sub-Saharan Africa that reported N_2O fluxes ranging between 0.22–71 kg N_2O–N ha^{-1} yr^{-1} (Kim et al. 2015; Musafiri et al. 2020). In their field experiment on maize in Kenya, Musafiri et al. (2020) measured increased soil N_2O emissions after inorganic fertilizer application; however, we did not observe this N_2O dynamics with our MF treatment. In their study, they also observed that these N_2O peaks following inorganic fertilization coincided with precipitation events. They explain that precipitation could have increased soil moisture, resulting in increased substrate availability, microbial activity, mineralization of C and N, and anaerobic microsites, and thereby resulting in enhanced N_2O fluxes. In our study, soil moisture levels were kept stable throughout the incubation period (60% of the soil water holding capacity) and therefore, we attribute our lower N_2O compared these field studies in part to the lack of periods with very high soil moisture levels.

As mentioned before, the rapid mineralization of organic fertilizers in tropical soils have been associated with increased greenhouse gas emissions (Agegnehu et al. 2017; Tsai and Chang 2019); however, the low CH_4 and N_2O emissions in this study, even after applying the maximum amount of compost allowed in Germany, show that the assumption that the application of large compost amounts leads to increased non-CO_2 emissions can be disputed.

4.3.2 Dynamics of plant-available nutrients

4.3.2.1 NH_4^+, NO_3^-, and available N

The NH_4^+ dynamics show a stable mineralization rate of organic matter throughout the incubation period, with a significant decrease for treatments HM+BC1, HM2, CM2 and CM+BC2, and no significant increase or decrease for the other treatments (Figure 4.3 a). The absence of a persistent NH_4^+ accumulation during incubation indicates a faster nitrification than mineralization rate, leading to a fast turnover to NO_3^- (Paul 2015). Good aeration promotes nitrification, as nitrifying bacteria and archaea are aerobic organisms and therefore, this rapid conversion of NH_4^+ to NO_2^- by *Nitrosomas*, and immediately followed by a second oxidation step to NO_3^- by *Nitrobacter* suggests well-aerated soils in our treatments (Paul 2015). The treatment with mineral fertilizer contained on average 38 mg NH_4^+-N kg^{-1} over the 180 incubation days, while we measured NH_4^+-N concentrations 97% lower in the other treatments and with no significant differences among them (Table 4.3).

The most abundant available N form in the different soil mixtures during the incubation experiment was NO_3^- (Table 4.3). We observed a relatively low nitrification activity with a small but significant NO_3^- increase in all treatments during the course of incubation (Figure 4.3 b). Thus, NO_3^- and NH_4^+ dynamics confirm that the nitrification rate under tropical conditions in well-aerated sandy soils is higher than the mineralization rate and, as a result, any kind of NH_4^+ formed during mineralization is immediately converted to NO_3^-. Over the 180 incubation days, we measured highest NO_3^--N in soil treatments MF and HM2, with an average value of 47 mg kg^{-1}, followed by CM2 and CM+BC2 with 42 mg kg^{-1}, and HM+BC2 with 39 mg kg^{-1} (Table 4.3). From days 120−180, compost treatments with the higher application rate had produced as much NO_3^- as the MF soil. Conversely, we measured a factor of 1.9 (22 mg kg^{-1}) and 4.6 (9 mg kg^{-1}) lower NO_3^--N content in the compost treatments with the lower application rate and control, respectively, compared to that of MF (Table 4.3). Since the N source added in the MF treatment consisted of completely available N, in the form of NH_4^+ and NO_3^-, the NH_4^+-N and NO_3^--N contents measured in this treatment during the incubation experiment were mostly from the mineral fertilizer itself and not the result of mineralization and nitrification processes of the organic matter contained in the soil.

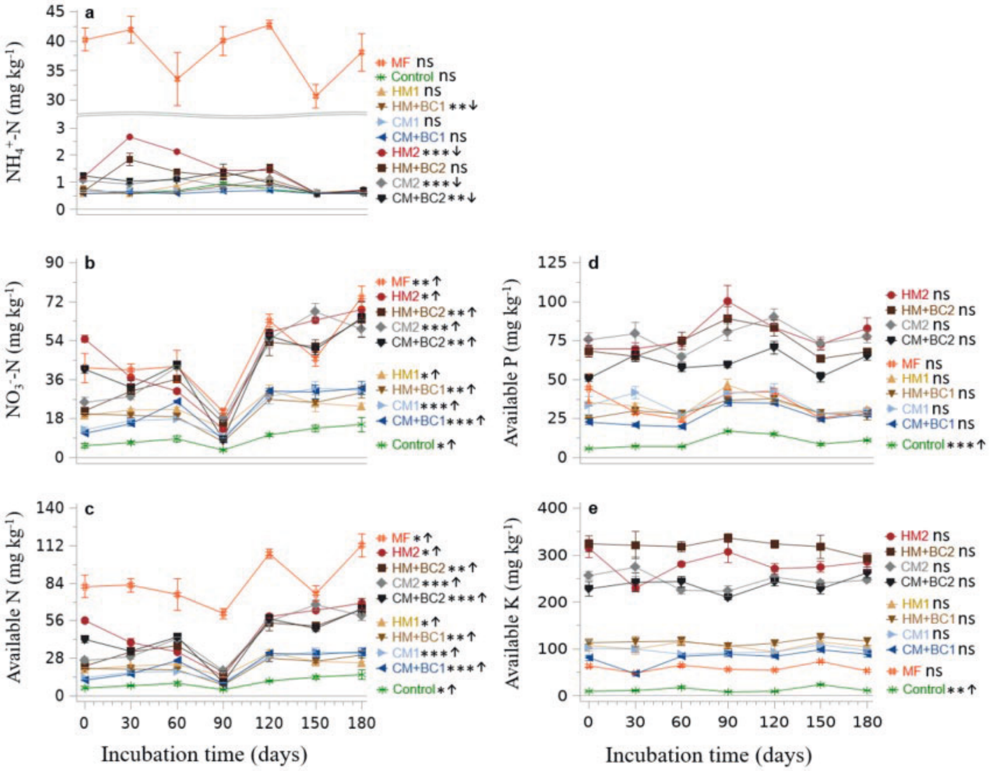


Figure 4.3. *(**a**) Dynamics of NH_4^+-N, (**b**) NO_3^--N, (**c**) available N, (**d**) available P, and (**e**) available K. Soil incubated at 25 °C, without any nutrient addition (control, green asterisk), with mineral fertilizer (MF, orange hash), with a lower compost application rate of: humanure (HM1, yellow triangle), humanure amended with biochar (HM+BC1, light brown triangle down), cattle manure (CM1, light blue triangle right), cattle manure amended with biochar (CM+BC1, blue triangle left), and with a higher compost application rate (three times more than the lower one) of: humanure (HM2, red circle), humanure amended with biochar (HM+BC2, brown square), cattle manure (CM2, gray diamond), cattle manure amended with biochar (CM+BC2, black home down). Values provided as means (± SE) with n=4. The significance between initial and final values is indicated with: ns (not significant, $p>0.05$), * ($p\leq0.05$), ** ($p\leq0.01$), or *** ($p\leq0.001$), and the symbols ↑ and ↓ indicate if the values increase or decrease, respectively.*

Throughout the incubation period, available N in the MF treatment accounted on average for 25% of the total initial N fraction, while a smaller share of 4−9% of the total N in the compost treatments was plant available (Table 4.3). In addition, the net soil N mineralization rates (final minus initial available N) during the 6-month incubation period at 25°C ranged between 4−42 mg kg^{-1}, which corresponds to 2−13% of the initial total N. This N mineralization increased the available N in the control, CM1, CM+BC1, HM+BC2, and CM2 on average by 59%, for MF, HM+BC1, and CM+BC2 by 31%, while the lowest increase of 18% was observed in HM1 and HM2 (Figure 4.3 c). Nevertheless, even with this increasing trend, it is worth noting that the available N fraction of the compost treatments did not reach that of the MF soil, which contained 1.8−2 times more available N than compost treatments with the higher application rate, and 3.8 times more than those with the lower application rate. This demonstrates that even with the maximum amount of compost allowed in Germany, the amount of available N provided by the compost even at incubation temperatures around 25°C over a period of sixth months is not enough to meet the crop N demand. Thus, the concern of increasing the risk of N losses by adding too much N with compost at rates exceeding the current German organic fertilizer regulations is not justified.

Table 4.3. *NH_4^+-N, NO_3^--N, available P and K, total N, P, K and C, and C:N ratio mean values (after repeated measures ANOVA) over the 180 days of the incubation experiment. Soil incubated at 25 °C, without any nutrient addition (control), with mineral fertilizer (MF), with a lower compost application rate of: humanure (HM1), humanure amended with biochar (HM+BC1), cattle manure (CM1), cattle manure amended with biochar (CM+BC1), and with a higher compost application rate (three times more than the lower one) of: humanure (HM2), humanure amended with biochar (HM+BC2), cattle manure (CM2), cattle manure amended with biochar (CM+BC2). Values provided as means (± SE) with n=4. Different superscript letters indicate significant differences at $p<0.05$ among treatments.*

Treatment	NH_4^+-N	NO_3^--N	Available P	Available K	N	P	K	C	C:N
				(mg kg^{-1})					ratio
Control	0.7^{a}	9.1^{a}	10.4^{a}	13.1^{a}	231^{a}	68^{a}	358^{a}	$1,511^{a}$	7.3^{a}
MF	30.1^{b}	46.9^{e}	33.9^{cd}	59.1^{b}	308^{ab}	94^{b}	407^{ab}	$1,487^{a}$	5.3^{ab}
HM1	0.8^{a}	22.5^{b}	33.9^{cd}	105.6^{de}	369^{bc}	85^{ab}	511^{abc}	$2,863^{ab}$	8.2^{bc}
HM+BC1	0.7^{a}	21.2^{b}	30.6^{c}	114.7^{e}	373^{bc}	96^{b}	523^{bc}	$3,908^{bc}$	10.7^{de}
CM1	0.7^{a}	21.3^{b}	34.9^{d}	96.9^{d}	401^{c}	95^{b}	491^{abc}	$3,117^{b}$	7.7^{b}
CM+BC1	0.6^{a}	22.1^{b}	26.6^{b}	81.4^{c}	331^{bc}	81^{ab}	436^{ab}	$3,464^{b}$	11.8^{e}
HM2	1.5^{a}	46.5^{e}	79.0^{g}	280.5^{g}	599^{e}	152^{d}	740^{d}	$5,623^{d}$	9.6^{cdf}
HM+BC2	1.1^{a}	39.1^{c}	73.2^{f}	319.2^{h}	582^{de}	140^{d}	942^{e}	$7,742^{e}$	13.7^{f}
CM2	0.9^{a}	42.2^{d}	77.2^{g}	245.6^{f}	503^{d}	119^{c}	636^{dc}	$4,927^{cd}$	10.1^{cde}
CM+BC2	1.0^{a}	42.2^{d}	60.2^{e}	236.6^{f}	517^{de}	134^{dc}	969^{e}	$6,014^{d}$	11.6^{de}

In compost-amended soils, it can be generally assumed that initial N plant availability is low, since >90% of the total compost N is bound in the organic N pool (Amlinger et al. 2003). Thus, N from organic fertilizers often shows little effect on crop growth in the year of application, due to the slow conversion of this organically bound N into available N, and the medium- and long-term N immobilization in soils (Amlinger et al. 2003; Meena 2020). This N deficiency gap in the first application years and the yield losses that arise with it make organic farming less attractive for farmers (Meena 2020). To come closer to or to reach the amount of available N necessary for optimal crop growth and yield – which here we assume is the level of available N in the MF treatment – the current maximum amount of organic fertilizer prescribed by the German Fertilizer Ordinance has to be increased. For depleted soils, with low organic C content, 3−5 times more compost could be applied every year during the first years not only to reach the available N desired but also to build organic matter in the soil (Kowaljow et al. 2017; Lorenz and Lal 2018; Steiner et al. 2007; Tsai and Chang 2020). Later on, once the available N delivered by the compost reaches the required level, farmers could gradually reduce the compost amount and adjust it, depending on the soil type and decomposition rate. In this regard it is important to also consider the unequal availability of the macronutrients contained in compost –normally being N the limiting nutrient– and therefore, special attention has to be given to the P and K inputs (further discussed in section 3.2.2; Maheshwari 2014; Meena 2020).

The relatively low nitrification activity and the absence of NH_4^+ accumulation during our study confirm that even after applying the maximum recommended compost amount to highly depleted soil under tropical conditions, the risk of over-fertilization and NO_3^- leaching in cropland soils might be low. Furthermore, in this study, all compost treatments, except for CM1, had C:N ratios that either provided a balanced N-mobilization and immobilization (9−11) or favored N-binding (11−14) over the 180 days of incubation (Table 4.3; Amlinger et al. 2003). Thus, since high compost application rates substantially increase available carbon substrates, which can be easily consumed by microorganisms, a sustained increase of the soil microbial biomass and a balanced N release and binding (Amlinger et al. 2003) can also be expected if even larger compost amounts than in our experiment are applied. Furthermore, the compost degradation rates with respect to C in this study also show that after one year under tropical temperature conditions, 15−43% of the large compost amount was degraded. This indicates that compost could represent a stable source of nutrients and therefore, the assumption that such large or even larger compost amounts would increase the risk of nutrient leaching can be disputed. Loss of nutrients from depleted agricultural soils in temperate regions would be even lower then, as any decrease in temperature will decrease the mineralization rate (Amlinger et al. 2003; Meena 2020).

Biochar had a significant impact on the mineral N dynamics only in the treatments containing humanure compost from the higher application rate, demonstrated by the significant reduction in the extractable NO_3^--N in HM+BC2 by 16% with respect to HM2 (Table 4.3). In their incubation study, Tsai and Chang (2020) also observed that the co-application of biochar and excessive compost, i.e. 5% by weight which is three times the recommended doses of organic compost in Taiwan, benefited the agricultural soils by improving NO_3^--N retention. Potential mechanisms that may have contributed to NO_3^- capture by the biochar's porous matrix include its interaction through conventional anion exchange capacity with functional groups and organo-mineral complexes, and through unconventional H-bonds created through the electron donation of π-orbitals of the polyaromatic systems to the electron-deficient hydrogen atoms in water of hydrated NO_3^- (Hagemann et al. 2017; Kammann et al. 2015). Application of biochar has also been demonstrated to modify the biological functionality of soils by providing a habitat for microorganisms due to its highly porous nature and thereby stimulating microbial N immobilization (Agegnehu et al. 2017). This sorption of NO_3^- on the biochar surface or the enhanced N immobilization suggest that co-composted biochar could potentially decrease NO_3^- leaching during field application of humanure compost (Kammann et al. 2015; Lehmann and Joseph 2009), and demonstrates the potential of co-composted biochar as a slow release fertilizer (Hagemann et al. 2017).

4.3.2.2 P and K

The available P and K fractions remained constant for all treatments throughout the incubation period, indicating that there is no significant mobilization or immobilization of these nutrients from or into the organic soil matrix (Figure 4.3 d, e). Contrary to available N, which regardless of the treatment was significantly higher in MF, plant-available P in the compost treatments with the lower application rate was at the same level as the MF treatment, while higher in compost treatments with the higher application rate, on average by a factor of 2 (Table 4.3). Regarding exchangeable K, already amending soil with the lower compost application rate supplied the soil with significantly larger amounts of available K compared to the control and MF treatments (Table 4.3).

