SYSTEM DYNAMICS ANALYSIS OF IMPACTS OF BIOSOLIDS AND BIOSOLIDS-DERIVED BIOCHAR LAND APPLICATION ON AGRICULTURAL SOIL QUALITY

by

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B.Env., Simon Fraser University, 2017

A THESIS SUBMITTED IN PARTIAL FULFILLMENT OF
THE REQUIREMENTS FOR THE DEGREE OF

MASTER OF SCIENCE

in

THE FACULTY OF GRADUATE AND POSTDOCTORAL STUDIES
(Resources, Environment and Sustainability)

THE UNIVERSITY OF BRITISH COLUMBIA
(Vancouver)

January 2020

The following individuals certify that they have read, and recommend to the Faculty of Graduate and Postdoctoral Studies for acceptance, a thesis entitled:

System dynamics analysis of impacts of biosolids and biosolids-derived biochar land application on agricultural soil quality

submitted by Connor Robinson in partial fulfilment of the requirements for

the degree of Master of Science

in Resources, Environment, and Sustainability

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Abstract

Modern sewage treatment results in the production of biosolids (treated sewage sludge), and more biosolids are produced as populations grow and minimum treatment standards increase. This leads to the question of what to do with the biosolids we create. For decision-makers determining a biosolids use strategy, choosing an appropriate use for their context is not a straightforward decision, as it involves weighing pros and cons and making trade-offs. One jurisdiction currently engaged in biosolids management decision-making is the Capital Regional District (CRD) in British Columbia, Canada. The CRD is constructing a new sewage treatment system which will be producing dried Class A biosolids by the end of 2020 and must determine a long-term use for these biosolids. Two biosolids use options that may be considered are (1) direct agricultural land application and (2) conversion to biochar for subsequent agricultural land application. This thesis applied a systems approach to comparing the impact of these two management options (land-applying either dried Class A biosolids or biochar derived from those biosolids) on agricultural soil quality. This was done by using system dynamics (SD) modelling. Biosolids land application led to a greater amount of plant available nitrogen and both stable and labile carbon in the soil, as well as greater crop yields, but also led to more nitrogen leaching and an increased presence of endocrine disrupting compounds. Biochar land application surprisingly did not lead to a greater amount of stable carbon, but did result in a lower amount of nitrogen leaching and no risk due to endocrine disrupting compounds, however also led to a minimal crop yield and meant nitrogen needed to be added to the soil. Reducing the application frequency of biochar did not have a significant influence on soil quality and crop growth, whereas adding mineral fertilizer had a substantial positive effect, and this second strategy should be investigated further. This study contributes to biosolids management decision-making by providing a comparative analysis of the use of biosolids or biosolids-derived biochar on agricultural land as a biosolids reuse option, from the perspective of soil quality.

Lay Summary

Sewage treatment produces a solid residual material – biosolids. There are many options for reusing biosolids; we can apply them on agricultural land as a fertilizer or can heat and convert into biochar, which can also be land-applied. In this thesis built a system dynamics model to simulate these two reuse options and compared their relative effects on agricultural soil quality. I demonstrate that biosolids increases the amounts of soil carbon and nitrogen, crop yield, and nitrogen leaching relative to biochar land application. Applying biochar alone did not produce a viable agricultural system and applying it less often did not help much but adding fertilizer with the biochar substantially improved the agricultural system's performance. This study contributes to the biosolids management decision-making process by comparing the use of biosolids and biosolids-derived biochar on agricultural land as a biosolids reuse option, from the perspective of agricultural soil quality.

Preface

This dissertation is an original intellectual product of the author, Connor Robinson.

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List of Definitions

Amendment	A material which is added to soil in order to improve its texture or fertility; for instance, manure, compost, fertilizer, biosolids, or biochar
Biochar	Biochar produced through pyrolysis of biosolids ('Biosolids-derived biochar')
Pyrolysis	A high-temperature thermal treatment, typically of organic matter-based materials, in the presence of little (or no) oxygen
Sector	A region of the system dynamics model characterized by a similar theme; in other words, a grouping of model elements
Scenario	A representation that describes the system of interest

List of Abbreviations

BC	British Columbia
C	Carbon
C/N	Carbon-to-nitrogen
CCME	Canadian Council of Ministers of the Environment
CCR	Carbon input from crop residues
CLD	Causal loop diagram
CRD	Capital Regional District
DCOM	Decomposition of organic matter
dw	Dry weight
EDC	Endocrine disrupting compound
EU	European Union
FL	Feedback loop
FSOM	Fast soil organic matter
IBI	International Biochar Initiative
N	Nitrogen
NE	Nitrogen effect
NL	Nitrogen loss by leaching
OC	Organic contaminant
ODM	Organic dry matter
OM	Organic matter
PAH	Polycyclic aromatic hydrocarbon
PAN	Plant available nitrogen
PCB	Polychlorinated biphenyl
PPCP	Pharmaceuticals and personal care products
RCY	Relative crop yield

RPB	Relative plant biomass
SD	System dynamics
SQI	Soil quality indicator
SOC	Soil organic carbon
SOM	Soil organic matter
SSOM	Slow soil organic matter
t	Metric tonne, i.e. 1,000 kg
TCC	Triclosan
TCS	Triclocarban
TS	Total solids
USBI	US Biochar Industry

Acknowledgements

Thank you to everyone who supported me on my journey to completing this thesis. I could not have done this work without the unwavering support from my family, friends and colleagues.

To my wife, Pauline Martens, thank you for your constant encouragement, patient listening, and reviewing of drafts of my work. Congratulations on completing this thesis together with me. To my family and friends, thank you for being by my side throughout this journey and for always being a sounding board.

To the wonderful community at the Institute for Resources, Environment, and Sustainability (IRES): your drive to make the future more sustainable all over the world inspires me beyond words. Special thanks to Marco Antonio Vázquez Pérez, Brianne Della Savia, Bronwyn McIlroy-Young, Dr. Sarah Mason-Renton, and Dr. Robin Harder – my Egesta Lab group colleagues, for your regular thoughtful encouragement and critique.

To my supervisory committee: Dr. Gunilla Öberg, Dr. Steve Conrad, and Dr. Mark Johnson. Gunilla, your curiosity and constructive criticism have pushed me to think differently about every problem I encounter, and your regular encouragement for me to think outside the box has developed in me a spirit of reflexiveness. Steve, your technical expertise has provided me with invaluable assistance throughout the development of my model, and your approachable attitude towards problem solving has helped me get over several speed-bumps along the way. Mark, you were there at critical moments in my thesis development, and your feedback helped me refine my research objective and approach.

To all the IRES staff: thank you for all the work you do to ensure the department operates efficiently. Thank you for always responding to my questions and creating such a warm, welcoming environment in which so many people are able to pursue their passion.

To the xʷməθkʷəy̓əm (Musqueam) People: this research – at this institution, in this location – would not have been possible if it were not for your countless years of stewardship of your sacred land by past and present generations. Thank you for allowing this institution and the people that utilize this land, such as myself, to be squatters on your traditional, ancestral, and un-ceded territory.

For Pauline

1. Introduction

1.1. The Sewage Treatment By-product Problem

Humans began treating sewage – a type of wastewater consisting of human waste and grey water from sinks, showers, and household appliances, due to the harm sewage poses when flowing into neighbouring rivers, lakes, and oceans. Sewage contains pollutants that if not removed can threaten human and environmental health (Shahidul Islam & Tanaka, 2004) and wildlife and its habitat (Chambers et al., 1997; Johnston & Roberts, 2009). To manage these pollutants and reduce the subsequent harm to human and ecological systems, communities have long employed a linear, end-of pipe paradigm which sees transporting sewage away from toilets towards outfalls where it is either treated or discharged directly away from areas of concerns (Öberg & del Morales, 2016). As more pollutants and their potential adverse effects are discovered, treatment and discharge standards have become more strict leading to increasing minimum treatment requirements (Lofrano & Brown, 2010), resulting in more materials being removed from sewage before the water is discharged.

The main purpose of sewage treatment is to ensure that the water discharged into the receiving environment is sufficiently clean. The main by-product from sewage treatment process is a semi-solid organic matter-rich substance known as 'biosolids' in North America and 'treated sewage sludge' in the European Union (Öberg & Mason-Renton, 2017) (for ease of reference, this thesis utilizes the term 'biosolids' throughout to describe either of these synonymous by-products). Many Canadian communities (Tessier, 2017), and others worldwide (Environmental Dynamics Inc., 2017), dispose of their biosolids as a solid waste in landfills and waste dumps, however the Federal government discourages this way of using of biosolids as it is a 'waste' of resources that can harm the environment (B.C. Ministry of Environment, 2017; Canadian Council of Ministers of the Environment, 2012). Moreover, given the organic rich composition of biosolids, communities are encouraged to reframe sewage under the paradigm of resource recovery, i.e. to consider it as a source of renewable resources (Berg et al., 2013; Rhyner, Schwartz, Wenger, & Kohrell, 1995).

Proponents of resource recovery have argued for moving away from the end-of-pipe sewage management paradigm towards one of managing waste as a resource, which supports public good and increases community sustainability (City of Vancouver, 2018; S. Lehmann, 2010). Additionally, scholars have argued for a circular resource recovery model where local resources

are utilized over imported ones (Maina, Kachrimanidou, & Koutinas, 2017; Olofsson & Börjesson, 2018). Although complete resource recovery is impossible to achieve due to losses in the system (and the term 'circular' thus a bit of a misnomer), the biosolids are a potential an ongoing, local source of materials. Biosolids, for instance, can be used to make compost, growing medium, or agricultural soil amendment (B.C. Ministry of Environment, 2016a, 2017). Consequently, many jurisdictions worldwide promote the use of local biosolids in agricultural land applications (Öberg & Mason-Renton, 2017).

However, despite potential resource opportunities, biosolids use tends to be a controversial topic due to the risks associated with potential pollutants (Goodman & Goodman, 2016; Mason-Renton, 2017). For instance, in the past in Sweden the use of biosolids in agriculture was contentious due to disagreement on safety margins for metals and on the urgency of reducing their flow into soil, uncertainty about the effects of and dose response to organic pollutants as well as disagreement on how they should be assessed, a lack of consensus on which (and if) pathogens will lead to issues after the biosolids are applied in agriculture, and which nutrient(s) is/are of most importance to recover (Bengtsson & Tillman, 2004). In some communities in British Columbia, biosolids use is controversial, and the reasons for this can include an overall disconnect from what happens after we flush the toilet, to a generally low level of public awareness about advantages and risks tied to biosolids use, and a strong influence of negative media coverage on community support (Whitehouse, 2018).

An alternative for communities concerned with risks associated with organic pollutants in biosolids is to process biosolids into biochar before using it (Schmidt & Wilson, 2014). Biochar is a charcoal-like substance that can be made from most organic-based substances (e.g., biosolids or crop waste biomass) (Biochar for Environmental Management, 2015). When discussing biochar, this thesis refers only to biochar produced from biosolids unless specifically noting biochar in the broader sense. The option space for biochar has expanded in recent years due to its properties and applications, technological advancements, and the growth of industries that can use biochar in their processes (Schmidt & Wilson, 2014). Applying biosolids or biochar on land involves making trade-offs between benefits and risks, or pros and cons.

Several frameworks are available for analyzing biosolids or biochar use in agricultural land applications. For instance, there are those based on sustainability (Overcash, Sims, Sims, &

Nieman, 2005; Pepper, Zerzghi, Brooks, & Gerba, 2008), economics (Kimberley, Wang, Wilks, Fisher, & Magesan, 2004; Lagae, Langemeier, Lybecker, & Barbarick, 2009), or human or ecological risk assessment (Brooks, McLaughlin, Gerba, & Pepper, 2012; Kumar, Hundal, Bastian, & Davis, 2017). Sustainability analyses tend to focus on multiple indicators (e.g. social, economic, environmental) to deduce what solution will have the lowest long-term impact on one or more of these indicators. Economic analyses generally provide an understanding of the financial costs and benefits to land application, while human health/ecological risk analyses generally provide an understanding of the relationship between exposure to certain substances and the related effects on human or ecological health. However, none of these approaches develop a deep understanding of the complex dynamics embodied by the behaviour of the system, which stems from the numerous interactions within the system itself.

This thesis examines some of the complex dynamics of biosolids and biochar land application by utilizing a system dynamics (SD) model to compare the relative change in agricultural soil quality resulting from land application of biosolids and land application of biochar. SD modelling is an evaluation approach for examining complex, dynamic systems as wholes and therefore facilitates the investigation of emergent properties and long-term dynamics of a system (Morecroft, 2010).

This thesis is organized into 6 chapters. The remainder of chapter 1 describes and discusses biosolids and biochar land application in more detail as well as introducing a systems approach as the underlying principle used in the study. Chapter 2 presents the overall problem statement and research objectives. Chapter 3 describes a detailed causal loop diagram (CLD) and an SD model for the evaluation of biosolids and biochar land application. The parameterized variables which define the SD model are described and modelled. Chapter 4 describes the SD model used in this study, elaborates on the process of setting up its initial conditions, and reflects on how the initial conditions fit what might be expected in reality. Chapter 5 applies the SD model to assess the application of biosolids and biochar in a case study of the Capital Regional District. Results from running alternative scenarios are presented and discussed. A summary of all thesis contributions and concluding remarks are presented in chapter 6. Chapter 6 also discusses limitations of the study and directions that future research might take to address several of those limitations.

1.2. Biosolids

In order to investigate biosolids reuse, we must first build an understanding of what biosolids are and how they can be used. This section provides some background on the creation process, composition, and classification of biosolids, as well as outlines the option space and benefits and risks associated with various biosolids use options.

***Biosolids*,** produced through sewage treatment, contain several potentially beneficial and harmful materials. Biosolids are rich in organic matter and nutrients (e.g., nitrogen, phosphorous, potassium, sulphur, and micronutrients); however, they also contain potentially harmful substances such as pathogens, and organic and inorganic contaminants (Sharma, Sarkar, Singh, & Singh, 2017). Biosolids can be used as a fertilizer and help to sustain healthy soils, and their use is regulated based on their concentration of harmful substances.

Land application of biosolids allows for materials (formerly crops) produced on agricultural land to be returned to agricultural land (in the form of biosolids) in a cyclical manner (Lu, He, & Stoffella, 2012; Sullivan, 2013). Biosolids provide nutrients and organic matter (which support plant establishment and growth), supplement and reduce fertilizer use, improve soil physical properties and make cultivation easier, promote soil development, and improve habitat for soil organisms (Kumar et al., 2017; Sullivan, Cogger, & Bary, 2015).

Biosolids agricultural land application also presents several risks. There are for example risks that relate directly to human health such as discomfort, illness, and concern around toxicity, as well as risks that relate to unknown or lesser understood health effects (Öberg & del Morales, 2016). There are also risks that relate to environmental health, including risks to soil productivity, aquatic ecosystems, and the climate (Öberg & del Morales, 2016).

In British Columbia, Canada, biosolids are regulated by the Organic Matter Recycling Regulation (OMRR) (British Columbia, 2002), which is modeled after US EPA Rule 503 (Öberg & del Morales, 2016). The OMRR classifies biosolids that are applied on land in BC into two groups – Class A and Class B (British Columbia, 2017a) – with Class A being more highly treated and containing fewer contaminants (and is thus less restricted in its use) relative to Class B which is generally less treated and contains more contaminants. Class A biosolids must have undergone a number of pathogen reduction processes and one vector attraction reduction process, and their trace element concentrations must be below a specified value (Table 1) as well as the levels of

certain pathogens or indicator organisms must be below a certain threshold (Table 2) (British Columbia, 2017a).

Table 1. Biosolids trace element regulatory limits in BC, the EU and the province of Carinthia in Austria. Class A limits in BC based on the Safety Guidelines for Fertilizers and Supplements (SGFS), Class B limits based on the Organic Matter Recycling Regulation (OMRR). All biosolids EU limits based on the EU biosolids rule, and Class 1 limits in Austria (Carinthia) based on reported values in academic literature.

	BC Limits		**Example EU Limits**	
Trace Element	**Class A** (mg/kg dw) [a]	**Class B** (mg/kg dw) [b]	**EU – All** (mg/kg dw) [c]	**Austria** (Carinthia) – **Class 1** (mg/kg dw) [d]
Arsenic (As)	17 – 33	75	–	–
Cadmium (Cd)	4.4 – 8.9	20	20 – 40	0.7
Chromium (Cr)	233 – 467	1,060	–	70
Cobalt (Co)	33 – 67	150	–	–
Copper (Cu)	167 – 333	2,200	1,000 – 1,750	70
Lead (Pb)	111 – 222	500	750 – 1,200	45
Mercury (Hg)	1.1 – 2.2	15	16 – 25	0.4
Molybdenum (Mb)	4.4 – 8.9	20	–	–
Nickel (Ni)	40 – 80	180	300 – 400	25
Selenium (Se)	3.1 – 6.2	14	–	–
Thallium (Ti)	1.1 – 2.2	–	–	–
Vanadium (V)	144 – 289	–	–	–
Zinc (Zn)	411 – 822	1,850	2,500 – 4,000	200

a British Columbia (2017a)
b British Columbia (2017b) – calculated using cumulative loading limits and the three biosolids application rates modeled in this study
c Council of the European Union (1986)
d Hudcová, Vymazal, and Rozkošný (2019)

Table 2. Biosolids pathogen regulatory limits in BC, the EU and the province of Carinthia in Austria. Class A and Class B limits in BC based on the SFGS. All biosolids EU limits based on the EU biosolids rule, and All biosolids limits in Austria (Carinthia) based on reported values in academic literature.

	BC Limits		**Example EU Limits**	
Pathogen / Indicator	**Class A** (d.w.) [a–c]	**Class B** (d.w.) [a, c]	EU (proposed) – **All** (d.w.) [d, e]	Austria (Carinthia) – **All** (d.w.) [f]
Faecal coliform	< 1,000 MPN / g	< 2,000,000 MPN / g	< 500 CFU *E. coli* / g	–
Salmonella	3 MPN per 4 g	–	No occurrence (50 g)	No occurrence (1 g)
Enteric viruses	< 1 PFU per 4 g	–	–	< 1,000 per g
Viable helminth eggs	< 1 per 4 g	–	–	No helminth eggs

a MPN = most probable numbers
b PFU = plaque forming units
c British Columbia (2017a)
d Council of the European Union (1986)
e CFU = colony forming units
f Hudcová et al. (2019)

According to the OMRR, in terms of trace elements Class B biosolids may contain from 5 times (in the cases of As, Cd, Co, Pb, Mb and Zn) to nearly 15 times (in the case of Cu) the amount in Class A biosolids (Table 1). The trace element concentration of Class A biosolids is regulated by the BC Government under the Fertilizer Act according to the Safety Guidelines for Fertilizers and Supplements (SGFS), and the limits are based on 45-year cumulative loading limits per hectare of land (British Columbia, 2017b). Class B biosolids are also regulated by the BC Government, according to their trace elements concentrations under the OMRR with the limits based on total amount of the trace element in the biosolids (British Columbia, 2017a).

The EU-wide trace element limits apply universally to member states, but these limits can be less stringent than the limits enforced by individual member states (Hudcová et al., 2019). Between 37% and 70% of member states have adopted lower limits for the most toxic heavy metals (Zn, Pb, Cu, Ni, Cd, and Hg), with one-third of member states adopting lower levels for all heavy metals (Hudcová et al., 2019). One such member state is Austria, where in the province of Carinthia the Class 1 biosolids standards are even more restrictive than those for Class A biosolids in BC (Hudcová et al., 2019) (Table 1). The stricter approach adopted by several member states has led to little (or no) biosolids land application in many of these countries – for instance, Croatia, Hungary, Malta, The Netherlands, Romania, Slovakia, and Slovenia all had less than 5% of their produced biosolids used in agriculture in 2014 (Hudcová et al., 2019).

Regulatory classification of biosolids varies by location. Locally, biosolids can be classified in terms of their pathogen content as either Class B or Class A. Class B biosolids are subject to fewer limits than Class A biosolids, and the limits are less stringent; for instance, the faecal coliform density limit for Class B is 2,000 times the limit for Class A and Class B biosolids do not have to meet reduction targets for *Salmonella*, enteric viruses, or viable helminth eggs (Table 2) (British Columbia, 2017a). Class A biosolids must contain equal to or less than 3 most probable numbers (MPN) of *Salmonella* per 4 g, less than 1 plaque-forming unit (PFU) of enteric viruses per 4 g, and less than 1 viable helminth egg per 4 g. In the EU, biosolids limits for bacteria (as indicators of faecal contamination) were proposed in 2,000, and although these limits were later withdrawn, EU member states determined their own standards for maximum concentrations of pathogens in biosolids (Hudcová et al., 2019). According to Hudcová et al. (2019), a province of Austria, for example, set limits for *Salmonella*, enteric viruses and viable helminth ova in its Class 1 biosolids that are more stringent than Class A limits in BC (Table 2).

A detailed comparison of the EU's with BC's regulations is not straightforward because there is inconsistency in the sets of indicators used across these jurisdictions; for example, the EU focused on the faecal coliform *E. coli* alone, whereas Austria does not consider faecal coliform at all, and BC considers all faecal coliforms in one group (Table 2) (Hudcová et al., 2019). Nonetheless, a quick comparison reveals that the EU limits are in certain cases more and in other cases less strict than BC's. For example, the EU requires no Salmonella to be present in 50 g of biosolids, while BC allows a small amount of Salmonella to be present in Class A biosolids, and Austria requires no helminth ova to be present while BC allows a small amount in Class A biosolids (Table 2) (Hudcová et al., 2019).

1.2.1. Option Space for Biosolids

Biosolids can be used in several ways, and one of the most common ways is as a soil amendment in agriculture. Biosolids can also be incinerated or pyrolyzed (with or without energy recovery) or gasified (*Biosolids Engineering and Management*, 2008). Deciding upon the 'best' management option for a given context is not straight-forward and there is not one 'correct' solution as trade-offs must be made; it becomes easier to make the trade-offs between various options when the pros and cons from the different options are made clear.

Agricultural land application of biosolids is common locally (B.C. Ministry of Environment, 2016a), nationally, and internationally. In Australia, for instance, about 75% of biosolids are applied in agriculture (Australian and New Zealand Biosolids Partnership (ANZBP), 2019) and in the EU about 37% of biosolids are used in the same way (Evans, 2012).

There are other options for the use of biosolids, such as incineration or pyrolysis (with or without energy recovery), gasification, and landfilling (B.C. Ministry of Environment, 2016a). Thermal treatments are often grouped together with landfilling as they generally result in net GHG emissions and the destruction of organic matter (both generally seen as a sub-optimal way to recover the resources embedded in biosolids), however this treatment method reduces the risk of pollution to soil and groundwater and, of more relevance to this thesis, the by-products from thermal treatments (e.g., biochar) could be applied in agriculture.

In BC, the majority of biosolids is used in some way on land, and only seven percent is disposed of in landfills. Of the produced biosolids that are used on land, nearly half is used in making compost or as a growing medium (both of which can be applied in agriculture), about ten percent is used directly in agriculture, and about fifteen percent is used in land reclamation, forestry and landfill closure (which is not the same as landfill disposal) combined (B.C. Ministry of Environment, 2017). Sixteen percent of B.C. biosolids is lagooned (treated in lagoons at wastewater treatment plants), while six percent is used in ways aside from those listed above (B.C. Ministry of Environment, 2017).

1.3. Biochar

In order to investigate biochar reuse, we must first build an understanding of what biochar is and how it can be used. This section provides background on the creation process, composition, and classification of biochar, as well as outlines the option space and benefits and risks associated with various biochar use options.

Biochar is a dark coloured, carbon-rich material produced by heating organic matter in a closed container in the presence of little to no air (*Biosolids Engineering and Management*, 2008). The thermal treatment process used to make biochar may be pyrolysis, gasification or incineration (Kookana, Sarmah, Van Zwieten, Krull, & Singh, 2011; Sohi, Krull, Lopez-Capel, & Bol, 2010), however in this thesis I only consider biochar produced through pyrolysis of biosolids – known as ***biosolids-derived biochar (biochar)***. Pyrolysis is described as ”the destructive distillation,

reduction, or thermal cracking and condensation of organic matter under heat and/or pressure in the absence of oxygen" (Williford, Chen, Wang, & Shammas, 2007). This process kills pathogens, sterilizing the end product (*Biosolids Treatment Processes*, 2007).

Humans have known about biochar for a long time, but current biochar research stems from the relatively recent (re)discovery of biochar in the *Terra Preta* soils of Brazil, which are highly fertile and enriched with charred biomass (Singh, Singh, & Cowie, 2010). Although global scale research and development of biochar for environmental management is a recent development, it has been studied since the mid-19th century (*Biochar for Environmental Management*, 2015).

Biochar can be made from a variety of different organic materials, and its properties are influenced by the feedstock type and pyrolysis conditions (e.g., heating rate, maximum temperature). The description or categorization of biochar rarely relies on limits or thresholds of its constituents (as does biosolids) but instead refers to the physicochemical properties of the biochar and its method of production (e.g., maximum temperature, heating rate, residence time). As biochar properties are quite heterogeneous due to varying feedstocks, researchers also classify biochar according to the feedstock used in production: lignin-rich (e.g., wood waste), cellulose-rich (e.g., plant residues), nuts/shells, manure/waste (e.g., poultry waste, cow manure, sewage, or biosolids), algae, and 'black carbon' (Aller, 2016; Singh et al., 2010).

When applied on agricultural land, biochar can aid agricultural productivity by providing nutrients and improving the soil, can help clean the environment by filtering out contaminants, and is a source of sustainable energy due to the pyrolysis gas (biogas) produced (Aller, 2016). Land-applied biochar can act as a long-term carbon sink (Ippolito, Laird, & Busscher, 2012), improve soil physicochemical properties and increase soil pH (Scharenbroch, Meza, Catania, & Fite, 2013), increase soil water holding capacity (C. Liu et al., 2016), reduce soil nitrogen and heavy metal leaching (Paramashivam, Dickinson, Clough, Horswell, & Robinson, 2017), reduce toxic element concentrations in plant tissues (Peng, Deng, Peng, & Yue, 2018), reduce soil ammonia volatilization and increase crop nitrogen use efficiency (Mandal et al., 2016), and increase crop productivity (Jeffery, Verheijen, van der Velde, & Bastos, 2011; Khan, Chao, Waqas, Arp, & Zhu, 2013). Biochar can also absorb potentially harmful substances, reducing their leaching rate and causing them to be more tightly bound to the soil (i.e., unavailable to plants) (Aller, 2016; Tan et al., 2014).

While studies suggest numerous benefits of biochar land application, there are also notable concerns. The sorption capacity of biochar could be detrimental if agrochemicals are taken up, as this reduces the effectiveness of the agrochemical in the long term (Aller, 2016). Some of the harmful substances originally presented in biosolids, such as heavy metals, can be found and may even be concentrated in the biochar (Hossain, Strezov, Chan, Ziolkowski, & Nelson, 2011; Van Wesenbeeck, Prins, Ronsse, & Antal, 2014) – these metals can stunt plant growth and harm soil biota (Chibuike & Obiora, 2014). Moreover, increased contaminants of emerging concern in biochar-amended soils may be detrimental in the short-run as there are indications that harmful substances may be better retained (Li et al., 2017), and in the long-run as nitrogen and organic matter begin to be lost from the soil once the absorptive capacity of the biochar is reached (Hussain et al., 2017).

Unlike biosolids, for which there are clearly defined regulatory limits pertaining to its constituents, and the regulation of biochar depends on how each jurisdiction defines it. In Ontario, for instance, biochar is defined as waste and wastes are not allowed to be applied on land; however, in BC biochar is defined neither as a waste nor as a soil amendment (Dennis, 2011). Due to the ambiguity surrounding its classification, in regions like BC biochar could theoretically be applied without restriction; however, the government could enact regulations that restrict this in the future.

In light of this lack of aligned regulations, the International Biochar Initiative (IBI) has created a set of guidelines for definition and testing of biochar that is used in soil; these IBI Biochar Standards relate to physicochemical properties of biochar only (International Biochar Initiative, 2015). The IBI has defined three test categories for biochar: a) basic utility properties *(required)*; b) toxicant assessment *(required)*; and c) advanced analysis and soil enhancement properties *(optional)* (International Biochar Initiative, 2015). A summary of these tests and the proposed test criteria is presented in Table 3.

Table 3. Proposed tests for all biochar to be used on soil according to the International Biochar Initiative (IBI) Biochar Standards. (A) Basic utility properties assessment and (B) toxicant assessment. Criteria for (A) based on IBI guidelines. Criteria for (B) international maximum allowable thresholds based on the IBI, and criteria for (B) Class A and Class B criteria based on the SGFS and OMRR, respectively.

Test	**Details for Assessment According the Test**			
A: Basic Utility Properties *(required)*	Parameter	Criterium [a]		Unit [b]
	C_{org}	10% minimum; Class 1: ≥60% Class 2: ≥30 to <60%; Class 3: ≥10% to 30%		% mass, d.w.
	$H:C_{org}$	0.7 minimum		molar ratio
B: Toxicant Assessment *(required)*	Parameter	(Range of) Maximum Allowable Thresholds*		Unit [b]
		International [a]	BC standards for Class A [c] / Class B [c, d]	
	Germination Inhibition Assay	**Pass/Fail**	–	–
	PAHs (sum of 16 US EPA PAHs)	**6 – 300**	–	mg/kg d.w.
	Dioxins/Furans	**17**	–	ng/kg WHO-TEQ d.w.
	PCBs	**0.2 – 1**	–	mg/kg d.w.
	Arsenic (As)	13 – 100	**25 / 75**	mg/kg d.w.
	Cadmium (Cd)	1.4 – 39	**6.7 / 20**	mg/kg d.w.
	Chromium (Cr)	39 – 1,200	**350 / 1,060**	mg/kg d.w.
	Cobalt (Co)	34 – 100	**50 / 150**	mg/kg d.w.
	Copper (Cu)	143 – 6,000	**250 / 2,200**	mg/kg d.w.
	Lead (Pb)	121 – 300	**170 / 500**	mg/kg d.w.
	Mercury (Hg)	1 – 17	**1.7 / 15**	mg/kg d.w.
	Molybdenum (Mo)	5 – 75	**6.7 / 20**	mg/kg d.w.
	Nickel (Ni)	47 – 420	**60 / 180**	mg/kg d.w.
	Selenium (Se)	2 – 200	**4.7 / 14**	mg/kg d.w.
	Thallium (Ti)	–	**1.7 / –**	mg/kg d.w.
	Vanadium (V)	–	**217 / –**	mg/kg d.w.
	Zinc (Zn)	416 – 7,400	**617 / 1,850**	mg/kg d.w.

a International Biochar Initiative (2015)
b d.w. = dry weight
c British Columbia (2017b)
d British Columbia (2017a)
** Note: for toxicant assessment (test category B), IBI recommends the parameters in the Parameter column be analyzed but recommends if there are local regulations for soil amendments that those be followed instead. Therefore, when identifying the 'range of maximum allowable thresholds' for toxicant assessment, I have bolded the values which should be used in BC; only for the upper four parameters should we refer to the IBI standards here in BC. The two values reported under 'BC standards' are the mean of Class A values and the Class B standard, both taken from Table 1.*

In terms of basic utility properties, the main parameter used to classify biochar is organic carbon, with a minimum of 10% organic carbon required for the material to be considered biochar (International Biochar Initiative, 2015). Biochar with at least 60% organic carbon is in Class 1, biochar with at least 30% to 60% organic carbon is Class 2, and biochar with at least 10% to 30% organic carbon is Class 3 (International Biochar Initiative, 2015). Moreover, the molar ratio of hydrogen to carbon in biochar indicates its carbon stability (lower H:C means carbon is more stable), and this value should be no greater than 0.7 (International Biochar Initiative, 2015). The other parameters that must be declared but have no set criteria are moisture, total ash, total nitrogen, pH, EC, liming (if pH >7) and particle size distribution (International Biochar Initiative, 2015).

In terms of toxins, the IBI proposes analysis of a set of parameters that is similar to those required for Class A and B biosolids in Canada, though IBI provides a range of internationally accepted values whereas in BC there are specific thresholds for the two classes of biosolids (British Columbia, 2017a, 2017b; International Biochar Initiative, 2015). The other parameters that must be measured are Boron (Bo), Chlorine (Cl), and Sodium (Na) (International Biochar Initiative, 2015). In cases where the local jurisdiction (in this case, BC) does not have defined regulatory limits for these parameters in soil amendments, IBI states that users should refer to international standards; thus, in BC one would utilize the international standards and tests when examining PAHs, Dioxins/Furans, PCBs, and performing the Germination Inhibition Assay test (International Biochar Initiative, 2015).

1.3.1. Option Space for Biochar

Biochar can be used in many different ways and in many different industries. Like with biosolids, deciding upon the 'best' option is not straight-forward as trade-offs must be made, but the trade-offs become easier to understand when the pros and cons from the different options are made clear.

Rationales for producing biochar include soil enhancement, waste management, mitigation of climate change, and energy production (*Biochar for Environmental Management*, 2015). Biochar can also be used in animal farming, in the building sector, for decontamination, for biogas production, in water treatment, and in other industrial applications like cement production (Schmidt & Wilson, 2014). As biochar production and regulation are not common worldwide in

the same way as for biosolids (due to wastewater treatment being so ubiquitous), data on the distribution of biochar uses is difficult to find.

Nonetheless, agricultural land application of biochar has been occurring for quite a long time (*Biochar for Environmental Management*, 2015). Recent research has demonstrated that biochar can act similar so a humic substance in the soil, mediating redox processes in biological processes (Lipczynska-Kochany, 2018). Biochar is produced for land application in BC (Canadian AgriChar Inc., 2017) and the United States (US Biochar Industry (USBI), 2018) and makes up the largest global market share for biochar (Grand View Research Inc., 2018).

1.4. System Dynamics as a Problem Evaluation Approach

Land application of biosolids and biochar involve a series of complex interactions within the soil system, the sum of which can be difficult to assess with a linear model. An event-oriented, linear approach sees the world as a complex succession of events (e.g., *'there is a nitrogen deficiency in the soil'*) and solutions as fixes (e.g., *'we must add nitrogen fertilizer'*), assuming each event has a cause and that altering the cause will change the event. One limitation to this type of thinking is that while each cause may have an effect, that cause may have another cause, events can have multiple causes, *et cetera*. This can lead to unintended consequences (e.g., *'there is excess nitrogen and it is being leached'*).

A systems approach seeks solutions designed in ways that are sensitive or responsive to feedbacks in their environment by seeing systems as wholes that have a purpose (i.e., exhibit behaviour) (Meadows, 2008). This way of seeing things was for example used by Jay Forrester in the 1960s in reference to industrial systems (Forrester, 1968) and significantly advanced by Donella Meadows in the early 1970s in the seminal report, *Limits to Growth* (Meadows, Meadows, Randers, & Behren, 1972). Viewing an agricultural system as a connected whole – an interconnected set of the elements deemed relevant which influence one another in the system (e.g., how carbon and nitrogen accumulation are interlinked and affect nutrient levels in soil) – aids in balancing complex objectives (e.g. *'I need enough nitrogen to grow [a certain amount] of my crop, but not so much that excess is susceptible to leaching'*).

Following a systems approach, SD modelling is a problem evaluation approach based on the idea that the system structure generates its behaviour (Stave, 2003), recognizing the interactions among distinct but interrelated system components that drive system behaviour (Mirchi, Madani, Watkins,

& Ahmad, 2012). An SD model simulates a system by representing it as a system of interconnected stocks (quantities) that are subjected to a series of differential equations which can cause increases or decreases in these stocks.

In order to analyze the long-term dynamics of the application of biosolids and biochar and their constituents to agricultural soil this thesis applies SD modelling to examine the dynamics of carbon, nitrogen, crop growth, and endocrine disruptors over time, simulating annual fertilizer application. This analysis follows the principles established by Meadows and others in that agricultural soil can be represented as an assemblage of distinct but interconnected components forming a larger system which exhibits behaviour which can be modeled and observed. In the case of biosolids and biochar land application, predictions of the dynamics of soil quality indicators inform decision-making at the farm level as well as regional agricultural health analysis.