Compared to available N, which only had a share of 4−9% of the total N, available P and K accounted for 34−66% and 19−39% of the total initial P and K fraction in the compost treatments, respectively. Even though the N, P, and K added in the MF treatment were added in principle in available form, i.e. as NO_3^--N, NH_4^+-N, P_2O_5, and K_2O, their dynamics show that not all the N, P, and K added was available after addition, with available P and K accounting for 36% and 15% of the total initial P and K fraction, respectively. Acid soils, such as the one used in this study (pH 4.9; Table 4.1), regulate plant availability of P and K by forming Al- and Fe-phosphates and absorbing K onto their surfaces (Glaser and Lehr 2019; Lehmann and Joseph 2009; Paul 2015). This retention is stronger in the case of the MF treatment, as with compost application the soil pH increases (Table 4.1) and therefore, the P sorption by Fe and Al oxides decreases (Glaser and Lehr 2019; Lehmann and Joseph 2009). On the other hand, with compost much larger total P and K fractions were added than in the MF treatment and therefore, a higher amount of available fractions of these nutrients could be extracted from these compost treatments (Table 4.1). This shows that P and K inputs to arable land through compost application can replace mineral P and K fertilizers and are especially suited for highly weathered and depleted soils in the tropics with very low P, K and organic matter contents (Kowaljow et al. 2017; Steiner et al. 2007).

Biochar amendment reduced the available P content of the treatments CM+BC1, CM+BC2 and HM+BC2 by 24%, 22%, and 7%, respectively, during incubation (Table 4.3). Conversely, we only detected a significant biochar effect on available K for treatment CM+BC1, i.e. at the lower application rate. As with nitrate, biochar has also the ability to absorb K, orthophosphate as well as organic P compounds and thereby, reduce the risk of P and K leaching from agricultural soils (Agegnehu et al. 2017; Glaser and Lehr 2019). Similar to nitrate, a mechanism identified by Hagemann et al. (2017) includes the interaction of nutrients with a heterogeneous, porous and hydrophilic organic coating present on co-composted biochar surfaces. This mechanistic evidence shows that the use of co-composted biochar has the potential to yield optimal agronomic benefits (Hagemann et al. 2017).

In their soil classification system for arable and grassland soils, VDLUFA (1999) and VDLUFA (2018) categorize soils into five classes (A−E) according to their available P and K contents. Concentrations of these available nutrients for soils of class C range from 31−60 mg kg^{-1} for P and from 50−80 mg kg^{-1} for K, and are considered ideal for achieving optimal yields. To maintain optimal yields as well as P and K contents, P and K fertilization amounts for class C soils should be according to the P and K removal of the harvested products. Soils of class A, like that in our study, are poor in available P (<1.5 mg kg^{-1}) and K (<30 mg kg^{-1}) and therefore, P and K fertilization, in comparison to that for C soils, should be substantially increased for attaining optimal yields until P and K contents of class C are reached in the long run.

In our incubation study, the available P and K inputs in all compost treatments were below the recommended ranges indicated by Landwirtschaftskammer Niedersachsen (2018) for growing maize. According to their fertilization recommendation, an amount of 70−84 kg ha^{-1} of available P and of 224−249 kg ha^{-1} of available K is required for class A soils in order to achieve medium yields for maize. This shows, as with available N, that even after adding the maximum compost amount allowed by the German Fertilizer Ordinance, i.e. three times the mineral fertilizer amount in total N terms every three years, the available P and K levels that are recommended to add for poor P and K class A soils were not exceeded. We thus recommend applying larger amounts of compost in the first years to nutrient-poor soils to alleviate K and P deficiency, especially in highly weathered and strongly leached soils of the tropics, such as Oxisols and Ultisols. Furthermore, the potential of compost to increase SOM and thereby to increase the amount of binding sites for mobile nutrients, could reduce the risk of leaching, and even further with biochar addition, as demonstrated by our results. Once available P and K levels according to class C soils are reached, the amount of compost applied has to be reduced and adjusted so that it replaces the amount of P and K taken by the plants. In this regard, Maheshwari (2014) reports findings from a long-term compost field experiment in southwestern Sydney that show lower levels of soluble P in the runoff water of soils treated with compost when compared to conventional farming practices. Nevertheless, in their environmental risk assessment they also conclude that soil P levels reached after large applications of garden organic compost, i.e. in their case 125 dry t ha^{-1}, can also build up over time and thereby reach an environmental limit. Thus, identifying management practices to prevent nutrient deficiency as well as excess are crucial components for increasing food security and avoiding the risk of leaching. Leaching would not only lead to a loss of valuable nutrients, but would also pose a risk to water quality and the environment (Glaser and Lehr 2019; Maheshwari 2014).It is also crucial to keep in mind that if every year for a long period, the amount of N required by the crop is added in the form of compost, after a certain time too much P and K will be added relative to N. Once this point is reached, it will not be feasible anymore to add compost as the only fertilizer, as the nutrients will most probably become imbalanced, and therefore, the missing N will have to be added through supplementary synthetic N fertilizer. The use of appropriate combination of mineral fertilizers along with compost has proved to be beneficial in improving and sustaining soil fertility (Maheshwari 2014; Meena 2020). This highlights the importance to use crop requirements, soil nutrient levels, soil properties as well as weather factors as criteria for determining compost application amounts (Agegnehu et al. 2017; Amlinger et al. 2003; Maheshwari 2014).

These P and K results, together with the N and C dynamics demonstrate for the first time that using human excreta-biochar compost, especially in large amounts, has the potential to quickly alleviate P and K soil deficiencies and to stabilize organic matter for longer time in soil, with low risk of nutrient leaching.

4.4 Conclusions

Our study provides for the first time an insight into the N, P, K and greenhouse gas dynamics when applying human excreta compost, with and without biochar, and especially when applying large compost amounts under tropical temperature conditions to a highly nutrient-depleted sandy soil. We compared these mineralization dynamics with those of cattle manure compost (with and without biochar) and mineral fertilizer. We found that after six months of incubation (i) 12−40% of compost added was mineralized, ii) co-composted biochar had a stabilizing effect on SOM, iii) biochar addition did not have a significant effect on CH_4 and N_2O emission, but significantly reduced the extractable nitrate and available P and K, and iv) the N availability of compost after applying the maximum compost amount allowed by the German Fertilizer Ordinance did not exceed 5−9% of the total N supply, and therefore, the risk of nitrate leaching was very low. Our findings open ways to improving soil health and food security with opportunities for climate change mitigation and for water pollution reduction through enhancing soil nutrient retention, not only with compost produced from cattle manure, but also from human excreta.

To maximize the agronomic and environmental benefits of large applications of compost further investigations are necessary, which require a critical evaluation of compost application restrictions in the national regulation schemes, including accounting for the immediate N availability of compost, as well as its mineralization dynamics in soil.

The implementation of carbon sequestration practices and nutrient supply strategies through large compost application amounts need to be prioritized in regions with highly degraded and depleted agricultural soils. Furthermore, the use of nutrient management practices, such as crop rotations with legumes, may offer further solutions for reducing P losses. This could also represent a potential strategy to cover the protein supply for animal cattle feed and thereby move towards a more sustainable economy.

As human excreta show high potential for providing N, P, and K for agriculture, options for recycling nutrients from human excreta must be further investigated, and legal restrictions on the use of this material for agricultural purposes should be reevaluated.

5

SYNOPSIS

5.1 Summary

Chapter 2 − Nutrient dynamics during composting of human excreta, cattle manure and organic waste affected by biochar:

Humans generate millions of tons of organic waste every day, including kitchen waste, cattle manure and human excreta. These kinds of waste are rich in macro- and micronutrients and organic compounds. However, this waste is not being managed in a way that allows us to derive value from its reuse, whilst limited access to mineral and organic fertilizers, and land degradation threaten food security, particularly in developing regions. Holistic approaches, such as ecological sanitation (EcoSan), may offer a new way of redefining human excreta as a valuable resource rather than waste. Thermophilic composting is one of the EcoSan options that offers a hygienically safe solution to converting human fecal material and urine, as well as animal manure, together with other organic waste materials such as vegetable scraps, into compost as a soil amendment. Moreover, given the lack of sanitation and appropriate waste management, as well as the health problems associated with it in many regions of developing countries, this integrated sanitation approach offers a new way of sanitation provision and waste management to communities that would otherwise not receive centralized sanitation due to financial and infrastructural limitations. Here, we composted human excreta with kitchen scraps, and separately cattle manure, as appropriate technology option by using a traditional compost box and sawdust, straw and biochar as bulking materials. The aim was to produce a non-phytotoxic compost free of pathogens, but rich in nutrients that can be used for improving soil fertility. To evaluate the feasibility of this appropriate-technology composting process, the suitability of the type of manure used, and the effect of biochar during composting, we followed the dynamics of key nutrients and physical and chemical parameters of four composting treatments: human excreta or cattle manure, with and without biochar.

Feasibility of the appropriate-technology composting process: Temperatures in the compost treatments reached values above 60°C for 7, 6, 5, and 8 consecutive days for treatments containing human excreta, human excreta with biochar, cattle manure, and cattle manure with biochar, respectively. The duration of these hygienization temperatures complied with the hygienic sanitation treatment requirements for organic waste according to the German Biowaste Ordinance. For all treatments, P and K were preserved during composting, whereas N, Ca and Mg losses were low or in the expected range. Moreover, final germination index values indicated the absence or only negligible concentrations of water-soluble phytotoxic compounds. Therefore, as assessed by the low N, P, K, Ca, and Mg losses, the temperature profile, and germination index, we implemented a well-running and hygienically safe thermophilic composting process by using a traditional compost box and the bulking materials sawdust, straw and biochar as appropriate technology option.

Comparing the type of manure used: In general, the dynamics of nutrients and different parameters during the composting process of both types of excreta behaved similarly, with a few exceptions. Cattle manure composts showed significantly higher final pH values than the humanure counterparts, which we associated with the 2.2−2.3 times larger final NO_3^- in the humanure treatments, which also suggests larger nitrification activity in these composts compared to the cattle manure ones. Even though contents of N, P, and K in human excreta are expected to be larger than in cattle manure, the latter showed significantly higher final TN contents, which we attributed to the larger sawdust amount we used during the collection of human excreta compared to that used for cattle manure. The larger amount of this recalcitrant C source in the humanure treatments was reflected in their higher TN losses compared to those from the cattle manure treatments. The higher amount of readily available C compounds and more balanced supply of C and N in the cattle manure treatments resulted in a better incorporation of N into the organic fractions or its immobilization by microbial biomass.

Evaluating the effect of biochar during composting: The use of biochar proved our hypotheses that the losses of organic matter, organic C, and nutrients would be reduced, and that biochar would increase thermophilic temperatures and aeration conditions during composting and the pH of the final composts. We found that biochar reduced losses of organic matter by 18−23% and C 33−42% in both humanure and cattle manure compost. In regard to N losses, amending human excreta with biochar reduced these losses by 49%, while by 100% in the case of cattle manure. Biochar also decreased the extractable amount of NO_3^- by 32−36% in the final composts, suggesting that amending composting mixtures with biochar may represent a strategy to decrease N losses from agricultural soils via nitrate leaching. When summing up temperatures for days with temperatures >55°C, we observed a significant increase by 24% for cattle manure with biochar with respect to its control without biochar, suggesting that biochar stimulates decomposition processes by improving aeration conditions. This improved air diffusion inside the compost mixture was also indicated by the significantly lower moisture content in the final biochar-amended composts. The addition of alkalinity via the ash content in biochar significantly increased the pH values of the final biochar-amended composts. In contrast, we did not see any biochar effect on the cation exchange capacity, electrical conductivity, germination index and on the total nutrient contents of the final composts.

Chapter 3 − Biochar addition reduces non-CO_2 greenhouse gas emissions during composting of human excreta and cattle manure

In 2017, approximately 4.2 billion people worldwide still lacked safely managed sanitation, including 1 billion people relying on pit latrines, and 673 million people that still defecated in the open, e.g. in street gutters, behind bushes or into open bodies of water. This inadequate sanitation and lack of a proper management of this critical human waste not only endanger public health, but also produce avoidable GHG emissions like CH_4 and N_2O, and a permanent drain of nutrients from agricultural and food systems. Meanwhile, as nutrients and gases from this organic material accumulate in our ecosystems, regions with the lowest sanitation coverage, such as Sub-Saharan Africa, also experience limited access to fertilizers and the most severe land degradation in the world. Ecological sanitation via thermophilic composting may tackle these challenges, as it represents a feasible way to treat and transform human excreta, and other animal manures into a pathogen-free and nutrient-rich fertilizer and soil amendment. Furthermore, properly treating these excreta, e.g., by combining the use of bulking materials, such as sawdust and straw, and additives, such as biochar, with this sanitation strategy may provide a potential way to reduce GHG emissions and to reduce the pile-turning workload. Here, we measured GHG and NH_3 emissions during the thermophilic composting of human excreta, and of cattle manure when using appropriate technology such as a traditional compost box, and sawdust, straw and biochar as bulking agents for pile aeration strategy. We particularly aimed to assess the impact that this appropriate-technology composting process has on the atmosphere and to evaluate climate change mitigation potential of biochar during the treatment of these excreta.

Impact of the appropriate-technology composting process on the atmosphere: We determined total GHG emissions over the 185 days of composting of 6921−8076 g CO_2-C_{eq} m^{-2}. However, since CO_2 is of biogenic origin and, therefore, climate-neutral, for the total GHG emissions only CH_4 and N_2O emissions were considered, which ranged from 918−3441 g CO_2-C_{eq} m^{-2}. When comparing the two types of manure used, we found higher CH_4 emissions in the cattle manure control compared to the humanure control, which we attributed to the larger amount of methanogens in the digestive tract of ruminant animals, and therefore, in their manure.

Evaluating the climate change mitigation potential of biochar during composting: We found that biochar reduced the emissions of N_2O by 56−57% and of CH_4 by 65−91% during composting, which we attributed mainly to the improved aeration caused by the biochar's highly porous structure, increased CH_4 diffusion and availability for methanotrophs, and NH_4^+ and NO_3^- adsorption by biochar and thereby reducing their availability for nitrification and denitrification, respectively. In contrast, we did not observe a significant effect of biochar on CO_2 and NH_3 emissions. In terms of non-CO_2 GHG emissions, we found that the addition of biochar reduced on average the sum of the emissions of CH_4 and N_2O by 2.2 kg of CO_2-C_{eq} for humanure with biochar and by 5.5 kg of CO_2-C_{eq} for cattle manure during the composting process, which corresponds to a non-CO_2-GHG emission mitigation of 51−71%. Our data suggest that thermophilic composting with biochar addition and appropriate technology represents a potential waste management practice that can contribute to climate change mitigation by reducing GHG emissions.

Chapter 4 − Nutrient and CO_2 dynamics after application of biochar-amended human excreta and cattle manure compost to sandy soil under tropical conditions:

Land degradation is occurring on 33% of the Earth's soils, in both low and high-income countries; however, its impact is especially severe in agricultural lands of developing tropical regions, particularly in areas of the world where proper sanitation and affordable mineral fertilizers are not available such as Sub-Saharan Africa. The depletion of soil organic matter (SOM) and nutrients not only poses a threat to global food security, but also has severe implications for our environment, including water quality and climate change through increased CO_2 emissions. Like animal manure, human excreta have a great fertilizer and soil amendment potential, as they not only contain essential plant nutrients, such as N, P, K and other micronutrients, but also organic matter. Thus, the use of human and animal excreta-derived compost, and especially of co-composted biochar, in agriculture has the potential to counteract soil degradation, contribute to climate change mitigation, increase soil nutrient retention and reduce the use of synthetic fertilizers. Here, we conducted a 6-month incubation experiment to follow the nutrient and GHG dynamics of four types of compost (human excreta or cattle manure compost, with and without co-composted biochar) at two application rates (170 kg total N ha^{-1}, and three times this amount) to a highly depleted sandy soil at tropical soil temperature (25°C) and compared them with those of a mineral fertilizer (MF) treatment. To elucidate the role of these composts as fertilizers and soil amendments, we specifically aimed to evaluate the mineralization dynamics, especially of large compost application amounts, and the potential that biochar-compost mixtures have in increasing C sequestration and reducing nutrient leaching in agricultural soils.

Mineralization dynamics during compost application in soil: Overall, we found degradation rates lower than 40% for all compost types over the 6-month incubation period at 25°C. Nitrogen availability in the high-rate compost treatments was comparable with the common-practice MF treatment, whereas P and K availability exceeded the MF treatment. In contrast, in the low-rate compost treatments, P and K availability was comparable to the MF treatment, whereas N availability was significantly below the MF level. This shows that the plant demand for P and K based on maize at tropical temperatures can be fully met with human excreta-derived compost application, however, a N deficiency gap between the amount of available N necessary for optimal crop growth and yield remains.

Biochar-compost mixtures' potential in increasing C sequestration and reducing nutrient leaching: While N_2O and CH_4 emission rates were generally very low, CO_2 emissions of the soil amended with the high rate of human excreta compost plus biochar were 36% lower than for the human excreta compost without biochar. In the same treatment, biochar decreased the extractable NO_3^- by 16% and available P by 7%, thereby reducing the risk of N and P leaching losses. Biochar addition also decreased available P in the biochar-cattle manure compost treatments by 22−24%. Our findings suggest that the use of co-composted biochar in agriculture has important implications for the future of a sustainable return of nutrients and organic matter to agricultural soils, with opportunities for C sequestration and nutrient retention.

5.2 Synthesis

Sanitation systems that aim to close the "human nutrient" loop by re-integrating nutrients from treated human feces back to the soil may offer synergistic solutions to multiple challenges in sanitation, health, water, and agriculture. Several studies have explored the use of circular sanitation systems (Langergraber and Muellegger 2005; Moya et al. 2019c; Simha and Ganesapillai 2017), the potential global or local nutrient recovery from urine and feces (Akram et al. 2019; Berendes et al. 2018; Harder et al. 2019; Trimmer and Guest 2018), the perception of the use of human excreta in agriculture (Gwara et al. 2021; Junghanns and Beery 2020), sustainable sanitation gaps in global climate policy and financing (Dickin et al. 2020), the treatment of human excreta via thermophilic composting (Preneta et al. 2013), from a hygienic (Berendes et al. 2015; Piceno et al. 2017) and GHG (McNicol et al. 2020; Ryals et al. 2019) point of view, and the use of human excreta derived products in agriculture (Liu et al. 2018; Moya et al. 2019b). However, to our knowledge, no studies exist that evaluate the entire process of such EcoSan concepts from a nutrient dynamics and GHG emissions perspective, i.e., since the moment these kinds of waste are treated until they are used in agriculture. Closing these knowledge gaps of the biochemistry of the nutrient cycle of human excreta could decrease the reluctance to break cultural taboos and accelerate the development and implementation of safely managed sanitation and of sustainable agricultural practices. This work complements the limited number of existing scientific studies on this topic by evaluating the feasibility of an appropriate-technology EcoSan concept as an approach to climate-smart agriculture. For this, we assess if this concept is suitable for safely and properly treating human and animal excreta together with other organic waste materials and biochar via thermophilic composting, and for producing and using compost for fertilization and soil amendment purposes (Figure 5.1).

For the collection of humanure (feces, urine and toilet paper), the constructed dry and container-based toilets and the use of sawdust as C-rich cover material proved to be a suitable alternative not only to increase access to basic sanitation, but to collect this critical waste, as the toilets were safe and easy to operate and no smell was perceived during their operation and when setting up the composting process.

Figure 5.1. *Our proposed ecological sanitation approach for climate-smart agriculture in Ethiopia, via thermophilic composting of human excreta with kitchen scraps and biochar.*

When evaluating our appropriate-technology thermophilic composting from a hygienic, nutrient, stability and GHG perspective, the duration of hygienization temperatures, low nutrient losses, absence or very low levels of phytotoxic substances, and the relatively low GHG emissions demonstrate its appropriateness as waste management option. We found that composting human excreta compared to other animal manures, such as that from cattle, does not lead to a significant increase in GHG emissions, and it provides also a high load of nutrients that can be transferred to the fields. In this regard, the P and K delivered by both compost application rates, i.e., the equivalent of 170 kg total N ha^{-1}, and three times this amount, showed that the plant demand for P and K based on maize at tropical temperatures can be fully covered through human excreta-derived compost application already with the lower application rate, and therefore could be especially suited for highly weathered and depleted soils in the tropics with very low P, K and organic matter contents. In contrast, there was a mismatch between available N from compost and the plant N demand calculated for one maize cropping season, with both compost application rates being able to provide only 25.7−56.3% of the available N of the mineral N fertilizer treatment. This demonstrates that even with the maximum amount of compost allowed in Germany, the available N provided by the compost even at incubation temperatures around 25°C over a period of sixth months is not enough to meet the crop N demand and that therefore the concern of over-fertilization and NO_3^- leaching in cropland soils with compost at rates exceeding the current German organic fertilizer regulations is not justified. For depleted soils, with low organic C content, we therefore recommend applying 3−5 times more compost every year during the first years not only to reach the available N desired but also to build up the soil organic matter pool. However, the available P and K delivered by the compost needs to also be considered to prevent the risk of leaching. Thus, identifying management practices to prevent nutrient deficiency as well as excess are crucial components for increasing food security and avoiding environmental pollution. Thus, EcoSan via thermophilic composting can have a substantial positive impact on sanitation on one hand, and on the other hand on improving the nutrient status of agricultural fields and to increase access to nutrient sources for developing regions. Nevertheless, for achieving optimal yields and to avoid excess of P and K, the application of compost will have to be combined with some supplementary mineral N fertilizer in the long run.

When applying the maximum compost amount allowed by the German fertilizer regulations, a total fresh weight of 94−95 tons of human excreta compost would be required (equivalent to 35−38 tons dry weight) to fertilize 1 ha of depleted sandy soil, like the one used in our incubation study, for growing maize for one cropping season (Appendix D: Table D.1). It is worth mentioning again that this high application rate is only required in the beginning, e.g. the first three years, as afterwards the mineralization of the organic material accumulated in the soil during those years will provide increasing amounts of N to the soil. To produce the amount of compost for those first application years, based on the amount of each feedstock used and the mass and N losses during our composting study, a fresh amount of 193−290 tons of humanure (this corresponds approximately to 99,742−149,871 excretions of human excreta per capita), 370−560 tons of kitchen waste, 18−27 tons of teff straw and for the biochar-compost mixture, an amount of 23 tons of biochar would be needed per hectare per year. The amount of total fresh compost needed to fertilize 1 ha of maize could be potentially reduced by reducing the N lost during the composting process of the human excreta treatments, e.g., by adding more easily degradable C sources, such as straw, to increase N immobilization (Reichel et al. 2018) and/or by collecting the leachate and returning it to the compost pile.

Concerning GHG emissions, we found that the contributions of CH_4 and N_2O during composting are moderate compared to that of CO_2, with a share of 0.3−2.6% for CH_4 and 13.7−35.8% for N_2O of the total cumulative CO_2-C_{eq} emissions. In contrast, CH_4 and N_2O production from all four types of composts at the two different application rates remained very low and did not play an important role during the soil incubation period. As in any decomposition process, it is expected that part of the OM of the original feedstocks becomes mineralized and converted to CO_2 during composting. However, these CO_2 emissions are climate-neutral as they are derived from OM that is relatively freshly produced from atmospheric CO_2. After assessing the total cumulative non-CO_2-C_{eq} emissions during production and application of compost to soil during a 6 months incubation period at a high application rate, an amount of 91 g of non-CO_2-C_{eq} was generated per kg of dry compost in the case of cattle manure control and of 70 g for human excreta. In contrast, both composts containing biochar generated 20 g of non-CO_2-C_{eq} per kg of dry compost.

When comparing the impact of the production of human excreta-derived compost on the greenhouse gas budget of the atmosphere with that of synthetic fertilizers, composting human excreta could save a substantial amount of energy during the process and, therefore, has a big CO_2 saving potential. Comparing the results of our study with urea as common N fertilizer, the production and use of 1 kg of urea-N would emit 11.2 kg of fossil fuel CO_2-eq per kg of urea-N (Fertilizers Europe 2011), while with human excreta-derived compost 4.6 and 1.6 kg of non-CO_2-eq per kg of compost-N with and without biochar, respectively, would be emitted (chapter 3: Table 3.1; Appendix D: Table D.2). Since CO_2 produced during the composting process and compost application in soil is of biogenic origin and, therefore, climate-neutral, we based this comparison on non-CO_2-eq emissions, i.e. we only considered CH_4 and N_2O emissions. Regarding transportation of the feedstocks to the composting facilities and then of the final product to the different farms, only a small CO_2 contribution is expected due to the short transport range, whereas for mineral N, P and K, a distance of thousands of kilometers and even an intercontinental transport is common. As already mentioned, the plant demand for P and K could be fully covered with appropriate compost application rates, and CO_2 emissions from P and K fertilizer mining and transport could thus be avoided. Therefore, composting would provide an even larger, especially transport-related C saving potential, but also associated to mining and production.

Other very important findings from our work are the benefits that biochar has on composting, including the increase of temperature during the thermophilic phase, making this treatment option even safer with respect to hygienization. Another positive effect of biochar is its reduction potential of CH_4 and N_2O emissions by a factor of 3−11 and 2, respectively, its NO_3^- binding potential, and its stabilization effect on C, leading to a lower C loss that is beyond the stable C content of the biochar. This stabilization effect of OM is, however, no indication of inhibition of microbial activity, as temperatures reached during the active phase of composting where significantly increased by biochar, suggesting a higher microbial activity during the thermophilic phase. During our incubation experiment, we also observed the potential that the addition of co-composted biochar to soil had on binding NO_3^-, stabilizing organic matter and decreasing C mineralization. Therefore, with biochar-compost mixtures, the buildup of humic substances in the soil could be further accelerated, and soils could quickly build up SOM, while retaining nutrients.

To effectively contribute to climate change mitigation and thereby successfully achieve the net-zero emissions scenario by 2050, sustainable soil C sequestration practices, e.g., through regular application of compost, need to be rapidly scaled up and implemented (Amelung et al. 2020). The findings from Clayton et al. (2021) highlight the importance of rapidly implementing sustainable agricultural practices that increase SOC. In this study, they observed a threshold of a SOC content of 1%, below which the efficiency of microorganisms in the immobilization of carbon, i.e., the microbial C use efficiency, was much lower than above the threshold, very likely due to the fact that the microbes needed more C for just maintaining their biomass and could not use it for growth. As a result, more CO_2 per gram of microbial biomass was emitted compared with soils with an SOC content above 1%. This lower efficiency implicates that if SOC content is not quickly increased above 1%, it is going to be very difficult to store more C in soils.

As already mentioned, excreta contain biogenic C that, when it becomes CO_2, it is not considered part of the human-caused emissions of CO_2 to the atmosphere. However, if some of this C can be kept from being converted to CO_2 and instead used as a sustainable soil carbon sequestration practice, a substantial contribution to removing C from the atmosphere can be achieved by applying compost to soils (Amelung et al. 2020), especially biochar-compost mixtures (Agegnehu et al. 2017; Glaser et al. 2001). In our 6-month soil incubation study, we found degradation rates of 24−40% of of the composting mixture and a relative reduction of this degradation rate of 50% with biochar. However, we cannot say for how many years this stability translates into, as this depends on the particular materials, soil and weather conditions, and other factors (Agegnehu et al. 2017; Amlinger et al. 2003; Maheshwari 2014). Nevertheless, even if after one year of compost application a fraction of C has become mineralized, a substantial amount of compost C will remain in the soil after one year even at soil temperature of 25°C. Thus, with a regular C input, especially combined with biochar, the SOC pool can build up over time.

One example of soils rich in SOC, reaching values of 4% or more, are garden soils called hortisols, which have been treated for hundreds of years with OM, such as manure, slurry, compost, kitchen scraps and even human excreta (Kuratorium Boden des Jahres 2017). A regular introduction of organic materials will not only increase the SOC content and thereby mitigate climate change, but will also change the characteristics of the soil, e.g., from an acidic, nutrient- and OM-poor sandy soil to a nutrient- and C-rich soil with neutral pH, which would not be possible to achieve just with mineral fertilizer (Maheshwari 2014). Concerning pH and its effect on nutrient availability, in our final composts with a pH of 7.4−8.6, the available P and K fractions amounted to 11−15% of the respective total P and K content. When applied to an acidic soil, the pH of the compost-soil mixture increased to 6.1−7.0, and the fraction of available P and K increased and contributed 23−87% of the total P and K fractions, respectively.

Our findings demonstrate the feasibility of this appropriate-technology EcoSan concept via thermophilic composting as an approach to climate-smart agriculture. This EcoSan concept offers potential solutions to critical sustainability challenges in sanitation, health, water, and agriculture, especially for developing regions with infrastructural and economic limitations; however, this integrated approach could be easily applicable to other regions worldwide. By addressing these challenges, this approach could help tackle many of the Sustainable Development Goals (SDG; UN 2020), which become more urgent as the effects of climate variability and change increase (WHO 2019a). In particular, this holistic approach may (1) increase access to safely managed sanitation and clean water (SDG 6: Clean water and sanitation), reducing public health issues and water pollution, (2) promote the recycling and reuse of organic materials, such as human excreta (SDG 12: Responsible consumption and production), (3) increase soil fertility and access to nutrient inputs for agriculture, sustainably increasing agricultural productivity (SDG 2: Zero hunger), (4) build resilience to climate change (SDG 11: Sustainable cities and communities), and mitigate climate change by reducing GHG emissions compared to alternative waste disposal methods, synthetic mineral fertilizer production and transport, and promoting C sequestration in agricultural ecosystems (SDG 13: Climate action).

5.3 Conclusions and outlook

Ecological sanitation via thermophilic composting of human excreta by using appropriate technology could be a feasible option for implementing a climate-smart agriculture concept that increases access to safely managed sanitation, improves waste management, soil health and food security, and opens ways for climate change mitigation in developing countries. We found that our appropriate-technology thermophilic composting was effective in minimizing GHG emissions and nutrient losses, at producing a stable compost rich in nutrients, and in treating human excreta in a hygienically safe way. With our research we also demonstrated that human excreta-derived compost, especially biochar-compost mixtures, may contribute to climate change mitigation, nutrient retention and to decrease the dependency of synthetic fertilizers, especially of mineral P and K. Ecological sanitation via thermophilic composting with biochar addition represents therefore a feasible integrated sanitation strategy and climate-smart approach that may further contribute to climate change mitigation by reducing GHG emissions in sanitation, waste management and through soil carbon sequestration in agricultural ecosystems. It additionally opens ways to improving soil health, food security and water quality by enhancing soil nutrient retention. Our findings have important implications for the future of a sustainable recycling of nutrients and organic matter from human excreta and their further reuse to improve soil fertility and agricultural productivity, particularly in areas of the world with financial and infrastructural limitations.

The focus of this PhD research work was on the community level of especially least developed countries, such as Ethiopia. However, our findings could provide support for the implementation of a diverse range of business models or initiatives at larger scale, for low- and middle-income as well as industrialized countries. By recognizing the value of human excreta, sanitation as well as other organic waste reuse-based businesses can completely reinvent the economics of sanitation and agriculture.