SD modelling has been widely used in the water and wastewater sector – for example, in modelling reactive nitrogen in wastewater systems (Nascimento, Kiperstok, Martin, Morato, & Cohim, 2018), the water-energy-carbon nexus of urban water systems (Chhipi-Shrestha, Hewage, & Sadiq, 2017), water resources management options in Singapore (Xi & Poh, 2013), and end-use water-energy interactions in London (De Stercke, Mijic, Buytaert, & Chaturvedi, 2018). SD modelling has also been applied in the agricultural sector to for example model plant biomass composition and soil organic matter, runoff, sediment transport and nutrient dynamics (Reuss & Innis, 1977; Thornley, Bergelson, & Parsons, 1995; Yeh, Wang, & Yu, 2006).

Although SD modelling has been used in a variety of contexts, often SD models encompass a wide breadth of topics at a coarse resolution so that the model can simulate a large system. However, for certain stakeholders, like farmers who manage agricultural fields for soil quality or health, a fine resolution model that provides specific, tangible outputs may be equally or perhaps even more useful in guiding the decisions made on their field.

To the best of my knowledge, this study is the first application of an SD model that specifically examines the impact of biosolids and biochar agricultural land application on soil quality, at a resolution that would be beneficial to farmers. This thesis contributes to the biosolids use literature through the application of SD modelling and is intended to provide decision-makers with information regarding the two biosolids use scenarios listed in chapter 2 (biosolids or biochar agricultural land application).

2. Research Objective

The objective of this research is to compare two hypothetical soil amendment land application scenarios from a soil quality point of view:

a) Agricultural land application of Class A biosolids

b) Agricultural land application of biochar produced from Class A biosolids (biochar; biosolids-derived biochar)

I address these objectives by using a case-study approach and by comparing how these two scenarios impact soil quality using as indicators: carbon, nitrogen, and endocrine disrupting compounds (EDC) in the soil as well as crop biomass. I assess the relative benefits and consequences related to how these indicators change over time under the two scenarios.

The motivation for this research lies within changing regulation that requires all wastewater treatment plants to treat their sewage to at least secondary level (reference to regulation). This thesis focuses on the Capital Regional District, (CRD) in British Columbia, which, because of the new regulation has been forced by the Federal Government to construct a new wastewater treatment plant. A new wastewater treatment system is presently being built and will soon be producing dried Class A biosolids, and the CRD has yet to determine the long-term beneficial use for these biosolids (Environmental Services Committee, 2018, January 10). Two potential options are direct land application of the biosolids, or pyrolysis of the biosolids to produce biochar, which could be applied in agriculture. Here, I compare direct land application of the biosolids, and biochar produced from those biosolids.

Biosolids and biochar land application lead to different effects on the soil system, and deciding which use is more beneficial is not a simple decision. A system dynamics (SD) modelling approach was therefore used to assess the pros and cons.

3. Methods

3.1. Case Study Background

To examine the application of biosolids and biochar, this study utilized as its case the development of a wastewater treatment facility on Vancouver Island in British Columbia, Canada to treat sewage collected within the Core Area of the CRD service area. The CRD is the regional government for 13 municipalities and three electoral areas on southern Vancouver Island and the nearby Gulf Islands (Figure 1) (Capital Regional District, 2018a). The CRD provides service, infrastructure and financing on regional issues involving municipalities and electoral areas, sub-regional issues that transcend two or more jurisdictions, and local issues within the electoral areas (Capital Regional District, 2018d). The sewage treatment expansion currently underway in the CRD is a trans-jurisdictional issue involving seven of the CRD's Core Area municipalities and two First Nations.

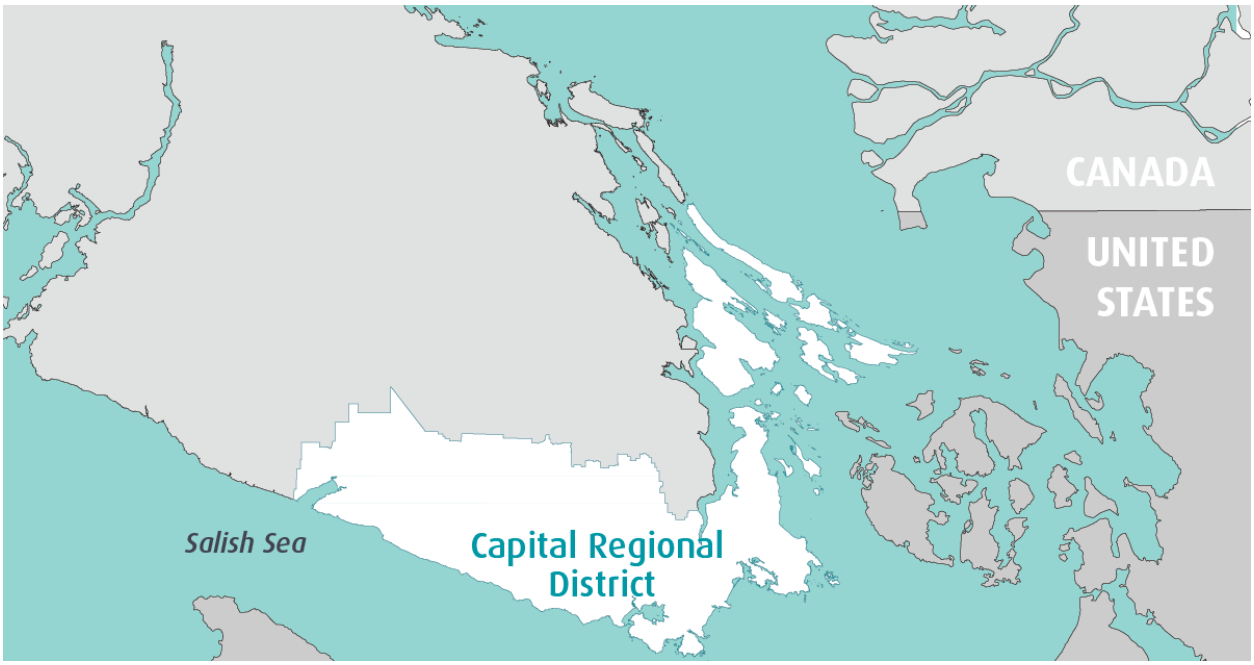

Figure 1. The Capital Regional District (CRD), located on the southern end of Vancouver Island (Capital Regional District, n.d.). The CRD is the regional government for 13 municipalities and three electoral areas.

The Core Area is a subset of CRD municipalities comprised of Victoria, Esquimalt, Saanich, Oak Bay, View Royal, Langford, and Colwood, plus the Esquimalt and Songhees First Nations (Figure 2) (Capital Regional District, 2018c). The seven municipalities are the most densely populated jurisdictions (excluding Sidney) in the CRD (Figure 3) and contain around 82% of the entire population of the CRD (Capital Regional District, 2017b); thus, the Core Area Wastewater

Treatment Project (CAWTP) will serve the majority of the region's population when it comes online by 2020.

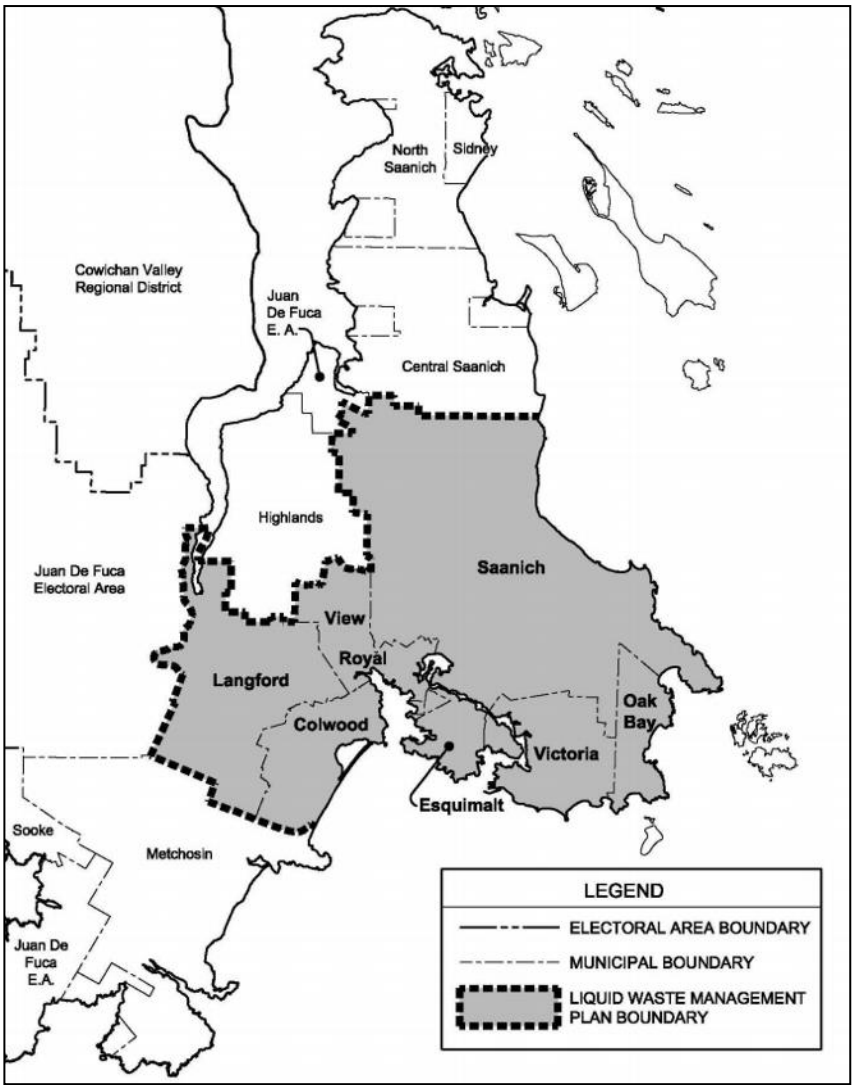

Figure 2. The Core Area in the CRD (modified from Capital Regional District (2019)). The Core Area is home to over four-fifths of the population of the CRD and will be serviced by the upcoming wastewater treatment project.

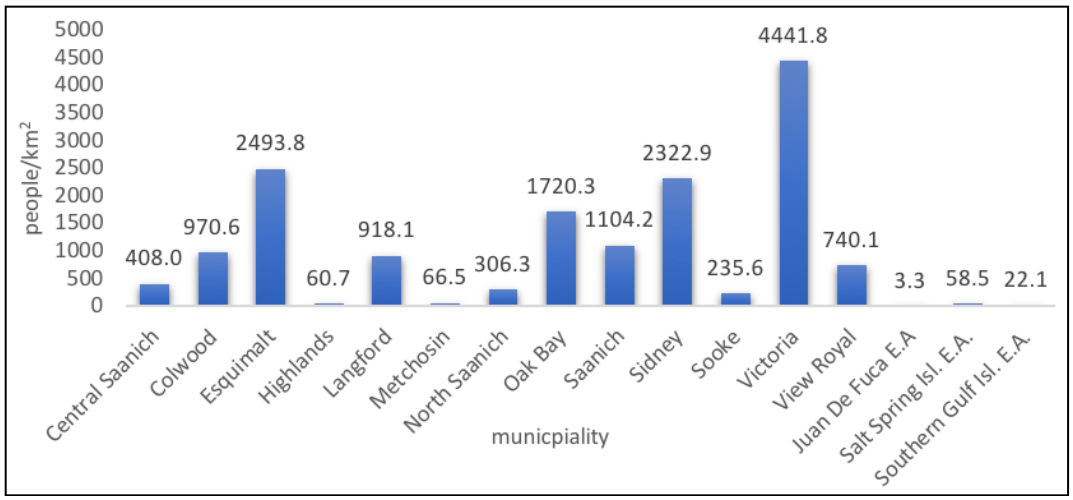

Figure 3. Population density of Core Area municipalities and electoral areas (data from Capital Regional District (2017b). The Core Area comprises seven of the top eight most densely populated municipalities.

Highlighting the development of sewage management in the CRD, Figure 4 illustrates the history of sewage management extending back to 1915 – with multiple decades of direct raw sewage discharge into the ocean followed by the current period of sewage treatment being implemented within Canada (Capital Regional District, 2017a). In the 1990s, the discussions about sewage treatment escalated leading to the formal planning for a wastewater treatment plant.

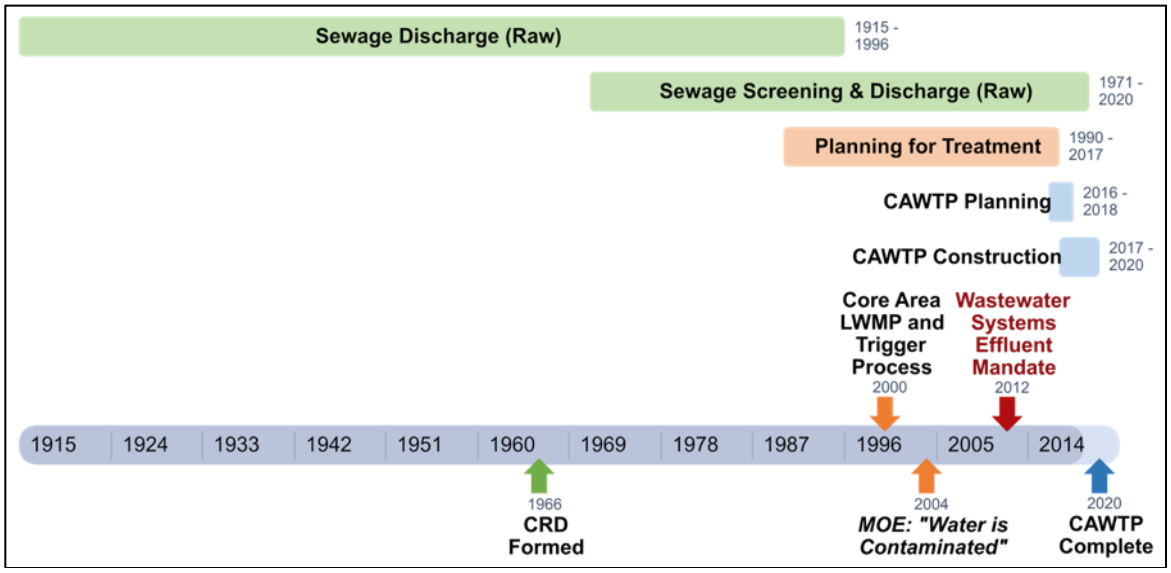

Figure 4. Timeline of sewage management in the CRD (Capital Regional District, 2017a).

The region still discharges raw (but screened) sewage into the ocean (green bars in the figure). In 1966 the CRD governance structure was officially formed leading to sewage development over the next two decades throughout the Core Area. Through the 1970s and 80s a monitoring system was constructed, and the system was upgraded with the construction of two pump stations. In the

1990s, planning for wastewater treatment commenced, and in 2004, the BC Ministry of Environment (MOE) requested that the CRD's Core Area Liquid Waste Management Plan (CALWMP) be amended to outline how and when the region would begin treating its wastewater.

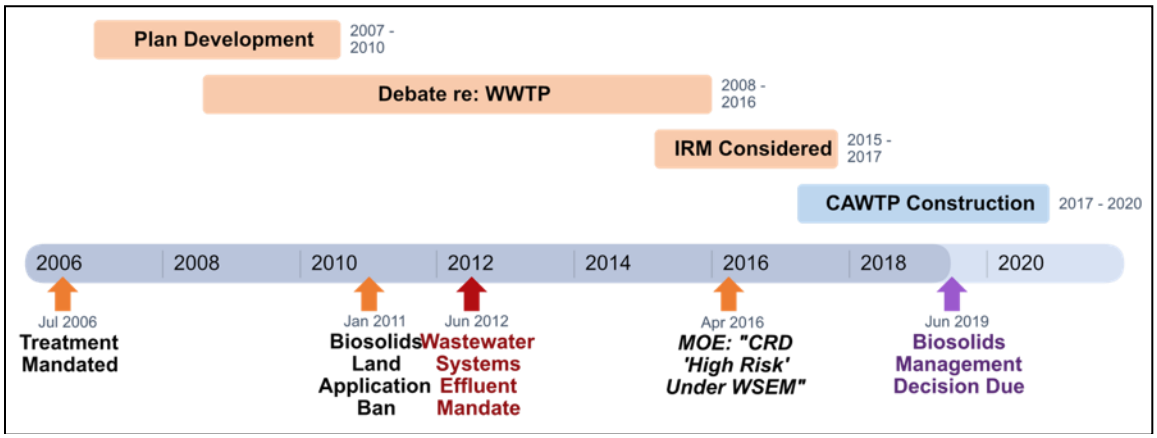

Figure 5. Timeline of biosolids and integrated resource management in the CRD. (Capital Regional District, 2017a)

Focusing in on the last 13 years, Figure 5 presents a recent timeline of sewage management in the CRD from 2006 to present that demonstrates an acceleration of sewage management decision-making for the region, which consequently led to the acceleration of the consideration of biosolids management.

In 2006, the Federal government mandated that the region begin planning for the construction of a WWTP. In the following three years an early plan was developed, and in 2010 biosolids planning began. A Biosolids Energy Centre (BEC) was being considered initially to utilize biosolids as an energy source, and in 2011 a biosolids land application ban was enacted (and is still in effect) by the CRD (CRD Environmental Sustainability Committee, 2011, May 25), preventing biosolids from being applied within land for any purpose in the CRD. From 2008 to 2016 there was an active debate among the municipalities within the CRD and members of the community regarding the siting of the WWTP. In July 2012, the Federal Wastewater Systems Effluent Regulations required all Canadian jurisdictions to have secondary treatment by the end of 2020 (Canada, 2012). Under this regulation, three of the CRD's wastewater management facilities (outfalls) would not comply without upgrades.

In 2016, the CRD Board approved the Core Area Wastewater Treatment Project Board (CAWTPB), which later that year forwarded a recommendation to the CRD for a single tertiary

treatment plant to be sited at McLoughlin Point. In December 2016, the CRD begun the process of selecting a proponent to construct its biosolids facility to be built at Hartland Landfill.

In its 2015-2018 Strategic Management Plan, the CRD had indicated that pursuing Integrated Resource Management (IRM) was a strategic priority. The CRD defines IRM as the integration of solid and liquid waste streams, combining currently landfilled or diverted material and biosolids, to maximize resource recovery by processing of some, or all, of those materials (Environmental Services Committee, 2018, January 10). However, due to uncertainty and disagreement surrounding IRM procurement this direction was terminated in early 2018 (Environmental Services Committee, 2018, January 10). In its place, the CRD decided to pursue procurement of a stand-alone Residuals Treatment Facility (RTF) which will be sited in the footprint of the Hartland Landfill. The RTF will produce dried Class A biosolids, which must be beneficially reused as per the CALWMP (Environmental Services Committee, 2018, January 10).

In December 2017, the preferred proponent for the RTF was announced by the CAWTPB, and the contract was awarded in February 2018. The RTF, to be located in the footprint of Hartland landfill in Saanich, will treat residual solids coming from the CRD's wastewater treatment plant into dried Class A biosolids (Capital Regional District, 2018b). Regardless of the long-term use of these biosolids, there is a need to develop an interim solution that does not include storage of the material at Hartland landfill (Environmental Services Committee, 2018, January 10).

The CRD presents an ideal case study for the evaluation of biosolids application because the CRD is set to start producing biosolids and is investigating the use of these biosolids. This consideration is set on the historical backdrop of a lack of attention towards the question of biosolids management. A number of consultant reports have been prepared for the CRD on this topic in recent years (Environmental Dynamics Inc., 2017; Stantec, 2016; SYLVIS, 2018), however these reports have been focused on comparing uses using metrics such as the Triple Bottom Line, providing an overview of feasibility and markets or reviewing the literature on uses, without performing any detailed model-based analyses of individual uses. This presents an opportunity to contribute to the policy discussion through the examination of two biosolid use scenarios using a SD modelling approach.

3.2. Research Approach: System Dynamics Modeling

In this section, I first explain the selection of a modeling approach, as well as the key soil quality indicators that inform the development of the 'Thesis Model' used for evaluating biosolid and biochar land application. I then illustrate the interactions and feedbacks between the soil quality systems of nitrogen, carbon, and EDCs through a causal loop diagram. I introduce the development of the Thesis Model by first describing the building blocks of an SD model and a referenced nutrient model developed by Bossel (2007) and how it provides a structural foundation for the development of the Thesis Model. I conclude by describing how I parametrized the Thesis Model so that it is ideally suited to study the application of biosolids and biochar in the CRD.

In this study I selected Stella as the modelling tool for evaluating the core interrelationships between soil quality parameters and to model the system dynamics in moving between different states of equilibrium embodied by compost and manure, biosolids, and biochar applications. I wanted to learn how to build a model from the ground up, and to avoid the necessity of learning how to program or code a software. Stella was therefore a suitable choice for my purposes because it does not force the modeler to learn how to code but instead utilizes a graphical interface through which the modeler defines the relationships by visually connecting elements and defining their relations to one another. This program provided the opportunity to develop an understanding of the underlying system structure informing soil quality, focusing on analyzing the system's response to a disturbance and gaining insight on the structure that generates its overall behavior (Barlas, 1996). Stella also provided the option, through its visual interface, to more easily communicate the results of this study to a broader audience.

3.2.1. Selection of Soil Quality Indicators

Given the desire to develop a model that would inform land application of biosolids and biochar in an agricultural setting, soil quality indicators were used as measure of system performance. Soil performs important functions, but these functions are difficult to quantify and are therefore often approximated with the use of soil quality indicators (SQI) (Lal, 2016). These attributes of a soil (e.g. physical, chemical, biological, and ecological) inform soil quality and functionality, which in turn inform the functions and ecosystem services the soil performs (Lal, 2016; Winder, 2003). It is worth nothing that *soil health* and *soil quality* should not be used interchangeably. Whereas soil quality refers to what soil does or its functions, soil health is related to the finite, dynamic and

biologically active nature of the soil and is directly connected to plant health (Lal, 2016). I selected nitrogen, carbon and EDCs as key soil quality indicators given their prevalence in literature and association with reference models. Table 4 provides a summary of the relative pros and cons of each selected soil quality parameter and are individually detailed in subsequent sections.

Table 4. Pros and cons of the selected parameter categories for the biosolids and biochar scenarios.

	Biosolids		**Biochar**	
Category	**Pro**	**Con**	**Pro**	**Con**
Carbon	Contains more 'fast' carbon	Contains less 'slow' carbon	Contains more 'slow' carbon	Contains less 'fast' carbon
Nitrogen	Provides nitrogen for soil and plants	Too much nitrogen can lead to leaching	Reduces nitrogen leaching	Contains little nitrogen
ECs	–	Contains EDCs	Contains less or no EDCs	–

3.2.1.1. Carbon

Soil organic matter (SOM) is seen as one of the main indicators of soil quality and key to maintaining soil functions (Wiesmeier et al., 2019). SOM improves the water retention and nutrient retaining capacity of the soil and promotes development of soil structure (Lal, 2016; J. Lehmann & Kleber, 2015). SOM can be expressed of as a continuum of gradually decomposing organic (carbon-based) compounds (soil organic carbon (SOC) and soil inorganic carbon (such as carbonates)), from plant and animal residues down to tiny molecules and ultimately to gas (J. Lehmann & Kleber, 2015). SOC is a key source of energy for microorganisms and is therefore a driver of several soil functions and ecosystem services (Lal, 2016). The three most important aspects of SOC are its quality, quantity, and the rate of turnover or dynamics (Lal, 2016).

In a broader sense, carbon in the soil is one of the many components of the global carbon cycle that has inputs, outputs, and transformations of different forms of carbon to, from and within it. The global carbon cycle includes the land, atmosphere, the oceans, freshwater, geological reservoirs (Prentice et al., 2001), and extends deep into the Earth's core (Dasgupta & Hirschmann, 2010).

This thesis focuses on land surface processes (shown as a dashed green box in Figure 6) – specifically, those involving plants and soils that are of high relevance when considering soil quality.

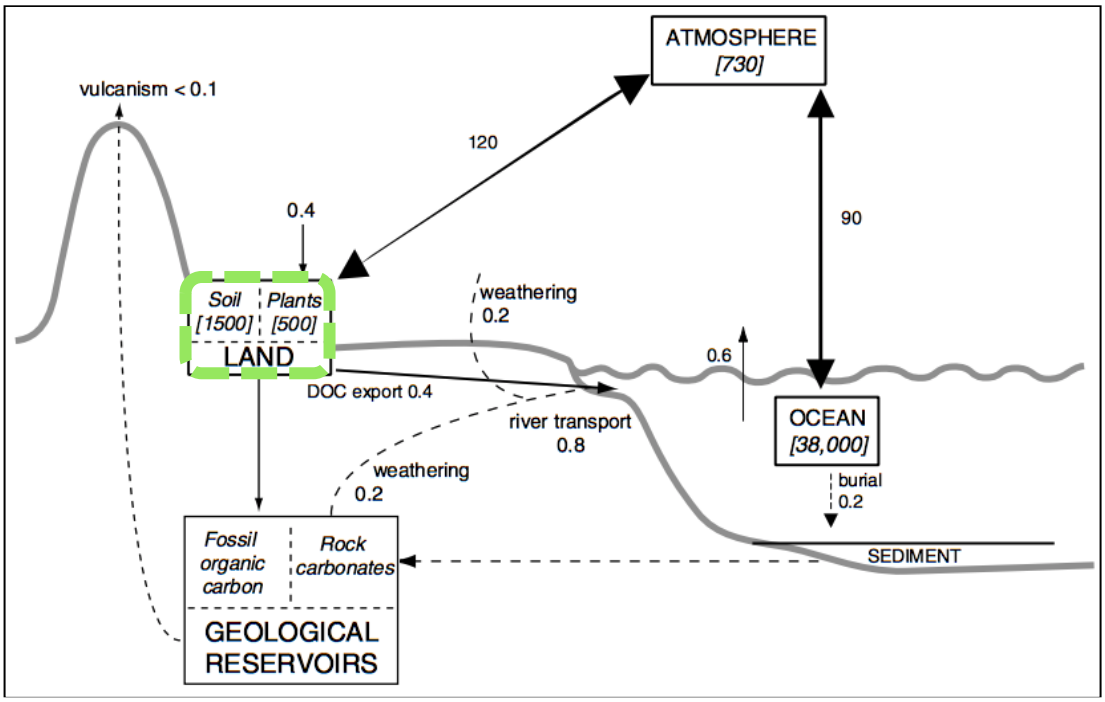

Figure 6. Main components and approximate distribution of carbon in its natural cycle. Values in Pg carbon yr-1. Modified from Prentice et al. (2001).

SOM and SOC can be approximated based on one another, and in this study, I model SOC – the carbon in soil organic matter. Both SOM and SOC can be used when describing the carbon content of soil, and the amount of SOC in SOM can range from 45% to 60% (Lal, 2016), though SOC is typically approximated from SOM as SOM/1.724, or around 0.58*SOM (Tian, Granato, Pietz, Carlson, & Abedin, 2006).

This thesis draws on Bossel (2007), who assumes there are two pools of SOM. While this is a simplification (Lal, 2016), this practice is common (J. Lehmann & Kleber, 2015) and for the sake of simplicity, I follow this approach, assuming that soil carbon dynamics can be represented by a 'fast' (i.e. labile) and a 'slow' (i.e. stable) pool of SOM (termed FSOM and SSOM, respectively). The 'slower' pool (SSOM) contains carbon, while the 'faster' pool (FSOM) contains both nitrogen and carbon. I assume the slower SOC pool (C-SSOM) is effectively inert (does not decompose and is only lost by erosion) and the faster pool (C-FSOM) does decompose, on the order of days or weeks, when conditions permit.

There is large variation in the amount of SOC in soils across the world, and a number of review studies provide a general baseline condition to which the results of this thesis can later be compared (Table 5). At the local scale, values have been reported from around 7,000 – 20,000 kg SOC ha-1,

at a national scale the value has been reported up to 75,000 t SOC ha^{-1}, and at a global scale the reported values range from around 37,000 – 62,000 t SOC ha^{-1} (though these values are based on three, one and two examples, respectively, and there are likely instances of lower and higher values at all scales) (Minasny et al., 2017; Neufeld & Paul, 2019; Sanderman, Hengl, & Fiske, 2018; Thomas et al., 2019; Y. Zhao, Krzic, Bulmer, & Schmidt, 2008).

Table 5. Examples of soil organic carbon (SOC) values (t SOC ha^{-1}) from local, national and international analyses.

Local		National		International	
Value	Description (Source)	Value	Description (Source)	Value	Description (Source)
6.8 – 20.4	Agricultural land in Delta, BC (Neufeld & Paul, 2019)	75	Canadian average, top 30 cm (Minasny et al., 2017)	36.9	Global average, top 30 cm (Minasny et al., 2017)
10.6	BC forest soils (Y. Zhao et al., 2008)			62	Global scale (Sanderman et al., 2018)
7.7	Baseline value (Thomas et al., 2019)				

3.2.1.2. Nitrogen

All plants use nitrogen for growth, and nitrogen is a component of proteins. Nitrogen is one of the main indicators of plant growth capability, as well as soil quality, as it improves crop growth and yield, the quality of the crop, photosynthesis, and more. Nitrogen deficiency has negative effects on plant growth (height, leaf area, dry weight) and photosynthetic rate (D. Zhao, Reddy, Kakani, & Reddy, 2005). At low nitrogen levels there is considered to be nitrogen stress or deficiency and at high levels there is nitrogen excess or sufficiency (J. Liu et al., 2010). However, excess nitrogen in the soil can be leached (removed by water) and this often ends up in water bodies where problems like eutrophication can occur.

Figure 7 illustrates the global nitrogen cycle, which includes the land, atmosphere, oceans and soils, with nitrogen cycling through soils and denitrification accounting for the biggest flows of nitrogen in the system (followed by leaching and transport to ocean, then deposition and volatilization of oxidized and reduced nitrogen) (Fowler et al., 2013). Within the broader system, soil nitrogen is one of the many components of the global nitrogen cycle that has inputs, outputs, and transformations of different forms of nitrogen to, from and within it.

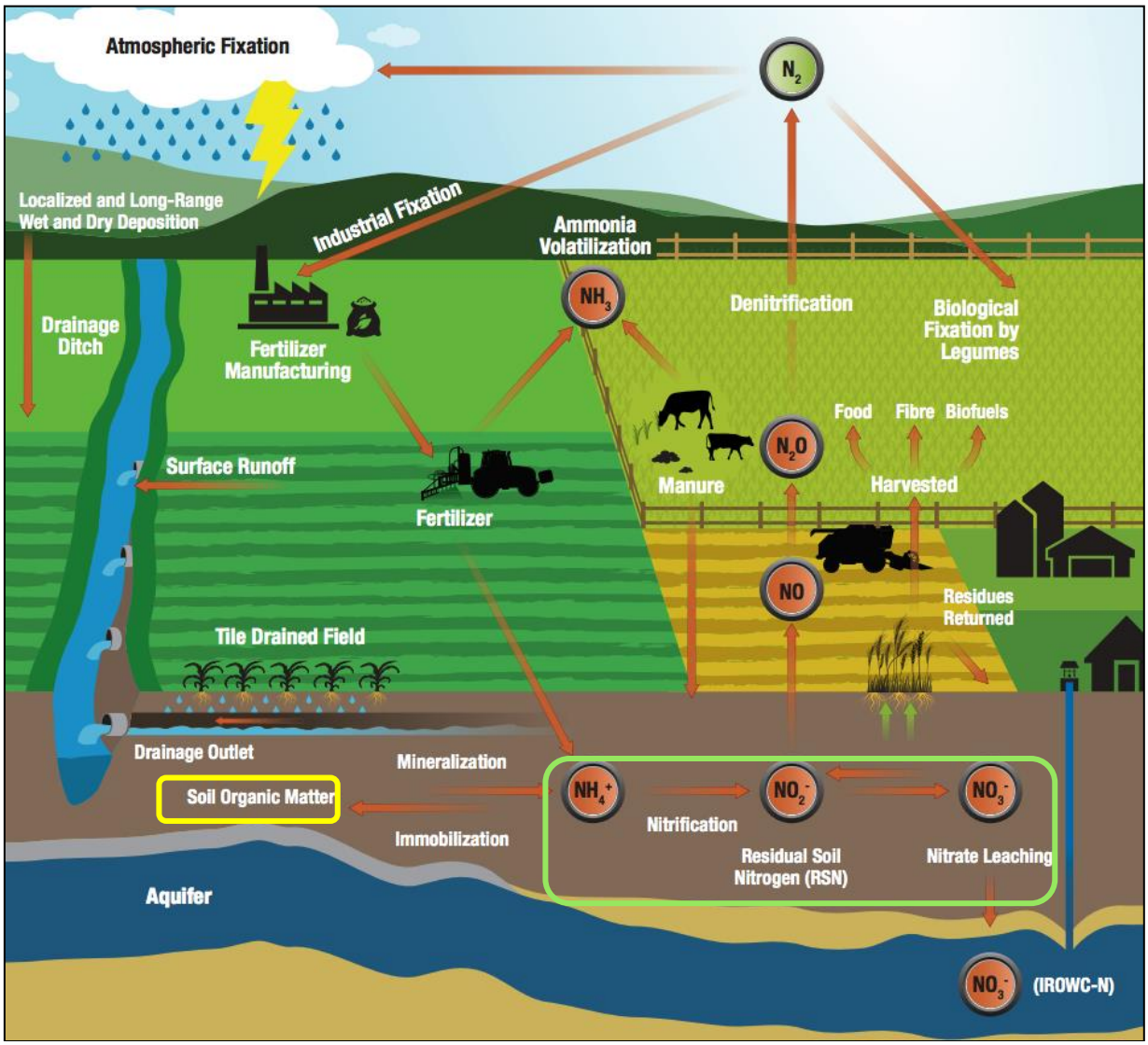

Figure 7. Main components of the nitrogen cycle in the context of an agricultural ecosystem. Modified from C. Drury et al. (2016).

Two of the main components of nitrogen in soil are plant available nitrogen and organic nitrogen. Nitrogen in soil that is bound to OM (carbon) is mainly held in the FSOM (SSOM contains little nitrogen) (Binkley & Hart, 1989). The remaining nitrogen in soil is inorganic – mainly nitrate (NO_3) and ammonia (NH_4), which is more readily available to plants than the nitrogen bound to FSOM (Schulten & Schnitzer, 1998), which is why inorganic nitrogen is commonly referred to as plant available nitrogen (PAN).

There are several pools of nitrogen in the soil (Rasouli, Whalen, & Madramootoo, 2014), but in this thesis I represent nitrogen dynamics using two pools of nitrogen: one pool bound to FSOM

(N-FSOM) and unavailable to plants, and another pool readily available to plants (PAN), which is the sum of NH_4 and NO_3.

There is a wide variation in the amount of PAN and organic nitrogen in soils across the world due to differences in natural processes and human practices, and there are several studies which provide a general baseline condition to which the results of this thesis can later be compared. Local values of PAN have been reported from around 18 – 98 kg ha^{-1} (though this is a small sample and there are likely examples of lower and higher PAN values, locally), at a larger scale the value of PAN can range from 9 – 63 kg ha^{-1}, and at a global scale the reported values include 11 and 51 kg ha^{-1} (Table 6). From the handful of values presented here, it seems that local (and national) PAN values may be above the global average (though this is a small sample).

Table 6. Examples of plant available nitrogen values (kg ha^{-1}) from local, national and international analyses.