With this work, we were able to address all the objectives. However, given the implications of this approach or other initiatives that also seek to close the human nutrient cycle and to break the cycle of disease, we identified the following gaps to make these sustainable, integrated sanitation/waste management/agriculture strategies successful:

i) ***More research on options for recycling nutrients from human excreta:*** Currently, there are only few studies in the scientific literature investigating the efficiency of treatment technologies in recovery of nutrients from human waste and removal of pathogen or other pollutants, such as pharmaceuticals and hormones.

ii) ***Legislations should include the use of human excreta for agricultural purposes:*** Legal restrictions on the use of this material for agricultural purposes should be re-evaluated, and clear policies need to be developed to ensure the safety and quality assurance of the production and use of human excreta-derived fertilizers. Moreover, specific schemes, such as EcoSan models using container-based dry-toilet systems, should be evaluated separately from, e.g., biosolids. Governmental support could help to enhance research, knowledge, awareness and social acceptance, and thereby overcome perceived barriers of the use of nutrients from human excreta.

iii) ***Better reporting on sanitation investment and status:*** The lack of reporting and data on sanitation shows the lack of prioritization that many countries especially in developing regions such as Sub-Saharan Africa have in this sector. Inadequate knowledge on the status of safe sanitation poses a significant health threat and can create additional costs. Therefore, better reporting is needed on sanitation investments to prioritize and implement sanitation solutions suitable for specific local and regional contexts.

iv) ***A critical evaluation of compost application restrictions in regulation schemes:*** Currently, regulations for the use of compost as fertilizer in many countries do not consider the slow release of organically bound nutrients and, for example in Germany, limit its application to a maximum amount of 510 kg of total N per hectare over a period of three years. To maximize the agronomic and environmental benefits of large compost application rates, further investigations are necessary. However, these need to be supported by legislations that recognize the fertilizer and long-term C sequestration potential of compost.

v) ***Prioritize C sequestration practices and nutrient supply strategies in regions with highly degraded and depleted agricultural soils:*** To quickly and successfully achieve the net-zero emissions scenario, sustainable SOC sequestration practices, such as regular application of large compost amounts, need to be prioritized in regions with highly degraded and depleted agricultural soils and with highly limited access to nutrients.

vi) ***Long-term field trials:*** Additional field trials over several crop seasons are required to evaluate the long-term effect of human excreta-derived compost, especially of large application amounts, on soil fertility and crop performance. These studies should also identify management practices that not only prevent nutrient deficiency but also excessive nutrient levels by considering crop requirements, soil nutrient levels, soil properties, and weather conditions as criteria for determining compost application amounts.

APPENDICES

Appendix A: supplemental material for chapter 2

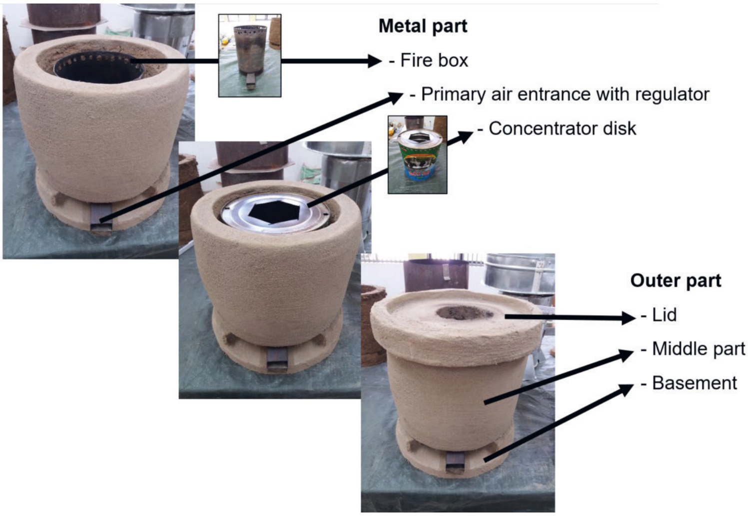


Figure A.1. *Components of the top-lit up-draft micro-gasifier (Noah stove; photos by Daniela Castro Herrera).*

Figure A.2. *Operation of the top-lit up-draft micro-gasifier (Noah stove; photos by Daniela Castro Herrera).*

Table A.1. *Physical and chemical properties (in dry weight) of the biochar used in this study. These were measured according to the guidelines of the European Biochar Certificate (ECB).*

Properties	Biochar	EBC threshold [a]	
		Basic	Premium
HTT [b] (°C)	500–600		
Water content (%)	7.04		
Ash content (% w/w)	8.41		
Volatile organic compounds (%)	10.36		
BET [c] surface (m^2 g^{-1})	278.18	> 150	
Bulk density (kg m^{-3})	265		
pH	10.69		
Salt (g kg^{-1})	1.122		
Elemental composition (%)			
H	1.61		
C	77.71	> 50	
Corg	76.82		
N	0.32		
O	4.88		
S	0.03		
Molar ratios			
H/C_{org}	0.25	< 0.7	
O/C_{org}	0.05	< 0.4	
Trace elements (mg kg^{-1})			
As	< 0.8	< 13	< 13
Pb	< 2	< 150	< 120
Cd	< 0.2	< 1.5	< 1
Cu	5	< 100	< 100
Ni	4	< 50	< 30
Hg	< 0.07	< 1	< 1
Zn	32	< 400	< 400
Cr	1	< 90	< 80
B	17		
Mn	1690		

Table A.1. *Continuation.*

Properties	Biochar	EBC threshold [a]	
		Basic	Premium
Other organic compounds			
PCBs[d] (ng kg^{-1})	0.0881	< 0.2	
PAHs[e] (mg kg^{-1})	10	< 12	< 4
Dioxins and furans[f] (mg kg^{-1})	0.497	< 20	

[a] According to EBC, 2012.

[b] HTT: Highest Treatment Temperature.

[c] BET: Brunauer, Emmett and Teller method.

[d] PCBs: Polychlorinated biphenyls. Reported as the sum of PCBs and WHO toxic equivalent (TEQ) using the WHO-TEFs(2005).

[e] PAHs: polycyclic aromatic hydrocarbons. Reported as the sum of the 16 priority PAHs pollutants according to the US Environmental Protection Agency (16 EPA-PAH).

[f] Reported as the sum of dioxins and furans and WHO toxic equivalent (TEQ) using the WHO-TEFs(2005).

Table A.2. *The main physical and chemical properties of the composting materials (in dry weight). Values provided as means (± SE) with n=4.*

Properties [a]	Humanure [b]	Cattle manure [b]	Vegetable scraps	Teff (*Eragrostis tef*) straw
MC (% fresh wt.)	75.7 ± 1.3	74.4 ± 1.5	92 ± 1 [c]	6.5 ± 0.1
TOC (%)	40.8 ± 3.0 [d]	42.5 ± 0.4	43.2 [e]	42.4 ± 0.9 [f]
TN (%)	0.61 ± 0.02	0.83 ± 0.11	3.6 [e]	1.0 ± 0.5 [f]
C:N ratio	67.1 ± 1.9 [d]	51.5 ± 7.5	12 [e]	42.4 [f]
pH	8.1 ± 0.3 [g]	8.2 ± 0.6 [h]	–	–
EC (mS cm^{-1})	1.53 ± 0.02	1.63 ± 0.01	–	–
CEC ($mmol_c$ kg^{-1})	7.4 ± 1.4	17.1 ± 2.1	–	–
NH_4–N (mg kg^{-1})	2830 ± 215	3428 ± 1041	–	–
NO_3–N (mg kg^{-1})	0.88 ± 0.20	0.53 ± 0.13	–	–
P_{av} (mg kg^{-1})	–	4120	–	–
K_{av} (mg kg^{-1})	–	4626	–	–

[a] MC, moisture content; TOC, total organic carbon; TN, total nitrogen; EC, electrical conductivity; CEC, cation exchange capacity; P_{av}, plant-available phosphorus; K_{av}, plant-available potassium; -, no data.

[b] Humanure consists of feces, urine, toilet paper and sawdust. Cattle manure consists of feces, urine and sawdust.

[c] (Lekcharoenkul, Tanongkankit, Chiewchan, & Devahastin, 2014).

[d] Calculated from the total initial TOC of the composting mixture for treatment CM.

[e] (Jenkins, 2005).

[f] (Chufo, Yuan, Zou, Pang, & Li, 2015).

[g] 1:10 w/v fresh sample:deionized water.

[h] 1:12.5 w/v fresh sample: 0.01 mol L^{-1} $CaCl_2$ solution.

Table A.3. *Description of the four compost treatments applied in this study per replicate*

Treatment	Description (kg in fresh weight, % in dry weight)
HM	Humanure (223 kg, 50%) + vegetable scraps (430 kg, 32%) + teff straw (20.5 kg, 18%)
HM+BC	Humanure (223 kg, 50%) + vegetable scraps (430 kg, 32%) + teff straw (20.5 kg, 18%) + biochar (27, 1 kg, 19%)
CM	Cattle manure (256 kg, 55%) + vegetable scraps (429 kg, 29%) + teff straw (20.5 kg, 16%)
CM+BC	Cattle manure (256 kg, 55%) + vegetable scraps (429 kg, 29%) + teff straw (20.5 kg, 16%) + biochar (27.1 kg, 19%)

Table A.4. *Initial and final fresh and dry pile weights of each treatment (all replicates combined).*

Treatment	Initial pile fresh weight (kg)	Final pile fresh weight (kg)	Initial pile dry weight (kg)	Final pile dry weight (kg)
HM	2883	289	536	245
HM+BC	2985	439	635	370
CM	2995	366	628	342
CM+BC	3124	493	669	404

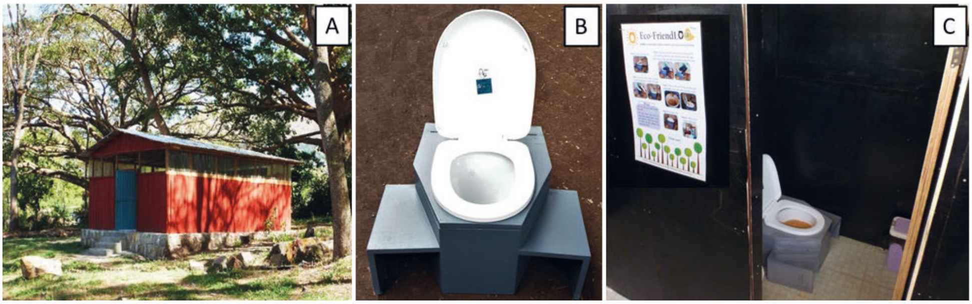


Figure A.3. **Ecological sanitation facility.** *(A) Ecological sanitation (EcoSan) toilet house, (B) and (C) EcoSan dry toilet.*

Figure A.4. *Mixing process for treatment 4: cattle manure, vegetable scraps, teff straw and biochar (photos by Daniela Castro Herrera).*

Figure A.5. *Filling up a composting box (photos by Daniela Castro Herrera).*

Figure A.6. **Compost box.** *(A) Wooden compost box used for the composting trials, (B) Compost box filled with compost material (day 0) and with a layer of 10–15 cm teff straw placed on the bottom, back, front, left, and right side for insulation purpose.*

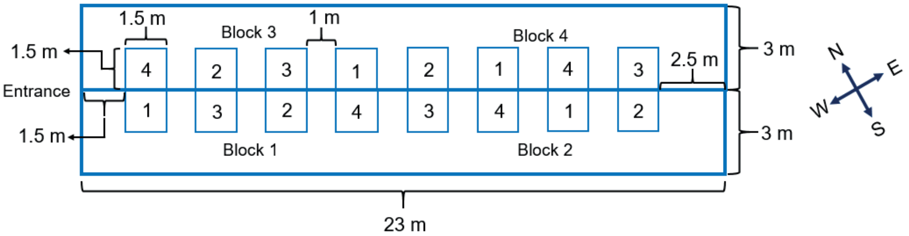


Figure A.7. *Position of each composting box (replicate) in the composting site. The numbers correspond to the treatment number.*

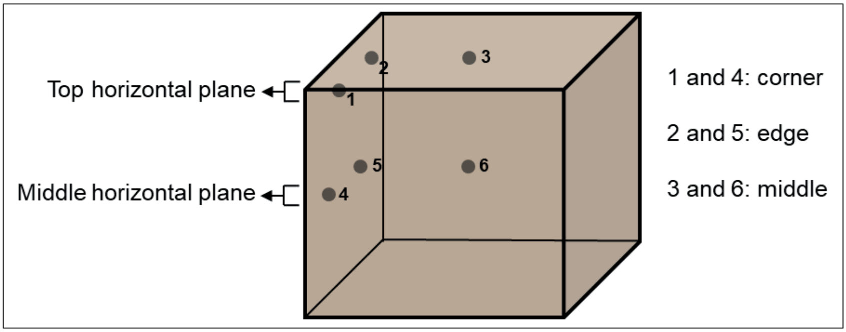


Figure A.8. ***Sampling.*** *Six different points inside the composting pile from which subsamples were collected.*

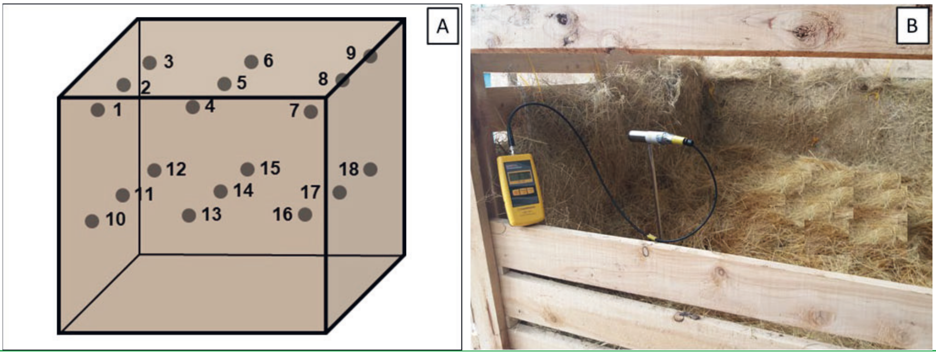


Figure A.9. ***Compost temperature measurements.*** *(A) Eighteen different points inside the composting pile where the temperature was measured, (B) Temperature measurement with compost thermometer.*

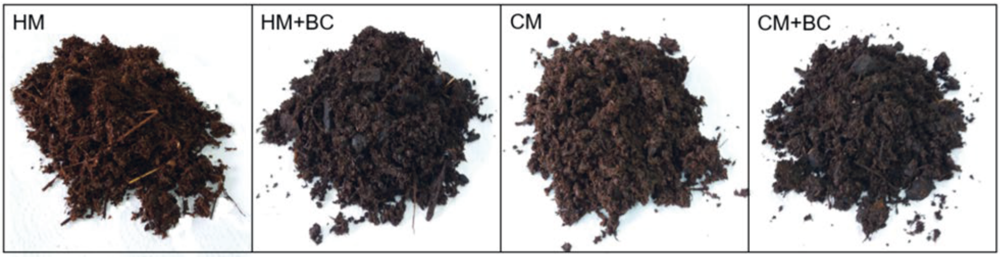


Figure A.10. *Photographs of final compost samples (day 185) for control (HM, CM) and biochar-amended treatments (HM+BC, CM+BC).*

Table A.5. *Temperatures (in °C) measured during composting for each replicate of each treatment.*

Day	Treatment 1					Treatment 2					Treatment 3					Treatment 4				
	1A	1B	1C	1D	Aver.	2A	2B	2C	2D	Aver.	3A	3B	3C	3D	Aver.	4A	4B	4C	4D	Aver.
0	27.1	24.2	23.9	24.4	24.9	27.5	24.6	27.0	25.8	26.2	27.5	28.1	27.6	25.0	27.1	28.2	26.6	26.4	24.7	26.5
1	43.8	55.0	56.8	55.0	52.6	43.8	57.9	58.1	57.8	54.4	43.8	54.8	50.6	50.0	49.8	52.7	55.8	56.4	54.9	55.0
2	53.7	61.4	60.6	66.5	60.6	53.0	64.3	63.9	65.4	61.7	53.0	56.6	59.7	56.6	56.5	62.4	60.5	58.3	62.2	60.9
3	53.7	65.8	62.8	67.2	64.5	60.7	64.0	65.0	67.2	64.2	60.7	58.4	60.9	60.3	60.1	68.4	65.5	64.3	65.4	65.9
4	53.7	65.3	64.4	66.2	65.1	64.0	64.6	62.4	64.0	63.7	64.0	57.1	63.4	61.9	61.6	68.2	64.3	62.8	65.8	65.3
5	53.7	64.9	62.9	66.1	64.9	64.1	63.4	60.2	63.3	62.8	64.1	57.8	64.2	62.2	62.1	67.4	64.5	59.7	64.6	64.1
6	53.7	65.1	62.4	63.9	64.2	64.4	62.4	58.5	62.4	61.9	64.4	56.2	61.9	60.6	60.8	69.5	64.1	61.5	65.2	65.1
7	53.7	63.6	60.5	62.5	63.1	63.2	61.9	58.0	59.8	60.7	63.2	54.5	61.7	60.3	59.9	61.1	62.5	59.3	63.9	61.7
8	53.7	62.5	59.7	61.2	61.9	59.8	60.4	56.3	59.4	59.0	59.8	52.2	59.8	58.2	57.5	62.7	61.5	59.3	62.3	61.4
9	53.7	61.7	57.7	59.6	59.7	58.8	57.6	54.7	58.4	57.4	58.8	51.1	57.7	57.0	56.2	61.6	60.0	57.8	61.5	60.2
10	53.7	58.2	56.9	57.7	57.8	57.1	57.4	53.7	57.0	56.3	57.1	48.9	55.8	55.3	54.3	62.5	57.7	55.6	59.6	58.8
11	53.7	57.5	54.2	56.1	56.3	56.5	53.9	52.0	55.4	54.5	56.5	49.0	54.7	54.5	53.7	60.9	57.5	53.8	58.3	57.6
12	53.7	54.8	53.2	54.1	54.8	55.0	52.7	50.9	53.7	53.1	55.0	47.5	53.3	53.8	52.4	60.2	54.5	52.0	56.3	55.8
13	53.7	54.0	50.5	53.2	53.7	53.6	50.1	48.3	52.2	51.0	53.6	46.8	52.8	53.6	51.7	58.9	52.6	51.0	54.9	54.4
14	53.7	52.9	49.1	45.2	50.8	51.4	49.7	47.6	45.9	48.6	51.4	48.0	49.5	53.0	50.5	57.6	50.8	49.2	52.5	52.5
15	53.7	50.3	47.2	45.1	48.8	48.9	48.4	46.8	44.6	47.2	48.9	46.5	48.4	46.4	47.5	54.5	49.3	47.2	45.4	49.1
16	53.7	48.7	46.4	44.6	47.7	47.2	47.9	43.0	41.2	44.8	47.2	46.3	44.6	45.0	45.8	52.8	47.8	46.0	43.9	47.6
17	53.7	48.5	44.8	43.6	46.6	46.0	45.9	41.9	40.7	43.6	46.0	45.3	45.9	43.3	45.1	50.7	47.7	43.9	42.9	46.3
18	53.7	46.7	43.4	42.4	45.3	44.6	43.5	38.1	39.9	41.5	44.6	45.5	44.1	43.2	44.3	49.6	45.8	42.9	41.3	44.9
19	53.7	45.4	42.2	41.1	43.7	45.0	42.7	38.7	38.5	41.2	45.0	44.9	41.8	42.0	43.4	48.2	43.9	41.1	40.8	43.5
20	53.7	44.7	41.5	40.1	42.5	43.4	41.3	36.6	37.2	39.6	43.4	44.5	43.1	40.6	42.9	49.5	43.3	40.3	39.9	43.2
21	53.7	43.4	38.9	38.5	40.9	43.6	40.6	34.7	35.4	38.6	43.6	44.0	41.3	38.6	41.9	46.6	41.7	38.6	39.7	41.7
22	53.7	42.8	37.2	38.4	40.7	42.5	40.6	33.0	36.2	38.1	42.5	43.4	40.9	36.3	40.8	47.0	41.9	37.1	37.0	40.8
23	53.7	40.9	35.8	35.1	39.1	44.3	38.6	38.6	32.9	38.6	44.3	45.4	39.6	38.2	41.9	44.6	42.8	35.9	38.3	40.4
24	53.7	41.5	35.1	33.1	38.9	42.8	39.6	31.7	30.9	36.2	42.8	41.9	31.6	34.4	37.7	45.4	40.4	34.3	35.3	38.9
25	53.7	38.6	31.4	31.5	36.1	44.0	36.6	29.1	29.5	34.8	44.0	43.6	39.0	33.9	40.1	42.3	40.1	34.0	33.5	37.5
26	53.7	38.1	31.0	30.6	36.1	42.6	36.3	27.8	28.4	33.8	42.6	42.1	36.3	32.5	38.4	42.9	38.8	30.9	31.9	36.1
27	53.7	36.6	28.9	28.9	34.1	42.8	34.4	26.7	26.9	32.7	42.8	42.4	34.9	31.1	37.8	39.6	37.7	29.9	30.1	34.3
28	53.7	36.7	27.3	28.2	33.9	41.2	33.8	26.2	26.1	31.8	41.2	41.1	33.1	30.8	36.5	38.3	36.4	28.8	29.4	33.2
31	53.7	32.5	24.0	24.1	30.1	37.7	29.5	23.5	23.5	28.5	37.7	40.6	29.0	27.5	33.7	37.2	34.2	25.7	26.2	30.8
34	53.7	29.3	22.0	22.4	27.6	37.7	26.6	22.7	22.6	27.4	37.7	37.9	26.3	24.5	31.6	33.1	31.0	23.5	23.8	27.9
37	53.7	24.5	21.1	21.6	25.3	35.4	23.0	21.8	22.1	25.6	35.4	35.7	24.3	23.0	29.6	31.8	26.9	23.1	23.3	26.2
40	53.7	23.2	21.3	20.4	24.6	34.3	22.0	21.5	20.8	24.6	34.3	33.0	23.7	21.7	28.2	31.0	25.0	22.8	21.9	25.1
43	53.7	22.5	20.0	20.2	23.3	32.6	21.8	20.8	20.9	24.0	32.6	31.6	22.3	20.9	26.8	27.7	24.6	21.6	21.6	23.9
46	53.7	22.9	20.3	20.2	22.9	30.5	22.0	21.0	20.6	23.5	30.5	31.1	22.6	20.0	26.0	25.8	24.1	21.8	21.0	23.2
49	53.7	21.8	20.3	19.6	22.0	27.6	21.5	20.6	19.7	22.4	27.6	28.7	21.9	19.3	24.4	24.4	23.7	21.5	20.2	22.5
52	53.7	21.2	20.2	18.9	21.3	24.9	20.9	21.0	19.3	21.5	24.9	25.9	22.2	19.5	23.1	23.5	22.9	22.3	20.2	22.2
55	53.7	21.2	20.1	18.6	20.6	23.2	21.1	20.5	19.0	20.9	23.2	24.9	22.2	18.9	22.3	23.4	23.1	22.2	19.5	22.1
58	53.7	21.4	20.3	19.5	20.7	23.1	21.1	20.9	19.8	21.2	23.1	23.4	21.5	19.5	21.9	22.8	22.6	21.9	20.3	21.9
61	53.7	20.1	19.8	19.2	20.1	23.0	20.3	20.2	19.7	20.8	23.0	22.6	21.3	19.6	21.6	22.9	22.2	21.5	20.3	21.7
64	53.7	20.4	18.7	19.5	20.0	23.0	20.2	19.6	19.9	20.7	23.0	22.2	21.3	19.8	21.6	22.9	22.3	21.8	20.4	21.9
67	53.7	20.4	19.0	19.6	20.1	22.5	20.2	19.8	20.3	20.7	22.5	21.4	21.0	20.4	21.4	21.9	21.8	21.0	21.2	21.5
70	53.7	20.4	18.4	19.5	19.6	22.3	21.9	18.9	20.3	20.9	22.3	21.1	20.5	20.8	21.2	22.3	21.1	20.2	21.3	21.2
73	53.7	20.7	19.1	19.1	19.8	22.0	20.6	19.4	19.8	20.5	22.0	21.8	20.3	20.3	21.1	22.0	21.3	20.2	20.8	21.1
76	53.7	19.5	19.0	19.4	19.7	21.5	19.9	19.4	19.9	20.2	21.5	21.1	20.0	20.5	20.8	21.5	21.1	20.3	20.7	20.9
79	53.7	19.8	19.0	19.0	19.5	21.5	19.6	n.m	19.9	20.4	21.5	21.4	n.m	19.9	20.9	20.8	21.0	n.m	20.0	20.6
82	53.7	19.4	19.0	19.1	19.5	21.2	19.5	20.0	20.0	20.2	21.2	20.9	20.3	20.4	20.7	20.9	20.3	20.4	20.6	20.6
85	53.7	19.0	19.1	19.3	19.6	21.4	19.4	19.5	19.9	20.0	21.4	20.8	20.9	20.2	20.8	20.7	20.1	20.7	20.1	20.4
87	53.7	18.8	18.9	19.6	19.4	22.1	18.8	19.8	20.1	20.2	22.1	20.3	20.8	20.3	20.9	21.3	19.8	21.1	20.2	20.6
101	53.7	19.9	18.9	18.9	19.2	20.0	19.9	19.6	19.5	19.8	20.0	20.7	19.9	19.8	20.1	19.5	19.9	19.6	19.7	19.6
129	53.7	19.0	18.8	0.0	19.0	19.8	19.0	19.3	0.0	19.3	19.8	19.9	19.6	n.m	19.8	18.9	19.4	19.6	n.m	19.3
143	53.7	19.0	n.m	0.0	18.9	19.8	19.0	n.m	0.0	19.4	19.8	20.0	n.m	n.m	19.9	19.6	19.6	n.m	n.m	19.6

n.m, not measured; aver., average.

Appendix B: supplementary material for chapter 3

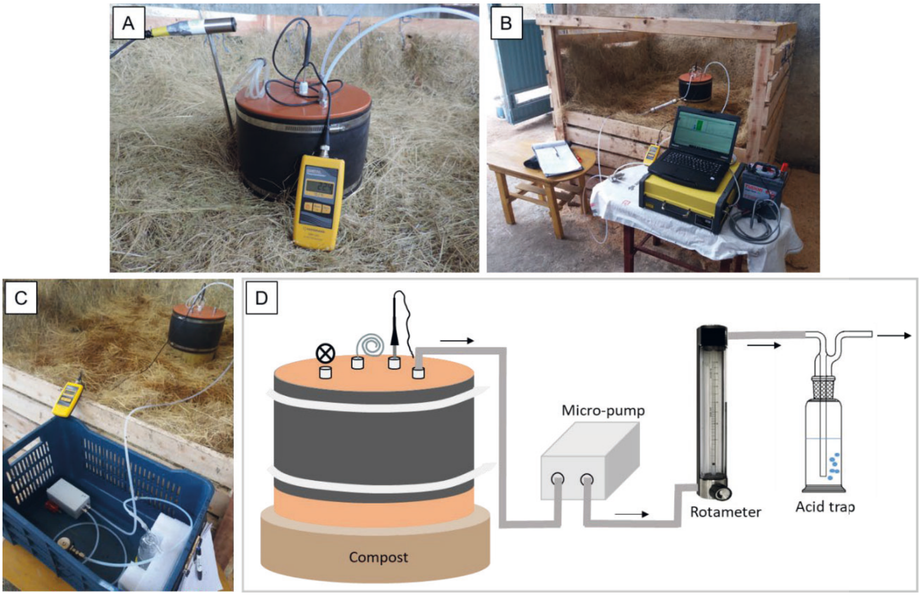


Figure B.1. **Gas flux measurements.** *(A) Static chamber, (B) GHG flux measurements, (C)* NH_3 *sampling, (D)* NH_3 *trapping system scheme.*

Table B.1. *Calculations to express cumulative greenhouse gas emissions as % of initial C or N using the data presented in the next studies in order to make them comparable with the results from our study.*

Hao et al. (2001)											
Reference	Table 2	Table 2	Table 2	Table 2	Table 1	Table 1	Calculated				
Treatment	Pile Weight (kg)	CO_2-C emissions (kg/Mg manure)	CH_4-C emissions (kg/Mg manure)	N_2O-N emissions (kg/Mg manure)	TC (g/kg)	TN (g/kg)	Pile Weight (kg)				
Static	9.55	73.8	6.3	0.11	342	17.7	9550				
Active	6.67	168	8.1	0.19	342	17.7	6670				
Continuation Hao et al. (2001)											
Treatment	CO_2-C emissions (kg)	CH_4-C emissions (kg)	N_2O-N emissions (kg)	TC (kg)	TN (kg)	CO_2-C (% of initial C)	CH_4-C (% of initial C)	N_2O-N (% of initial N)			
Static	91.20	704.79	60.17	0.01	31.19	909090.91	7025226.08	192.89			
Active	44.49	1120.56	54.03	0.01	15.22	526315.79	13256529.63	355.09			
Ahn et al. (2001)											
Reference	Table 2	Table 2	Table 3	Table 3	Table 3	Calculated	Calculated	Calculated			
Treatment	Initial total C mass (kg)	Initial total N mass (kg)	CO_2-C emissions (kg)	CH_4-C emissions (kg)	N_2O-N emissions (kg)	CO_2-C (% of initial C)	CH_4-C (% of initial C)	N_2O-N (% of initial N)			
Static	133	5.0	38.5	0.9	0.0053	28.9	0.7	0.1			
Active	133	5.0	42.4	1.3	0.0185	31.9	1.0	0.4			
Maeda et al. (2013)											
Reference	Table 2	Table 2	Calculated	Table 2	Calculated	Table 2	Calculated	Table 2	Calculated		
Replicate	Pile Weight (kg)	TS (DM) (%)	TS (DM) (kg)	VS (OM) (% TS)	VS (OM) (kg)	N (g N/kg TS)	N kg/pile	C/N	TOC (kg/pile)		
1	4460	26.3	1172.98	87.8	1029.88	23.4	27.45	22.8	625.81		
2	4700	28.3	1330.1	87	1157.19	20.8	27.67	23	636.32		
3	4480	31.3	1402.24	87.1	1221.35	21.6	30.29	27.5	832.93		
				Mean	1136.14	Mean	28.47	Mean	698.35		
Continuation Maeda et al. (2013)											
Reference	Table 3	Calculated*	Table 3	Calculated**	Calculated	Calculated	Calculated	Table 3	Table 3	Calculated	Calculated
Parameter	CH_4 emissions (kg/pile)	CH_4-C emissions (kg/pile)	CO_2 emissions (kg/pile)	CO_2-C emissions (kg/pile)	CH_4-C (% of initial C)	CO_2-C (% of initial C)	Initial N (kg/pile)	N_2O-N emissions (kg)	NH_3-N emissions (kg)	N_2O-N (% of initial N)	NH_3-N (% of initial N)
Values	3.9	2.925	434.7	118.54	0.42	16.97	28.47	0.06	0.8439	0.21	2.96
*12/16 = 75% C in CH_4											
**12/44 = 27.27% C in CO_2											

Appendix C: supplementary material for chapter 4

Table C.1. *Treatments and amount of fertilizer added for the incubation experiment (n=4).*

Treatment		Description	Fertilizer amount (g)
1	S	Soil (control)	0
2	MF	Mineral fertilizer	0.19
3	HM1	Humanure compost amount 1	4.98
4	HM1+BC	Biochar-amended humanure compost amount 1	5.01
5	CM1	Cattle manure compost amount 1	3.46
6	CM1+BC	Biochar-amended cattle manure compost amount 1	3.45
7	HM2	Humanure compost amount 2	14.94
8	HM2+BC	Biochar-amended humanure compost amount 2	15.03
9	CM2	Cattle manure compost amount 2	10.38
10	CM2+BC	Biochar-amended cattle manure compost amount 2	10.35