Local		National		International	
Value	Description (Source)	Value	Description (Source)	Value	Description (Source)
18 – 98 (PAN)	Spring manure (Zebarth, Kowalenko, & Harding, 2007)	9 – 25 (RSN)	National, average (C. F. Drury, Yang, J.Y., De Jong, R. , 2011)	11 (PAN)	Rice paddy in China (base test) (Yan et al., 2018)
21 – 92 (PAN)	Fraser Valley soil (Kowalenko, 1987)	42 – 63 (RSN)	Pacific Maritime, average (C. F. Drury, Yang, J.Y., De Jong, R. , 2011)	51 (PAN)	International trends (avg.) Stenger, Priesack, and Beese (1998)

3.2.1.3. C/N ratio

The amount of carbon and nitrogen in the soil individually provide information about soil quality, as does the ratio of carbon to nitrogen in the SOM (called the C/N ratio). The C/N ratio is important to consider because it has a direct influence on decomposition of plant residues and also nitrogen cycling (in this thesis, nitrogen loss from the stock of PAN). The generally accepted range for a desirable C/N ratio is 20 – 30, and the ideal C/N ratio for a microbial diet (and therefore soil health, from a biological perspective), is 24. Soils typically have C/N ratios ranging from around 10 – 26 (Batjes, 1996). A higher C/N ratio (e.g., above 30) results in less OM decomposition and slower N cycling, whereas a lower C/N ratio (e.g., below 20) results in the inverse. Also, by influencing PAN loss, the C/N ratio also influences N availability to plants. The C/N ratio is not represented as a stock in the Thesis Model, but the C/N ratio is determined based on the two stocks mentioned above (C-FSOM and N-FSOM). In this thesis, the C/N ratio is calculated based on the ratio of C-

FSOM to N-FSOM; I exclude C-SSOM in this ratio as in the Thesis Model the C/N ratio influences decomposition and nitrogen loss, both of which are related to the 'stable' fraction of the SOM.

3.2.1.4. Endocrine Disrupting Compounds

The SD model developed in this thesis also uses EDCs as a parameter that indicates soil health. EDCs are substances that interact with the hormonal system and is a subcategory of organic contaminants (OC). OC is an umbrella term which describes various substances that come from plants or animals as well as several waste products. Under the umbrella of OC are contaminants of emerging concern (CEC), which are substances or organisms not currently monitored in the environment but can end up there and are believed to result in negative consequences for human and/or environmental health. OC also encompasses pharmaceuticals and personal care products (PPCP), plasticizers, and pesticides. EDCs are OCs as well but are a specific set of compounds that can interfere with hormonal systems; therefore, an EDC will always be an OC, and an EDC could be (but is not necessarily) a CEC, PPCP, plasticizer, and/or pesticide.

Relative to carbon and nitrogen, there has been comparatively little research done on OCs, which is a research field that emerged around the 1950s (Noguera-Oviedo & Aga, 2016). While OC research has been ongoing for decades, it has only been in the past three decades that their potential harmful properties have been widely identified as a priority by policy makers and subsequent precaution over their presence in the environment has been exercised (Noguera-Oviedo & Aga, 2016). As a result, the review studies on OCs tend to cover a shorter time span than the reviews that have been done for carbon and nitrogen. Still, there have been tens of thousands of publications in this area in the past few decades, about half of which relate to EDCs, as demonstrated in Figure 8.

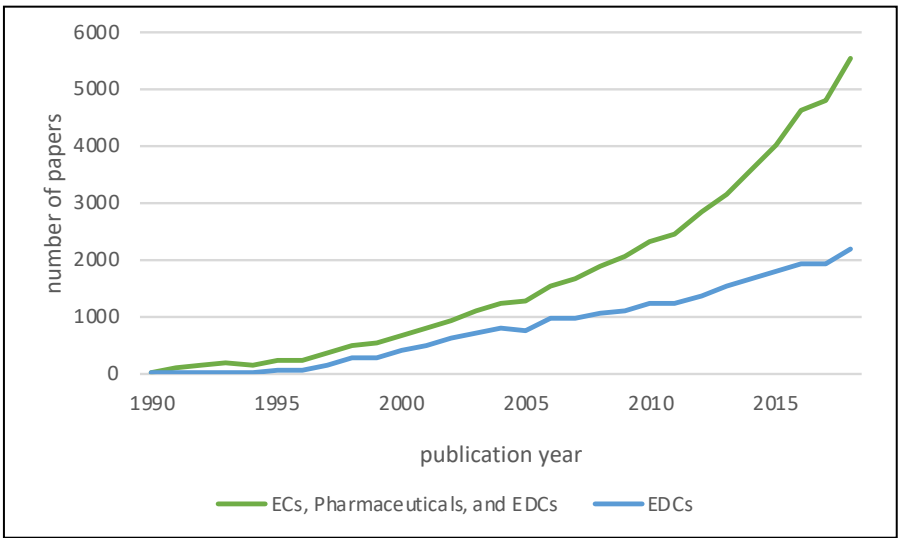

Figure 8. Number of yearly scientific articles across two groups of search phrases – (1) CECs, Pharmaceuticals, and EDCs (green line); versus (2) EDCs only (blue line) – by Topic, in the Web of Science, from 1990 to 2018.*

** Search phrase for (1): ((pharmaceutic* OR drug OR drugs) AND (effluent* OR wastewater* OR waste water* OR river* OR stream* OR sedimen* OR (water NEAR/8 pollut*))) OR ((emergent or emerging) NEAR/5 contamina*) OR endocrin* NEAR/4 disrupt* OR estrogenic NEAR/4 (compound* OR chemical*). Search phrase for (2): (endocrin* NEAR/4 disrupt* OR estrogenic NEAR/4 (compound* OR chemical*)).*

EDCs are compounds which alter the endocrine (hormone) system of an organism, leading to negative effects in the organism, its offspring, or (sub)populations of that organism (Bouchard, 2017). This grouping encompasses compounds such as Bisphenol A, Phthalates, Polybrominated diphenyl ethers, parabens (Monneret, 2017), and others. In contrast to carbon and nitrogen, EDCs are a loosely defined group of vastly different substances, making understanding and modelling their behaviour as a whole a different challenge than for carbon and nitrogen.

The reason for concern over EDCs is that they negatively affect the hormonal system with a potential impact on reproduction, growth, metabolism, and obesity at the individual, population and subpopulation level (Monneret, 2017). However, there is controversy regarding which substances that should be included in the group as it is not sufficient that they interact with the hormonal system, they must also cause an adverse effect. A central question is how to define and measure 'adverse effects', as the relationship between dose and effect is often non-linear and depends on various factors such as life stage and species, and the methodologies of scientific studies testing dose-response relationships range from long-term to single exposures – all of which makes regulating EDCs a complex issue that is still in its early stages.

EDCs are often present in biosolids and can therefore potentially also be present in biochar, depending on the properties of the EDC and the treatment process used to make biochar. Presence

does, however, not equal risk. Knowledge about the potential impact of these substances is sparse and questions have been raised whether they might have the potential to influence soil quality and health, plant composition and growth, and the health of organisms that consume those plants. EDCs are important in the context of biosolids and biochar agricultural land application because the uncertainties surrounding their potential harm is a major reason for public concerns about land application.

There are thousands of identified EDCs, so in this thesis, as a sample of EDCs to examine, I selected Triclosan (TCS) and Triclocarban (TCC). These two EDCs are chlorinated antibacterial compounds found in several household cleaning supplies, personal care products such as toothpastes, bar soaps and body washes as well as clothing, toys, building materials and paints (Halden et al., 2017; Lozano, Rice, Ramirez, & Torrents, 2010). The reason I selected these two EDCs is because they are present in relatively high concentrations in municipal biosolids (e.g., median values of TCS and TCC were 6,085 and 1,930 ng/g TS dw [in Canada], and 7,615 and 21,050 ng/g TS dw [in the US], while seven other common EDCs are present in anywhere from 12.5 to 5,910 ng/g TS dw [in both Canada and the US]) (La Guardia et al., 2003; Monteith, Dong, Parker, & Sterne, 2010), and they have been assigned to the *'High Risk'* category of compounds by the Provincial government (B.C. Ministry of Environment, 2016b). These two EDCs pose a potential hazard, have low solubility in water (especially TCC) and are non-volatile (Henderson, 2013).

One study of a probabilistic modelling terrestrial ecological risk evaluation for TCS in land-applied biosolids found that TCS poses only a short-lived high risk to soil microbes in the mixed zone in the worst case exposure scenario and only under certain soil conditions, and a moderate risk to soil microbes in the top 30 cm, plants, and small rodents; this relatively low overall risk is primarily due to the rapid degradation of TCS (Fuchsman et al., 2010). Another study consisting of a physical experiment of TCS exposure to various organisms found that TCC poses a high risk to a rodent and a shorebird in the worst case scenario and a moderate risk to larger predators, a rodent, a shorebird and earthworms in all other scenarios (Snyder, O'Connor, & McAvoy, 2011). It has been documented that these substances are partially degraded in biosolids and that the degradation products also may be bioactive (Lozano, Rice, Ramirez, & Torrents, 2013; Wu, Spongberg, & Witter, 2009). Due to the lack of data and the difficulty in including the degradation products, I

have in this study assumed that none of these degradation products are formed, and that when these substances are degraded in the soil, they are fully mineralized to carbon dioxide and water.

In a 33-yearlong study of annual biosolids application (Xia et al., 2010) the average background amount of TCS was found to be 39 micrograms/ha (3.9 x 10^{-8} kg TCS ha^{-1}), and TCC was found to be 811 micrograms/ha (8.11 x 10^{-8} kg TCC ha^{-1}), in the upper 30 cm of soil. I assume these are the background (initial) amounts of TCS and TCC in the soil in the Thesis Model.

3.2.2. Causal Loop Diagram

In the SD model developed in this thesis, each indicator is described in a separate 'area', or sub-system. A high-level illustration of the interactions between the nitrogen, carbon, EDC, crop, and biosolids and biochar amendment sub-systems is shown in the Causal Loop Diagram (CLD, Figure 9). The polygon overlaps illustrate where areas cross over from one sub-system to another. For instance, the carbon and nitrogen sub-systems are linked via the C/N ratio, and the crop sub-system crosses over into the carbon, nitrogen, and EDC sub-systems.

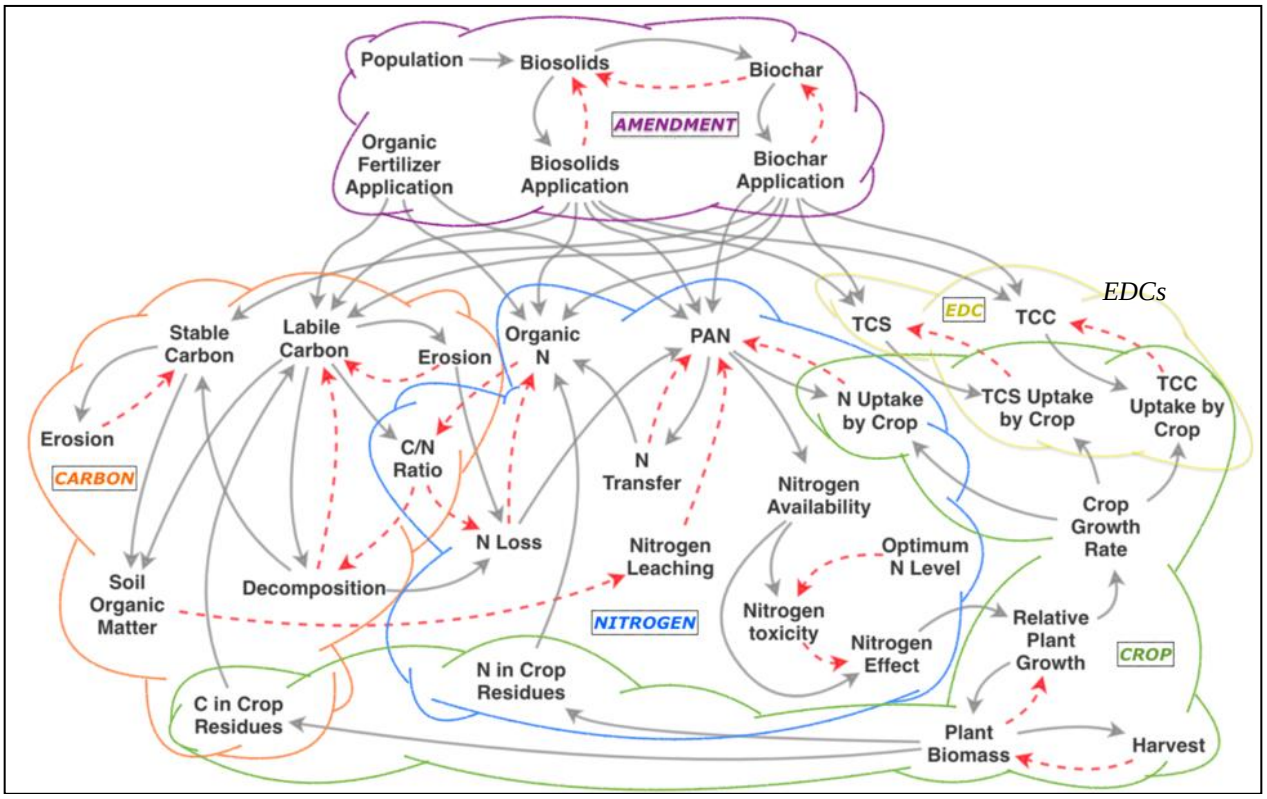

Figure 9. Causal loop diagram of soil amendments, nitrogen, carbon, crop, and EDCs in a biosolids and biochar agricultural land application context.

Solid grey and red dashed arrows represent the positive and negative influence of a variable on the other, respectively. With a positive link from variable A to B, a change in the value of A results in a *same-direction* change in the value of B (for instance, an *increase* in income results in an *increase* in wealth). With a negative link from variable A to B, there is a converse change in the value of B (for instance, a *decrease* in the price of a commodity/resource results in an *increase* in its purchase/consumption). Two types of feedback loops (FLs) are illustrated in the CLD; reinforcing (R) loops, representing positive feedback, and balancing (B) loops driving counteractive feedback.

Focusing in on the FLs that influence the key system elements – C-FSOM, C-SSOM, N-FSOM, PAN and relative plant biomass (RPB) – paints a more complete picture of the feedbacks that drive the system's behaviour. While there are many FLs in this system, there is not enough room here to delve into each one. Instead, the purpose of the following paragraphs is to highlight some examples of core behaviour and feedbacks within different parts of the system.

On the whole, there is a high degree of complexity in this system, and this is demonstrated by the many feedbacks between carbon, nitrogen, and crop components. There are 37 FLs in the CLD of the Thesis Model that contain more than two elements. Here, I only present FLs containing more than two model elements because FLs containing two elements are unlikely to drive system behaviour in the same way larger FLs can.

The FLs illustrate that there are some key elements in the system such as relative nitrogen availability and the C/N ratio. The area of the CLD directly after the relative nitrogen availability shows that there is an optimal level of nitrogen in the soil relative to the amount the crop can take up, and below that threshold there is a positive influence of more nitrogen on plant growth but below that threshold there is a negative influence. The C/N ratio is influential because it ties together elements from several different areas of the system and is involved in the majority of FLs in the system.

The C/N ratio results in there being many pairs of nearly identical balancing and reinforcing FLs – one of the FLs follows the direct link from relative nitrogen availability to the nitrogen effect, and one of the FLs follows the 'nitrogen toxicity' pathway from relative nitrogen availability to the nitrogen effect. (Note: 'nitrogen toxicity' is an element of the CLD but does not exist as an explicit element in the Thesis Model, as this effect is embodied by the graphical function that defines the nitrogen effect in the Thesis Model – below a certain threshold nitrogen has a positive

effect on plant growth, but above that threshold nitrogen has a negative effect, precisely at the threshold plant growth is greatest). The result is a pair of FLs, one of which is balancing and one of which is reinforcing, and this pair of FLs means the system is constantly working towards the optimum nitrogen level that optimizes plant growth. These FLs termed 'balancing and reinforcing' (BR) loops are presented here in one figure that shows both pathways. While each FL has a balancing and a reinforcing pathway, in each case I identify under which condition (nitrogen toxicity or no nitrogen toxicity) the FL is balancing, and under which condition is it reinforcing. In the following figures, thick lines/arrows highlight the linkages that form the particular loop being described.

Balancing and reinforcing loop (BR1) shows that labile carbon and PAN are both either balanced or reinforced via crop growth, which informs the amount of carbon in the crops and thus decomposition and nitrogen loss from organic nitrogen to PAN (Figure 10). With no nitrogen toxicity this loop is balancing, and with toxicity it is reinforcing.

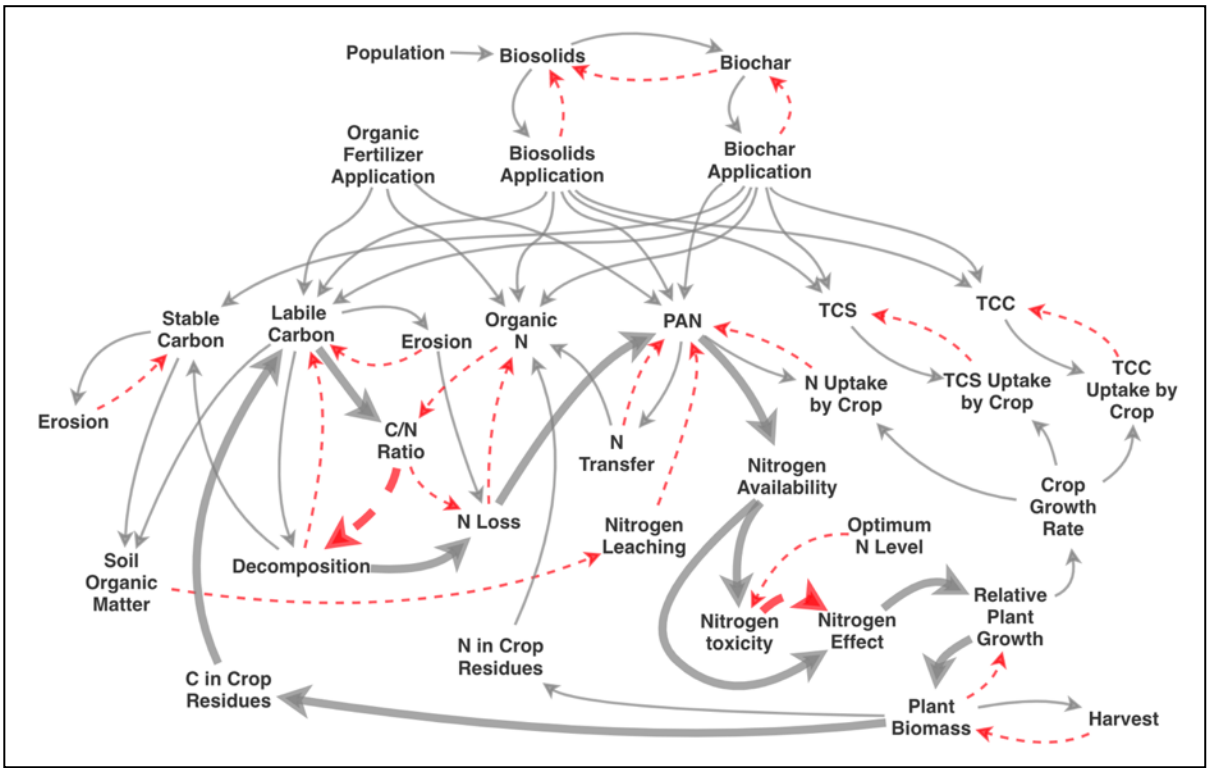

Figure 10. Balancing and reinforcing loop 1 (BR1).

Balancing and reinforcing loop (BR2) shows that stable carbon is also involved in a FL similar to the previous (BR1) one, and the amount of stable carbon has a direct influence on the amount of PAN via reduced nitrogen leaching (Figure 11). With no nitrogen toxicity this loop is balancing, and with toxicity it is reinforcing.

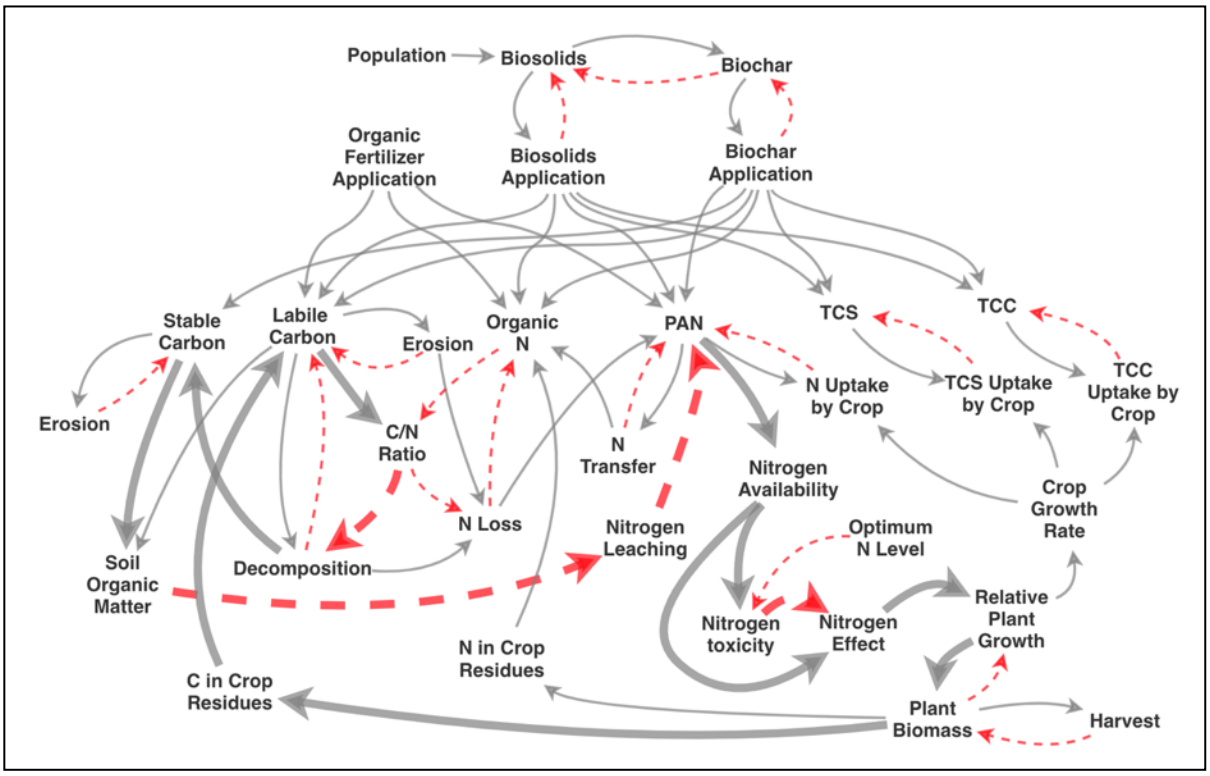

Figure 11. Balancing and reinforcing loop 2 (BR2).

Balancing and reinforcing loop (BR3) shows that the effects from the previous FL (BR2) are counteracted by labile carbon directly influencing decomposition. With no nitrogen toxicity, this loop is reinforcing, and with toxicity it is balancing – this is the opposite from loop (BR2).

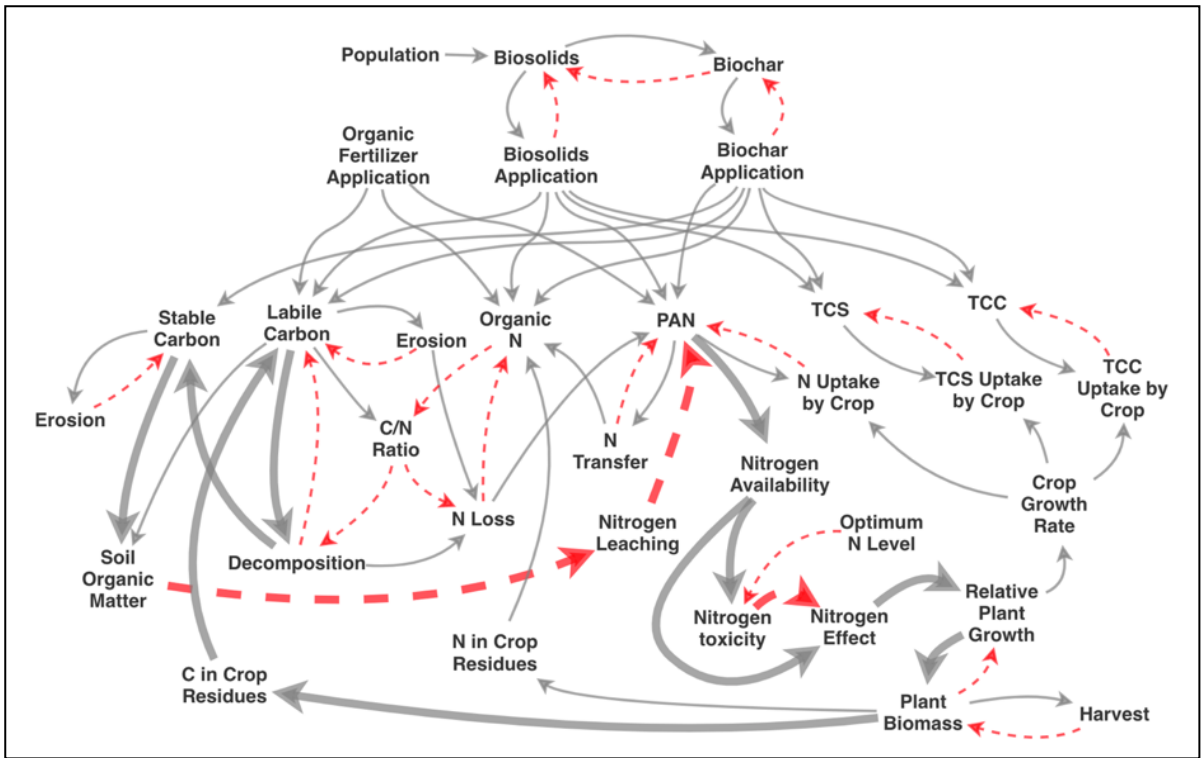

Figure 12. Balancing and reinforcing loop 3 (BR3).

Balancing and reinforcing loop (BR4) shows that PAN and organic nitrogen are both either balanced or reinforced by PAN via its influence on crop growth, which informs the amount of organic nitrogen and thus decomposition which directly informs nitrogen loss from organic nitrogen to PAN. With no nitrogen toxicity, this loop is reinforcing and with toxicity it is balancing.

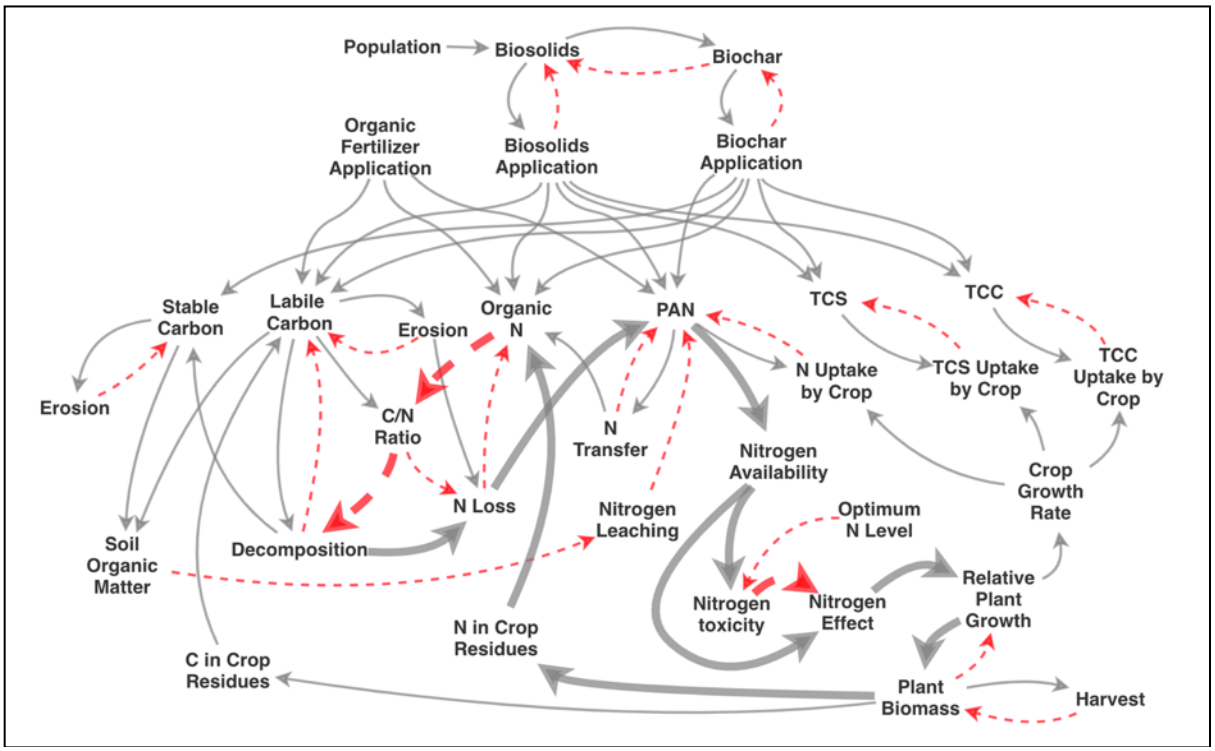

Figure 13. Balancing and reinforcing loop 4 (BR4).

Balancing and reinforcing loop (BR5) shows that PAN is either balanced or reinforced due to its influence on crop growth, which increases nitrogen uptake by the crop. With no nitrogen toxicity, this loop is balancing and with toxicity it is reinforcing.

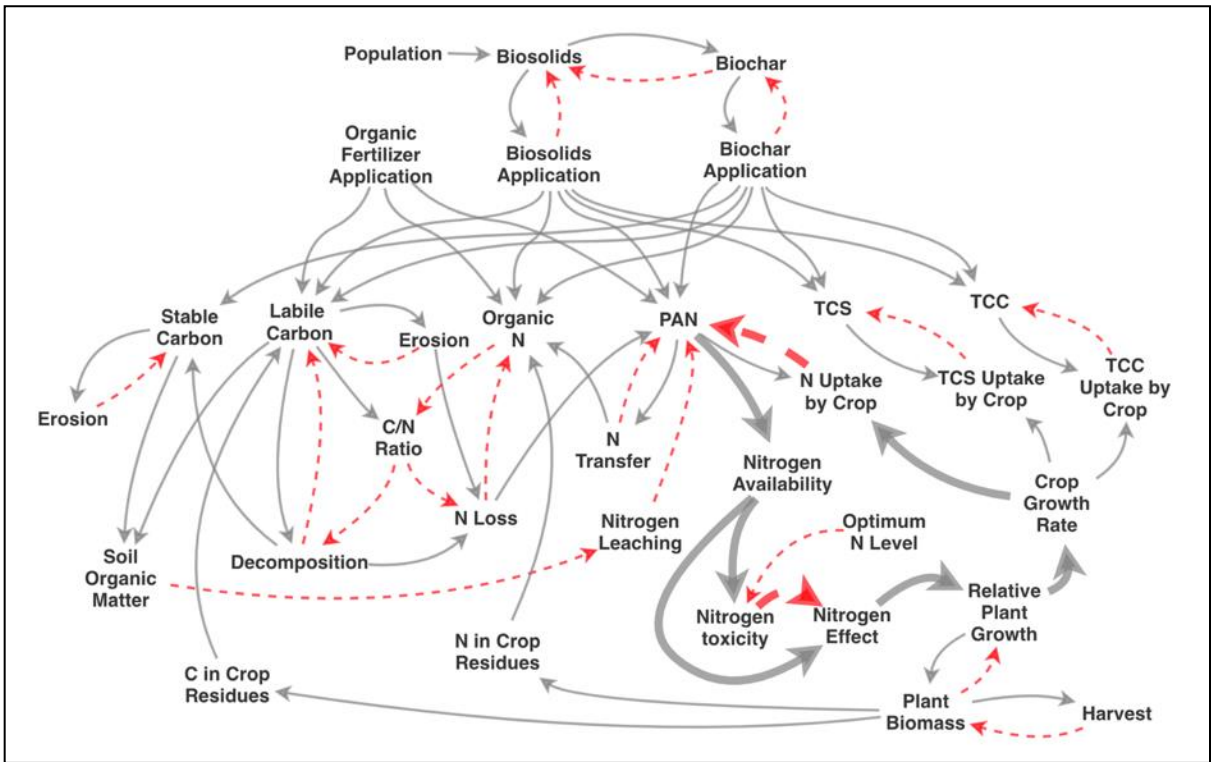

Figure 14. Balancing and reinforcing loop 5 (BR5).

Reinforcing loop (R1) shows that PAN is reinforced as it decreases nitrogen leaching indirectly by increasing soil organic matter. This FL also shows that organic nitrogen is reinforced as it increases nitrogen transfer from PAN indirectly by decreasing leaching. However, this reinforcement of organic nitrogen is counteracted by balancing loop (B1) which shows that organic nitrogen is reduced as it increases nitrogen loss indirectly by increasing decomposition. Further, this FL shows that stable carbon is reinforced as it increases decomposition by indirectly decreasing the C/N ratio. However, this reinforcement of stable carbon is counteracted – by (BR2) when there is no nitrogen toxicity and by (BR3) when there is toxicity.

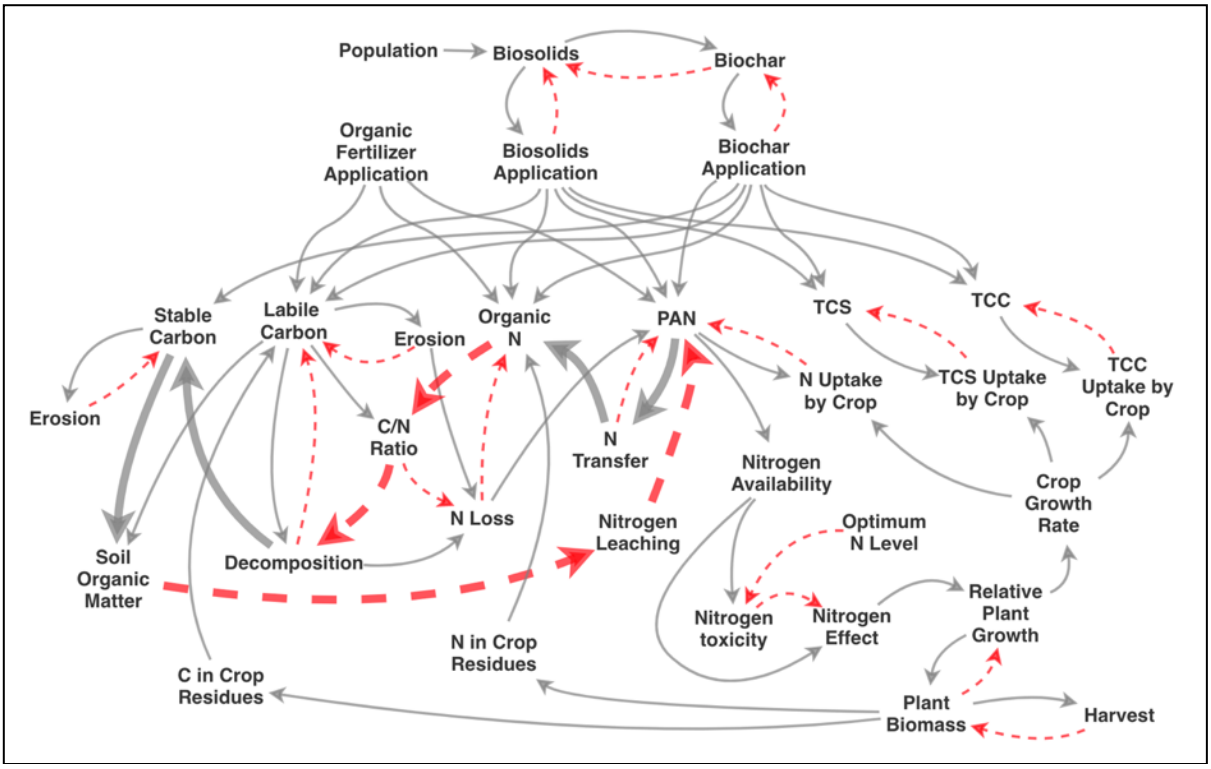

Figure 15. Reinforcing loop 1 (R1).