Figure C.1. *Incubator with soil samples (280 PVC tubes).*

Figure C.2. *G2508 Picarro gas analyzer.*

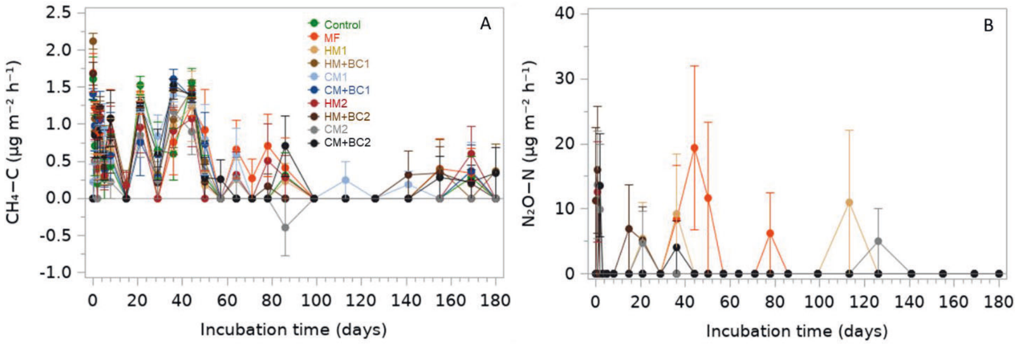


Figure C.3. ***CH_4 and N_2O emission rate dynamics.*** *Soil incubated at 25 °C, without any nutrient addition (control, green), with mineral fertilizer (MF, orange), with a lower compost application rate of: humanure (HM1, yellow), humanure amended with biochar (HM+BC1, light brown), cattle manure (CM1, light blue t), cattle manure amended with biochar (CM+BC1, blue), and with a higher compost application rate (three times more than the lower one) of: humanure (HM2, red), humanure amended with biochar (HM+BC2, brown), cattle manure (CM2, gray), cattle manure amended with biochar (CM+BC2, black). Values provided as means (± SE) with n=4.*

Appendix D: supplementary material for chapter 5

Table D.1. *Application amount of humanre and cattle manure compost with and without biochar.*

Treatment	Fresh weight compost (g)	Moisture content (%)	Dry weight compost (g)	Fresh weight (ton/ha)*	Dry weight (ton/ha)*
HM1	4.98	63.1	1.84	31	12
HM1+BC	5.01	59.8	2.01	32	13
CM1	3.46	63.3	1.27	22	8
CM1+BC	3.45	60.6	1.36	22	9
HM2	14.94	63.1	5.51	94	35
HM2+BC	15.03	59.8	6.04	95	38
CM2	10.38	63.3	3.81	65	24
CM2+BC	10.35	60.6	4.08	65	26

*PVC tubes used for the soil incubation experiment: diameter 4.5cm, area 15.9 cm^2.

Table D.2. *Amount of non-CO_2-eq emitted per kg of compost-N during the production of the different types of compost.*

Treatment	Cumulative non-CO_2-Ceq (g m^{-2})[a]	Cumulative non-CO_2-Ceq (g)[b]	Multiplied by four replicates	kg of dry compost produced[c]	% N[d]	kg dry N	kg of non-CO_2-Ceq / kg N dry compost
HM	1895	4264	17055	245	1.5	3.7	4.6
HM+BC	918	2066	8262	370	1.4	5.2	1.6
CM	3441	7742	30969	342	2.2	7.5	4.1
CM+BC	900	2025	8100	404	2.1	8.5	1.0

[a] Taken from table 3.1 (Chapter 3)

[b] Multiplied by the area of the compost box: 2.25 m^2.

[c] From Table A.4.

[d] From Table 2.1 (Chapter 2)

REFERENCES

Adekiya AO, Ejue WS, Olayanju A, Dunsin O, Aboyeji CM, Aremu C, Adegbite K, Akinpelu O (2020) Different organic manure sources and NPK fertilizer on soil chemical properties, growth, yield and quality of okra. Sci Rep 10:16083. https://doi.org/10.1038/s41598-020-73291-x

Agegnehu G, Srivastava AK, Bird MI (2017) The role of biochar and biochar-compost in improving soil quality and crop performance: A review. Applied Soil Ecology 119:156–170. https://doi.org/10.1016/j.apsoil.2017.06.008

Agyarko-Mintah E, Cowie A, van Zwieten L, Singh BP, Smillie R, Harden S, Fornasier F (2017) Biochar lowers ammonia emission and improves nitrogen retention in poultry litter composting. Waste Manag 61:129–137. https://doi.org/10.1016/j.wasman.2016.12.009

Ahn HK, Mulbry W, White JW, Kondrad SL (2011) Pile mixing increases greenhouse gas emissions during composting of dairy manure. Bioresour Technol 102:2904–2909. https://doi.org/10.1016/j.biortech.2010.10.142

Akram U, Quttineh N-H, Wennergren U, Tonderski K, Metson GS (2019) Enhancing nutrient recycling from excreta to meet crop nutrient needs in Sweden - a spatial analysis. Sci Rep 9:10264. https://doi.org/10.1038/s41598-019-46706-7

Alewell C, Ringeval B, Ballabio C, Robinson DA, Panagos P, Borrelli P (2020) Global phosphorus shortage will be aggravated by soil erosion. Nature communications 11:4546. https://doi.org/10.1038/s41467-020-18326-7

Amelung W, Bossio D, Vries W de, Kögel-Knabner I, Lehmann J, Amundson R, Bol R, Collins C, Lal R, Leifeld J, Minasny B, Pan G, Paustian K, Rumpel C, Sanderman J, van Groenigen JW, Mooney S, van Wesemael B, Wander M, Chabbi A (2020) Towards a global-scale soil climate mitigation strategy. Nature communications 11:5427. https://doi.org/10.1038/s41467-020-18887-7

Amlinger F, Götz B, Dreher P, Geszti J, Weissteiner C (2003) Nitrogen in biowaste and yard waste compost: dynamics of mobilisation and availability—a review. European Journal of Soil Biology 39:107–116. https://doi.org/10.1016/S1164-5563(03)00026-8

Awasthi MK, Wang M, Chen H, Wang Q, Zhao J, Ren X, Li D-S, Awasthi SK, Shen F, Li R, Zhang Z (2017) Heterogeneity of biochar amendment to improve the carbon and nitrogen sequestration through reduce the greenhouse gases emissions during sewage sludge composting. Bioresour Technol 224:428–438. https://doi.org/10.1016/j.biortech.2016.11.014

Barker AV, Pilbeam DJ (eds) (2015) Handbook of Plant Nutrition, 2nd edn. CRC Press

Berendes D, Levy K, Knee J, Handzel T, Hill VR (2015) Ascaris and Escherichia coli Inactivation in an Ecological Sanitation System in Port-au-Prince, Haiti. PLoS One 10:e0125336. https://doi.org/10.1371/journal.pone.0125336

Berendes DM, Yang PJ, Lai A, Hu D, Brown J (2018) Estimation of global recoverable human and animal faecal biomass. Nat Sustain 1:679–685. https://doi.org/10.1038/s41893-018-0167-0

Bernal MP, Alburquerque JA, Moral R (2009) Composting of animal manures and chemical criteria for compost maturity assessment. A review. Bioresour Technol 100:5444–5453. https://doi.org/10.1016/j.biortech.2008.11.027

Bijleveld M (2003) The possibilities for ecological sanitation in the Netherlands. Master, University of Twente

Birzer C, Medwell P, Wilkey J, West T, Higgins M, MacFarlane G, Read M (2013) An analysis of combustion from a top-lit up-draft (TLUD) cookstove. Journal of Humanitarian Engineering 2:1–7

Bracken P, von Münch E, Panesar A (2009) Tackling the Urban Waste and Food Crises Simultaneously and Sustainably - Examples from the Philippines and Burkina Faso. Presented at 34th WEDC International Conference, Addis Ababa, Ethiopia. https://www.susana.org/en/knowledge-hub/resources-and-publications/library/details/899

Brown S, Kruger C, Subler S (2008) Greenhouse gas balance for composting operations. J Environ Qual 37:1396–1410. https://doi.org/10.2134/jeq2007.0453

Bundesministerium der Justiz und für Verbraucherschutz (2017) Verordnung über die Verwertung von Bioabfällen auf landwirtschaftlich, forstwirtschaftlich und gärtnerisch genutzten Böden (Bioabfallverordnung - BioAbfV): BioAbfV. Bundesministerium der Justiz und für Verbraucherschutz

Bundesministerium der Justiz und für Verbraucherschutz, Bundesamts für Justiz (2012) Verordnung über das Inverkehrbringen von Düngemitteln, Bodenhilfsstoffen, Kultursubstraten und Pflanzenhilfsmitteln (Düngemittelverordnung - DüMV): DüMV

Bundesministerium der Justiz und für Verbraucherschutz, Bundesamts für Justiz (2017) Verordnung über die Anwendung von Düngemitteln, Bodenhilfsstoffen, Kultursubstraten und Pflanzenhilfsmitteln nach den Grundsätzen der guten fachlichen Praxis beim Düngen (Düngeverordnung - DüV): DüV

California Compost Quality Council (2001) Compost maturity index. www.ccqc.org

Cao Y, Wang X, Bai Z, Chadwick D, Misselbrook T, G. Sommer S, Qin W, Ma L (2019) Mitigation of ammonia, nitrous oxide and methane emissions during solid waste composting with different additives: A meta-analysis. Journal of Cleaner Production 235:626–635. https://doi.org/10.1016/j.jclepro.2019.06.288

Cha JS, Park SH, Jung S-C, Ryu C, Jeon J-K, Shin M-C, Park Y-K (2016) Production and utilization of biochar: A review. Journal of Industrial and Engineering Chemistry 40:1–15. https://doi.org/10.1016/j.jiec.2016.06.002

Chadwick DR (2005) Emissions of ammonia, nitrous oxide and methane from cattle manure heaps: effect of compaction and covering. Atmospheric Environment 39:787–799. https://doi.org/10.1016/j.atmosenv.2004.10.012

Chen W, Liao X, Wu Y, Liang JB, Mi J, Huang J, Zhang H, Wu Y, Qiao Z, Li X, Wang Y (2017) Effects of different types of biochar on methane and ammonia mitigation during layer manure composting. Waste Manag 61:506–515. https://doi.org/10.1016/j.wasman.2017.01.014

Chen YX, Huang X-D, Han Z-Y, Huang X, Hu B, Shi D-Z, Wu W-X (2010) Effects of bamboo charcoal and bamboo vinegar on nitrogen conservation and heavy metals immobility during pig manure composting. Chemosphere 78:1177–1181. https://doi.org/10.1016/j.chemosphere.2009.12.029

Chowdhury MA, Neergaard A de, Jensen LS (2014) Composting of solids separated from anaerobically digested animal manure: Effect of different bulking agents and mixing ratios on emissions of greenhouse gases and ammonia. Biosystems Engineering 124:63–77. https://doi.org/10.1016/j.biosystemseng.2014.06.003

Clayton J, Lemanski K, Bonkowski M (2021) Shifts in soil microbial stoichiometry and metabolic quotient provide evidence for a critical tipping point at 1% soil organic carbon in an agricultural post-mining chronosequence. Biol Fertil Soils 57:435–446. https://doi.org/10.1007/s00374-020-01532-2

Coalition Clean Baltic (2009) The Swedish Eco-Sanitation Experience: Case studies of succesful projects implementing alternative techniques for wastewater treatment in Sweden. https://ccb.se/publications/the-swedish-eco-sanitation-experience/

Collivignarelli M, Abbà A, Frattarola A, Carnevale Miino M, Padovani S, Katsoyiannis I, Torretta V (2019) Legislation for the Reuse of Biosolids on Agricultural Land in Europe: Overview. Sustainability 11:6015. https://doi.org/10.3390/su11216015

Cordell D, Drangert J-O, White S (2009) The story of phosphorus: Global food security and food for thought. Global Environmental Change 19:292–305. https://doi.org/10.1016/j.gloenvcha.2008.10.009

Czekała W, Dach J, Malińska K, Przybył J, Myszura M (2015) Dynamics of gaseous emissions during composting of sewage sludge with maize straw as bulking agent. J. Ecol. Eng. 16:108–114. https://doi.org/10.12911/22998993/2943

Dalzell HW, Riddlestone AJ, Gray KR, Thurairajan K (1987) Soil management: compost production and use in tropical and subtropical environments.: FAO Soils Bulletin No. 56. http://www.fao.org/3/s8930e/s8930e.pdf

Deutsche Bundesstiftung Umwelt (2015) Terra Preta sanitation 1: Background, Principles and Innovations. DBU Deutsche Bundesstiftung Umwelt, Osnabrück

Dickin S, Bayoumi M, Giné R, Andersson K, Jiménez A (2020) Sustainable sanitation and gaps in global climate policy and financing. npj Clean Water 3. https://doi.org/10.1038/s41545-020-0072-8

Dunst G (2015) Kompostierung und Erdenherstellung: Praxisbuch und Anleitung für: Hausgarten, Landwirtschaft, Kommune und Profi. Sonnenerde-Gerald Dunst Kulturerden GmbH.

Ecodesign: The bottom line (2012). Nature 486:186–189. https://doi.org/10.1038/486186a

EBC (2012) European Biochar Certificate - Guidelines for a Sustainable Production of Biochar, 6th edn. European Biochar Foundation (EBC), Arbaz, Switzerland

Eghball B, Power JF, John EG, Doran JW (1997) Nutrient, Carbon, and Mass Loss during Composting of Beef Cattle Feedlot Manure 26:189–193

El Kader NA, Robin P, Paillat J-M, Leterme P (2007) Turning, compacting and the addition of water as factors affecting gaseous emissions in farm manure composting. Bioresour Technol 98:2619–2628. https://doi.org/10.1016/j.biortech.2006.07.035

Elser J, Bennett E (2011) A broken biogeochemical cycle. Nature 478:29–31. https://doi.org/10.1038/478029a

Ersson O, King K (2019) The Kailash Ecovillage project converting human excreta into organic foodstuffs and sanitized compost using new international building codes for compost toilet and urine diversion systems. Blue-Green Systems 1:33–54. https://doi.org/10.2166/bgs.2019.192

European Compost Network (2021) Biowaste Generates Organic Matter: Improving Soils with Compost. https://www.compostnetwork.info/policy/from-waste/biowaste-generates-organic-matter/

European Union (2013) Soil atlas of Africa. JRC soil atlas series. European Union, Luxembourg, Luxembourg

European Union (2015) Soil atlas of Latin America and the Caribbean. EUR, 25402 EN. European Union, Luxembourg

FAO (2015) Status of the World's Soil Resources: Main Report. Food and Agriculture Organization of the United Nations, Rome

FAO (2016) Ethiopia Climate-Smart Agriculture Scoping Study. Food and Agriculture Organization of the United Nations, Addis Ababa, Ethiopia

FAO (2017) Soil organic carbon: The hidden potential. Food and Agriculture Organization of the United Nations, Rome

FAO (2019) Global Symposium on Soil Erosion: Key messages. Food and Agriculture Organization of the United Nations

FAO (2021) Climate-Smart Agriculture. http://www.fao.org/climate-smart-agriculture/en/

Fertilizers Europe (2011) Energy efficiency and greenhouse gas emissions in European nitrogen fertilizer production and use

Fisher D, Glaser B (2012) Synergisms between Compost and Biochar for Sustainable Soil Amelioration. In: Kumar S, Bharti A (eds) Management of Organic Waste, pp 167–198

Glaser B, Haumaier L, Guggenberger G, Zech W (2001) The 'Terra Preta' phenomenon: a model for sustainable agriculture in the humid tropics. Naturwissenschaften 88:37–41. https://doi.org/10.1007/s001140000193

Glaser B (2007) Prehistorically modified soils of central Amazonia: a model for sustainable agriculture in the twenty-first century. Philos Trans R Soc Lond B Biol Sci 362:187–196. https://doi.org/10.1098/rstb.2006.1978

Glaser B, Lehr V-I (2019) Biochar effects on phosphorus availability in agricultural soils: A meta-analysis. Sci Rep 9:9338. https://doi.org/10.1038/s41598-019-45693-z

Godlewska P, Schmidt HP, Ok YS, Oleszczuk P (2017) Biochar for composting improvement and contaminants reduction. A review. Bioresour Technol 246:193–202. https://doi.org/10.1016/j.biortech.2017.07.095

Gottschall R (1992) Kompostierung: Optimale Aufbereitung und Verwendung organischer Materialen im ökologischen Landau, 5th edn. C. F. Müller, Karlsruhe.