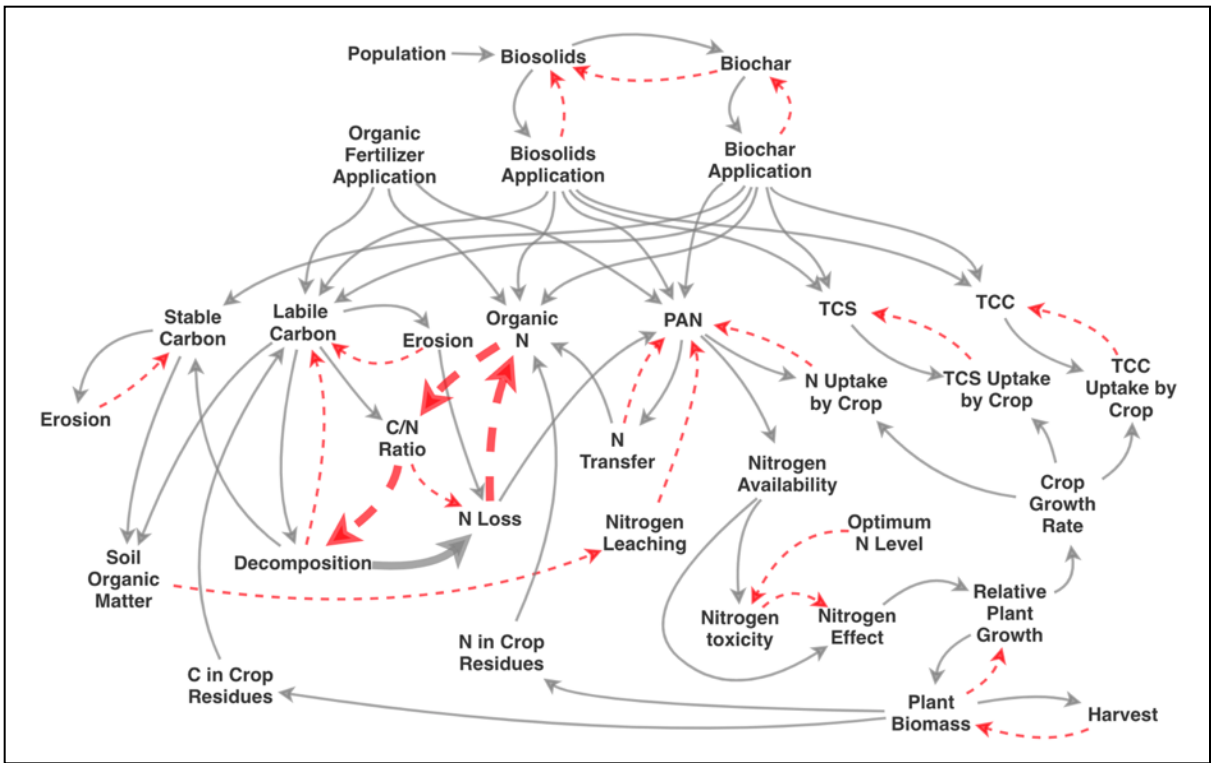

Figure 16. Balancing loop 1 (B1).

While this is not a comprehensive list of all FLs in the system, these FLs illustrate some of the key balancing and reinforcing mechanisms working with one another to generate system behaviour. In the following section, this CLD is translated into a complete SD model for simulation of long-term dynamics under the scenarios being tested in this thesis.

3.2.3. System Dynamics Model Elements

System dynamics models are built by a limited set of core elements: stocks, flows, converters, connectors, and clouds. Figure 17 illustrates a basic, low-resolution model of a reservoir of water being used for human use and being replenished by precipitation expressed as a stock and flow diagram with the four core elements of a system dynamics model. Stocks represent an accumulations of model elements (physical or otherwise) or the current state of the system and are generally modeled as a quantity of something – represented as water in the reservoir in Figure 17. Flows represent the inputs and outputs to and from stocks over time, thus capturing system activity and rates of change in a system – in Figure 17 shown as removal for human use and precipitation. Clouds represent system boundaries and are assumed to be infinite sources or sinks that attach from or to stocks via flows, and whether a cloud is a source or sink depends on the direction of flow relative to the cloud. Converters are the typically more isolated model elements that inform

other model components or modify flows and other converters – converters are time of year, water use per capita, and population shown in Figure 17. Connectors are used to link convertors, stocks and flows in an SD model to other model elements – these are the arrows in the model. Connectors define the basis for model interrelations and allow for calculations to be made based on those connections. For instance, inflows and outflows can be influenced by the stock itself (for example, if less water in the reservoir results in less removal for human use), or other variables such as the time of year (in the case of precipitation) or the population density of an urban area (in the case of water removal for human use). In the Thesis Model, the core stocks and flows pertain to soil quality indicators, described in the following sections.

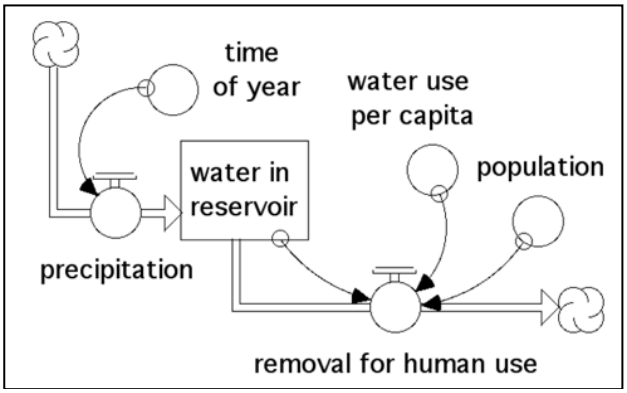

Figure 17. A simple system dynamics model around a reservoir, influenced by precipitation and human use.

3.2.4. Description of the Bossel Model and Assumptions

The structural foundation for the system dynamics model (Thesis Model) used in this study is to a large extent based on the *Z311 Nutrient dynamics* model developed by Bossel (2007). The Z311 Nutrient dynamics model targets the complex dynamics of a soil system, as well as trade-offs between organic and conventional agriculture. The 'Bossel Model' represents the primary processes of the soil nitrogen system and its interactions with the processes of plant growth and organic matter decomposition. The Bossel Model provides a fundamental framework for modelling the slow and fast dynamics of carbon and nitrogen, and how these soil constituents relate to and inform plant growth, as well as identifying feedbacks between carbon, nitrogen, and plant biomass. The Bossel Model can be used to illustrate how time scales of processes within the soil vary; e.g. certain components like nitrogen have faster dynamics, while other soil components like carbon have slower dynamics. The Bossel Model can also be used to illustrate that while more nitrogen can be required to achieve high yields, this can come with the trade-off of potential

fertilizer and harvest losses, and environmental damage if timing of fertilization is not in harmony with plant growth. The Bossel Model is specifically geared to the coupled nature of carbon and nitrogen stocks and their transformations, and crop growth.

The Bossel Model does not, however, contain the specificity required to model the case study of the CRD biosolids/biochar agricultural land application. For example, most of the sources that Bossel used are in German, so one can assume they are based on the German context, and all references were published between 1978 and 1988. Bossel notes that his model is based on a limited amount of data, drawing upon just six references: *Standard values for agriculture and horticulture* (Ruhrstickstoff, 1983), *Outline of plant nutrition* (Finck, 1982), *Ecology of plants* (Larcher, 1980), and *Textbook of soil science* (Scheffer/Schachtsabel, 1979); as well as two English textbooks on soils and plants: *Russell's Soil Condition and Plant Growth* (Wild, 1988) and *Soils and* Fertility (Thompson & Troeh, 1978). The majority of these references are textbooks, and the Bossel Model is thus rooted in assumptions that were accepted at the time when the model was first developed (near the end of the 1990s). Because Bossel does not clearly identify which of his model parameters are based on which data sources, I assumed there may be a number of parameters in the Bossel Model that were estimated based on my knowledge or knowledge passed on by experts in the field, and that these parameters may therefore also be based on the German context.

The five key stocks in the Bossel Model are: carbon in readily decomposable OM, carbon in stable OM, organic nitrogen, plant available nitrogen (PAN), and relative plant biomass. There are many relations between these stocks. For example, the crop biomass informs the amount of carbon and nitrogen in crop residues, which in turn inform stocks of slow carbon and organic nitrogen, respectively. An outflow for one stock (loss of PAN due to transfer to N-FSOM) informs an inflow of another (gain of N-FSOM by transfer from PAN). The C/N ratio informs several different parameters (e.g., decomposition of fast carbon and organic nitrogen, nitrogen transfer from PAN to organic nitrogen, and erosion of organic nitrogen).

3.2.5. System Dynamics Model Development and Parameterization

I used Stella® Version 10.1.2 (isee systems, 2019) in macOS High Sierra 10.13.6 to code a system dynamics model representing the dynamics of carbon, nitrogen, and two EDCs in a CRD agricultural soil influenced by annual application of organic and inorganic soil amendments.

After developing an understanding of the parameters of interest for this thesis, I created the Thesis Model based specifically on the CRD case study to support the research objectives. In constructing the model, I updated as many of the Bossel Model parameters as possible with case-specific and contemporary data sources. The CLD presented in section 3.2.2 provides a visual illustration of the relationships in the Thesis Model, while Appendix A provides the technical definition of the relations between Thesis Model elements, and Appendix B provides a visual illustration of the complete stock and flow diagram of the Thesis Model. The following section describes the key elements of the systems dynamics Thesis Model. A fully detailed description of the Thesis Model is available in Appendix A including the full names, units, equations or values, descriptions, and references (where applicable) for each model parameter.

In the Thesis Model, the amount (stock) of C-FSOM in the soil depends on the inputs (biosolids, biochar, crop residues, and organic fertilizers), and the outputs (OM decomposition and erosion) (Figure 47). In the Thesis Model, the amount (stock) of C-SSOM in the soil depends on the inputs (biochar and OM decomposition), and the output (losses, based on the relative erosion loss). This assumes that C-SSOM does not undergo decomposition (is only lost through erosion), and that biosolids only contain C-FSOM (not C-SSOM). The first assumption is based on the Bossel Model and the second is based on both the Bossel Model and other research. The Bossel Model includes application of compost and manure, but those applications only inform C-FSOM, not C-SSOM. Moreover, research has shown much of the carbon in biochar is converted into fixed carbon (i.e., the non-volatile matter in coal minus the ash) through the process of pyrolysis, and biochar has been found to be resistant to decomposition (Ippolito et al., 2012) and can act like SSOM in the soil (Lipczynska-Kochany, 2018). I therefore assume biochar to contribute to the stable carbon formed through the process of decomposition, and thus provide input for C-SSOM.

As in the Bossel Model, the fraction of OM in the soil is calculated based on the total amount of carbon in both C-FSOM and C-SSOM. In the Thesis Model, as in the Bossel Model, the relative erosion loss influences both stocks of carbon, and the amount of carbon in the different amendments is calculated based on the amount of amendment applied, the organic dry matter (ODM) fraction of the amendment, and the amount of carbon per OM.

I constructed the erosion and C/N sectors in the Thesis Model identical to the Bossel Model (Figure 45 and Figure 46). As in the Bossel Model, I modeled the relative erosion loss based on the depth

of the plow layer, soil loss by erosion and soil specific weight. In the depiction of erosion in these models, erosion includes all surface soil erosion (wind and water). As in the Bossel Model, I model the C/N ratio in the soil as the ratio of C-FSOM to N-FSOM, so the C/N ratio does not account for PAN or C-SSOM.

As in the Bossel Model, OM decomposition in the Thesis Model is driven by the C/N ratio (Figure 48); as the C/N ratio decreases towards 20, the decomposition factor increases towards 1, and below 20, it remains at 1. The Bossel Model assumes that only 25% of C-FSOM is transferred as input to C-SSOM each year and the remainder is lost from the system (into the air as carbon dioxide); I take on this assumption for the Thesis Model. As in the Bossel Model, OM decomposition is also impacted by the amount of C-FSOM and seasonal time (because incoming solar radiation changes with the season, influencing soil temperature), which is based on the yearly time; essentially, seasonality is mimicked by a sine function and that function informs the amount of decomposition. Around the beginning of the year, OM decomposition is at its lowest, and it peaks just before the middle of the year. As in the Bossel Model, I built this fluctuation in to the function for the flow of OM decomposition in the Thesis Model.

Based on the Bossel Model, with additions of biosolids and biochar to make the Thesis Model relevant for this context, the amount (stock) of PAN in the soil depends on the inputs (biosolids, biochar, mineral fertilizer, the atmosphere, and OM decomposition), and the outputs (leaching, plant uptake and transfer to OM) (Figure 49). I calculated PAN in the biochar and biosolids based on the amount of amendment applied and the PAN fraction of the amendment. Based on the Bossel Model and the additions of biosolids and biochar to make the model relevant for my thesis, the amount (stock) of N-FSOM in the soil depends on the inputs (biosolids, biochar, crop residues, organic fertilizer, and nitrogen input by transfer), and the outputs (OM decomposition and erosion). As in the Bossel Model, the C/N ratio in the Thesis Model influenced the amount of nitrogen loss by erosion and decomposition, as well as nitrogen input by transfer.

As in the Bossel Model, relative nitrogen availability in the Thesis Model is a function of the amount of PAN in the soil and the maximum crop yield as well as the crop's specific nitrogen uptake. The amount of nitrogen in crop residues is a function of the relative plant biomass and the maximum amount of nitrogen in those crop residues. This assumes the crop's nitrogen content reaches its maximum.

Likewise, the amount of leaching (erosion) is dependent on the fraction of OM in the soil, with leaching decreasing as the fraction of OM increased. The nitrogen transfer function is based on the C/N ratio in the soil (Figure 50); as the C/N ratio decreased towards 20, the nitrogen transfer function decreased, and below 20 it remained at 0. Nitrogen leaching is influenced by the fraction of organic matter in the soil (C-FSOM and C-SSOM combined) (Figure 51). With 0% OM in the soil, all nitrogen is leached, but this declines rapidly to 20% nitrogen leaching at 10% OM and to 10% nitrogen leaching at 100% OM.

Similarly, the amount of RPB – a measure of the amount of plant biomass relative to the maximum amount of crop residues for a specific crop – depends on the input (crop growth), and the output (harvest) (Figure 52). The nitrogen effect, which influences crop growth, is a function of relative nitrogen availability (RNA) in the soil (Figure 53). Crop growth occurs from early April to early September, at which time the crop is harvested (this differs from the Bossel Model, which uses a shorter growing season more in line with the German growing season because Bossel is from Germany). Nitrogen uptake by plants is based on the growth rate and the maximum crop yield, while the yield of total biomass is based on RPB and the maximum crop yield per area for that particular crop. At the time of harvest, the amount of total biomass is used to determine how much crop residues remain in the field after harvest. The crop harvest is transferred into the stock of relative crop yield (RCY) which has no outflows and is thus an accumulation of all the years' crop yield from RPB.

Additionally, relative erosion loss is calculated based on the depth of the plow layer, soil loss by erosion and soil specific weight. This measure of erosion accounted for all surface soil erosion (wind and water). Note that relative erosion loss is a unitless parameter and is thus assumed to be constant. Erosion from certain stocks (e.g., C-FSOM, N-FSOM, and C-SSOM), on the other hand, are flows that have units (e.g., kg ha^{-1} $year^{-1}$) and are based on the value of the stock itself as well as the relative erosion loss.

In the Thesis Model, the amount (stock) of TCS in the soil depends on the inputs (biosolids and biochar), and the outputs (crop uptake, photodegradation and microbial degradation) (Figure 54). The stock of TCC in the soil depends on the same, except there is no photodegradation (Figure 55). Annual microbial and photo-degradation rates were based on the half-lives of these two EDCs

for each type of degradation, and plant uptake rates were based on findings of final concentrations of these EDCs in plant tissue in a study of TCC and TCS plant uptake (Henderson, 2013).

Similarly, the size of the population (stock) depends on the input of growth, which is based on the region's overall population projection which incorporates birth, deaths, immigration and emigration. As the outputs (death, emigration) are incorporated into the growth rate, the Thesis Model not include any outputs from the stock of population (Figure 56).

Likewise, the stock of biosolids storage depends on the amount of biosolids produced, and the outputs (removal for land application, biochar production, and excess biosolids distribution) (Figure 56). The production of biosolids is based on the population size and per capita biosolids production, with the limitation being the treatment capacity of the RTF, and excess biosolids is distributed from storage. The stock of biochar storage depends on biochar production, and the outputs (removal for land application and excess biochar distribution) (Figure 56). Biochar production is based on the amount of biosolids and the yield of biochar per unit of biosolids, and excess biochar is distributed from storage.

Additionally, removal of biosolids and biochar from their respective stocks for application were based on the demand to satisfy the application rate (which is based on the agronomic application rate and the agricultural land area), and excess biosolids distribution occurred when the amendment storage capacity is met. The flows of biosolids and biochar production, as well as biosolids land application are all able to be turned 'on' or 'off' by using the switches (Figure 56 – pink model elements). The amount of py-oil, py-gas, and biogas produced depend on biochar production and biosolids production.

Finally, the model input parameters sector contains several constant converters, conversion factors, and crop-specific parameters (Figure 43), while the switches sector allows the model user to easily change between the Subscenarios (Figure 44). Each of the parameters in the switches sector is meant to be set to either 0 or 1, depending on the Subscenario, and I describe these model settings in section 3.3.

3.2.6. Model Validation

Given the lack of empirical data I relied upon a relativist/holistic viewpoint for model development and validation, i.e. a model is one of many possible ways of representing a real-world situation

(Barlas, 1996). Without empirical data the validation of the model cannot be made entirely objective, formal, or quantitative and relies upon ensuring the structure of the reference model closely approximates real world behaviour and that model behaviour does not violate any governing system principles (Barlas, 1996). Structural validity is then about testing to what extent the two systems (the model and the real system) are indistinguishable (Aumann, 2007).

Thus, to validate the Thesis Model I examined the model for its dimensional consistency, sensitivity extremes, and structural appropriateness. To test the model structure, I performed a dimensional consistency test, ensuring all the units and scale of all equations matches with the parameters they are used to calculate. To test the model behaviour, I performed a test of sensitivity to extremes (Barlas, 1996; Mirchi et al., 2012).

First, as a test of dimensional consistency, I used Stella to check that the units of all parameters were consistent with one another. Dimensional consistency is the same as having the units on both sides of an equation match up. An example of dimensionally consistent parameters in the Thesis Model is C_in_compost (kg ha^{-1}), which is calculated as the product of compost_applied (kg ha^{-1}), and C_per_OM (a unitless ratio) and ODM_fraction_in_compost (a unitless ratio). An example of dimensionally inconsistent parameters (which Stella would not allow) would be if in the preceding example C_per_OM was in units of kg; this would cause the units of C_in_compost to not be consistent with the units of the parameters that algebraically inform it. Stella contains a tool called 'Check Units' which can be utilized before attempting to run the model, and this by using this tool I was able to determine that the units throughout the model are dimensionally consistent.

Second, a test of sensitivity to extremes I set several of values to 0% of the intended value, and allowed the model to run, comparing the results with expected system behavior under these extreme conditions. For instance, Figure 18 illustrates the dynamics of biosolids-related flows when the biosolids agronomic application rate is set to 0: the result is that all of the biosolids-related flows are also 0. As another example, Figure 19 illustrates that when the relative nitrogen availability is set to 0, no plant growth occurs. Further, Figure 20 illustrates that when nitrogen loss by leaching and nitrogen transfer to OM are both set to 0, PAN grows consistently for the entire model period. Each of these sensitivity to extremes tests demonstrated that the model behaves as expected under extreme conditions.

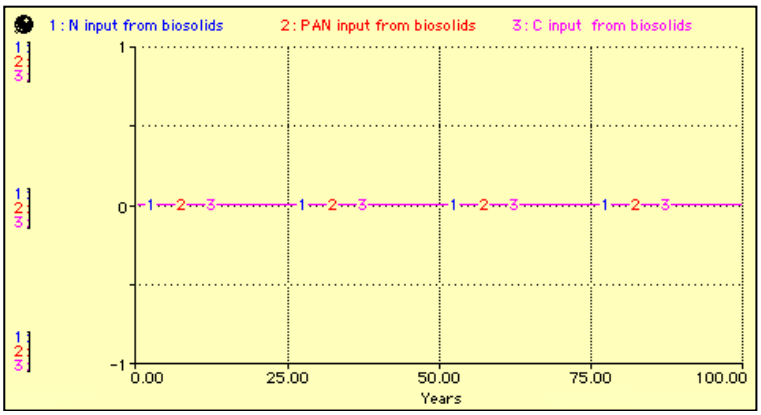

Figure 18. Model output when biosolids application rate is set to 0.

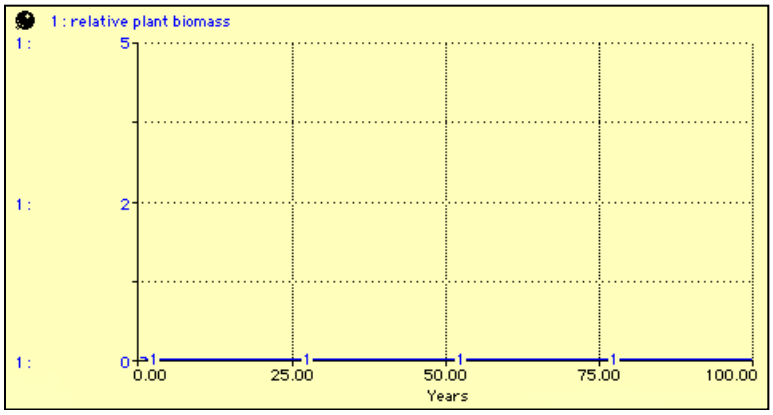

Figure 19. Model output when relative nitrogen availability is set to 0.

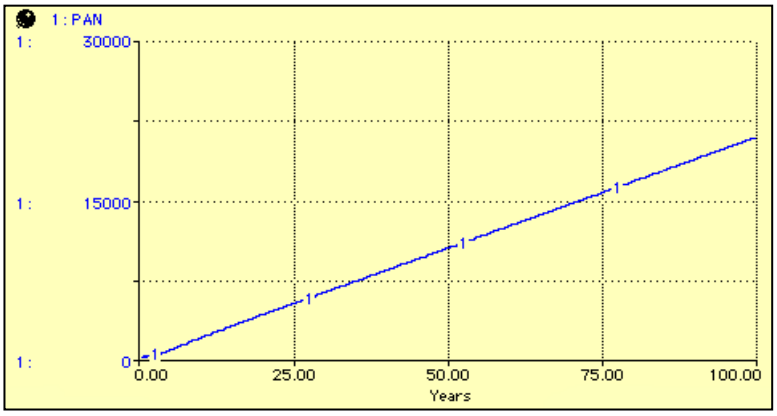

Figure 20. Model output when nitrogen leaching and nitrogen transfer are set to 0.

3.3. Modeling the Case Study

3.3.1. Scenarios and Subscenarios

To apply the model to the case study site I prepared the three scenarios illustrated in Table 7. The parameters in the left column are always multiplied with the parameters they inform in the model, so when any of these is set to 0 the parameter it informs will also be 0, but when set to 1 it does not alter the parameter value.

Table 7. Model scenarios: A) baseline (compost and manure applied); B) biosolids; C) biochar.

	Scenario		
Parameter	**A**	**B**	**C**
Compost_and_manure_on_or_off	1	0	0
Biosolids_production_on_or_off	0	1	1
Biosolids_land_application_on_or_off	0	1	0
Biochar_land_application_on_or_off	0	0	1

Scenario A, the baseline scenario, assumes *compost and manure* are applied but biosolids and biochar are not applied. Scenario B, the biosolids scenario, assumes *only biosolids* are applied and neither biochar, compost, nor manure are applied. Scenario C, the biochar scenario, assumes *only biochar* is applied and neither biosolids, compost, nor manure are applied. For the biosolids and biochar scenarios, there are three sub-scenarios tested, all of which are described in the following section. I assume that in all scenarios no mineral fertilizer is applied.

3.3.2. Application Rates

For the Biosolids and Biochar Scenarios, I applied the model across three different Subscenarios defined by different application rates, labelled according to the Scenario (either B or C) and the Application Rate (either 1, 2, or 3; Table 8). The Low (1), Medium (2) and High (3) rates for the biosolids and biochar scenarios are based on industry practice (for biosolids) and academic literature (for biochar) and were chosen given the lack of available data for the CRD study area. The biosolids application rates fall within the range of what is used by Metro Vancouver, whose biosolids are applied at around 17,000 kg ha^{-1} (Kempe, 2019). The biochar application rates, while not based on a specific local example, fall within the range of what has been proven useful in other contexts, with rates in the range of 4,000 – 16,000 kg ha^{-1} having positive (or at least no negative) effects on crop productivity (Major, 2010).

Table 8. Application rates (kg ha-1) in the three sub-scenarios for biosolids and biochar.

	Scenario	
Application Rate	**Biosolids (B)**	**Biochar (C)**
Low (1)	10,000 (B1)	5,000 (C1)
Medium (2)	15,000 (B2)	10,000 (C2)
High (3)	20,000 (B3)	15,000 (C3)

Note that the absolute increases from Rate 1 to 2 and Rate 2 to 3 are the same for scenarios B and C, but the relative increases from Rate 1 are different for Rates 2 and 3. Sub-scenarios B2 and B3 use **1.5***[Rate 1] and **2***[Rate 1], while sub-scenarios C2/D2 and C3/D3 use **2***[Rate 1] and **3***[Rate 1], respectively. Because the absolute increase from Rate 1 to 2 and Rate 2 to 3 is the same within the biosolids and biochar scenarios, respectively, one can determine which scenario is most 'effective' at influencing the stocks of interest in the desired direction. For example, if the difference in a stock value from Low to Medium was larger than Medium to High it would suggest the increase in application rate was disproportionally ineffective at increasing that stock value, whereas if the inverse was true it would suggest the increased application rate was disproportionally effective at increasing that stock value.

These application rates are based on the literature and expert approximation due to the lack of available data specifically for the CRD. As a result, this led to a disconnect between available land for crop growth and the application given rates in this study. If the CRD were to produce and land apply biosolids and biochar at the rates in this study, it is worth considering how much of these amendments could be applied on CRD cropland. Table 9 presents the percentage of CRD agricultural land used for crop growth that would be required in order to satisfy these purposes.

Table 9. Percentage of CRD land area required to accommodate the amount of amendment applied under each application rate. Based on initial population size, the amount of biosolids produced per capita, biochar per unit of biosolids, and total agricultural land area used for crops and tame hay.

	Scenario	
Application Rate	**Biosolids**	**Biochar**
Low	645%	810%
Medium	970%	1610%
High	1290%	2420%

This study suggests that even if all of the arable land used for crops in the CRD was to be utilized for the purposes of the land application scenarios in this study, it would not be possible for the CRD to land-apply all of its biosolids or biochar on this type of land in its jurisdiction. This

suggests that while land application of biosolids or biochar may be considered, there needs to be a way to use the majority of these materials that could not be accommodated by CRD crop- and tame hay-growing land alone. Using these amendments on this type of land would require an additional 5.5 to 23 times more of this land. For reference, all of Vancouver Island has about 3 times more of this type of land than the CRD, the Lower Mainland (Southwest) has almost 11 times more, the Cariboo has around 16 times more, and all of BC has around 133 times more; therefore, although the CRD may not have the capacity to utilize their biosolids or biochar in this way within their jurisdiction, other local jurisdictions and the province as a whole have enough of this type of land to accommodate CRD biosolids or biochar.

3.3.3. Crop-specific Parameters

For all scenarios, I assumed that a two-year, corn-alfalfa crop rotation scheme is followed. I assumed this rotation scheme because it is more realistic for a farmer to rotate through crops than to grow the same crop year after year, and there have been known benefits of the corn-alfalfa crop rotation scheme for at least nearly four decades (Miller, 1983). Moreover, corn and tame hay (e.g., alfalfa) comprise the majority of crop biomass grown on agricultural land in the CRD (B.C. Ministry of Agriculture, 2016).

I assumed the crop rotation starts with corn, followed by alfalfa, and then the rotation repeats starting with corn again. I chose corn because 22% of crop production in the CRD from 2010 to 2014 was corn for fodder (B.C. Ministry of Agriculture, 2016). I chose alfalfa because tame hay (i.e., leguminous fodder, for which alfalfa has been used for millennia (Bolton, 1962)) made up over two-thirds of CRD crop production from 2010 to 2014 (B.C. Ministry of Agriculture, 2016).

Different crops have different characteristics which identify them, several of which pertain to the maximum possible amount of crop growth, crop biomass composition, and nitrogen uptake and fixation. Table 10 illustrates the crop-specific parameters for corn and alfalfa. In the Thesis Model, every year at the beginning of the crop growth period these parameters change between the values in the columns labeled ‘Corn’ and ‘Alfalfa’. A detailed description of these parameters and their data sources is presented in Appendix A.

Table 10. Crop-specific parameters for each of the two crops in the rotation. (rounded)*

	Value	
Parameter	**Corn**	**Alfalfa**
Max_crop_yield [kg ha$_{-1}$]	7,800	7,000
*Specific_nitrogen_uptake [–] **	0.007	0.03
Harvest_specific_weight [–]	1.15	1.14
Max_crop_residues [kg ha$_{-1}$]	2,000	4,000
Amount_straw_and_leaves [kg ha$_{-1}$]	8,600	0
CN_ratio_in_straw [–]	57	15
Max_nitrogen_in_crop_residues [kg ha$_{-1}$]	39	75
Nitrogen_fixation [–]	0	0.025

4. Model Description

Several structural changes were made to the Bossel Model in preparing the Thesis Model. For instance, I changed the initial model conditions, several parameter values, the model structure and its settings, as well as added many parameters so the model fit the present research context. A detailed summary of all changes is provided in Table 17 (Appendix A).

The Thesis Model was used to test the influence of a disturbance to the system, in the form of a shifting equilibrium. The initial model condition is assumed to be a dynamic equilibrium – i.e. I assume that the time preceding the simulation period is characterized by a state of dynamic equilibrium, under the conditions of the baseline scenario – and at the beginning of the simulation (t = 0) I change the conditions of the model such that the system must respond to that disturbance and lead to a new dynamic equilibrium. This allows me to effectively test how the 'baseline' system responds to a disturbance, i.e. changing the land application practice between different soil amendments.

To allow the Thesis Model to represent a dynamic equilibrium situation from the beginning, I altered the starting value of the four stocks to which flows had also been added (Table 17) – this 'setting up' of the model is described in the following paragraph. The stocks of relative plant biomass (RPB) and relative crop yield (RCY), as well as their starting values, are the same as in the Bossel Model. I developed the new stocks of TCS, TCC, population, biosolids storage, and biochar storage and the flows that inform these stocks (Table 17), with the starting value of population based on Capital Regional District (2017b) data, the starting values of TCS and TCC based on academic literature, and the initial values of biosolids and biochar storage assumed to be 0 (Table 11).

Table 11. New stocks added to the model, their inflows, outflows and initial values. (approximate)*

Stock	**Inflows (+) and Outflows (–)**	**Initial Value**
TCS_in_soil (kg ha-1)	+ TCS_sorption_from_biosolids, TCS_sorption_from_biochar – TCS_plant_uptake, TCS_microbial_degradation, TCS_photo_degradation	0.000000039 (Xia et al., 2010)
TCC_in_soil (kg ha-1)	+ TCC_sorption_from_biosolids, TCC_sorption_from_biochar – TCC_plant_uptake, TCS_microbial_degradation,	0.0000000811 (Xia et al., 2010)
Population (people)	+ growth	312700* (Capital Regional District (2017b))
biosolids_storage (kg)	+ biosolids_production – biosolids_removed_for_application, biochar_production, excess_biosolids_distribution	0
biochar_storage (kg)	+ biochar_production – biochar_removed_for_application, excess_biochar_distribution	0

Aside from the alterations to structure, the Bossel Model represents a single year, whereas the Thesis Model simulates annual land application for 100 years to examine long-term dynamics of the system. In order to model the dynamics of this system at a finer scale, the Thesis Model uses a time-step of 0.01 years (i.e., 3.6525 days); the time-step of one-hundredth of a year was chosen because a time-step of one-tenth of a year would not produce many data points while one-thousandth of a year would both provide more data points than are needed to see the short-term dynamics in which I am interested and would be computationally time consuming. (A time-step is the increment of time in which the model computes stock values and can be thought of as the temporal resolution of the model. With a smaller time-step comes more data points (outputs) and with more data points the trend(s) in model output dynamics can be more clearly discerned.) The time-step of just over half a week is relatively small, allowing for the model to simulate processes at a sub-weekly scale; this better matches the time-scale of the processes modeled in this study.

One intention of the Thesis Model was to explore soil quality indicators through multiple states of dynamic changes (including equilibrium) so several changes to the Bossel Model were needed for the baseline to reach equilibrium. Early model calibration runs revealed that the initial stock values for the baseline represented a system that was not in equilibrium. As it would have been difficult to draw interpretations and compare between two scenarios with a baseline that was fluctuating, adjustment to the baseline conditions in the Bossel Model were required.

To estimate (starting) equilibrium conditions, the baseline scenario (Scenario A) was run until it reached equilibrium. Figure 21 and Figure 22 illustrate that original baseline conditions reached

equilibrium within the first three or four decades. I determined the 'equilibrium' initial values for the stocks of C-FSOM, C-SSOM, N-FSOM and PAN based on the average value from years 80 – 100 (the final 20 years) of the simulation. The calibrated stock values are presented in Table 12, allowing for a more straight-forward comparison of the biosolids and biochar scenarios through the use of a comparison between the two scenarios and the baseline. Details on how these starting conditions were developed are developed below.

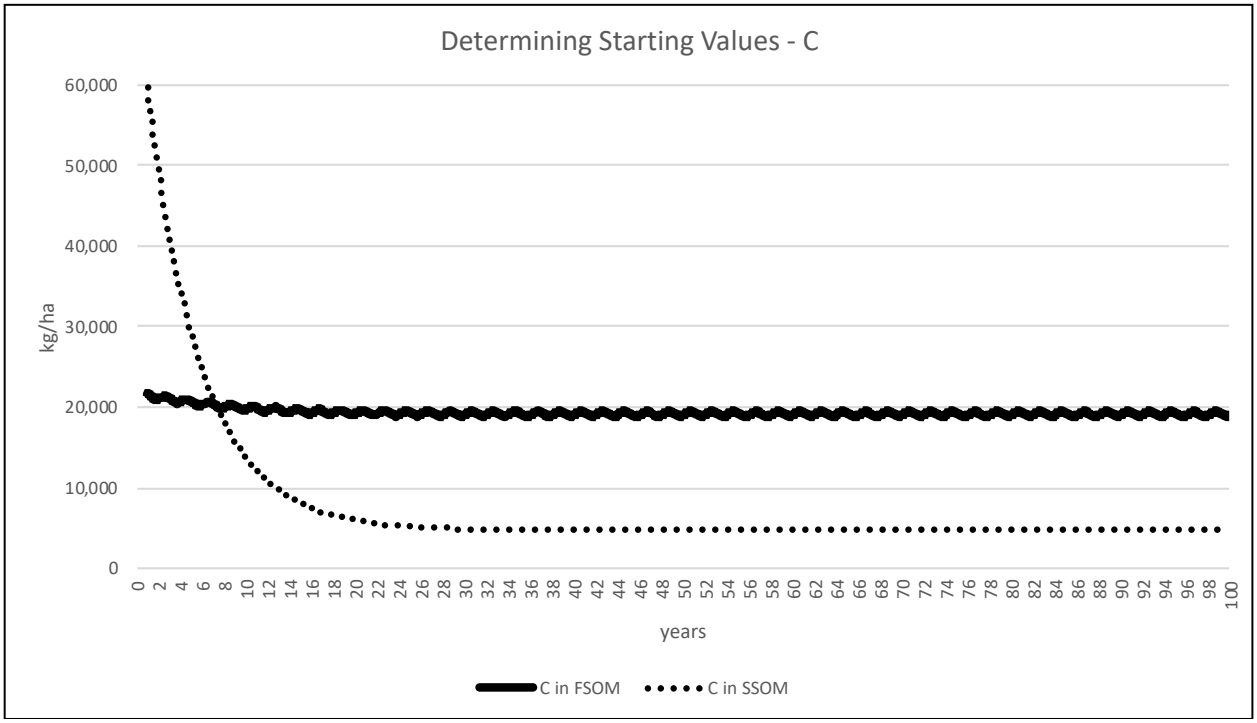

Figure 21. 'Calibration' model runs used to determine the initial stock values for carbon stocks, based on the baseline scenario conditions.

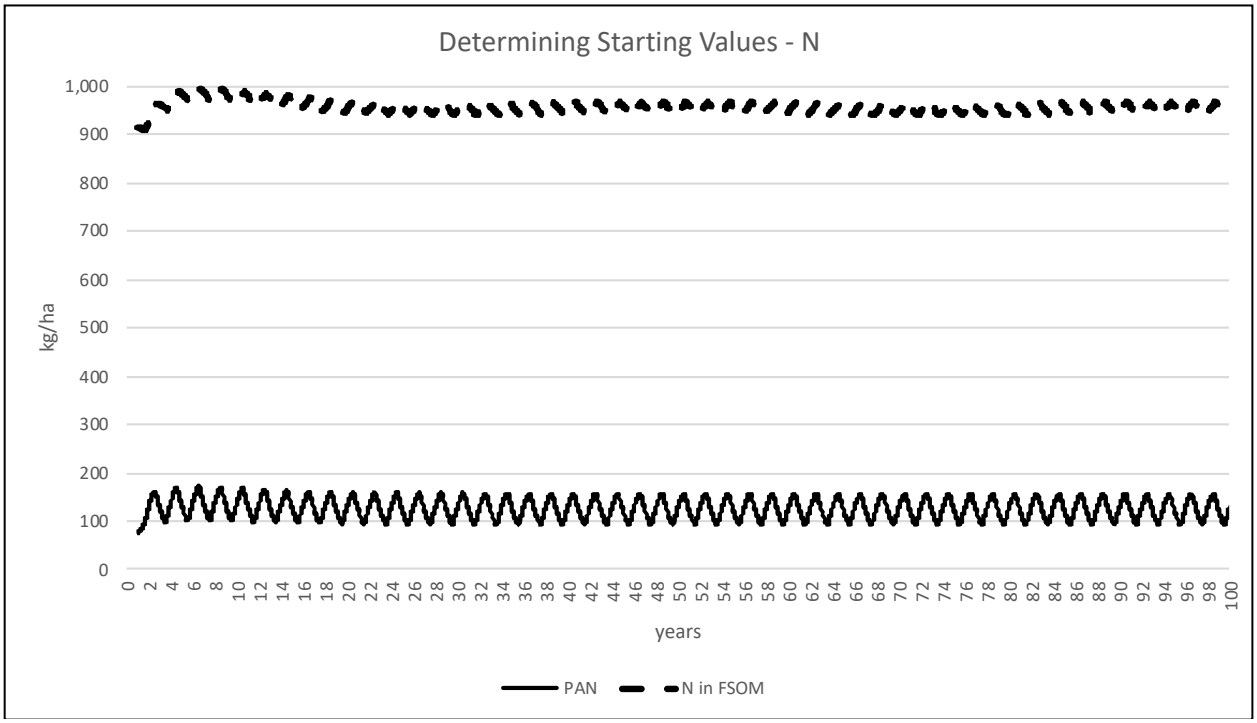

Figure 22. 'Calibration' model runs used to determine the initial stock values for nitrogen stocks, based on the baseline scenario conditions.

Table 12. Stocks in the Bossel Model to which I have added flows and whose initial values I modified.
**rounded – see Appendix A for the exact details of all parameters*

		Initial Formula or Value	
Stock	Inflows Added	**Original**	**Calibrated**
carbon_ in_FSOM (kg ha-1)	C_input_from_ biosolids, nutrient_C_input_ from_biochar	ORGANIC__MATTER_IN_SOIL*(1-INITIAL_FRACTION_SSOM_IN__ORGANIC_MATTER)*DEPTH_OF__PLOW_LAYER*sqm__per_ha*ORGANIC_MATTER_SPECIFIC_WEIGHT*C_per_OM	19200*
carbon_ in_SSOM (kg ha-1)	Permanent_C_ input_from_ biochar	(ORGANIC__MATTER_IN_SOIL*(1-INITIAL_FRACTION_SSOM_IN__ORGANIC_MATTER)*DEPTH_OF__PLOW_LAYER*sqm__per_ha*ORGANIC_MATTER_SPECIFIC_WEIGHT*C_per_OM)*(INITIAL_FRACTION_SSOM_IN__ORGANIC_MATTER/(1-INITIAL_FRACTION_SSOM_IN__ORGANIC_MATTER))	4750*
nitrogen_ in_FSOM (kg ha-1)	N_input_from_ biosolids, N_input_from_ biochar,	initial_C__in_FSOM/INITIAL_CN_IN_FSOM	953*
plant_ available_ nitrogen (kg ha-1)	PAN_input_from_ biosolids, PAN_input_from_ biochar	50	125*

First, the calibrated initial value of PAN, 125 kg ha-1, seems somewhat high relative to other studies, which found values ranging from 9 – 98 kg ha-1. The calibrated value of PAN is roughly 28% greater than the maximum (98 kg ha-1) of the range of values found in previous literature. This difference may be due to the Thesis Model either underestimating the amount of PAN loss (by leaching, crop uptake, or transfer to OM) or overestimating the amount of PAN input (by atmospheric deposition or OM decomposition), though it is unclear which parameter(s) is (are) creating this difference.

Second, the CN ratio in this calibrated model condition was around 20.2, based on the values of C-FSOM and N-FSOM. This is within the range of normal values for an agricultural soil and could be found in an agricultural field, and is within the commonly desired range of 20 – 30 though att he lower end of this range. Batjes (1996) found the C/N ratios of soils to range from 9.9 (arid Yermosols) to 25.8 (Histosols); and Batjes (1996) also found Podzols contain the second-highest C/N ratios (after Histosols), so the calibrated C/N ratio in the Thesis Model is likely close to what may be expected in the CRD given that the agricultural soils of the CRD are mainly Podzols.

Third, the sum of C-SSOM and C-FSOM was around 24,000 t SOC ha^{-1}. This amount of total carbon in the soil seems appropriate relative to other sources – for example, according to the *Global Soil Organic Carbon Map* (FAO & ITPS, 2019), soils in the Pacific Northwest United States commonly have 10,000 – 30,000 kg SOC ha^{-1} (I did not include Canada in this value due to the low spatial resolution of the data set from FAO & ITPS).

I added an inflow from biosolids and biochar to C-FSOM, N-FSOM and PAN, and an inflow from biochar to the stock of C-SSOM. This implies the assumption that biochar contains C-SSOM, C-FSOM, N-FSOM and PAN, whereas biosolids contain the same except for C-SSOM. The assumption that biosolids do not contain C-SSOM is based on the structure of the Bossel Model, in which the organic soil amendments (compost and manure) do not directly inform the stock of C-SSOM. A description of each of these flows is provided in Table 18 (Appendix A).

Several of the flows based on the Bossel Model were indirectly changed as a result of altering the parameters that inform them (the changes in those parameters are described below). I assumed the input to the soil can be from land-application of biosolids, biochar, organic fertilizers, mineral fertilizers, crop residues and atmospheric deposition, and that the output can be due to decomposition of OM (dominated by microbial degradation), plant uptake, photodegradation, harvest, and leaching (erosion). A detailed summary of the flows – names, units, equations, descriptions, and references (where applicable) – is provided in Appendix A. A summary of the converters which were altered from the Bossel Model, their original and new values, the reasoning for the alterations, and the sources used (where applicable) are shown in Table 13 and Table 14.

Table 13 presents the constant values, while Table 14 presents converters that were changed from constant values in the Bossel Model to formulas in the Thesis Model. The complete Thesis Model is visually illustrated in Appendix B. In Table 14 the only unitless parameter is nitrogen fixation, and this constant value is multiplied by others to inform the nitrogen uptake by plants (kg ha^{-1} yr^{-1}), which is also calculated based on the growth rate (kg ha^{-1} yr^{-1}) and specific nitrogen uptake (kg nitrogen kg^{-1} yield, i.e., unitless as it is kg kg^{-1}).

*Table 13. Constant converters modified from Bossel (2007), their sources (where applicable) and the reasoning behind the changes. * rounded.*

		Value			
Converter	**Unit**	**Original**	***New***	**Reasoning**	**Source**
nitrogen_fertilizer_applied	kg ha^{-1}	80	**0**	Removed – focus here is on the organic amendments	N/A
compost_applied	kg ha^{-1}	10,000	**5,000**	Halved so the total amount of compost/manure in A is more within the range of the total organic amendment in B and C	N/A
ODM_fraction_in_compost	–	0.35	**0.42***	Organic dry matter fraction of compost, average of 9 locally made composts	(Alba, Buzuk, & Thompson, 2019)
CN_in_compost	kg C / kg N	15	**16.7***	C/N ratio in the compost, average of 9 locally made composts	(Alba et al., 2019)
manure_applied	kg ha^{-1}	15,000	**7,500**	Halved so the total amount of compost/manure in A is more within the range of the total organic amendment in B and C	N/A
ODM_fraction_in_manure	–	0.25	**0.25***	Organic dry matter fraction of the manure, median value for solid beef manure	(Government of Manitoba, 2015)
CN_in_manure	kg C / kg N	20	**14.6***	C/N ratio in the manure, median value for solid beef manure	(Government of Manitoba, 2015)
begin_growth_period	years	0.3	**0.25**	Crop growth begins at 0.25 years (early April) in Victoria, based on the timing of the beginning of the growing season	(Canada, 2019a)
harvest_time	years	0.6	**0.7**	Crop harvest occurs at 0.7 years (early September) in Victoria, based on the long growing season which extends into October	(Canada, 2019b)
soil_specific_weight	kg/m^3	1,600	**1,510**	Soil bulk density (mass per unit volume), mean across 33 BC sites	(Y. Zhao et al., 2008)
mineral_fertilizer_date	years	0.25	**0.22**	Mineral fertilizer date; assumed to be in mid-March based on thebegin growth date	(Canada, 2019a)
leaching_rate	1 $years^{-1}$	0.3	**1**	Proportion of PAN leached	N/A
organic_fertilizer_date	years	0.2	**0.18**	Assumed to be in early March, based on the begin growth date	(Canada, 2019a)
organic_matter_specific_weight	kg/m^3	650	**15,00**	Organic matter relative density (mass per unit volume)	N/A
C_per_OM	–	0.47	**0.58**	Proportion of carbon in organic matter (OM = organic C * 1.724)	N/A

Table 14. Constant converters modified from Bossel (2007) to be formulas, and their sources and reasoning. Rather than the parameters being set to one constant value, they switch between two values at 0.25 years into every year (when crop growth starts).

Converter	Unit	Old	***New***	Reasoning	Source
Max_ crop_ yield	kg ha^{-1}	6500	**IF CROP__BIYEARLY__TIME >= 1.25 OR CROP__BIYEARLY__TIME < 0.25 THEN 7000 ELSE 7800**	Maximum crop yield varies with the crop; for corn, the value is 7800 kg ha^{-1} while for alfalfa it is 7,000	(Bossel, 2007)
Specific_ nitrogen_ uptake	kg N / kg yield	0.029	**IF CROP__BIYEARLY__TIME >= 1.25 OR CROP__BIYEARLY__TIME < 0.25 THEN 0.029 ELSE 0.032**	Specific nitrogen uptake (kg N / kg yield) varies with the crop; for corn, the value is 0.032 while for alfalfa it is 0.029	(Bossel, 2007)
Harvest_ specific_ weight	kg fresh weight / kg dry matter	1.15	**IF CROP__BIYEARLY__TIME >= 1.25 OR CROP__BIYEARLY__TIME < 0.25 THEN 1.14 ELSE 1.15**	Harvest yield specific weight (kg fresh weight / kg dry matter) varies with the crop; for corn, the value is 1.15 while for alfalfa it is 1.14	(Bossel, 2007)
Max_ crop_ residues	kg ha^{-1}	1700	**IF CROP__BIYEARLY__TIME >= 1.25 OR CROP__BIYEARLY__TIME < 0.25 THEN 4000 ELSE 2000**	The maximum amount of crop residues varies with the crop; for corn, the value is 2,000 kg/ha while for alfalfa it is 4,000 kg/ha	(Bossel, 2007)
Amount_ straw_ and_ leaves	kg ha^{-1}	7700	**IF CROP__BIYEARLY__TIME >= 1.25 OR CROP__BIYEARLY__TIME < 0.25 THEN 6900 ELSE 8600**	The amount of straw and leaves; for corn, the value is 8,600 kg/ha while for alfalfa it is 6,900 kg/ha	(Bossel, 2007)
CN_ ratio_ in_straw	molecu lar ratio	80	**IF CROP__BIYEARLY__TIME >= 1.25 OR CROP__BIYEARLY__TIME < 0.25 THEN 13 ELSE 57**	The C/N ratio in straw varies with the crop; for corn, the value is 57 while for alfalfa it is 13	(USDA Natural Resources Conservation Service, 2011)
Max_ nitrogen_ in_crop_ residues	kg ha^{-1}	17	**IF CROP__BIYEARLY__TIME >= 1.25 OR CROP__BIYEARLY__TIME < 0.25 THEN 126 ELSE 39**	The maximum amount of nitrogen in crop residues varies with the crop; for corn, the value is 39 kg/ha while for alfalfa it is 126 kg/ha	(Bossel, 2007; Torma, Vilček, Lošák, Kužel, & Martensson, 2017)
Nitrogen_ fixation	–	0	**IF CROP__BIYEARLY__TIME >= 1.25 OR CROP__BIYEARLY__TIME < 0.25 THEN 0.025 ELSE 0**	Nitrogen fixation depends on the crop; for corn, the value is 0 while for alfalfa it is 0.068	(Bossel, 2007)
nitrogen_f ertilizer_a pplied	kg ha^{-1}	80	**IF CROP__BIYEARLY__TIME >= 1.25 OR CROP__BIYEARLY__TIME < 0.25 THEN 0 ELSE 137**	Fertilizer application varies with the crop; for corn, the value is 137 kg/ha – for alfalfa it is 0	(Yang et al., 2008)

5. Results and Discussion

This chapter presents a discussion of the modelling results of the three modelling scenarios: Baseline, Biosolids, and Biochar application. Results are presented in order of the key areas of interest in the model: carbon, nitrogen, the carbon-to-nitrogen (C/N) ratio, EDCs, and plant biomass. The C/N ratio is an important element in the model due to its impact on OM decomposition and nitrogen cycling, so the C/N ratio is presented in its own section because it does not fall under only one category (e.g., carbon or nitrogen).

In these results, I report rounded values, with the values rounded to between one and four significant digits depending on the value itself. For example, values in the thousands like carbon are rounded to the hundreds, whereas values in the tens and hundreds like Nitrogen are rounded to the ones, and values in the tenths like EDCs are rounded to the hundredths. Results are discussed in the format of a comparison between the final condition of each scenario and the baseline scenario. In general, in the last 20 years of the simulation, the majority of key stock and parameter values have reached steady state equilibrium, so the average of the last 20 years of the model period is taken to be the final value for each parameter. Additional investigations – into what happens when the application frequency for biochar is decreased and when biochar application is supplemented with mineral fertilizer application – are discussed in the last two sections.

5.1. Carbon

Figure 23 illustrates the accumulation of carbon in the stabile soil organic matter (C-SSOM) [kg ha^{-1}] across the three study scenarios. These results indicate that the **baseline** condition of applying compost and manure presents a final stable carbon accumulation of 4,750 kg C-SSOM ha^{-1}. Applying **biosolids** instead of compost and manure at a rate of 10,000, 15,000, and 20,000 kg biosolids ha^{-1} yr^{-1} increases the accumulation of C-SSOM by 1,140, 3,180, and 5,270 kg C-SSOM ha^{-1} (+24%, +67%, and +111%), respectively. Applying **biochar** instead of compost and manure at a rate of 10,000 and 15,000 kg biochar ha^{-1} yr^{-1} increases the accumulation of C-SSOM by 920 and 3,840 kg C-SSOM ha^{-1} (+19% and +81%), respectively, while an addition of 5,000 kg biochar ha^{-1} yr^{-1} decreases the accumulation of C-SSOM by 1,910 kg C-SSOM ha^{-1} (–40%).

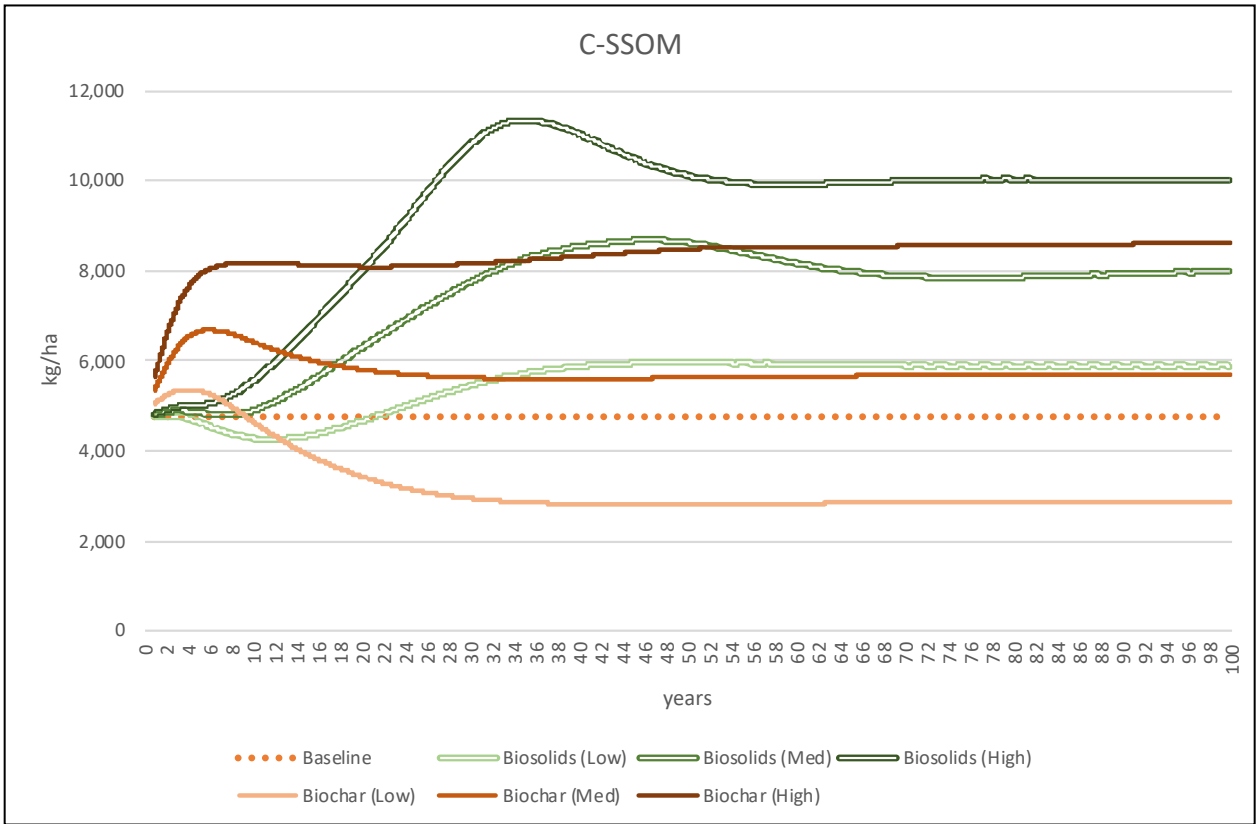

Figure 23. Carbon in the SSOM – moving annual average.

Figure 24 illustrates the rate of OM decomposition (DCOM) [kg ha$_{-1}$ yr$_{-1}$] across the three study scenarios. These results indicate that the **baseline** condition of applying compost and manure presents a rate of 3,830 kg OM DECOMPOSITION ha$_{-1}$ yr$_{-1}$ upon reaching steady state conditions. Applying **biosolids** instead of compost and manure at a rate of 10,000, 15,000, and 20,000 kg biosolids ha$_{-1}$ yr$_{-1}$ increases the rate of OM DECOMPOSITION by 920, 2,590, and 4,240 kg OM DECOMPOSITION ha$_{-1}$ yr$_{-1}$ (+24%, +68%, and +112%), respectively. Applying **biochar** instead of compost and manure at a rate of 5,000, 10,000, and 15,000 kg biochar ha$_{-1}$ yr$_{-1}$ decreases the rate of OM DECOMPOSITION by 3,140, 2,460, and 1,690 kg OM DECOMPOSITION ha$_{-1}$ yr$_{-1}$ (–82%, –64%, and –44%), respectively.

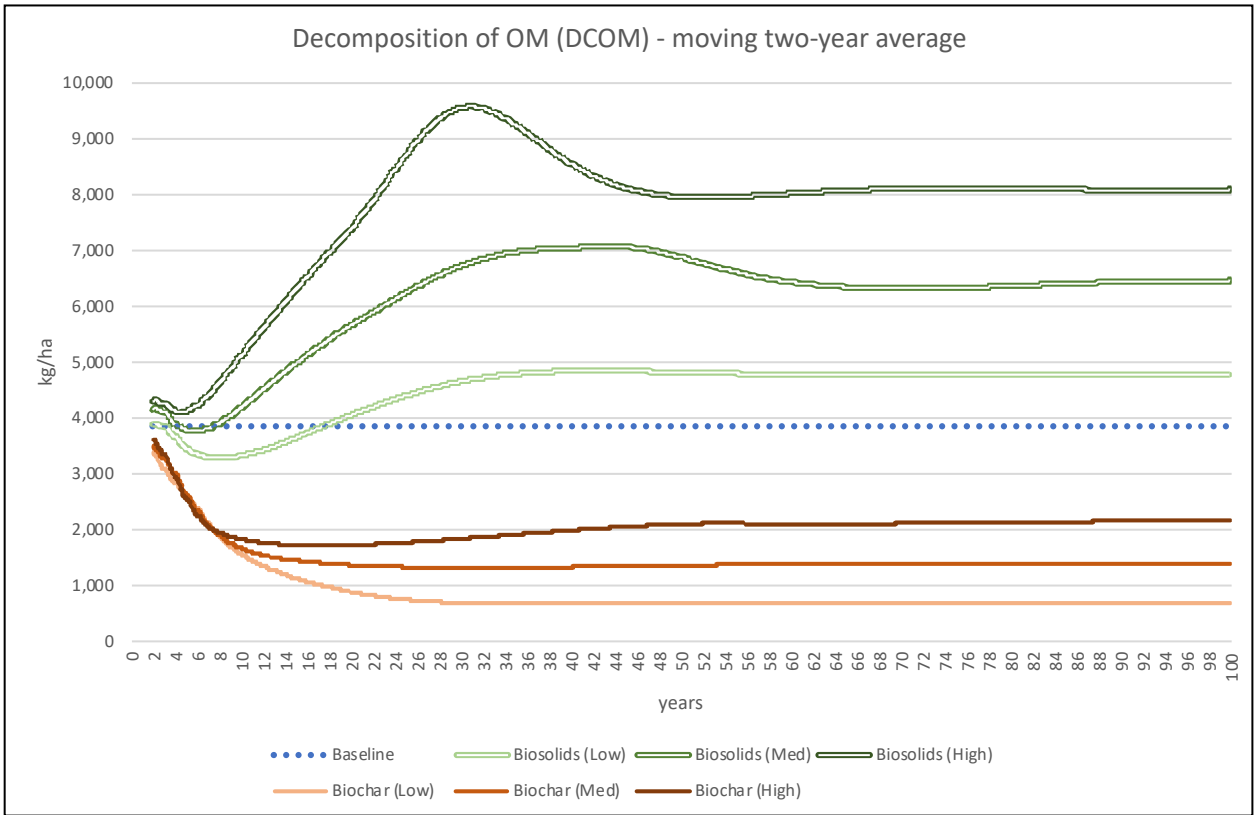

Figure 24. Decomposition of OM – moving biannual average.

Figure 25 illustrates the accumulation of carbon in the labile soil organic matter (C-FSOM) [kg ha$_{-1}$] across the three study scenarios. These results indicate that the baseline condition of applying compost and manure presents a carbon accumulation of 19,200 kg C-FSOM ha$_{-1}$ upon reaching steady state conditions. Applying **biosolids** instead of compost and manure at a rate of 10,000, 15,000, and 20,000 kg biosolids ha$_{-1}$ yr$_{-1}$ increases the accumulation of C-FSOM by 22,800, 30,600, and 38,900 kg C-FSOM ha$_{-1}$ (+119%, +159%, +203%), respectively. Applying **biochar** instead of compost and manure at a rate of 15,000 kg biochar ha$_{-1}$ yr$_{-1}$ increases the accumulation of C-FSOM by 11,900 kg C-FSOM ha-1 (+62%), while an addition of 5,000 and 10,000 kg biochar ha$_{-1}$ yr$_{-1}$ decreases the accumulation of C-FSOM by 12,800 and 2,500 kg C-FSOM ha$_{-1}$ (–67% and –13%), respectively.

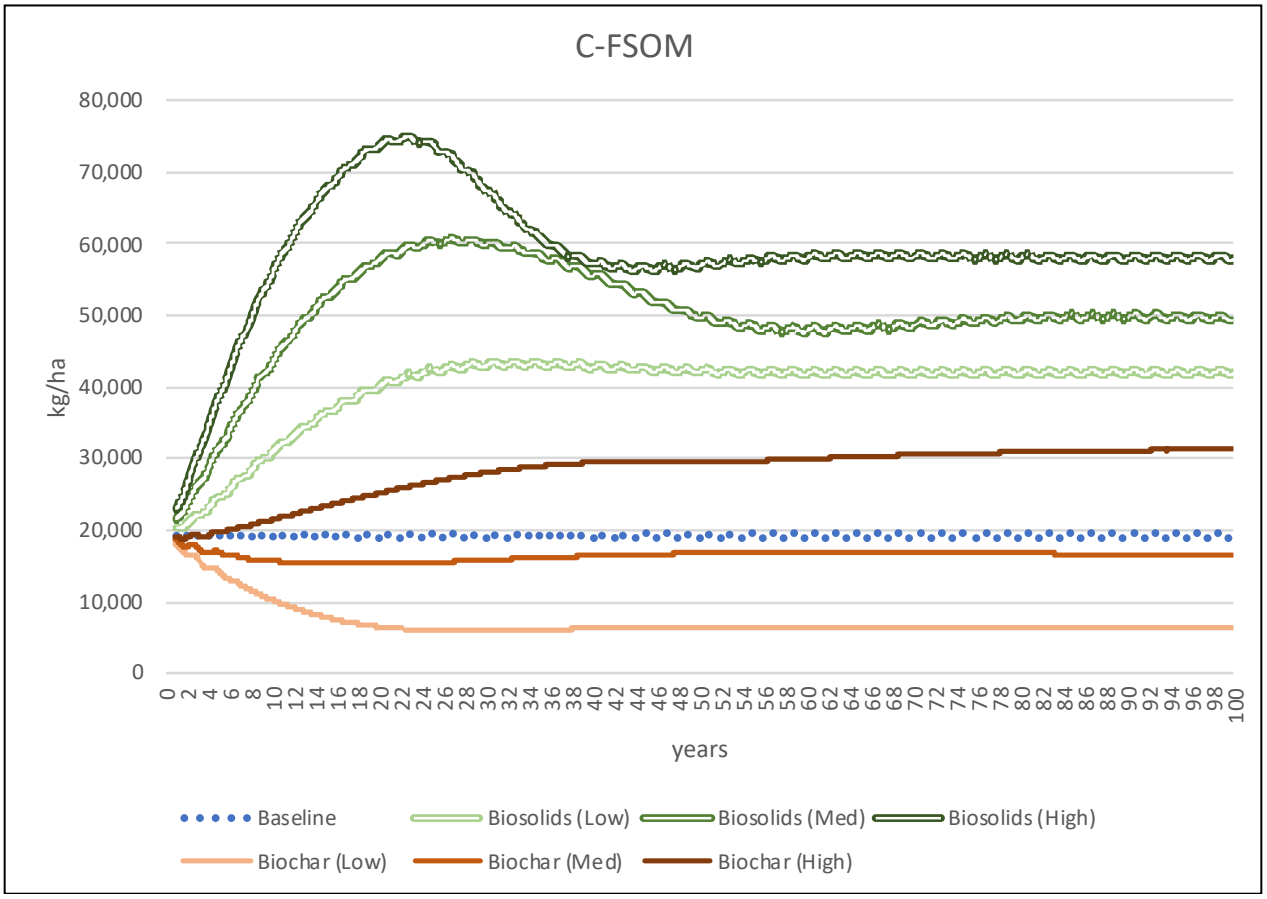

Figure 25. Carbon in the FSOM – moving biannual average.

Figure 26 illustrates the rate of carbon input from crop residues (CCR) [kg ha$_{-1}$ yr$_{-1}$] across the three study scenarios. These results indicate that the **baseline** condition of applying compost and manure presents a final crop residue input rate of 1,560 kg CCR ha$_{-1}$ yr$_{-1}$. Applying **biosolids** instead of compost and manure at a rate of 15,000 and 20,000 kg biosolids ha$_{-1}$ yr$_{-1}$ decreases the rate of CCR by 74, 66, and 59 kg CCR ha$_{-1}$ yr$_{-1}$ (–4.7%, –4.2%, and –3.8%%), respectively. Applying **biochar** instead of compost and manure at a rate of 5,000 and 10,000 kg biochar ha$_{-1}$ yr$_{-1}$ each decreases the rate of CCR by 1,560 kg CCR ha$_{-1}$ yr$_{-1}$ (–100%), while an addition of 15,000 kg biochar ha$_{-1}$ yr$_{-1}$ decreases CCR by 1,450 kg CCR ha$_{-1}$ yr$_{-1}$ (–93%).

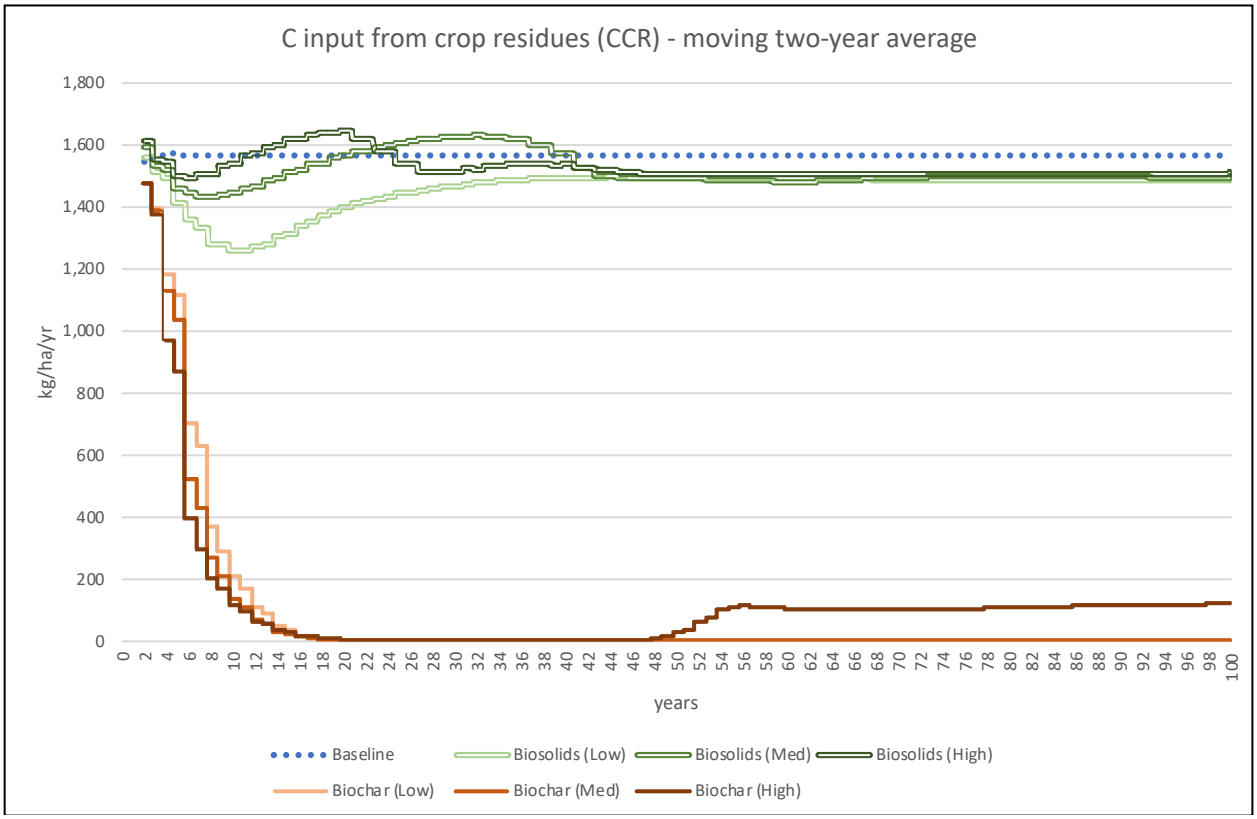

Figure 26. Carbon input from crop residues – moving biannual average.

5.1.1. Reflection: Carbon

In the Baseline scenario there was 4,750 kg stable carbon ha-1 and 19,200 kg labile carbon ha-1, meaning the total of around 24,000 kg SOC ha-1 was made up of around 20% stable carbon and 80% labile carbon. However, the expected ratio of stable to labile carbon in soils is typically higher; for instance, Sahoo, Singh, Gogoi, Kenye, and Sahoo (2019) studied active and passive SOC fractions across many land use types in India and consistently found around 40% stable and 60% labile carbon across the various land uses. This difference between their results and mine likely exists because in the current baseline scenario, the two soil amendments being applied -– compost and manure – are high in labile carbon, and in reality, they may not be applied repeatedly at the same rate every year, so as this continues for several decades in the simulation the balance of soil carbon content shifts away from the stable carbon fraction towards labile carbon.

Applying biosolids results in greater labile carbon accumulation than in the baseline scenario, with average values ranging from 42,000 – 58,000 kg labile carbon ha-1. Applying biosolids also results in greater stable carbon accumulation than in the baseline scenario, with average values ranging

from 5,900 – 10,000 kg ha^{-1}. These results agree with previous research that has found an increase of stable carbon accumulation resulting from biosolids application, however the extent of the increase may be on the low end of the spectrum. For instance, Wijesekara et al. (2017) performed a study of biosolids application and found increases of stable carbon accumulation in the range of +70% to +300% (whereas the current results suggest an increase from +24% to +112%). The extent of the increase may in part be low because the current baseline scenario incorporates compost and manure application while the control plots in the study by Wijesekara et al. (2017) were unamended soils.