Guo X-X, Liu H-T, Zhang J (2020) The role of biochar in organic waste composting and soil improvement: A review. Waste Manag 102:884–899. https://doi.org/10.1016/j.wasman.2019.12.003

Gwara S, Wale E, Odindo A, Buckley C (2021) Attitudes and Perceptions on the Agricultural Use of Human Excreta and Human Excreta Derived Materials: A Scoping Review. Agriculture 11:153. https://doi.org/10.3390/agriculture11020153

Hagemann N, Joseph S, Schmidt H-P, Kammann CI, Harter J, Borch T, Young RB, Varga K, Taherymoosavi S, Elliott KW, McKenna A, Albu M, Mayrhofer C, Obst M, Conte P, Dieguez-Alonso A, Orsetti S, Subdiaga E, Behrens S, Kappler A (2017) Organic coating on biochar explains its nutrient retention and stimulation of soil fertility. Nature communications 8:1089. https://doi.org/10.1038/s41467-017-01123-0

Hagemann N, Orsetti S, Subdiaga E, La Rosa JM de, Knicker H, Schmidt H-P, Kappler A, Behrens S (2018) Effect of biochar amendment on compost organic matter composition following aerobic composting of manure. Sci Total Environ 613-614:20–29. https://doi.org/10.1016/j.scitotenv.2017.08.161

Hao X, Chang C, Larney FJ, Travis GR (2001) Greenhouse Gas Emissions during Cattle Feedlot Manure Composting. J Environ Qual 30:376–386

Harder R, Wielemaker R, Larsen TA, Zeeman G, Öberg G (2019) Recycling nutrients contained in human excreta to agriculture: Pathways, processes, and products. Crit Rev Environ Sci Technol 49:695–743. https://doi.org/10.1080/10643389.2018.1558889

He X, Chen L, Han L, Liu N, Cui R, Yin H, Huang G (2017) Evaluation of biochar powder on oxygen supply efficiency and global warming potential during mainstream large-scale aerobic composting. Bioresour Technol 245:309–317. https://doi.org/10.1016/j.biortech.2017.08.076

Heinonen-Tanski H, van Wijk-Sijbesma C (2005) Human excreta for plant production. Bioresour Technol 96:403–411. https://doi.org/10.1016/j.biortech.2003.10.036

Hellmann B, Zelles L, Palojärvi A, Bai Q (1997) Emission of Climate-Relevant Trace Gases and Succession of Microbial Communities during Open-Windrow Composting. Applied and Environmental Microbiology 63:1011–1018

Hestrin R, Enders A, Lehmann J (2020) Ammonia volatilization from composting with oxidized biochar. J. environ. qual. https://doi.org/10.1002/jeq2.20154

Hossain MZ, Bahar MM, Sarkar B, Donne SW, Ok YS, Palansooriya KN, Kirkham MB, Chowdhury S, Bolan N (2020) Biochar and its importance on nutrient dynamics in soil and plant. Biochar 2:379–420. https://doi.org/10.1007/s42773-020-00065-z

Hu M, Fan B, Sun W, Wang H, Qu B, Zhu S (2016) Constructing the ecological sanitation: a review on technology and methods. Journal of Cleaner Production 125:1–21. https://doi.org/10.1016/j.jclepro.2016.03.012

Hua L, Wu W, Liu Y, McBride MB, Chen Y (2009) Reduction of nitrogen loss and Cu and Zn mobility during sludge composting with bamboo charcoal amendment. Environ Sci Pollut Res Int 16:1–9. https://doi.org/10.1007/s11356-008-0041-0

Huuhtanen S, Laukkanen A (2006) A guide to sanitation and hygiene for those working in developing countries. http://global.tkk.fi/course_material/DryToilet_Guide.pdf

IPCC (2006) 2006 IPCC guidelines for national greenhouse gas inventories. IPCC, Japan

IPCC (2013) Climate change 2013: The physical science basis: Working Group I contribution to the Fifth assessment report of the Intergovernmental Panel on Climate Change. Cambridge University Press, New York

Janczak D, Malińska K, Czekała W, Cáceres R, Lewicki A, Dach J (2017) Biochar to reduce ammonia emissions in gaseous and liquid phase during composting of poultry manure with wheat straw. Waste Manag 66:36–45. https://doi.org/10.1016/j.wasman.2017.04.033

Jenkins JC (2005) The humanure handbook: A guide to composting human manure. Jenkins Publishing, Grove City, Pennsylvania

Jensen PKM, Phuc PD, Knudsen LG, Dalsgaard A, Konradsen F (2008) Hygiene versus fertiliser: the use of human excreta in agriculture--a Vietnamese example. Int J Hyg Environ Health 211:432–439. https://doi.org/10.1016/j.ijheh.2007.08.011

Jia X, Wang M, Wenqiao Y, Xiaotang J, Baozhu Y (2016) The influence of biochar addition on chicken manure composting and associated methane and carbon dioxide emissions. BioResources 11(2):5255–5264

Joseph S, Kammann CI, Shepherd JG, Conte P, Schmidt H-P, Hagemann N, Rich AM, Marjo CE, Allen J, Munroe P, Mitchell DRG, Donne S, SPOKAS K, Graber ER (2017) Microstructural and associated chemical changes during the composting of a high temperature biochar: Mechanisms for nitrate, phosphate and other nutrient retention and release. Sci Total Environ 618:1210–1223. https://doi.org/10.1016/j.scitotenv.2017.09.200

Joseph SD, Camps-Arbestain M, Lin Y, Munroe P, Chia CH, Hook J, van Zwieten L, Kimber S, Cowie A, Singh BP, Lehmann J, Foidl N, Smernik RJ, Amonette JE (2010) An investigation into the reactions of biochar in soil. Soil Res. 48:501. https://doi.org/10.1071/SR10009

Junghanns J, Beery T (2020) Ecological Sanitation and Sustainable Nutrient Recovery Education: Considering the Three Fixes for Environmental Problem-Solving. Sustainability 12:3587. https://doi.org/10.3390/su12093587

Kamman C, IPPOLITO J, Hagemann N, Borchard N, CAYUELA ML, ESTAVILLO JM, FUERTES-MENDIZABAL T, JEFFERY S, KERN J, NOVAK J, RASSE D, Saarino S, Schmidt H-P, SPOKAS K, Wrage-Mönning N (2017) Biochar as a tool to reduce the agricultural greenhouse-gas burden – knowns, unknowns and future research needs. Journal of Environmental Engineering and Landscape Management 25:114–139. https://doi.org/10.3846/16486897.2017.1319375

Kammann CI, Schmidt H-P, Messerschmidt N, Linsel S, Steffens D, Müller C, Koyro H-W, Conte P, Joseph S, Stephen J (2015) Plant growth improvement mediated by nitrate capture in co-composted biochar. Sci Rep 5:11080. https://doi.org/10.1038/srep11080

Keener HM, Dick WA, Hoitink HAJ (2000) Composting and Beneficial Utilization of Composted By-Product Materials. In: Power JF, Dick WA, Kashmanian RM, Sims JT, Wright RJ, Dawson MD, Bezdicek D (eds) Land Application of Agricultural, Industrial, and Municipal By-Products. Soil Science Society of America, Madison, WI, USA, pp 315–341

Kim D-G, Thomas AD, Pelster D, Rosenstock TS, Sanz-Cobena A (2015) Reviews and syntheses: Greenhouse gas emissions in natural and agricultural lands in sub-Saharan Africa: synthesis of available data and suggestions for further studies. Biogeosciences Discussions 12:16479–16526. https://doi.org/10.5194/bgd-12-16479-2015

Kim SY, Pramanik P, Bodelier PLE, Kim PJ (2014) Cattle Manure Enhances Methanogens Diversity and Methane Emissions Compared to Swine Manure under Rice Paddy. PLoS One 9:e113593. https://doi.org/10.1371/journal.pone.0113593

Kowaljow E, Gonzalez-Polo M, Mazzarino MJ (2017) Understanding compost effects on water availability in a degraded sandy soil of Patagonia. Environ Earth Sci 76. https://doi.org/10.1007/s12665-017-6573-1

Kramer S (2011) The SOIL Guide to Ecological Sanitation. http://www.oursoil.org/resources

Kuratorium Boden des Jahres (2017) Boden des Jahres 2021: Gartenboden. https://boden-des-jahres.de/archiv/2017-gartenboden/

Landwirtschaftskammer Niedersachsen (2018) Empfehlungen zur Grunddüngung. Landwirtschaftskammer Niedersachsen

Landwirtschaftskammer Nordrhein-Westfalen (2015) Stickstoffdüngung im Mais. Landwirtschaftskammer Nordrhein-Westfalen

Langergraber G, Muellegger E (2005) Ecological Sanitation--a way to solve global sanitation problems? Environ Int 31:433–444. https://doi.org/10.1016/j.envint.2004.08.006

Lehmann J, Joseph S (2009) Biochar for environmental management: Science and technology. Earthscan, London, Sterling VA

Li R, Wang Q, Zhang Z, Zhang G, Li Z, Wang L, Zheng J (2015) Nutrient transformation during aerobic composting of pig manure with biochar prepared at different temperatures. Environ Technol 36:815–826. https://doi.org/10.1080/09593330.2014.963692

Li S, Song L, Jin Y, Liu S, Shen Q, Zou J (2016) Linking N_2O emission from biochar-amended composting process to the abundance of denitrify (*nirK* and *nosZ*) bacteria community. AMB Express 6:37. https://doi.org/10.1186/s13568-016-0208-x

Liu S, Berns AE, Vereecken H, Di Wu, Brüggemann N (2017a) Interactive effects of MnO2, organic matter and pH on abiotic formation of N2O from hydroxylamine in artificial soil mixtures. Sci Rep 7:39590. https://doi.org/10.1038/srep39590

Liu N, Zhou J, Han L, Ma S, Sun X, Huang G (2017b) Role and multi-scale characterization of bamboo biochar during poultry manure aerobic composting. Bioresour Technol 241:190–199. https://doi.org/10.1016/j.biortech.2017.03.144

Liu D, Xie B, Dong C, Liu G, Hu D, Qin Y, Li H, Liu H (2018) Effect of fertilizer prepared from human feces and straw on germination, growth and development of wheat. Acta Astronautica 145:76–82. https://doi.org/10.1016/j.actaastro.2018.01.014

López-Zavala MÁ (2019) Fate of Carbon in Composting Process. In: Funamizu N (ed) Resource-Oriented Agro-sanitation Systems. Springer Japan, Tokyo, pp 39–51

Lorenz K, Lal R (2018) Carbon Sequestration in Agricultural Ecosystems. Springer International Publishing, Cham

Maeda K, Hanajima D, Morioka R, Toyoda S, Yoshida N, Osada T (2013) Mitigation of greenhouse gas emission from the cattle manure composting process by use of a bulking agent. Soil Science and Plant Nutrition 59:96–106. https://doi.org/10.1080/00380768.2012.733868

Magid J, Eilersen AM, Wrisberg S, Henze M (2006) Possibilities and barriers for recirculation of nutrients and organic matter from urban to rural areas: A technical theoretical framework applied to the medium-sized town Hillerød, Denmark. Ecological Engineering 28:44–54. https://doi.org/10.1016/j.ecoleng.2006.03.009

Maheshwari DK (2014) Composting for Sustainable Agriculture, vol 3. Springer International Publishing, Cham

Malisie AF, Prihandrijanti M, Otterpohl R (2007) The potential of nutrient reuse from a source-separated domestic wastewater system in Indonesia--case study: ecological sanitation pilot plant in Surabaya. Water Sci Technol 56:141–148. https://doi.org/10.2166/wst.2007.566

Manning DAC (2010) Mineral sources of potassium for plant nutrition. A review. Agron. Sustain. Dev. 30:281–294. https://doi.org/10.1051/agro/2009023

Martins O, Dewes T (1992) Loss of Nitrogenous Compounds during Composting of Animal Wastes. Bioresour Technol 42:103–111

McNicol G, Jeliazovski J, François JJ, Kramer S, Ryals R (2020) Climate change mitigation potential in sanitation via off-site composting of human waste. Nat. Clim. Chang. 10:545–549. https://doi.org/10.1038/s41558-020-0782-4

Meena RS (2020) Nutrient Dynamics for Sustainable Crop Production. Springer Singapore, Singapore

Moreno-Caselles J, Moral R, Perez-Murcia M, Perez-Espinosa A, Rufete B (2002) Nutrient value of animal manures in fron of environmentl hazards. Communications in Soil Science and Plant Analysis 33:3023–3032. https://doi.org/10.1081/CSS-120014499

Morgan P (2007) Toilets That Make Compost: Low-cost, sanitary toilets that produce valuable compost for crops in an African contex

Moya B, Parker A, Sakrabani R (2019a) Challenges to the use of fertilisers derived from human excreta: The case of vegetable exports from Kenya to Europe and influence of certification systems. Food Policy 85:72–78. https://doi.org/10.1016/j.foodpol.2019.05.001

Moya B, Parker A, Sakrabani R, Mesa B (2019b) Evaluating the Efficacy of Fertilisers Derived from Human Excreta in Agriculture and Their Perception in Antananarivo, Madagascar. Waste Biomass Valor 10:941–952. https://doi.org/10.1007/s12649-017-0113-9

Moya B, Sakrabani R, Parker A (2019c) Realizing the Circular Economy for Sanitation: Assessing Enabling Conditions and Barriers to the Commercialization of Human Excreta Derived Fertilizer in Haiti and Kenya. Sustainability 11:3154. https://doi.org/10.3390/su11113154

Musafiri CM, Macharia JM, Kiboi MN, Ng'etich OK, Shisanya CA, Okeyo JM, Mugendi DN, Okwuosa EA, Ngetich FK (2020) Soil greenhouse gas fluxes from maize cropping system under different soil fertility management technologies in Kenya. Agriculture, Ecosystems & Environment 301:107064. https://doi.org/10.1016/j.agee.2020.107064

Nguyen DH, Biala J, Grace PR, Scheer C, Rowlings DW (2014) Greenhouse gas emissions from sub-tropical Greenhouse gas emissions from sub-tropical agricultural soils after addition of organic by-products 3

Nkonya E, Mirzabaev A, Braun J von (2016) Economics of Land Degradation and Improvement – A Global Assessment for Sustainable Development. Springer International Publishing, Cham

Orner KD, Mihelcic JR (2018) A review of sanitation technologies to achieve multiple sustainable development goals that promote resource recovery. Environ. Sci.: Water Res. Technol. 4:16–32. https://doi.org/10.1039/C7EW00195A

Paul EA (ed) (2015) Soil Microbiology, Ecology and Biochemistry. Elsevier

Peigné J, Girardin P (2004) Environmental Impacts of Farm-Scale Composting Practices. Water, Air, and Soil Pollution 153:45–68

Piceno YM, Pecora-Black G, Kramer S, Roy M, Reid FC, Dubinsky EA, Andersen GL (2017) Bacterial community structure transformed after thermophilically composting human waste in Haiti. PLoS One 12:e0177626. https://doi.org/10.1371/journal.pone.0177626

Preneta N, Kramer S, Magloire B, Noel JM (2013) Thermophilic co-composting of human wastes in Haiti. Journal of Water, Sanitation and Hygiene for Development 3:649–654. https://doi.org/10.2166/washdev.2013.145

Prost K, Borchard N, Siemens J, Kautz T, Séquaris J-M, Möller A, Amelung W (2013) Biochar affected by composting with farmyard manure. J Environ Qual 42:164–172. https://doi.org/10.2134/jeq2012.0064

Rauschning G, Berger W, Ebeling B, Schöpe A (2009) Ökologische Siedlung mit Komposttoiletten Allermöhe, Hamburg, Deutschland (in German) - Ecological settlement with compost toilets in Allermöhe Hamburg, Germany. Case study of sustainable sanitation projects.