Applying biochar results in the lowest rates of crop residue input, and the highest biosolids application rate results in the greatest rate of OM decomposition of the three biosolids scenarios. Applying biochar also results in lower stable carbon accumulation than with biosolids, which may seem counterintuitive given that biochar is known to have a greater proportion of this stable carbon pool than biosolids as pyrolysis removes most of the labile carbon in the organic material (Gonzaga et al., 2017; Hossain et al., 2011; Bhatta Bhawana Kaudal, Aponte, & Brodie, 2018). It has been demonstrated that biochar is effective at sequestering carbon in a stable form in the soil (Ippolito et al., 2012; Kookana et al., 2011; Laird, 2008; Woolf, Amonette, Street-Perrott, Lehmann, & Joseph, 2010), but the current results suggest this is not necessarily true in the long-term if biochar alone is applied to the soil. These results also agree with previous research which has shown that biochar addition can have a positive effect on carbon mineralization in the short term, but can also lead to negative priming (i.e. lower mineralization of labile carbon to stable carbon) in soils, perhaps due to stabilization of labile carbon by biochar (Cross & Sohi, 2011). The current results of stable carbon dynamics show that biochar addition to the soil results in a substantial increase in stable carbon in the first decade but the effect tapers off; this agrees with the findings of Cross and Sohi (2011) who demonstrated that carbon mineralization was higher in biochar-amended soils than unamended soils and the effect was speculated to be caused by *rapid* mineralization of the *relatively small* labile carbon fraction (ranging from 0.25% – 1.0% of its mass) of the biochar. Only one of the biochar scenarios (the highest application rate) results in an accumulation of labile carbon that is greater than the baseline, and this makes sense as biochar contains little labile carbon. the results of this study agree with previous research on the effect of biochar on the decomposition of OM. For example, Cheng, Hill, Bastami, and Jones (2017) performed a field experiment of biochar addition to soil and found it to suppress the decomposition of complex OM in the soil,

with higher biochar production temperatures resulting in the greatest suppression of OM decomposition. The current results also agree with Dempster, Gleeson, Solaiman, Jones, and Murphy (2011) who performed an experiment on the effects of applying eucalyptus biochar on soils growing wheat and concluded that there was decreased decomposition of SOM under biochar application, Additionally, because crop residue input is directly informed by the amount of plant biomass in the Thesis Model, the reasoning for the observed dynamics in plant biomass discussed below (i.e. that biosolids tend to be more effective in increasing crop biomass) applies here as well.

Overall, the model may underestimate stable carbon or overestimate labile carbon because in all scenarios there appears to be an imbalance of labile carbon relative to stable carbon. In the baseline scenario there was 24,000 kg total SOC ha^{-1}, in the biosolids scenarios there was between 52,000 and 68,000 kg total SOC ha^{-1}, and in the biochar scenarios there was between 9,200 and 40,000 kg total SOC ha^{-1}. Put differently, in the baseline scenario total SOC was made up of 20% stable carbon, in the high biosolids scenario it was 15% stable carbon, and in the high biochar scenario it was 22% stable carbon. It is not common to see such a high proportion of labile carbon and low proportion of stable carbon in agricultural soils, so this disagreement in the proportions of stable carbon and labile carbon stocks suggests additional investigation is required to clarify the imbalance of these stocks in the Thesis Model.

In addition, this study shows that soil carbon stocks should be managed in tandem as there are interrelations between the stocks which cause them to limit one another's growth in certain cases. The current results suggest that due to the interrelation between accumulations of slow and fast carbon, when there are deficiencies in one accumulation (in this case, labile carbon) relative to the other (in this case, stable carbon), the growth of one of the accumulations may be limited. Figure 23 illustrates that the relatively high amount of stable carbon in the early years of the biochar scenarios cannot be supported by the other components of the system. In the first five years of the simulation, the biochar scenarios result in more stable carbon than all other scenarios, but there quickly appears to be a 'leveling' effect which limits the increase of stable carbon. In contrast, when biosolids are applied, stable carbon takes longer to reach an equilibrium. Higher biosolids application rates are also less and less effective at boosting stable carbon accumulation: even though the increment in amount of amendment applied is the same between each scenario, the increase in stable carbon accumulation from the medium to high biosolids application is less than that between the low and medium application rates; in contrast, the increment in stable carbon

accumulation from the low to medium and medium to high biochar application rates are roughly the same, with the increment from the medium to high biochar application rate being largest.

There also appears to be a carbon saturation effect exemplified within the Thesis Model, which is in line with what the literature on soil carbon sequestration finds (Chung, Grove, & Six, 2008; Stewart, Paustian, Conant, Plante, & Six, 2007; Stewart, Plante, Paustian, Conant, & Six, 2008). When biosolids are applied, the trend in carbon accumulation is an overshoot followed by apparently reaching a dynamic equilibrium. This is in line with research on carbon saturation which has demonstrated a non-linear relationship between carbon addition to certain carbon stocks and the accumulation of carbon in said stocks (Stewart et al., 2007). The relationship between increased carbon application and its accumulation is at first somewhat linear, but eventually becomes asymptotic as the influence of more addition becomes less and less effective at increasing that particular stock (Stewart et al., 2007). This occurs in the Thesis Model when there being a large stock of carbon results in that stock experiencing increased losses, for example in the form of increased erosion. In contrast, when biochar is applied the growth pattern is an initial increase followed by an exponential approach to its dynamic equilibrium, and some of the biochar scenarios result in decreased carbon stocks. As discussed above, stable carbon accumulation is likely slowed by the deficiency of biochar in fast carbon, and not caused by the carbon saturation effect that occurs when biosolids are applied.

5.2. Nitrogen

Figure 27 illustrates the accumulation of plant available nitrogen (PAN) [kg ha^{-1}] across the three study scenarios. These results indicate that the **baseline** condition of applying compost and manure presents a nitrogen accumulation of 125 kg PAN ha^{-1} upon reaching steady state conditions. Applying **biosolids** instead of compost and manure at a rate of 15,000 and 20,000 kg biosolids ha^{-1} yr^{-1} increases the accumulation of PAN by 71 and 173 kg PAN ha^{-1} (+57% and +138%), respectively, while an addition of 10,000 kg biosolids ha^{-1} yr^{-1} decreases the accumulation of PAN by 19 kg ha^{-1} (–15%). Applying **biochar** instead of compost and manure at a rate of 5,000, 10,000 and 15,000 kg biochar ha^{-1} yr^{-1} decreases the accumulation of PAN by 91, 79, and 70 kg PAN ha^{-1} (–72%, –63%, and –56%), respectively.

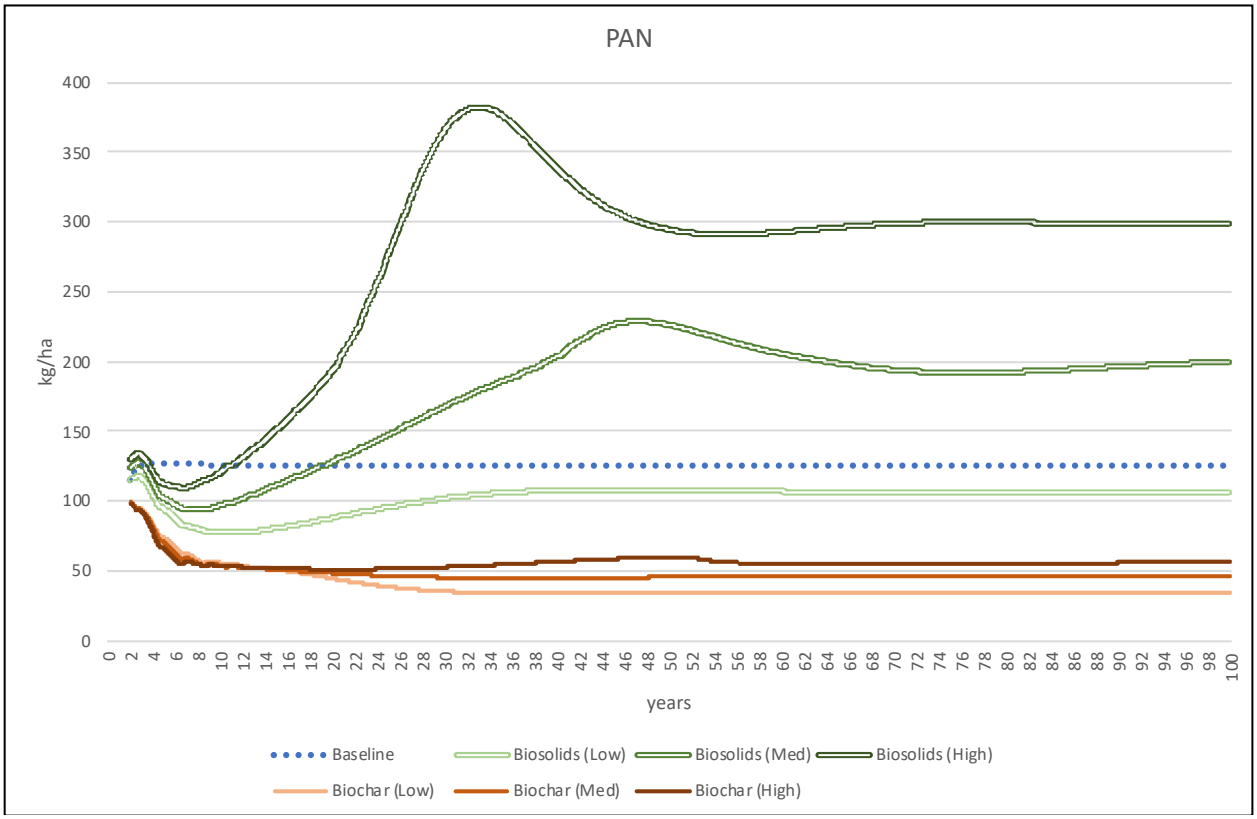

Figure 27. Plant available nitrogen – moving biannual average.

Figure 28 illustrates the flow of nitrogen loss by leaching (NL) [kg ha$_{-1}$ yr$_{-1}$] across the three study scenarios. These results indicate that the **baseline** condition of applying compost and manure presents a nitrogen flow of 108 kg NL ha$_{-1}$ yr$_{-1}$ upon reaching steady state conditions. Applying **biosolids** instead of compost and manure at a rate of 15,000 and 20,000 kg biosolids ha$_{-1}$ yr$_{-1}$ increases the flow of PAN by 23 and 73 kg NL ha$_{-1}$ yr$_{-1}$ (+21% and +68%), respectively, while an addition of 10,000 kg biosolids ha$_{-1}$ yr$_{-1}$ decreases the flow of NL by 31 kg ha$_{-1}$ (–29%). Applying **biochar** instead of compost and manure at a rate of 5,000, 10,000 and 15,000 kg biochar ha$_{-1}$ yr$_{-1}$ decreases the flow of NL by 75, 68, and 55 kg NL ha$_{-1}$ yr$_{-1}$ (–70%, –62%, and –60%), respectively.

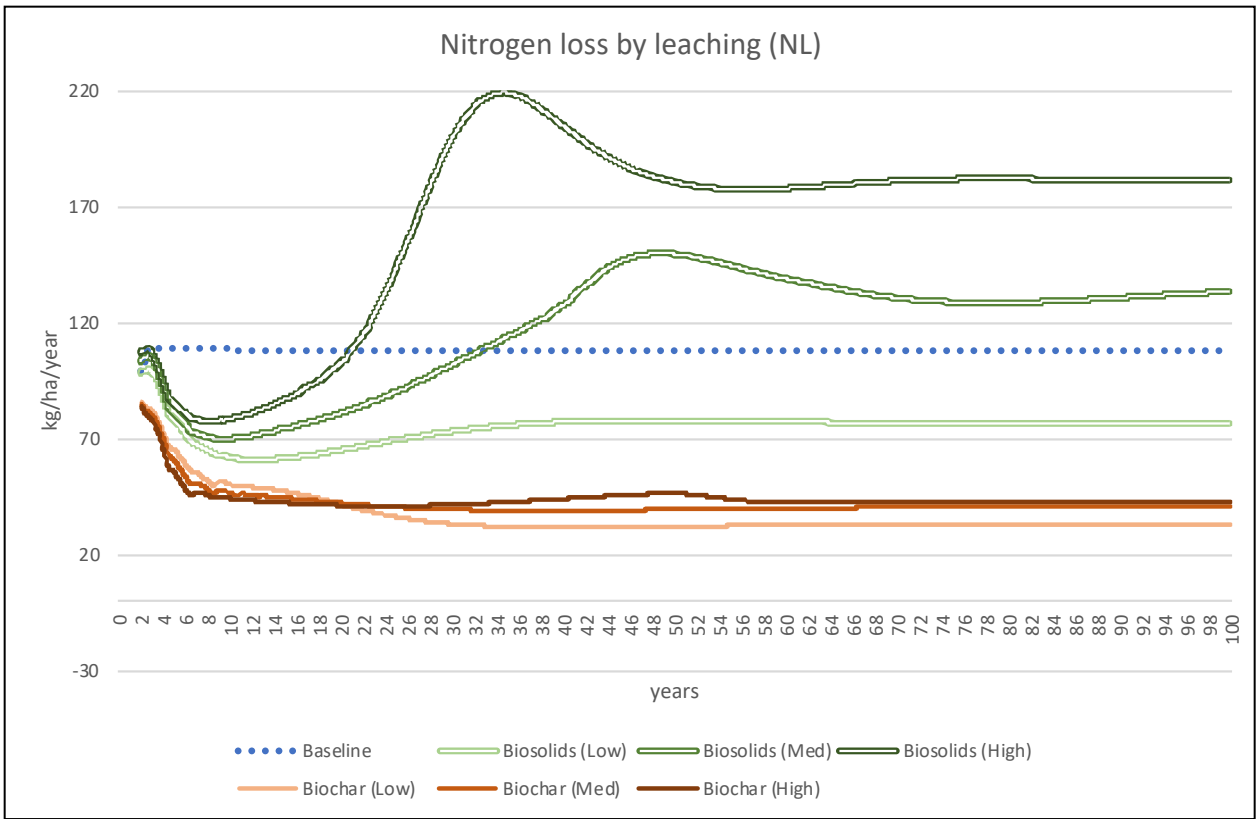

Figure 28. Nitrogen loss by leaching – moving annual average.

Figure 29 illustrates the accumulation of nitrogen in the labile soil organic matter (N-FSOM) [kg ha_{-1}] across the three study scenarios. These results indicate that the **baseline** condition of applying compost and manure presents a nitrogen accumulation of 950 kg N-FSOM ha_{-1} upon reaching steady state conditions. Applying **biosolids** instead of compost and manure at a rate of 10,000, 15,000, and 20,000 kg biosolids ha_{-1} yr_{-1} increases the accumulation of N-FSOM by 480, 820, and 1,160 kg N-FSOM ha_{-1} (+51%, +86%, and +122%), respectively. Applying **biochar** instead of compost and manure at a rate of 5,000, 10,000, and 15,000 kg biochar ha_{-1} yr_{-1} decreases the accumulation of N-FSOM by 740, 450, and 80 kg N-FSOM ha_{-1} (–77%, –47%, and –8%), respectively.

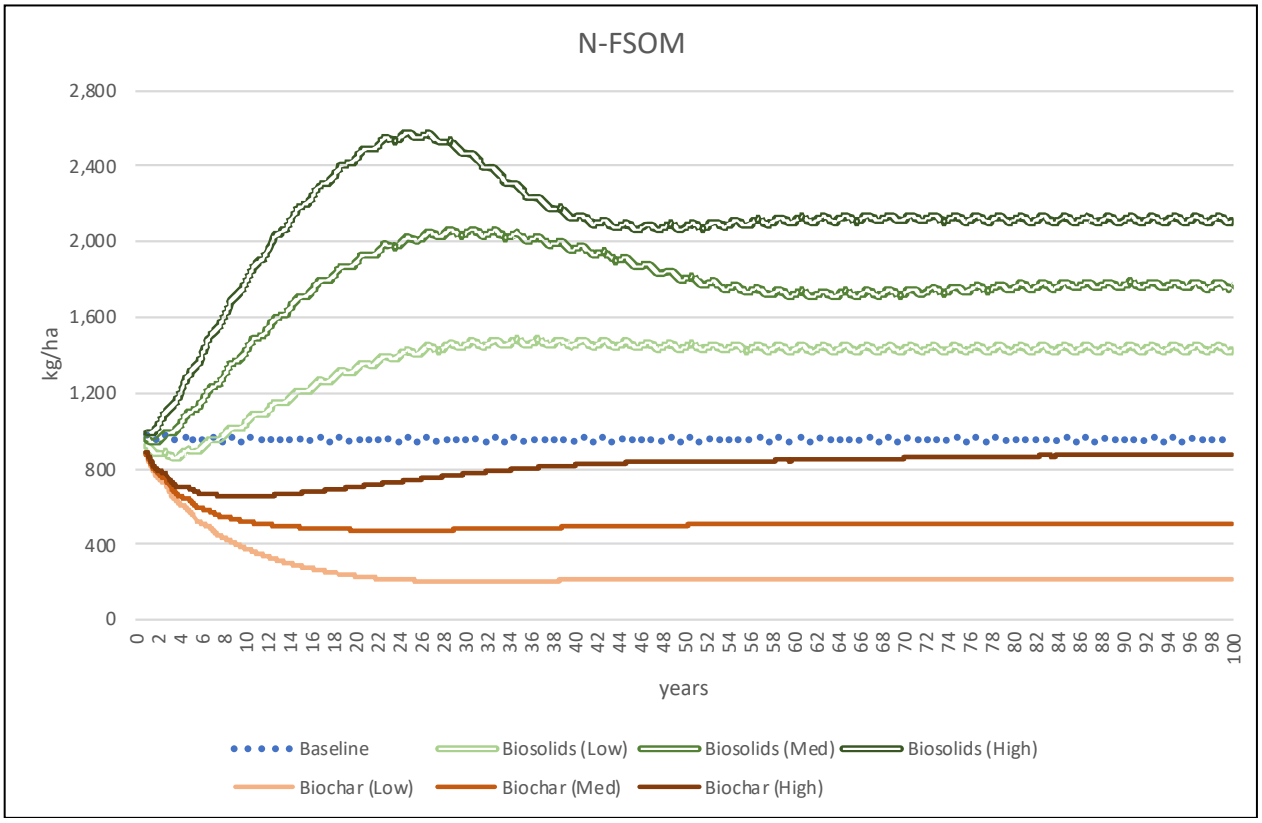

Figure 29. Nitrogen in the FSOM – moving annual average.

5.2.1. Reflection: Nitrogen

By the end of the simulation in the baseline scenario there was an average of 125 kg plant available nitrogen ha^{-1} in the soil. The expected amount of plant available nitrogen found in soils locally, nationally and internationally ranges from 9 – 100 kg plant available nitrogen ha^{-1} (C. F. Drury, Yang, J.Y., De Jong, R. , 2011; Kowalenko, 1987); Stenger et al. (1998); (Yan et al., 2018; Zebarth et al., 2007). Therefore, the baseline scenario is likely an above average representation of the amount of plant available nitrogen that would be in the soil. However, this deviation could perhaps be expected as the baseline scenario in the Thesis Model involves a high amount of organic soil amendment which is rich in nitrogen, and despite the absence of mineral fertilizer application, the yearly application of compost and manure provides an above-average amount of nitrogen to the soil over time. By the end of the simulation in the baseline scenario there was an average of 950 kg organic nitrogen ha^{-1} in the soil. The expected amount of soil organic nitrogen in total soil nitrogen is generally around 90% (Schulten & Schnitzer, 1998), and because 950 kg organic nitrogen ha^{-1} and 125 kg plant available nitrogen ha^{-1} makes 1075 kg total nitrogen ha^{-1}, organic

nitrogen in the baseline scenario is around 88% of total soil nitrogen; this is within the range of what would be expected in an agricultural soil, i.e. around 90% (Schulten & Schnitzer, 1998).

Applying biosolids generally results in greater plant available nitrogen accumulation than in the baseline scenario, with average values ranging from 100 – 300 kg plant available nitrogen ha^{-1}. Applying biosolids also results in greater organic nitrogen accumulation than in the baseline scenario, with average values ranging from 1,400 – 2,100 kg organic nitrogen ha^{-1}. These results agree with previous research which has demonstrated biosolids application increases soil plant available and organic nitrogen stocks (Cogger, Bary, Fransen, & Sullivan, 2001; Jin, Potter, Johnson, Harmel, & Arnold, 2015; Scharenbroch et al., 2013). In terms of the proportion of organic nitrogen in total soil nitrogen, applying biosolids results in the proportion of organic nitrogen being increased by less than 1% to 5%, with higher application rates resulting in a lower proportion of organic nitrogen and higher proportion of plant available nitrogen.

Applying biochar results in lower plant available nitrogen accumulation than in the baseline scenario, with average values ranging from 35 – 56 kg plant available nitrogen ha^{-1}. This amount of plant available nitrogen is typical for agricultural soils, so one might expect crops to grow well in these scenarios, however, as will be explained later, this is not the case due to the dependency of crop growth on the amount of carbon relative to nitrogen in the soil and the excess of carbon present in the biochar scenarios. Applying biochar also results in lower organic nitrogen accumulation than in the baseline scenario, with average values ranging from 220 – 870 kg organic nitrogen ha^{-1}. In terms of the proportion of organic nitrogen in total soil nitrogen, applying biochar results in the proportion being shifted by –2% to +6%, with higher application rates resulting in a higher proportion of organic nitrogen and lower proportion of plant available nitrogen. In terms of nitrogen leaching, the same pattern as the pattern of plant available nitrogen dynamics is present, but the trend is less pronounced (e.g., the medium biosolids application results in a similar rate of nitrogen leaching to the baseline, whereas the low biosolids application rate has plant available nitrogen accumulation that is closest to the baseline).

While biosolids application reduces nitrogen leaching relative to compost and manure application, biochar application results in the lowest amount of nitrogen leaching. There is less nitrogen leaching with higher levels of total carbon (i.e., stable carbon plus labile carbon) in the soil, and because when biosolids are applied there is more total carbon in the soil than in the baseline, this

leads to a lower rate of nitrogen leaching. However, there is even less nitrogen leaching when biochar is applied because the amount of plant available nitrogen in the soil in the biochar scenario is already low (and nitrogen leaching is a factor of plant available nitrogen in the thesis model, so a low amount of plant available nitrogen means a low amount of nitrogen leaching). This result agrees with previous research which has found that biochar can significantly decrease nitrogen leaching from the soil (Kookana et al., 2011; Xu, Tan, Wang, & Gai, 2016). For instance, Paramashivam et al. (2016) performed experiments on biochar addition to soil and demonstrated its addition can decrease nitrogen leaching to a greater extent than biosolids. the range of decreases in nitrogen leaching found in this thesis (–60% to –70%) is around middle of the range of decreases (–10% to –96%) found in the study by Paramashivam et al. (2016), suggesting that our results are in alignment. In a study of 'urban' (municipal) biochar and its use as a growing medium, Bhawana Bhatta Kaudal, Chen, Madhavan, Downie, and Weatherley (2016) found that the biochar treatments significantly reduced the amount of nitrate and ammonium in the leachate: compared to the 0% biochar treatment, the 100% biochar treatment reduced total plant available nitrogen by 88%, which is in alignment with the current results (though the decrease in the current study is lower). Knowles, Robinson, Contangelo, and Clucas (2011) experimented with biochar addition to soil for mitigating nitrogen losses and found that biochar treatments reduced nitrogen leaching, with decreases in the range of 42% to 50%.

The current results agree with previous research on the effects of biosolids versus biochar application on plant available nitrogen in the soil and concurrent plant growth. For example, Scharenbroch et al. (2013) performed an experiment on the effects of biosolids and biochar on tree growth and found biosolids performed better than biochar, leading to greater plant available nitrogen in the soil and subsequent greater tree growth than biochar. However, the biosolids treatment in the study by Scharenbroch et al. (2013) did not perform significantly better than biochar in terms of plant available nitrogen, as the biochar treatment resulted in just 36% less plant available nitrogen than biosolids. This difference in our results is likely due to the fact that the study by Scharenbroch et al. (2013) was based on a short time scale, while the current study is of a long-term simulation. Indeed, during the early years of the present simulation, plant available nitrogen is not significantly lower in the Biochar scenarios than in the biosolids scenarios, which is in agreement with the findings of Scharenbroch et al. (2013), however later in the simulation in this study it does become significantly lower. A study by Bhatta Bhawana Kaudal et al. (2018)

found all biochar treatments led to more significant reductions in soil ammonium than the control (no biochar) treatment; this agrees with the current result that biochar addition reduces plant available nitrogen, but the extent of the reduction is greater in the current study. This difference in our results may be attributed to the fact that the study by Bhatta Bhawana Kaudal et al. (2018) utilized composted pine bark in all treatments and utilized fertilizer in several treatments, whereas in my biochar scenarios there was nothing else added, so their study conditions and mine differed considerably in the amount of total plant available nitrogen being added to the soil.

When biochar is applied, some form of plant available nitrogen needs to be added to the soil if the goal is to reach the same plant available nitrogen accumulation as the baseline scenario, and biosolids application may also lead to excess plant available nitrogen and nitrogen leaching in the soil. This nitrogen leaching is believed to be counteracted by the application of organic based soil amendments (J. Lehmann & Kleber, 2015), but the present results show that the amount of amendment applied must be determined carefully in order to reduce the potential for excess nitrogen leaching. Applying biosolids at all of the tested rates results in greater organic nitrogen accumulation than the baseline scenario, while applying biochar results in less organic nitrogen accumulation than the baseline (though the highest biochar application rate results in organic nitrogen accumulation that is relatively close to the baseline). The observed patterns and trends for the rates of loss and gain that inform organic nitrogen are similar to those of labile carbon because the flows into and out of organic nitrogen in the thesis model are calculated based on carbon-based flows that also inform labile carbon. These results align with previous research on the effects of biosolids and biochar application on soil organic nitrogen dynamics. Scharenbroch et al. (2013) performed an experiment on biosolids and biochar land application and found the biosolids treatment performed better at increasing the amount of organic nitrogen in the soil. Prommer et al. (2014) did a field experiment of biochar application on arable land in Austria and found that biochar application reduced the rate of soil organic nitrogen cycling; this is in agreement with the current result that the stock of organic nitrogen was lowest in the biochar scenarios, because nitrogen cycling drives the conversion of plant available nitrogen to organic nitrogen and nitrogen cycling was reduced in the biochar scenario.

5.3. C/N Ratio

Derived from two of the main stocks in the model, the carbon-to-nitrogen (C/N) ratio influences many other parameters in the model and is involved in most of the feedback loops in the system. For instance, 11 of 12 main reinforcing loops and 7 of 11 main balancing loops include the C/N ratio. Because the C/N ratio is a central component of the model, its dynamics are presented here in a separate section.

Figure 30 illustrates the C/N ratio across the three study scenarios. These results indicate that the **baseline** condition of applying compost and manure presents a carbon-to-nitrogen ratio of 20 upon reaching steady state conditions. Applying **biosolids** instead of compost and manure at a rate of 10,000, 15,000, and 20,000 kg biosolids ha^{-1} yr^{-1} increases the C/N ratio to 29, 28, and 27 (+45%, +40%, and +36%), respectively. Applying **biochar** instead of compost and manure at a rate of 5,000, 10,000, and 15,000 kg biochar ha^{-1} yr^{-1} increases the C/N ratio to 30, 33, and 36 (+47%, 64%, and +77%), respectively.

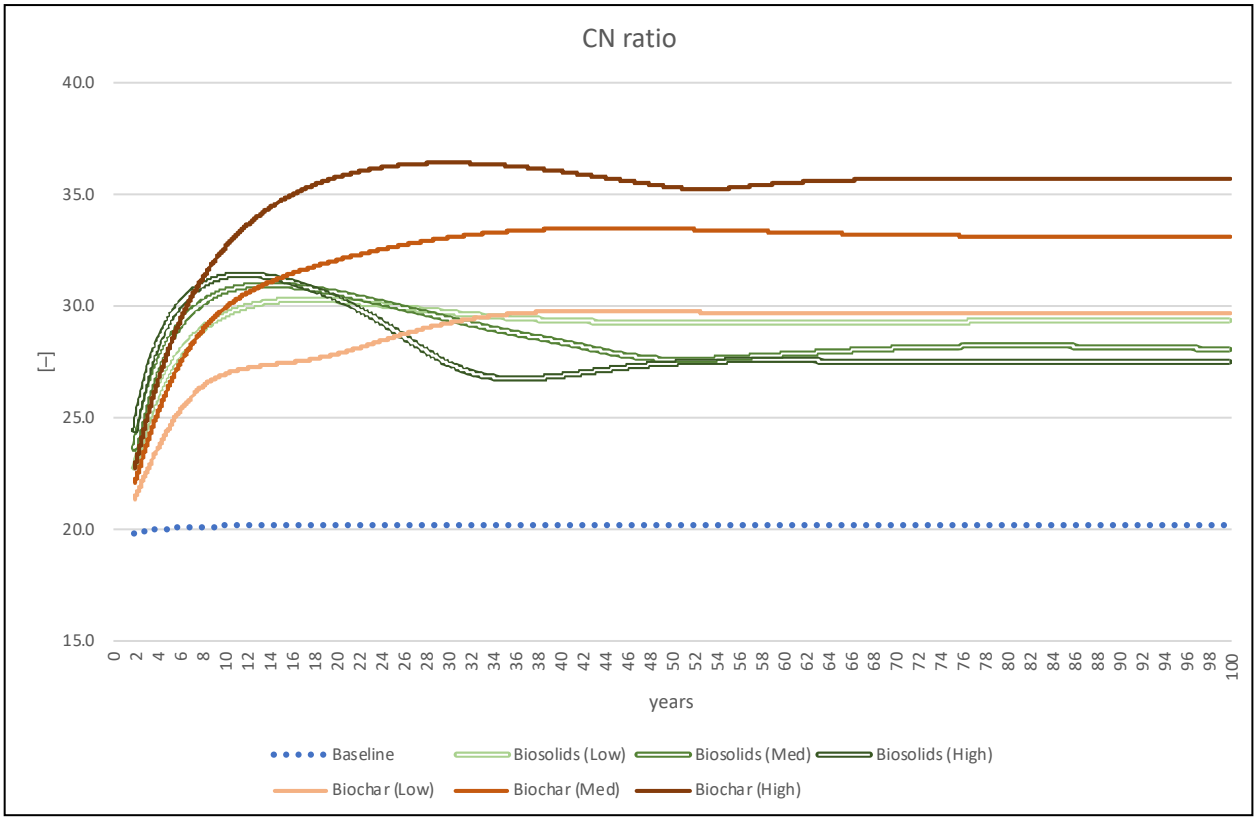

Figure 30. C/N ratio in the soil.

5.3.1. Reflection: C/N Ratio

By the end of the simulation in the baseline scenario there was an average carbon-to-nitrogen ratio of 20. This carbon-to-nitrogen ratio is within the range of 10 – 26 that commonly occurs in soils to which no amendments are applied (Batjes, 1996). In the current baseline scenario there are organic amendments, however, which suggests that there is a mechanism operating within soils which acts to balance the carbon-to-nitrogen ratio regardless of what is or is not applied to the soil.

Applying biosolids results in the carbon-to-nitrogen ratio being increased relative to the baseline scenario, with the final average carbon-to-nitrogen ratio ranging from 27 – 29, and higher biosolids application rates result in lower carbon-to-nitrogen ratios. This result agrees with previous research which has found an inverse correlation between biosolids application rates and the soil carbon-to-nitrogen ratio. For instance, (Jin et al., 2015) experimented with biosolids application and demonstrated a higher biosolids application rate led to a decreased carbon-to-nitrogen ratio. Although all biosolids scenarios result in a carbon-to-nitrogen ratio above the 'ideal' value of 24, they are all within the desired range of 20 – 30.

Applying biochar results in the carbon-to-nitrogen ratio being substantially increased relative to the baseline scenario, with the final average carbon-to-nitrogen ratio ranging from 30 – 36, and higher biochar application rates result in a higher carbon-to-nitrogen ratio (the opposite effect from biosolids). This result agrees with previous research which has found a positive correlation between biochar application rates and the soil carbon-to-nitrogen ratio. For instance, Karer, Wilmmer, Zehetner, Kloss, and Soja (2013) tested the effects of biochar application to temperate soil on nutrient dynamics and crop growth, and found higher biochar application rates resulted in higher soil carbon-to-nitrogen ratios. Only one of the biochar application rates results in the carbon-to-nitrogen ratio being narrowly inside the ideal range of 20 – 30, and the other two result in the carbon-to-nitrogen ratio being so high that it is prohibitive to crop growth.

The carbon-to-nitrogen ratio influences nitrogen turnover, and there is said to be an 'ideal' carbon-to-nitrogen ratio for optimizing nitrogen dynamics in the soil. The decay of SSOM is accompanied by mineralization and immobilization of plant available nitrogen, both of which are influenced by the carbon-to-nitrogen ratio; at a carbon-to-nitrogen ratio below 20 there is a net gain of plant available nitrogen, between 20 and 30 there is neither gain nor loss, and above 30 there is a net loss of plant available nitrogen from the soil. More precisely, the 'ideal' carbon-to-nitrogen ratio

for a microbial diet is said to be 24 (USDA Natural Resources Conservation Service, 2011). The only scenarios in which the carbon-to-nitrogen ratio falls outside the 'ideal' range is under the medium and high biochar application rates and the baseline scenario, although the carbon-to-nitrogen ratios under the lowest biosolids and biochar application rates are close to 30. Further, whereas the higher biochar application rates result in an increased carbon-to-nitrogen ratio, higher biosolids application rates result in a lowered carbon-to-nitrogen ratio.

The current results agree with previous research which has demonstrated both biosolids and biochar increase the carbon-to-nitrogen ratio, but that biochar increases it to a greater extent. Sharma et al. (2017) reviewed the agricultural utilization of biosolids and found that biosolids applied to soil increases soil organic carbon content, which would also increase the soil carbon-to-nitrogen ratio (assuming nitrogen remains constant). Laird et al. (2010) performed an experiment on biochar application to agricultural soil and found that biochar application led to higher carbon-to-nitrogen ratios in the soil than manure application. Bhatta Bhawana Kaudal et al. (2018) investigated the use of biochar as a growing medium by testing its addition to soil, examining the effects on soil quality and plant growth, and found that the biochar treatments increased the soil carbon-to-nitrogen ratio by 4% (increasing from 50 to 52) to 15% (increasing from 27 to 31) from the beginning to the end of their study.

5.4. Crop Biomass

Figure 31 illustrates the dynamics of relative plant biomass (RPB) across the three study scenarios. These results indicate that the **baseline** condition of applying compost and manure presents an average RPB of 0.16 upon reaching steady state conditions. Applying **biosolids** instead of compost and manure at a rate of 15,000 and 20,000 kg biosolids ha^{-1} yr^{-1} increases the average RPB by 0.02 and 0.03 (+13% and +15%), respectively, while an addition of 10,000 kg biosolids ha^{-1} yr^{-1} decreases the average RPB by 0.007 (–4%). Applying **biochar** instead of compost and manure at a rate of 5,000 and 10,000 kg biochar ha^{-1} yr^{-1} each decreases the average RPB by 0.16 (–100%), while an application of 15,000 kg biochar ha^{-1} yr^{-1} decreases the average RPB by 0.15 (–92%).