Reichel R, Wei J, Islam MS, Schmid C, Wissel H, Schröder P, Schloter M, Brüggemann N (2018) Potential of Wheat Straw, Spruce Sawdust, and Lignin as High Organic Carbon Soil Amendments to Improve Agricultural Nitrogen Retention Capacity: An Incubation Study. Front Plant Sci 9:900. https://doi.org/10.3389/fpls.2018.00900

Reid M (2020) Sanitation and climate. Nat. Clim. Chang. 10:496–497. https://doi.org/10.1038/s41558-020-0787-z

Reid MC, Guan K, Wagner F, Mauzerall DL (2014) Global methane emissions from pit latrines. Environ Sci Technol 48:8727–8734. https://doi.org/10.1021/es501549h

Rose C, Parker A, Jefferson B, Cartmell E (2015) The Characterization of Feces and Urine: A Review of the Literature to Inform Advanced Treatment Technology. Crit Rev Environ Sci Technol 45:1827–1879. https://doi.org/10.1080/10643389.2014.1000761

Roth C (2014) Micro-gasification: cooking with gas from dry biomass: An introduction to concepts and applications of wood-gas burning technologies for cooking. Deutsche Gesellschaft für Internationale Zusammenarbeit, Eschborn, Germany

Ryals R, McNicol G, Porder S, Kramer S (2019) Greenhouse gas fluxes from human waste management pathways in Haiti. Journal of Cleaner Production 226:106–113. https://doi.org/10.1016/j.jclepro.2019.04.079

Rynk R, van de Kamp M, Willson GB, Singley ME, Richard TL, Kolega JJ, Gouin FR, Laliberty L, Kay D, Murphy DW, Hoitink HAJ, Brinton WF (1992) On-Farm Composting Handbook. Cooperative Extension, Ithaca, New York

Scholz, Sembres, Roberts, Whitman, Wilson, Lehmann (eds) (2014) Biochar Systems for Smallholders in Developing Countries: Leveraging Current Knowledge and Exploring Future Potential for Climate-Smart Agriculture. The World Bank, Washington DC

Senesi N (1989) Composted materials as organic fertilisers. Science of the Total Environment 81/82:521–542

Shuval HI, Gunnerson CG, Julius DS (1981) Appropriate Technology for Water and Sanitation -: Night-soil Composting. The World Bank

Sianipar CPM, Dowaki K, Yudoko G (2013) Seven Pillars of Survivability: Appropriate Technology with a Human Face. European Journal of Sustainable Development 2:1–8

Simha P, Ganesapillai M (2017) Ecological Sanitation and nutrient recovery from human urine: How far have we come? A review. Sustainable Environment Research 27:107–116. https://doi.org/10.1016/j.serj.2016.12.001

Sommer SG (2001) Effect of composting on nutrient loss and nitrogen availability of cattle deep litter. European Journal of Agronomy 14:123–133

Sommer SG, Møller HB (2000) Emission of greenhouse gases during composting of deep litter from pig production – effect of straw content. J. Agric. Sci. 134:327–335. https://doi.org/10.1017/S0021859699007625

Somorin TO (2020) Valorisation of Human Excreta for Recovery of Energy and High-Value Products: A Mini-Review. In: Michael O. Daramola, Augustine O. Ayeni (eds) Valorization of Biomass to Value-Added Commodities. Springer, Switzerland, pp 341–370

Sonoki T, Furukawa T, Jindo K, Suto K, Aoyama M, Sánchez-Monedero MA (2013) Influence of biochar addition on methane metabolism during thermophilic phase of composting. Journal of Basic Microbiology 53:617–621

Steiner C, Teixeira WG, Lehmann J, Nehls T, Macêdo JLV de, Blum WEH, Zech W (2007) Long term effects of manure, charcoal and mineral fertilization on crop production and fertility on a highly weathered Central Amazonian upland soil. Plant Soil 291:275–290. https://doi.org/10.1007/s11104-007-9193-9

Steiner C, Das KC, Melear N, Lakly D (2010) Reducing nitrogen loss during poultry litter composting using biochar. J Environ Qual 39:1236–1242. https://doi.org/10.2134/jeq2009.0337

Strokal M, Kroeze C (2014) Nitrous oxide (N_2O) emissions from human waste in 1970–2050. Current Opinion in Environmental Sustainability 9-10:108–121. https://doi.org/10.1016/j.cosust.2014.09.008

Tiquia SM, Tam NFY, Hodgkiss IJ (1996) Effects of composting on phytotoxicity of spent pig-manure sawdust litter. Environmental Pollution 93:249–256

Trimmer JT, Guest JS (2018) Recirculation of human-derived nutrients from cities to agriculture across six continents. Nat Sustain 1:427–435. https://doi.org/10.1038/s41893-018-0118-9

Tsai C-C, Chang Y-F (2019) Carbon Dynamics and Fertility in Biochar-Amended Soils with Excessive Compost Application. Agronomy 9:511. https://doi.org/10.3390/agronomy9090511

Tsai C-C, Chang Y-F (2020) Nitrogen Availability in Biochar-Amended Soils with Excessive Compost Application. Agronomy 10:444. https://doi.org/10.3390/agronomy10030444

Turner BL, Leytem AB (2004) Phosphorus compounds in sequential extracts of animal manures: chemical speciation and a novel fractionation procedure. Environ Sci Technol 38:6101–6108. https://doi.org/10.1021/es0493042

UN (2020a) Sub-Saharan Africa: SDG 6 snapshot in Sub-Saharan Africa

UN (2020b) Sustainable development Goals Report

USDA (2000) National Engineering Handbook: Environmental Engineering, Part 637: Chapter 2 Composting. U.S. Department of Agriculture, Natural Resources Conservation Service

Vandecasteele B, Willekens K, Steel H, D'Hose T, van Waes C, Bert W (2017) Feedstock Mixture Composition as Key Factor for C/P Ratio and Phosphorus Availability in Composts: Role of Biodegradation Potential, Biochar Amendment and Calcium Content. Waste Biomass Valor 8:2553–2567. https://doi.org/10.1007/s12649-016-9762-3

VDLUFA (1999) Standpunkt: Kalium-Düngung nach Bodenuntersuchung Kalium-Düngung nach Bodenuntersuchung und Pflanzenbedarf Richtwerte für die Gehaltsklasse C. VDLUFA

VDLUFA (2014) VDLUFA-Methodenbuch Band II.2. Die Untersuchung von Sekundärrohstoffdüngern, Kultursubstraten und Bödenhilfsstoffen. VDLUFA, Darmstadt, Germany

VDLUFA (2018) Standpunkt: Phosphordüngung nach Bodenuntersuchung und Pflanzenbedarf. VDLUFA

von Münch E, Ingle R (2012) Compilation of 25 case studies on sustainable sanitation projects from Africa

Wang C, Lu H, Dong D, Deng H, Strong PJ, Wang H, Wu W (2013) Insight into the effects of biochar on manure composting: evidence supporting the relationship between N_2O emissions and denitrifying community. Environ Sci Technol 47:7341–7349. https://doi.org/10.1021/es305293h

Wang X, Zhao Y, Wang H, Zhao X, Cui H, Wei Z (2017) Reducing nitrogen loss and phytotoxicity during beer vinasse composting with biochar addition. Waste Manag 61:150–156. https://doi.org/10.1016/j.wasman.2016.12.024

Wang Q, Awasthi MK, Ren X, Zhao J, Li R, Wang Z, Wang M, Chen H, Zhang Z (2018) Combining biochar, zeolite and wood vinegar for composting of pig manure: The effect on greenhouse gas emission and nitrogen conservation. Waste Manag 74:221–230. https://doi.org/10.1016/j.wasman.2018.01.015

Wang L, Li Y, Prasher SO, Yan B, Ou Y, Cui H, Cui Y (2019) Organic matter, a critical factor to immobilize phosphorus, copper, and zinc during composting under various initial C/N ratios. Bioresour Technol 289:121745. https://doi.org/10.1016/j.biortech.2019.121745

Werner C, Panesar A, Rüd SB, Olt CU (2009) Ecological sanitation: Principles, technologies and project examples for sustainable wastewater and excreta management. Desalination 248:392–401. https://doi.org/10.1016/j.desal.2008.05.080

WHO (2006) WHO guidelines for the safe use of wastewater, excreta and greywater: Volume IV: Excreta and greywater use in agriculture. World Health Organization

WHO (2018) Household air pollution and health. World Health Organization

WHO (2019a) Discussion paper: Climate, Sanitation and Health. https://www.who.int/publications/i/item/climate-sanitation-and-health

WHO (2019b) Sanitation: Key facts. https://www.who.int/news-room/fact-sheets/detail/sanitation

WHO and UNICEF (2020) State of the World's Sanitation: An urgent call to transfor sanitation for better health, environments, economies and societies. United Nations Children’s Fund; World Health Organization

Wiedner K, Fischer D, Walther S, Criscuoli I, Favilli F, Nelle O, Glaser B (2015) Acceleration of Biochar Surface Oxidation during Composting? J Agric Food Chem 63:3830–3837. https://doi.org/10.1021/acs.jafc.5b00846

Wu S, He H, Inthapanya X, Yang C, Lu L, Zeng G, Han Z (2017a) Role of biochar on composting of organic wastes and remediation of contaminated soils-a review. Environ Sci Pollut Res Int 24:16560–16577. https://doi.org/10.1007/s11356-017-9168-1

Wu H, Lai C, Zeng G, Liang J, Chen J, Xu J, Dai J, Li X, Liu J, Chen M, Lu L, Hu L, Wan J (2017b) The interactions of composting and biochar and their implications for soil amendment and pollution remediation: a review. Crit Rev Biotechnol 37:754–764. https://doi.org/10.1080/07388551.2016.1232696

Xiao R, Awasthi MK, Li R, Park J, Pensky SM, Wang Q, Wang JJ, Zhang Z (2017) Recent developments in biochar utilization as an additive in organic solid waste composting: A review. Bioresour Technol 246:203–213. https://doi.org/10.1016/j.biortech.2017.07.090

Xingyong J, Meng W, Wenqiao Y, Xiaotang J, Baozhu Y (2016) The influence of Biochar Addition on Chicken Manure Composting and Associated Methane and Carbon Dioxide Emissions. BioResources 11(2):5255–5264

Yanai Y, Toyota K, Okazaki M (2007) Effects of charcoal addition on N_2O emissions from soil resulting from rewetting air-dried soil in short-term laboratory experiments. Soil Science and Plant Nutrition 53:181–188. https://doi.org/10.1111/j.1747-0765.2007.00123.x

Zaha C, Dumitrescu L, Manciulea I (2013) Correlations betweent composting conditions and characteristics of compost as biofertilizer. Bulletin of theTransilvania University of Braşov Series I: Engineering Sciences 6 (55) No. 1

Zhang L, Sun X (2014) Changes in physical, chemical, and microbiological properties during the two-stage co-composting of green waste with spent mushroom compost and biochar. Bioresour Technol 171:274–284. https://doi.org/10.1016/j.biortech.2014.08.079

Zhou C, Liu J, Wang R, Yang W, Jin J (2010) Ecological-economic assessment of ecological sanitation development in the cities of Chinese Loess Plateau. Ecological Complexity 7:162–169. https://doi.org/10.1016/j.ecocom.2009.10.001

Zhou S, Wen X, Cao Z, Cheng R, Qian Y, Mi J, Wang Y, Liao X, Ma B, Zou Y, Wu Y (2021) Modified cornstalk biochar can reduce ammonia emissions from compost by increasing the number of ammonia-oxidizing bacteria and decreasing urease activity. Bioresour Technol 319

Band / Volume 559
Distribution of Relaxation Times for Analysis of Solid Oxide Fuel Cell Stacks
K. Fitzek (2021), 237 pp
ISBN: 978-3-95806-599-4

Band / Volume 560
Machine Learning in Modeling of the Dynamics of Polymer Electrolyte Fuel Cells
W. Zou (2021), vii, 139 pp
ISBN: 978-3-95806-601-4

Band / Volume 561
Introduction Strategies for Hydrogen Infrastructure
S. Cerniauskas (2021), viii, 179 pp
ISBN: 978-3-95806-602-1

Band / Volume 562
Noble Metal Coated Porous Transport Layers for Polymer Electrolyte Membrane Water Electrolysis
C. Liu (2021), 139 pp
ISBN: 978-3-95806-603-8

Band / Volume 563
Modeling and Diagnosis of the Stratospheric Circulation
E. J. Charlesworth (2021), v, 103, A2 pp
ISBN: 978-3-95806-605-2

Band / Volume 564
Potentialfeldmessungen zur Qualitätsbewertung von Bipolarplatten
M. Sietmann (2021), ix, 160 pp
ISBN: 978-3-95806-606-9

Band / Volume 565
Sequential and coupled inversion of time-lapse borehole GPR measurements for vadose zone model parameterization
Y. Yu (2022), XX, 121 pp
ISBN: 978-3-95806-607-6

Band / Volume 566
Cirrus clouds in the extratropical tropopause and lowermost stratosphere region
I. Bartolomé García (2022), iii, 155 pp
ISBN: 978-3-95806-610-6

Band / Volume 567
Stationary and Transient Behaviour of Polymer Electrolyte Fuel Cells
Y. Shi (2022), viii, 172 pp
ISBN: 978-3-95806-611-3

Band / Volume 568
14th Carolus Magnus Summer School on Plasma and Fusion Energy Physics
D. Reiser (Ed.), (2022), 207 pp
ISBN: 978-3-95806-613-7

Band / Volume 569
Spectral induced polarization of calcite precipitation in porous media
S. Izumoto (2022), xviii, 106 pp
ISBN: 978-3-95806-614-4

Band / Volume 570
Technische und ökonomische Bewertung der Polymer-Elektrolyt-Membran Elektrolyse
S. M. Saba (2022), IV, 263 pp
ISBN: 978-3-95806-615-1

Band / Volume 571
Advanced Sintering of Garnet-Based Ceramic Composite Cathodes for All-Solid-State Lithium Batteries
M. Ihrig (2022), VIII, 160 pp
ISBN: 978-3-95806-616-8

Band / Volume 572
Developing an integrated value-based institutional framework for analyzing nexus governance challenges – the case study of Germany
C. Märker (2022), 290 pp
ISBN: 978-3-95806-617-5

Band / Volume 573
Ecological sanitation via thermophilic co-composting of humanure and biochar as an approach to climate-smart agriculture
D. Castro Herrera (2022), XVIII, 127 pp
ISBN: 978-3-95806-622-9

Weitere ***Schriften des Verlags im Forschungszentrum Jülich*** unter
http://wwwzb1.fz-juelich.de/verlagextern1/index.asp

Energie & Umwelt / Energy & Environment
Band / Volume 573
ISBN 978-3-95806-622-9

Mitglied der Helmholtz-Gemeinschaft

JÜLICH
Forschungszentrum