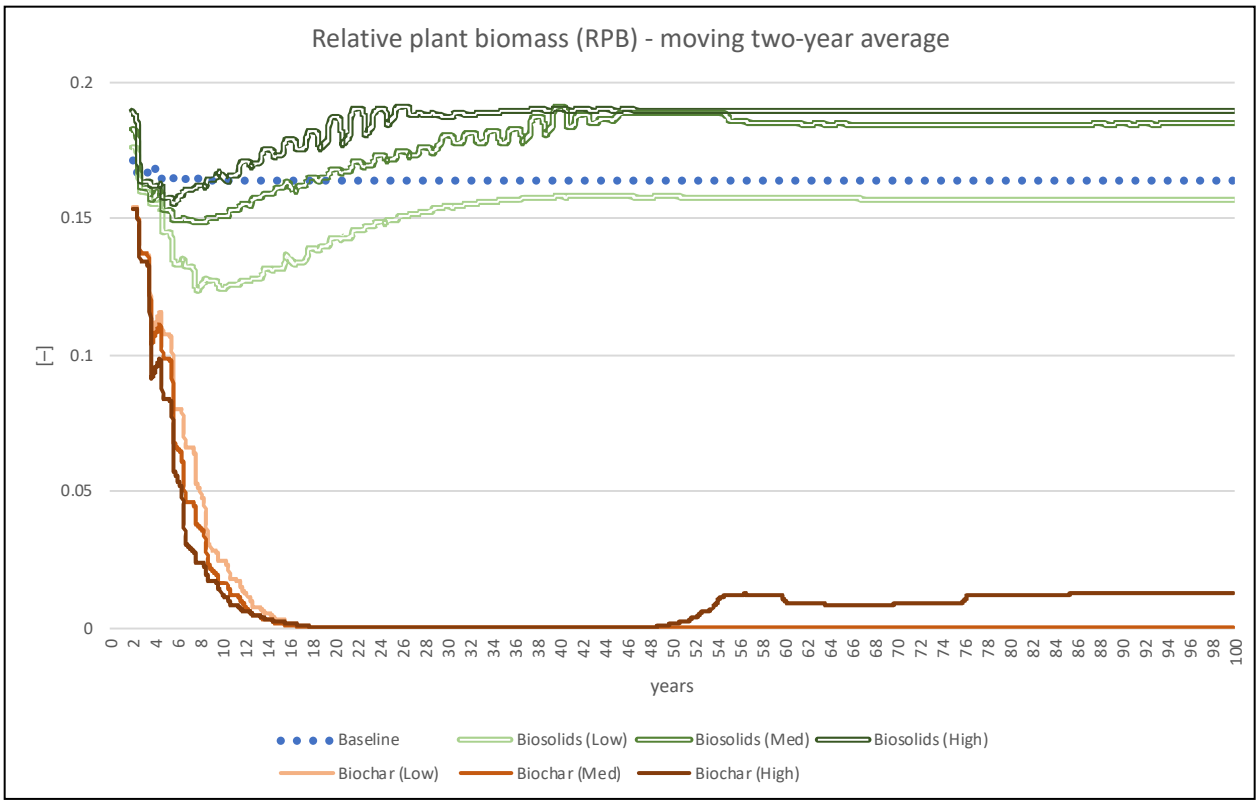

Figure 31. Relative plant biomass – moving two-year average.

Figure 32 illustrates the nitrogen effect (NE) across the three study scenarios. These results indicate that the **baseline** condition of applying compost and manure presents an NE of 0.73 upon reaching steady state conditions. Applying **biosolids** instead of compost and manure at a rate of 15,000 and 20,000 kg biosolids ha^{-1} yr^{-1} increases the NE by 0.21 and 0.23 (+29% and +31%), respectively, while an addition of 10,000 kg biosolids ha^{-1} yr^{-1} decreases NE by 0.07 (–9%). Applying **biochar** instead of compost and manure at a rate of 5,000, 10,000, and 15,000 kg biochar ha^{-1} yr^{-1} decreases NE by 0.58, 0.51, and 0.43 (–79%, –69%, and –59%), respectively.

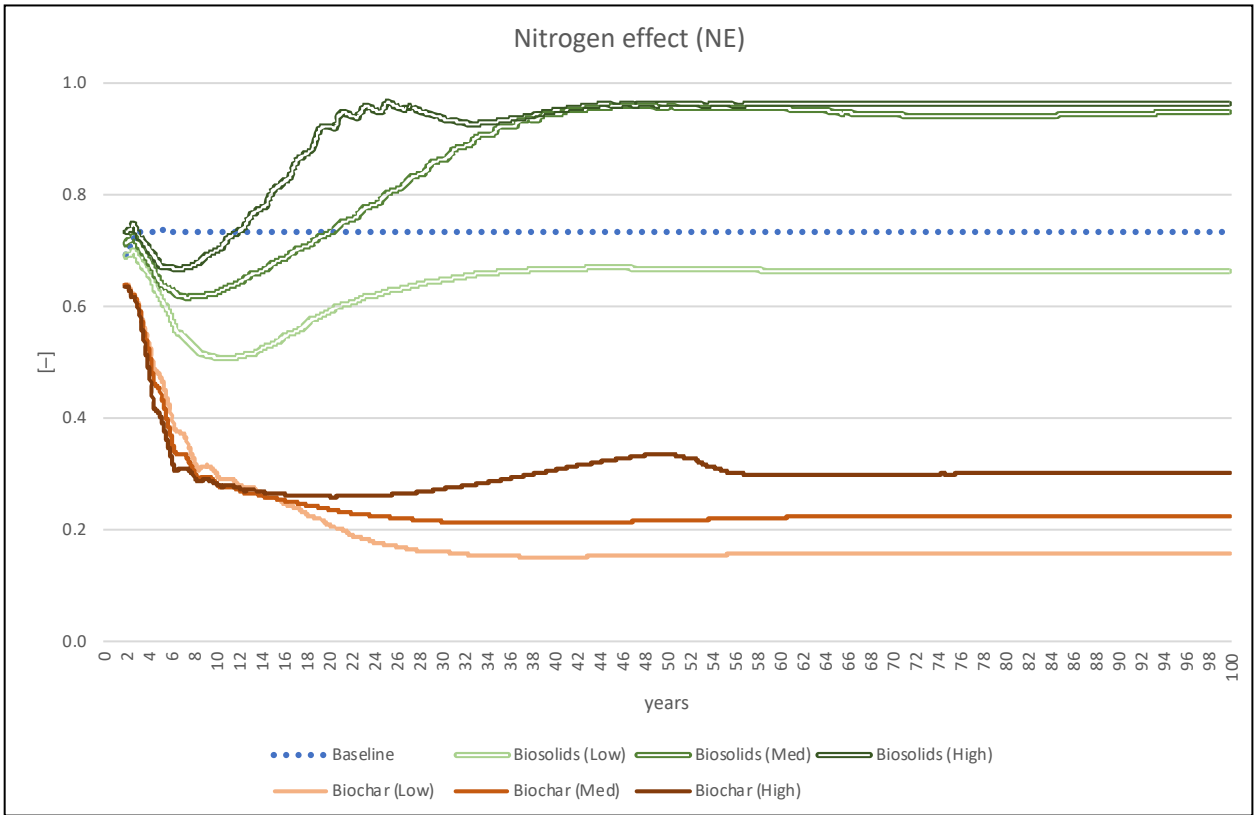

Figure 32. Nitrogen effect.

5.4.1. Reflection: Crop Biomass and Yield

By the end of the simulation in the baseline scenario there was an average plant biomass of 0.16, meaning the amount of crop biomass was at an average of 16% of its possible maximum. Determining whether this plant biomass value is in line with previous research is not straightforward, as this parameter is not commonly used in other studies. The value of plant biomass represents the amount of crop biomass relative to the maximum possible amount, and although a specific maximum value has been chosen for crops in the thesis model there is wide variation in reality and therefore also a wide range of possible maximum crop biomass values, which makes standardized measurement difficult.

Applying biosolids results in plant biomass being generally increased relative to the baseline scenario, with final average values ranging from 0.15 to 0.19. These results are in line with previous research which has found biosolids application can increase crop yield. For example, Sharma et al. (2017) reviewed the agricultural utilization of biosolids and found that its application consistently boosts plant growth. The lowest biosolids application rate in this study resulted in

lower plant biomass than the baseline, but this makes sense as the amount of organic amendment applied (10,000 kg biosolids ha^{-1}) at the lowest application rate is lower than the total amount of organic amendment applied in the baseline scenario (12,500 kg combined of compost and manure ha^{-1}).

Applying biochar results in plant biomass being substantially decreased relative to the baseline scenario, with final average values ranging from 0 to 0.1. These results are in line with previous research which has demonstrated biochar application decreases crop yield. For example, Karer et al. (2013) studied biochar application to two temperate soils and found that biochar application resulted in decreases of crop yield ranging from 37% to 72% by one year after the application. Based on the one-time application rates tested by Karer et al. (2013) (24,000 and 72,000 kg biochar ha^{-1}), in the Thesis Model it would take only 1.6 years (i.e. 24,000 kg ha^{-1} total, 15,000 kg ha^{-1} yr^{-1} applied) to 14.4 years (i.e. 72,000 kg ha^{-1} total, 5,000 kg ha^{-1} yr^{-1} applied) of annual biochar application to have applied the same amount in total. The current results on the effect of the application of biochar alone are therefore likely in agreement with those of Karer et al. (2013), demonstrating that plant biomass decreases by about 50% by around 8 years into the simulation.

In general, applying biosolids results in a greater crop yield than the baseline scenario, with only the lowest biosolids application rate resulting in less plant biomass than the baseline. Crop growth rate is influenced by the nitrogen effect, which is a function of the relative availability of plant available nitrogen in the soil, so if plant available nitrogen is either too scarce or too plentiful the crop grows less than optimally. Applying biosolids at the highest rate results in the greatest nitrogen effect because the availability of plant available nitrogen in the soil is closest to the optimal value of 1. In contrast, the biochar scenarios resulted in the nitrogen effect to range from around 0.15 to 0.3, which results in little crop growth. These results agree with previous research which suggests both biosolids and biochar can have a positive effect on crop yield, but the effect of biosolids tends to be more positive. Hussain et al. (2017) reviewed studies of biochar used for crop production and its potential risks and benefits, and demonstrated biochar tends to increase crop productivity and this increase is generally in the range of 30–40% (own approximation; reported values ranged from –30% to +294%). However, the studies reviewed by Hussain et al. (2017) were typically short-term experiments of single biochar applications and several of those also included application of mineral fertilizer in addition to biochar. It is therefore not surprising that this study finds little positive influence of repeatedly adding biochar alone to the soil for a

long time period. Filiberto and Gaunt (2013) performed a review of the influence of biochar application on soil productivity in which they identified several studies which found a negative influence of biochar additions on crop productivity; the negative influence occurred at high one-time application rates upwards of 100,000 kg biochar ha^{-1}, which agrees with the current results that after several years of 5,000 – 15,000 kg biochar ha^{-1} applied, a negative influence on OM decomposition occurs. McHenry (2011) reviewed studies on biochar and agricultural yield results and identified several that demonstrated biochar addition alone can lead to negative influences on crop growth and yield. Hossain, Strezov, Chan, and Nelson (2010) analyzed the agronomic properties of wastewater biochar and the effects of its addition to soil on soil quality and plant growth, finding that the biochar treatment increased yield over the control plot, however did not improve yield as much as the fertilizer alone, which agrees with the current findings that biosolids (an organic rich fertilizer) improves yield more than biochar. It is important to note that the control plot in the study by Hossain et al. (2010) differed significantly from my Baseline scenario, in that my baseline included compost and manure additions whereas their control plot did not include any soil amendments.

On the other hand, biosolids have generally been shown to have a more positive impact on crop productivity. For instance, Sharma et al. (2017) reviewed the agricultural use of biosolids and its potential effects on soil and plant growth, and demonstrated that biosolids tended to increase crop productivity and this increase was generally in the range of 50% – 70% (own approximation, reported values range from –22% to +137%). In a field experiment of the effects of biochar and biosolids land application on tree growth, Scharenbroch et al. (2013) found the biosolids treatment performed better than biochar, leading to greater total plant biomass. In contrast to the results of this study, however, Downie (2011) found that biochar addition to soil increased crop yield; but their study included mineral fertilizer application in addition to biochar, the study was of a relatively short-term experiment of three years, and the control plot had no soil amendment applied to it. The current results showing biochar application results in decreased crop yield in the early years of the simulation may still therefore agree with the findings of Downie (2011), because even though plant biomass is decreasing it takes about 10 years for it to bottom out – so, if this study had tested a ‘no amendment’ scenario, these biochar scenarios would likely perform better than it.

5.5. Endocrine Disrupting Compounds

Figure 33 and Figure 34 illustrate the accumulation of Triclosan (TCS) across the three study scenarios. Only one year is shown in Figure 33 because the pattern is the same for each year, and Figure 34 presents a moving annual average (beginning after 100 time-steps, or one year). These results indicate that the **baseline** condition of applying compost and manure presents a TCS accumulation of 0 kg TCS ha^{-1}. Applying **biosolids** (scenario 2) instead of compost and manure at a rate of 10,000, 15,000 and 20,000 kg biosolids ha^{-1} yr^{-1} cause TCS to reach an average of 0.3, 0.4, and 0.6 kg ha^{-1}, respectively (with the maximum values being 8, 12 and 16 kg ha^{-1}, respectively), by the end of the model period. In the biosolids scenarios, TCS spikes when biosolids are applied at t = 0.18 (the 65th day) in the year and is completely gone from the soil by 113 to 128 days (0.31 to 0.35 of a year) after biosolids are applied each year. Applying **biochar** (scenario 3) instead of compost and manure at a rate of 5,000, 10,000, and 15,000 kg biochar ha^{-1} yr^{-1} each result in a TCS accumulation of 0 kg TCS ha^{-1}.

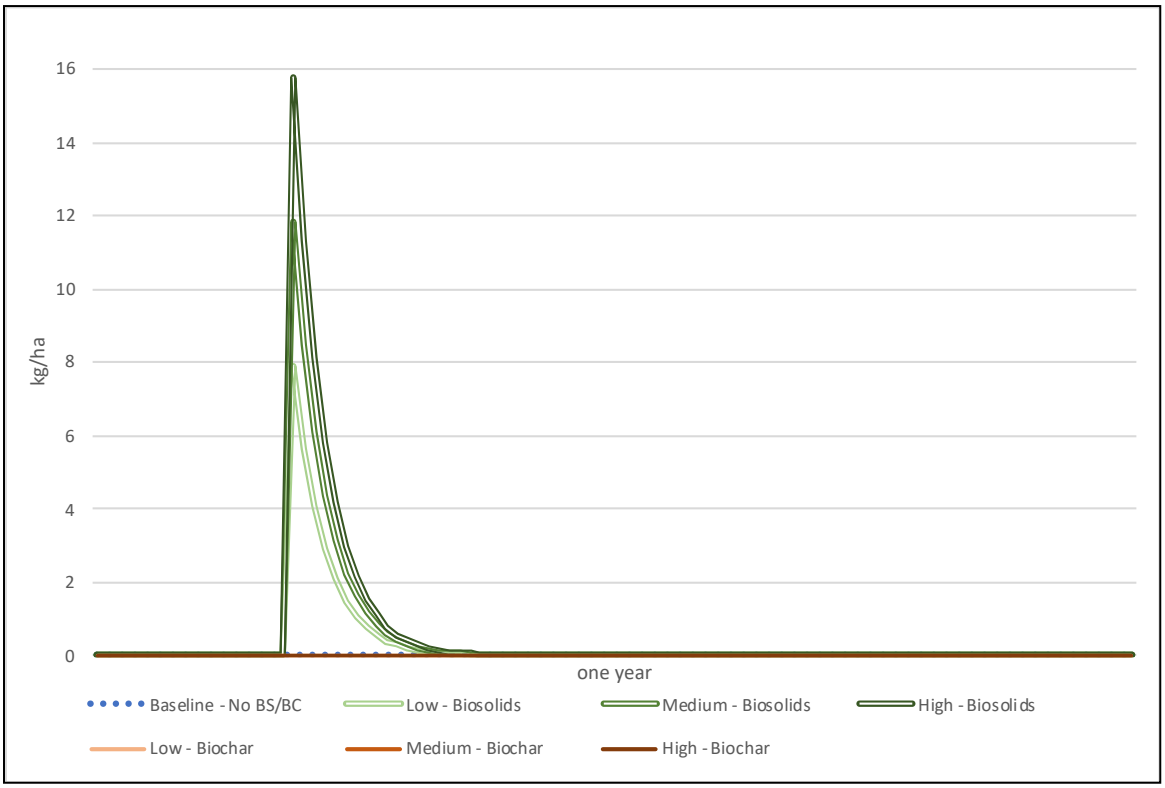

Figure 33. Triclosan in the soil over the final year in the simulation.

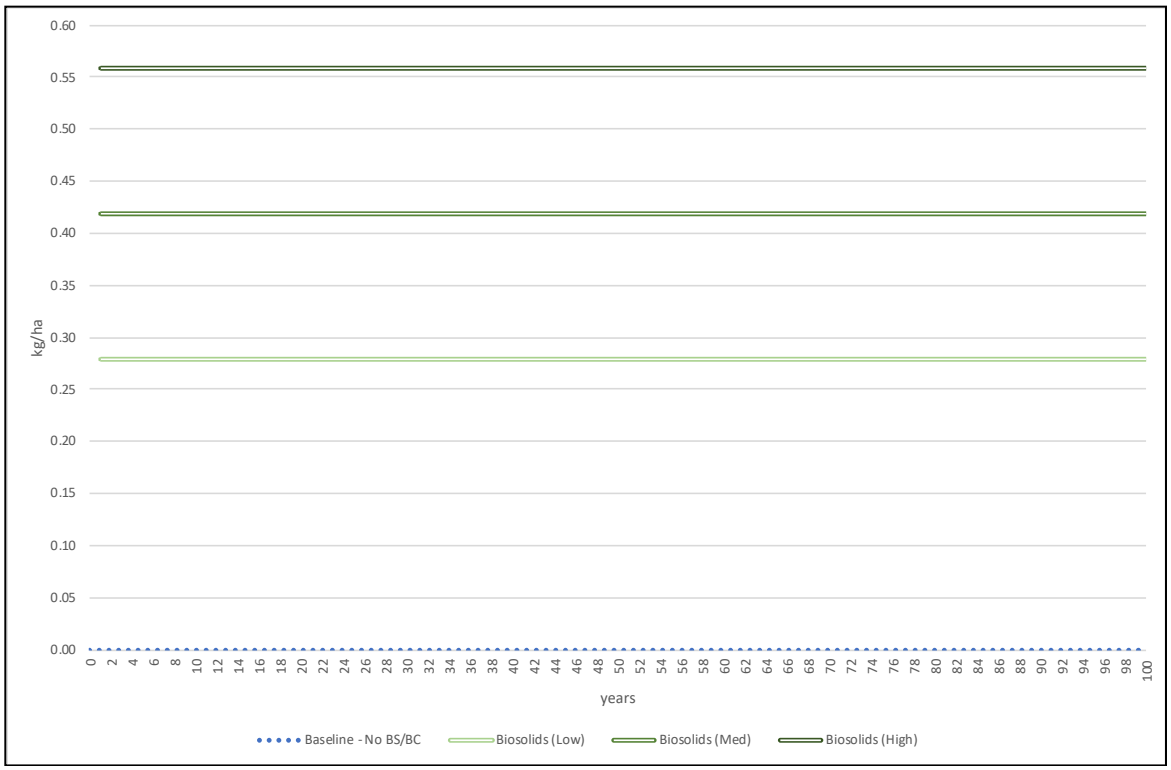

Figure 34. Triclosan in the soil – moving annual average.

Figure 35 and Figure 36 illustrate the accumulation of Triclocarban (TCC) across the three study scenarios. Only one year is shown in Figure 35 because the pattern is the same for each year, and Figure 36 presents a moving annual average (beginning after 100 time-steps, i.e. one year). These results indicate that the baseline condition of applying compost and manure presents a TCC accumulation of 0 kg TCC ha^{-1}. Applying biosolids instead of compost and manure at a rate of 10,000, 15,000 and 20,000 kg biosolids ha^{-1} yr^{-1} cause TCC to reach an average of 1.0, 1.5, and 2.0 kg ha^{-1}, respectively (with the maximum values being 2.7, 4.1 and 5.5 kg ha^{-1}, respectively), by the end of the model period. In the biosolids scenarios, TCC spikes when biosolids are applied at t = 0.18 (the 65th day) in the year and is never completely gone from the soil (only reaches as low as 93% of the maximum value). Applying biochar instead of compost and manure at a rate of 5,000, 10,000, or 15,000 kg biochar ha^{-1} yr^{-1} results in a TCC accumulation of 0 kg TCC ha^{-1}.

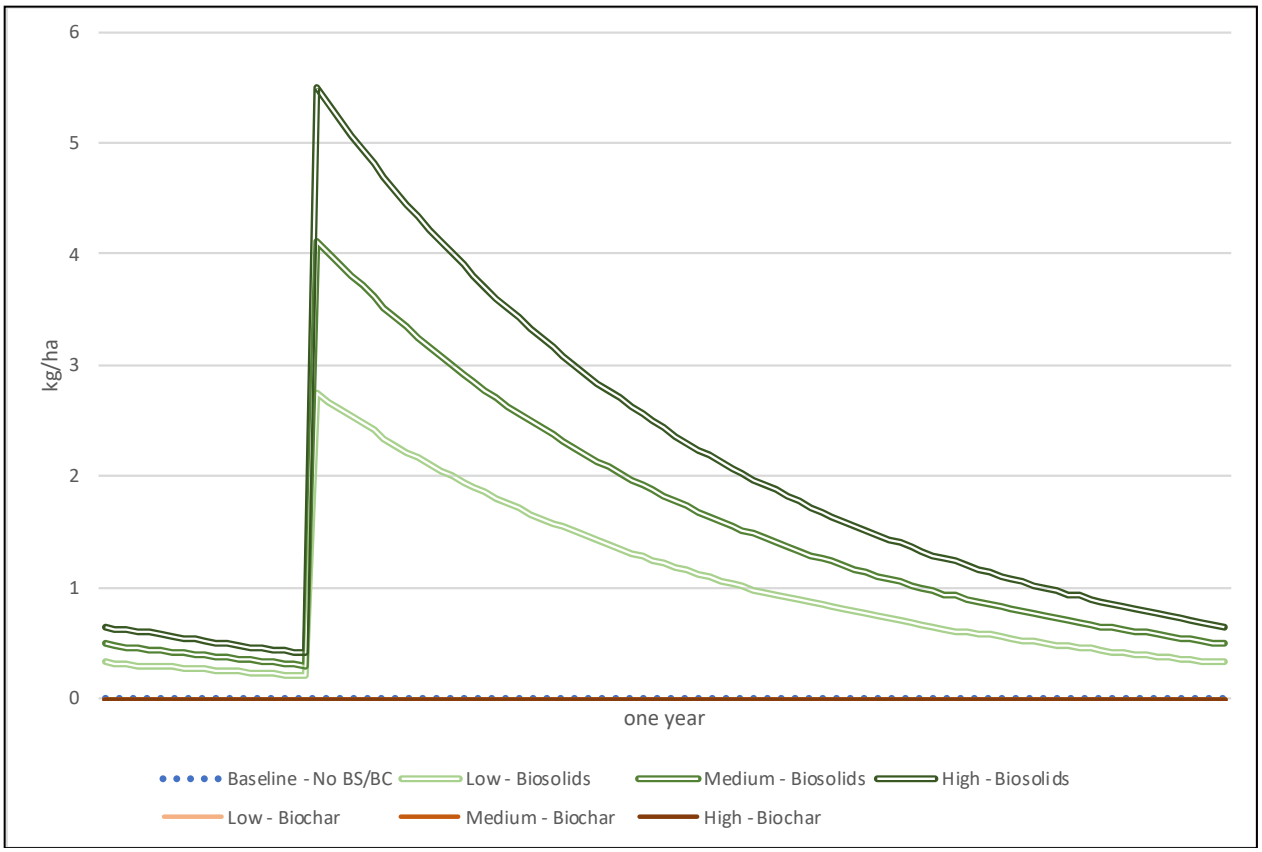

Figure 35. Triclocarban in the soil over the final year in the simulation.

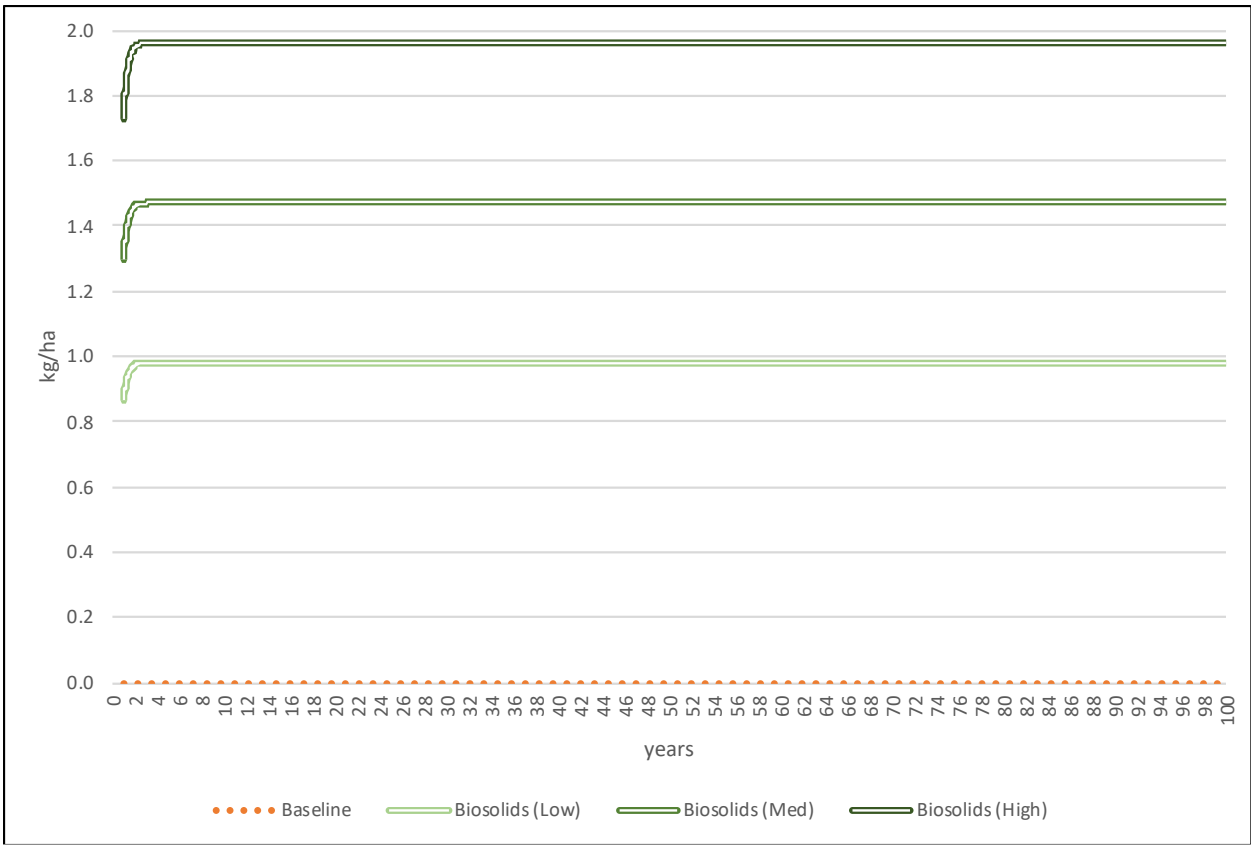

Figure 36. Triclocarban in the soil – moving annual average.

5.5.1. Reflection: Endocrine Disrupting Compounds

Applying compost and manure or biochar results in Triclosan remaining constant at 0. This makes sense because there is assumed to be no Triclosan present in all three of those amendments. On the other hand, applying biosolids results in annual Triclosan spikes followed by exponential decreases to 0. As a result of the rapid microbial and photo-degradation rates of Triclosan, the compound is nearly gone from the soil during the crop growth period. For instance, during the time between the start of crop growth and its harvest, Triclosan reaches 5% of its maximum value on the 29th day after biosolids application and reaches 0% on the 55th day after biosolids application. This result agrees with previous research on soil Triclosan dynamics after biosolids application. Fuchsman et al. (2010) performed a review of terrestrial risk evaluation of Triclosan in land-applied biosolids and found that Triclosan was fully degraded after one third of a year with annual biosolids applications when the half-life based on biodegradation was 2 weeks. The results of this study agree with those of Fuchsman et al. (2010) likely because the half-lives for Triclosan used in this study (17 and 19 days for photodegradation and microbial degradation, respectively)

are also in the range of two weeks, and furthermore, the dynamics of Triclosan displayed in the model run outputs closely mirror those found by Fuchsman et al. (2010).

Similar to with Triclosan, in the biochar and baseline scenarios Triclocarban is constant at 0. This makes sense for the same reason as above. On the other hand, applying biosolids results in annual spikes in Triclocarban followed by exponential decreases, but not to 0 (as in the case of Triclosan). As a result of the relatively slower degradation rate of Triclocarban, this compound is still present in the soil during crop growth. During the time between crop growth beginning and harvest time, Triclocarban decreases from 79% of its maximum value to 37% of its maximum. This result agrees with previous research on soil Triclocarban dynamics. For instance, several studies have demonstrated that Triclocarban seems to persist for a long time relative to Triclosan in the soil (resulting in bioaccumulation in earthworms) (Gbomina Jr., 2015; Higgins, Paesani, Chalew, Halden, & Hundal, 2011; Wu et al., 2009). Because Triclocarban persists in the soil year-over-year, its average value increases slowly over time, and this is exemplified by the slopes of the trendlines: under the Low, Medium and High biosolids application rates, the trendlines for Triclocarban have slopes of 2.6×10^{-8}, 3.8×10^{-8}, and 5.5×10^{-8}, respectively.

Overall, these results suggest that in the scenario where EDCs are present (i.e. the biosolids scenario), Triclosan levels may not pose a risk due to how rapidly Triclosan is degraded and removed from the soil, and Triclocarban is persistent in the soil during the crop growth period and accumulates slowly over time. However, despite its persistence in the soil, the Triclocarban exposure to consumers is likely still low in comparison to typical daily exposure rates; for example, research has found that human exposure to Triclocarban from consumption of pumpkin or zucchini grown directly on biosolids-amended soil was far less than exposure from product use (but greater than exposure from drinking tap water) (Gbomina Jr., 2015). This relatively low exposure from eating vegetables grown on biosolids-amended soil could be explained by the fact that these two EDCs are used in household products that consumers put directly on and in their bodies such as shampoo, soap, deodorant, cosmetics, lotions and creams, mouth rinses, and toothpaste (Clarke & Smith, 2011).

5.6. Additional Investigation: Biochar Application Frequencies

Over the course of the experiment, it became clear that applying biochar on an annual basis would not lead to a viable agricultural system due to the excess carbon and high C/N ratio that restricted

crop growth. This led to the 'what if' question of what would happen to the system if biochar was not applied every year, but instead every five or ten years. The Biochar scenario was modified to create two new scenarios to which the original Biochar scenario could be compared: one in which biochar was applied at an interval of five years and one in which it was applied every ten years. The following section presents some of the noteworthy changes to key stocks and parameters observed when the application frequency is reduced and discusses general implications for the interpretation of the findings of this study.

Figure 37 shows the C/N ratio in all scenarios tested, including the explorative tests of 5 and 10-year biochar application frequencies (showing the average of the final 20 years of the simulation). The result of applying biochar once every 5 and 10 years is the lowering of the C/N ratio to within the desirable range of 20 – 30. Figure 38 shows the moving two-year average of the C/N ratio over the simulation period for the three biochar scenarios (annual, 5-year, and 10-year application frequencies) under the medium application rate (10,000 kg biochar ha^{-1}). While Figure 37 shows the average C/N ratio in the medium application rate is less in the 5-year application frequency scenario and even less in the 10-year scenario, Figure 38 shows that there are higher spikes of the C/N ratio under less frequent application.

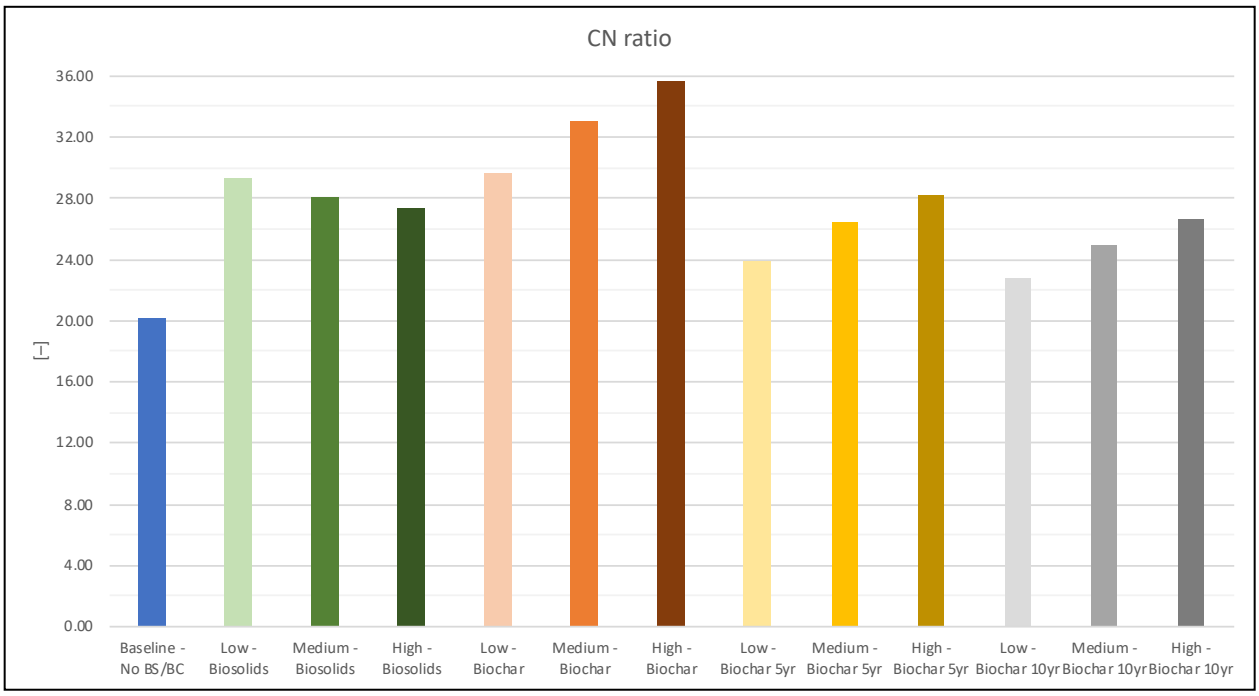

Figure 37. Final carbon-to-nitrogen ratio for all scenarios tested (baseline, biosolids, and biochar), including 5 and 10-year biochar application frequencies; average of the final 20 years.

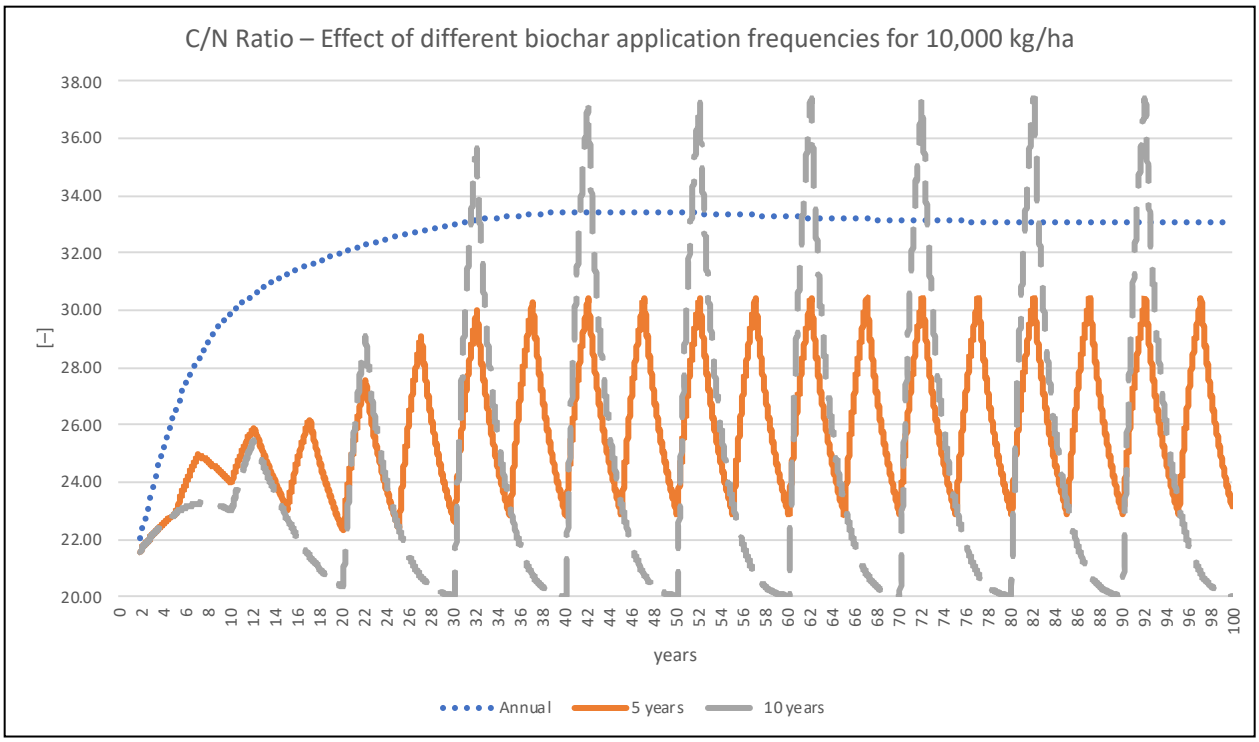

Figure 38. Carbon-to-nitrogen ratio under decreased frequency of biochar application: the effect of 5 and 10-year applications of 10,000 kg/ha versus annual applications.

Figure 39 shows the nitrogen effect (NE) in all scenarios tested, including the explorative tests of 5-year and 10-year biochar application frequencies (showing the average of the final 20 years of the simulation). The result of applying biochar once every 5 and 10 years is a substantial decrease of the NE. Figure 40 shows the moving two-year average of the NE over the simulation period for the three biochar scenarios (annual, 5-year, and 10-year application frequency) under the medium application rate (10,000 kg biochar ha-1). Together, Figure 39 and Figure 40 show that the average NE is lower in the 5-year biochar application frequency scenario and even less in the 10-year scenario, relative to annual application.

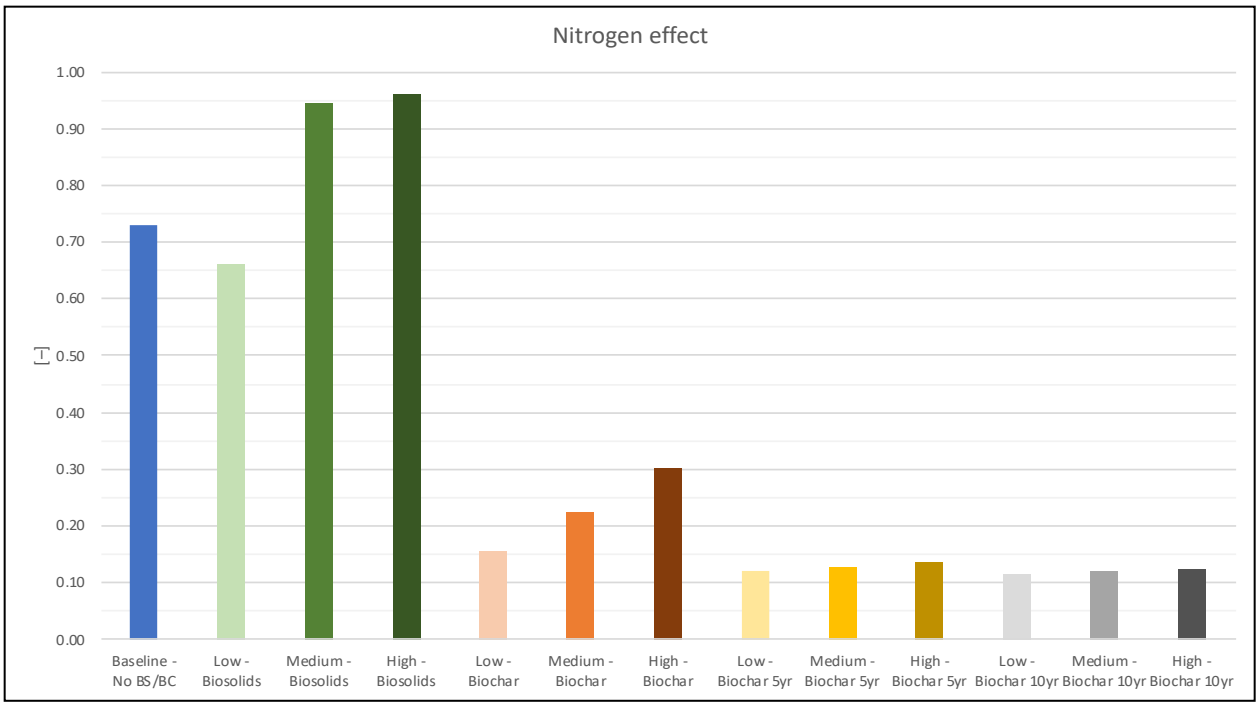

Figure 39. Nitrogen effect for all scenarios tested, including 5 and 10-year biochar application frequencies; average of the final 20 years.

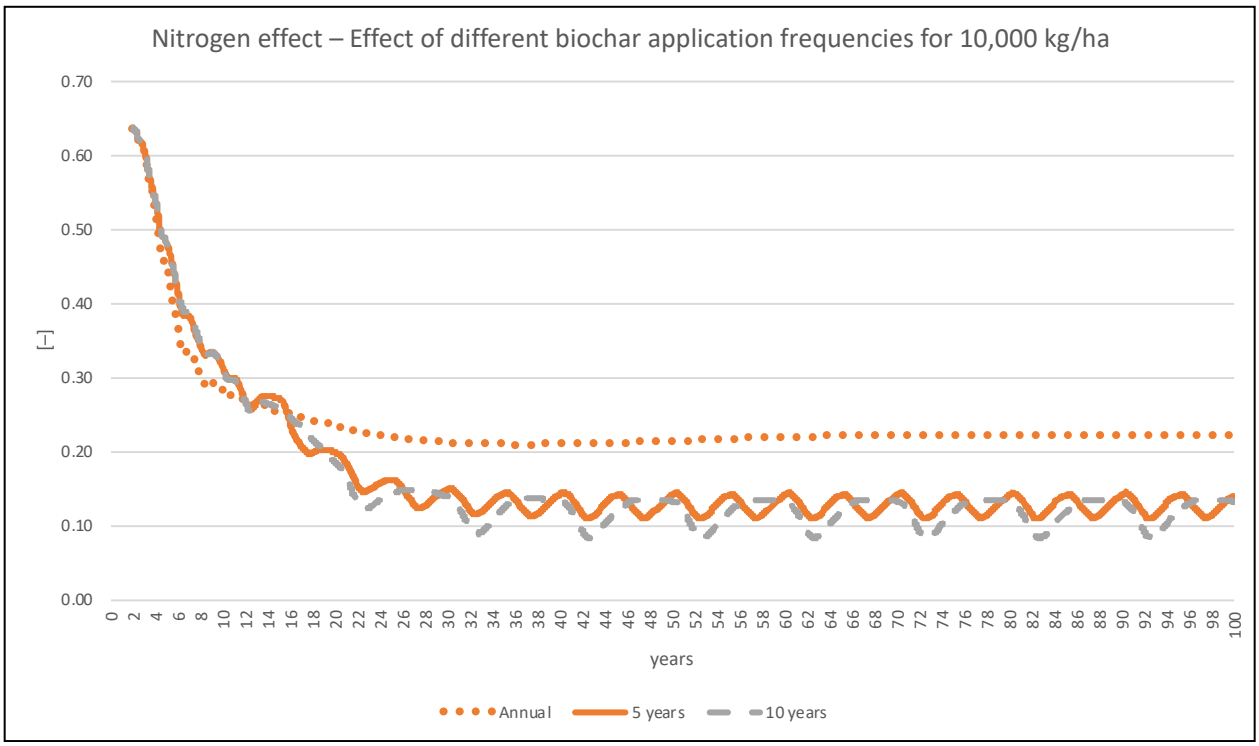

Figure 40. Nitrogen effect under decreased frequency of biochar application: the effect of 5 and 10-year applications of 10,000 kg/ha versus annual applications.

Figure 41 shows the amount of PAN in all scenarios tested, including the explorative tests of 5-year and 10-year biochar application frequencies (showing the average of the final 20 years of the simulation). The result of applying biochar once every 5 and 10 years is the further lowering of PAN to within the range of 20 – 31 kg PAN ha^{-1} – all of which are lower than in the annual biochar application scenario with the lowest application amount tested.

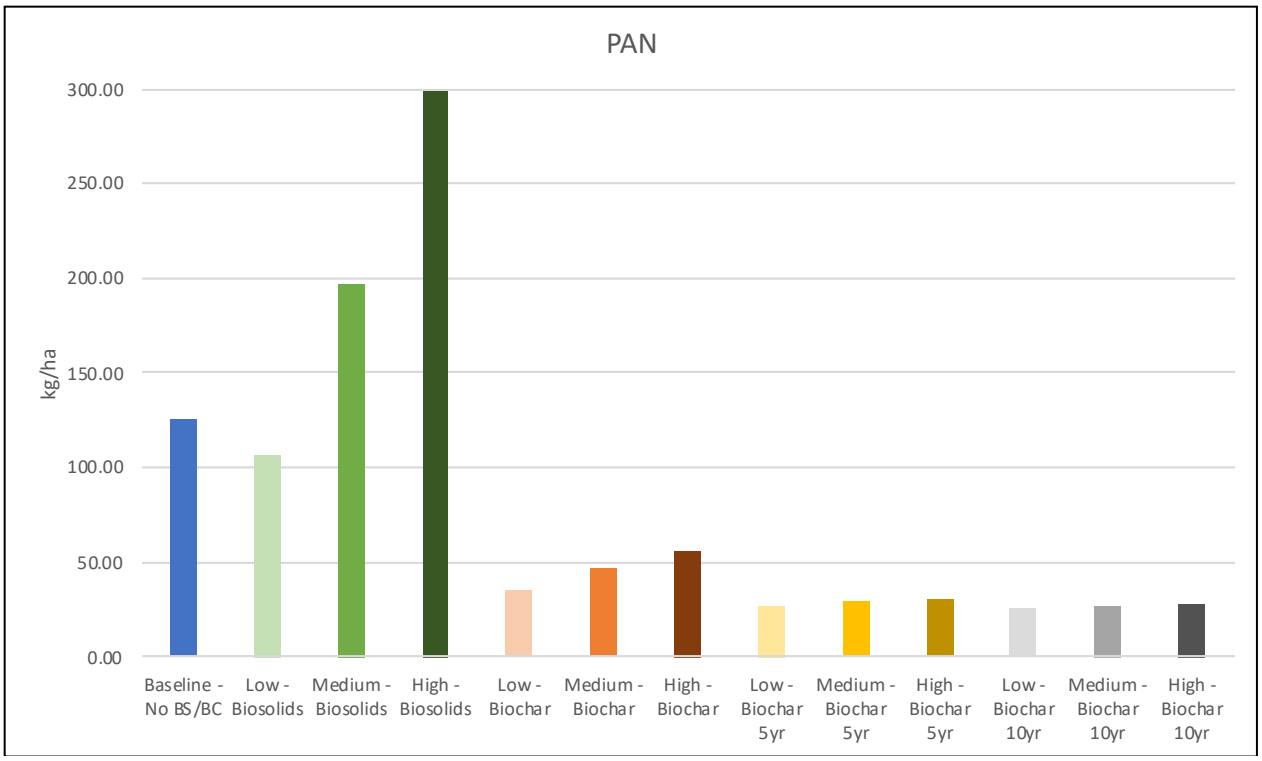

Figure 41. PAN for all scenarios tested, including 5 and 10-year biochar application frequencies; average of the final 20 years.

In summary, decreasing the application frequency does appear to improve the C/N ratio (i.e. reduce the C/N ratio to within the ideal range of 20 – 30) but does not improve plant growth, as is evidenced by the lower NE and lower levels of PAN in the 5-year and 10-year scenarios. This demonstrates that simply decreasing the application frequency is not enough to make the agricultural system viable for crop growth, due to a lack of nitrogen for crop growth. A brief investigation on the effects of adding mineral fertilizer in addition to biochar was undertaken (presented in the following section) however further detailed investigation was outside the scope of this study. The current results suggest that repeatedly applying biochar alone is unlikely to create a viable agricultural system in the long-term, regardless of the application frequency.

5.7. Additional Investigation: Mineral Fertilizer and Biochar

After investigating the biochar scenario and additionally looking at the effect of different application frequencies, it was clear that this biochar-only scenario led to there being excess carbon in the soil, a high C/N ratio, and subsequent detrimental effects on crop growth. This led me to investigate another biochar scenario, in which mineral fertilizer was applied in addition to using the 5-year and 10-year application frequencies used in the above investigation (Table 15). The following briefly describes the differences between these scenarios and the baseline and compare the results with those of the biochar scenario.

Table 15. Scenario D.

	Scenario			
Parameter	**A**	**B**	**C**	**D**
Compost_and_manure_on_or_off	1	0	0	0
Biosolids_production_on_or_off	0	1	1	1
Biosolids_land_application_on_or_off	0	1	0	0
Biochar_land_application_on_or_off	0	0	1	1
Mineral_fertilizer_on_or_off	0	0	0	1

5.7.1. Carbon

The application of mineral fertilizer in addition to biochar results in the stock of C-SSOM, as well as DCOM, being shifted in the positive direction, relative to the biochar-only scenarios. Applying mineral fertilizer and biochar instead of compost and manure at a rate of 10,000 and 15,000 kg biochar ha^{-1} yr^{-1} increases the accumulation of C-SSOM by 2550 and 5450 kg C-SSOM ha^{-1} (+54% and +115%), respectively, while an addition of 5,000 kg biochar ha^{-1} yr^{-1} decreases C-SSOM by 340 kg C-SSOM ha^{-1} (–7%) – i.e. there was **19% to 56% more stable carbon** than in the biochar-only scenarios. Applying mineral fertilizer and biochar instead of compost and manure at a rate of 5,000, 10,000, and 15,000 kg biochar ha^{-1} yr^{-1}decreases the rate of DCOM by 1870, 1140, and 410 kg DCOM ha^{-1} yr^{-1} (–49%, –30% and –11%), respectively – i.e. there was **60% to 185% more decomposition** than in the biochar-only scenarios,

The application of mineral fertilizer in addition to biochar results in the accumulation of C-FSOM being shifted in the positive direction in general, and the accumulation of CCR being substantially increased, relative to the biochar-only scenario. Applying mineral fertilizer and biochar instead of compost and manure at a rate of 10,000 and 15,000 kg biochar ha^{-1} yr^{-1} increases the accumulation

of C-FSOM by 1300 and 11500 kg C-FSOM ha^{-1} (+7% and +60%), respectively, while an addition of 5,000 kg biochar ha^{-1} yr^{-1} decreases C-FSOM by 6800 kg C-FSOM ha^{-1} (–35%) – i.e. there was **94% more to 1.4% less labile carbon** than in the biochar-only scenarios. Applying mineral fertilizer and biochar instead of compost and manure at a rate of 5,000, 10,000, and 15,000 kg biochar ha^{-1} yr^{-1} decreases the rate of CCR by 278, 234, and 182 kg CCR ha^{-1} yr^{-1} (–18%, –15% and –12%), respectively (**far greater crop residue carbon input** than in the biochar-only scenarios).

5.7.2. Nitrogen

The application of mineral fertilizer in addition to biochar results in the stock of PAN being up to three times that of the biochar-alone scenario, as well as NL being increased and the variation in NL decreasing (e.g., decreases in NL range from –65% to –75% in the biochar-only scenario, and –12% to –14% in the combined biochar and fertilizer scenarios). Applying mineral fertilizer and biochar instead of compost and manure at a rate of 5,000, 10,000, and 15,000 kg biochar ha^{-1} yr^{-1} decreases the accumulation of PAN by 21, 13, and 4 kg PAN ha^{-1} (–17%, –10%, and –3%), respectively – i.e. there was **120% to 200% more plant available nitrogen** than in the biochar-only scenarios. Applying mineral fertilizer and biochar instead of compost and manure at a rate of 5,000, 10,000, and 15,000 kg biochar ha^{-1} yr^{-1} decreases the flow of NL by 14, 123, and 15 kg NL ha^{-1} yr^{-1} (–13%, –12%, and –14%), respectively – i.e. there was **115% to 190% more nitrogen leached** than in the biochar-only scenarios.

The application of mineral fertilizer in addition to biochar results in the average stock of N-FSOM being shifted in the positive direction, relative to the application of biochar alone. Applying mineral fertilizer and biochar instead of compost and manure at a rate of 5,000 and 10,000 kg biochar ha^{-1} yr^{-1} decreases the accumulation of N-FSOM by 480 and 220 kg N-FSOM ha^{-1} (–50% and –23%), respectively, while an addition of 15,000 kg biochar ha^{-1} yr^{-1} increases the accumulation of N-FSOM by 93 kg N-FSOM ha^{-1} (+10%) – i.e. there was **20% to 120% more organic nitrogen** than in the biochar-only scenarios.

5.7.3. C/N Ratio

The application of mineral fertilizer in addition to biochar results in the C/N ratio being shifted in the negative direction, relative to the application of biochar alone; this shift is desirable as it brings the C/N ratios under the medium and high biochar application rates within the range of 20 – 30 (in

contrast, only one of the biochar scenarios results in the C/N ratio being within that range). Applying mineral fertilizer and biochar instead of compost and manure at a rate of 10,000 and 15,000 kg biochar ha^{-1} yr^{-1} increases the C/N ratio to 26, 28, and 29 (+30%, +39%, and +46%), respectively – i.e. the **carbon-to-nitrogen ratio was 12% to 18% lower** than in the biochar-only scenarios.

5.7.4. Crop Biomass and Yield

The application of mineral fertilizer in addition to biochar results in the stocks of RPB and RCY, as well as the average NE, being substantially shifted in the positive direction, relative to the application of biochar alone. Applying mineral fertilizer and biochar instead of compost and manure at a rate of 5,000, 10,000, and 15,000 kg biochar ha^{-1} yr^{-1} decreases the accumulation of RPB by 0.04, 0.03, and 0.02 (–24%, –18%, and –12%), respectively (**far greater relative plant biomass** than in the biochar-only scenarios. Applying mineral fertilizer and biochar instead of compost and manure at a rate of 5,000, 10,000, and 15,000 kg biochar ha^{-1} yr^{-1} decreases the final RCY by 21, 18, and 16 (–27%, –24%, and –20%), respectively – i.e. the **total amount of crop harvested over the model period was 920% to 1120% greater** than in the biochar-only scenario. Applying mineral fertilizer and biochar (scenario 4) instead of compost and manure at a rate of 5,000, 10,000, and 15,000 kg biochar ha^{-1} yr^{-1} decreases NE by 0.1, 0.07, and 0.04 (–15%, –10%, and –5%), respectively – i.e. the **average nitrogen effect was 130% to 300% greater** than in the biochar-only scenarios. This demonstrates that biochar may be effective at improving crop yield if mineral fertilizer is added as well.

5.7.5. Endocrine Disrupting Compounds

There is no difference in the stocks of TCS and TCC when mineral fertilizer is added in addition to biochar. Applying mineral fertilizer and biochar instead of compost and manure at a rate of 5,000, 10,000, and 15,000 kg biochar ha^{-1} yr^{-1} each result in TCS and TCC accumulations of 0 kg TCS ha^{-1}, which is the same as in scenario carbon.

6. Conclusions

6.1. Summary

Overall, the results of this study suggest that biosolids application leads to better soil quality and plant growth than biochar application. Applying biosolids results in more soil organic and plant available nitrogen, a closer to optimal C/N ratio, and greater crop yields. Among the biosolids scenarios, the highest application rate – 20,000 kg biosolids ha^{-1} yr^{-1} – results in the highest crop biomass, though crop biomass under the medium biosolids application rate is just slightly lower. Although applying biochar alone does not result in a thriving or viable agricultural system, a brief investigation into applying biochar and mineral fertilizer suggests that applying both may significantly improve the state of the agricultural system relative to applying biochar alone.

Despite biochar containing more stable carbon than biosolids, applying biochar alone does not increase the stock of stable carbon in the soil substantially given the co-dependencies of the two carbon stocks (labile and stable) on one another. Whereas biochar application increases the C/N ratio to such a degree that crop growth is limited, biosolids application creates a balanced soil composition with which the crop can thrive. Relative to the baseline scenario of compost and manure application, which leads to the C/N ratio being on the low end of the range (of 20 – 30)

Biosolids application results in more plant available and organic nitrogen than biochar application, with the trade-off of greater nitrogen leaching. In terms of crop growth, the medium and high biosolids application rates perform nearly the same with the high application rate performing noticeably better than the medium rate. Interestingly, the nitrogen effect under the medium and high biosolids application rates is about the same as with the high application rate –considering this, in light of the above observations demonstrates that the relative nitrogen availability in the soil is not the only factor contributing to the performance of a given scenario.

Additional investigations into the effects of changing the frequency of biochar application to once every five and ten years, as well as the effect of adding nitrogen fertilizer in addition to annual biochar application, revealed that adding fertilizer is integral for the creation of a viable agricultural system that utilizes biochar as a soil amendment. While there was some positive effect of decreasing the biochar application frequency, the positive changes were far less pronounced than when fertilizer was simply added in addition to biochar. The results of this investigation

suggest that if the goal is to apply biochar, one should only apply it on an annual basis if nitrogen fertilizer is also added.

6.2. Developments in the Case Study Region

At the outset of this research, the CRD was in the process of determining a short-term use for its biosolids. The CRD has since determined what the short-term use will be: biosolids will be provided to cement manufacturers in the Lower Mainland where the biosolids will be used as a renewable fuel source and the ash from the incineration process is to be incorporated into the cement product.

Nonetheless, the lessons learned from this thesis may be applied to the CRD case in the long-term, or to other contexts. Despite the fact that the CRD has determined the short-term use for its biosolids, the regional government has yet to determine what the long-term use will be. Furthermore, other jurisdictions are likely to face the same question of how to use their biosolids now and in the near future, so this comparison of two biosolids use pathways could assist decision-makers who are considering land application of biosolids or biosolids-derived biochar.

6.3. Limitations & Extensions

6.3.1. Limitations

To facilitate model development and limit the scope of the study, assumptions were made that should be further explored. These assumptions have influence in most sectors of the model and include: specific assumptions such as the crop rotation scheme being a two-year rotation of corn and alfalfa (to base the model on a realistic agricultural practice); more general assumptions such as the lack of feedback or connections between EDCs and crop growth (to avoid basing a relation on a topic for which there is not much data); and core assumptions such as the initial stocks being in dynamic equilibrium (to provide a stable baseline for comparison of scenario results). I describe these and other assumptions in the following paragraphs, and in the following section I propose extensions of this research which would address several limitations of this study.

I assumed that the initial stock values were in dynamic equilibrium from the beginning of the model period, based on the baseline. Every farmer in the region was assumed to grow the same thing at the same time and follow the same crop rotation scheme: corn then alfalfa, repeated. The entire area of agricultural land in the CRD was assumed to be allocated to growing these crops.

The same amount of amendment was assumed to be added every year, meaning the annual application rate is not based on the composition of the soil. Living organisms, soil moisture and pH were assumed to not influence the system, as were any other soil properties like porosity or cation exchange capacity. EDCs were assumed to function independently from carbon and nitrogen in the soil as well as the crop, meaning there were no feedbacks from EDCs to other parts of the system. Soil nutrient and EDC holding capacity, as well as nitrogen fixation, were assumed to be independent from carbon in the soil.

A base assumption of the Thesis Model is that that the amendments that inform stocks in the model (biosolids, biochar, compost, manure, mineral fertilizer) all act on the stocks in the same way – i.e., for example, the carbon in biosolids, biochar, compost and manure undergo the same processes at the same time scale. In reality, there would be a temporal delay before the different forms of carbon were released/mobilized and therefore each substance would act slightly differently from one another. Another core assumption of the Thesis Model is that there is one application of soil amendments every year, however not all farmers apply all soil amendments every year.

Due to these assumptions around the core of the Thesis Model, the model is not an exact representation of reality and this is exemplified by several model outputs. For example, the literature suggests there is an average of around 50% labile carbon and 50% stable carbon in soils, but the Thesis Model shows a higher proportion of labile carbon. Observations like this are incongruous with physical conditions and thus require further study or development with the use of empirical data once it becomes available.

This model may underestimate OM decomposition and overestimate stable carbon loss by erosion and may also underestimate the stock of stable carbon in the soil (which could be due to the rate of stable carbon loss by erosion being too high). Erosion loss is the only form of stable carbon loss and it is determined based on the stock of stable carbon as well as the relative erosion loss, which is a constant value. As a result, the loss of stable carbon is proportional to the amount of stable carbon in the soil. This relationship is worth exploring and investigating in greater detail by expanding on this area of the model.

The Thesis Model does not distinguish between stocks of NH_4 and NO_3 (which are two different forms of PAN), and instead groups these two compounds together. Since NH_4 and NO_3 behave

and interact with soil, crops, and the environment in different ways, isolating these two forms of PAN could be a worthwhile extension of the model.

This model also does not distinguish between more than two stocks of carbon, and this is a simplification of the real-world situation which makes comparing the results of this study to others somewhat challenging. For instance, the current results for carbon stocks suggest there is a carbon saturation effect when biosolids are applied, but studies of carbon saturation typically distinguish carbon in the soil into more individual components (e.g., free/coarse particulate OM, silt-plus-clay carbon, microaggregate carbon, and non-hydrolysable carbon) (Chung et al., 2008; Stewart et al., 2008). Therefore, it is difficult to compare the present results with those of other studies, and I am only able to comment on the general trend.

Another key assumption is that there is homogenous distribution of the stocks of soil constituents across the entire area being modeled (e.g., one hectare of land to the depth of the plow layer). The Thesis Model is not a distribution model, so there has not been an effort to understand how the constituents are distributed over the area of the model – this could be the subject of another study. The aim here was to understand in general terms how much of these constituents is spread across the land, without considering heterogeneous distribution.

6.3.2. Extensions

My research demonstrates a method for analyzing the long-term dynamics of biosolids and biochar agricultural land application on soil quality that can be applied wherever a choice is being made between utilizing these two soil amendments. From a methodological perspective, the application of system dynamics modelling to this subject at this fine resolution – at the scale of biophysical soil processes – advances simulation modelling techniques and their applicability to biosolids decision-making. In addition, the system dynamics model I developed advances the study of systems approaches for understanding agronomic dynamics and reinforces calls to consider the effects of biosolids and biochar use in the long-run (Archontoulis et al., 2016; Lu et al., 2012; Schroder et al., 2008). In the following paragraphs, I offer several suggestions for logical and useful extensions to my research.

Ideally, the model would be expanded to include more resource users, or stakeholders in the biosolids supply chain. The model currently only represents biophysical aspects of soil quality within the agricultural system, but in reality, there are other key stakeholders and potential trade-

offs involved in biosolids decision-making. Bringing more of these actors and options into the analysis would allow for a better understanding of what is at stake.

This model was developed in isolation from the CRD's internal policy activities and no contact was made with decision-makers to inform the modelling experiment. However, often a system dynamics model is useful when developed for a specific real-world scenario; this way, a decision-maker can gain insights that are of great relevance to their situation. Therefore, future research should incorporate these decision-makers' goals or objectives in the model, and these could be elicited through discussions with the relevant parties during the research design stage.

Another logical and useful expansion to the model would be to look at coupling different systems with the biophysical system presently being modelled. If the Thesis Model was expanded to include more systems such as social, economic, energy and biological systems, the resulting pros and cons might change the outcomes of the study.

The role of social aspects such as public acceptance in influencing support for a proposed project could be substantial, so social systems could be incorporated into the model to allow for the exploration of the influence of environmental on social dynamics, and vice versa. There could for example be feedback mechanisms whereby levels of EDCs in the soil inform the level of public acceptance, which could in turn inform the application rate for the soil amendment in that scenario, and such feedbacks should be thoroughly explored in order to build a better picture of the true complexity of the system that incorporates the human dimension.

The role of economic dimensions such as whether the soil amendment is provided to the farmer for free or sold at a cost, the cost of distribution of excess amendments, or economic incentives such as carbon credits, could influence a farmer's willingness to use the amendment. Incorporating feedback mechanisms, for example whereby the cost influences the farmer's willingness to apply, may result in a model that better represents reality. The inclusion of a cost could end up changing the balance of pros and cons and therefore potentially the preferred use strategy may be different.

The production of biosolids and biochar produces energy in the form of py-gas and py-oil, both of which can be sold and used by third parties as a renewable fuel source. By incorporating a more complex energy system into the model, the capture and sale of energy may potentially change which use strategy is preferred (economic incentive). Biochar application might end up being

preferred as there would be more energy captured due to the additional energy produced through pyrolysis (i.e., further energy is produced at the RTF).

In the soil, there is of course an entire host of living organisms, from microbes to earthworms, all of which contribute to and interact with the soil and its constituents in a number of important ways. For instance, EDCs may accumulate in microbes as they consume organic matter containing EDCs, and earthworms influence soil structure (e.g., soil specific weight). By incorporating these biological elements, the model may be able to replicate the dynamics of the stocks as it would better represent the real world.

Field testing or experiments would better ground the model in reality, as the values for parameters in the model could be set to represent the real system even better. Agricultural soil testing in the CRD could be done to calibrate the initial values for the main stocks in the model. Furthermore, soil testing would allow for the accurate determination of agronomic application rates for the soil amendments; application rates could be determined based on actual amount of Nitrogen in the soil and other field-specific parameters.

This study showed that application of biochar alone did not result in a thriving agricultural system, suggesting that biochar needs to be added in conjunction with a fertilizer containing nitrogen. Previous studies (e.g., Paramashivam (2015)) have found that a mixture of biosolids and biochar performed better than both biosolids and biochar alone, however due to a lack of available studies and relevant data this mixture was not included in the present analysis. Future studies should investigate the influence of adding nitrogen in conjunction with biochar, whether in the form of mineral fertilizer, compost, manure, or even biosolids.

6.4. Concluding Remarks

The following table summarizes the results of testing biosolids and biochar application scenarios. Biosolids application results in increased plant available nitrogen, organic nitrogen, and labile carbon stocks, however these scenarios do carry the risks associated with increased nitrogen leaching and the presence of EDCs, some of which are removed from the soil each year and others which are not and therefore accumulate over time. Biosolids application also results in the greatest plant biomass and crop yield, with the highest application rate performing the best.

Table 16. Key results of this study

Parameter Category	Biosolids (B)		Biochar (C)	
	Pros	**Cons**	**Pros**	**Cons**
Nitrogen	↑ Plant Available	↑ Leaching	↓ Leaching	Needs to be added
Carbon	↑ Fast SOM	–	↑ Slow SOM	–
EDCs	–	Greater hazard	No change in hazard	–
Crop	Greatest yields	–	–	Lowest yields
C/N Ratio	Ideal	–	–	Too high

Biochar application may be preferred for storing stable carbon (if negative effects on crop productivity are ignored) and minimizing the rate of nitrogen leaching and risk due to EDCs. This finding is in line with previous research which has found biochar is an ideal long-term carbon sequestration mechanism that allows us to offset GHG emissions (Sohi et al., 2010) and can cause reduced nitrogen leaching from the soil (Yuan, Lu, Wang, Chen, & Lei, 2016).

In this thesis I have demonstrated that biosolids application may be the preferred strategy from the perspective of soil quality. Farmers would be keen to optimize the C/N ratio in the soil, maximize yields, and (indirectly) reduce the cost of mineral fertilizer. It is up to the individual stakeholders (farmers) to weigh the pros and cons to determine whether the risks posed by EDCs and nitrogen leaching are acceptable, given the alternative which is a biochar scenario where these two risks are much lower or non-existent but the yields are lower and PAN addition is needed. Perhaps the inclusion of an economic incentive, for example in the form of carbon credits, in the model would shift the balance of pros and cons in favour of biochar due to the increased amount of carbon storage and the related economic benefits to the farmer. However, maintaining an ideal C/N ratio and improving crop yield may be of greater importance to farmers than the economic incentive of carbon credits.

6.5. Contribution

In this thesis I have presented an application of system dynamics modelling for biosolids decision-making, comparing two potential options the CRD, or other regions, may be considering (i.e., agricultural land application of biosolids or biochar). This provides biosolids decision-makers with a fine-resolution analysis of the long-term dynamics of some parameters of interest to (one of, if not the only) primary stakeholders in agricultural land application: farmers.

This thesis has demonstrated that system dynamics modelling can be used for gaining an understanding of the long-term dynamics of certain agricultural soil constituents (specifically, carbon, nitrogen and EDCs) and plant growth, as well as broadening our understanding of an agricultural system and its overall structure under biosolids or biochar application. This modelling approach can be used to compare between these two biosolids management options in other jurisdictions. This thesis contributes to knowledge of the benefits and potential drawbacks, in terms of soil constituents and quality, of biosolids and biochar land application in agriculture.

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Appendices

Appendix A

Details and References Used in Parameterization

NOTE: in the tables in this Appendix, white cells denote parameters that I kept unchanged, orange cells denote parameters that I directly altered from Bossel (2007), while green cells denote parameters that I developed myself.

Table 17. Stocks in the model.

ID	Name	Unit	Inflows [+] and outflows [–]	Description	Initial Value	Reference(s)
S1	carbon_in_FSOM	kg ha^{-1}	+ C_input_from_biosolids, nutrient_C_input_from_biochar, C_input_from_organic_fertilizer, and C_input_from_crop_residues – Decomposition_of_OM, C_loss_by_erosion	Carbon in quickly decomposable soil organic matter	19191.4390	Bossel (2007)
S2	carbon_in_SSOM	kg ha^{-1}	+ Permanent_C_input_from_biochar, C_input_to_SSOM – C_losses_of_SSOM	Carbon in non-decomposable soil organic matter	4750.1333	Bossel (2007)
S3	nitrogen_in_FSOM	kg ha^{-1}	+ N_input_from_biosolids, N_input_from_biochar, N_input_from_organic_fertilizer, N_input_from_crop_residues, N_input_by_transfer – N_loss_from_erosion, N_loss_from_OM_decomposition	Nitrogen in quickly decomposable soil organic matter	952.7246	Bossel (2007)
S4	plant_available_ nitrogen	kg ha^{-1}	+ PAN_input_from_biosolids, PAN_input_from_biochar, PAN_input_from_mineral_fertilizer, N_from_atmosphere, N_input_from_OM_decomposition – N_loss_by_leaching, N_loss_by_transfer_to_OM, N_uptake_by_plants	Nitrogen that is plant-available (NH_4, $N0_3$)	125.4892	Bossel (2007)

ID	Name	Unit	Inflows [+] and outflows [–]	Description	Initial Value	Reference(s)
S5	relative_plant_biomass	–	+ relative_growth_rate – harvest	Relative measure of crop biomass in the field	0.01	Bossel (2007)
S6	relative_crop_yield	–	+ harvest	Total amount of crop harvested	0	Bossel (2007)
S7	TCS_in_soil	kg ha^{-1}	+ TCS_sorption_from_biosolids, TCS_sorption_from_biochar – TCS_plant_uptake, TCS_microbial_degradation, TCS_photo_degradation	Amount of Triclosan in the soil	3.9×10^{-8}	(Xia et al., 2010)
S8	TCC_in_soil	kg ha^{-1}	+ TCC_sorption_from_biosolids, TCC_sorption_from_biochar – TCC_plant_uptake, TCS_microbial_degradation,	Amount of Triclocarban in the soil	8.11×10^{-8}	(Xia et al., 2010)
S9	population	people	+ growth	Amount of people in the Core Area, 2020	312668.75	Capital Regional District (2017b)
S10	biosolids_storage	kg	+ biosolids_production – biosolids_removed_for_application, biochar_production, excess_biosolids_distribution	Amount of biosolids in storage	0	–
S11	biochar_storage	kg	+ biochar_production – biochar_removed_for_application, excess_biochar_distribution	Amount of biochar in storage	0	–

Table 18. Flows in the model.

ID	Name	Unit	Equation	Description	Reference(s)
F1	C_input_from_ biosolids	kg ha^{-1} year^{-1}	IF (ABS(YEARLY__TIME-ORGANIC_FERTILIZER_DATE) < (DT/2)) THEN biosolids_land_application_on_or_off*((C_in_biosolids)/DT) ELSE (0)	Carbon inflow from biosolids into C-FSOM	–
F2	nutrient_C_input_ from_biochar	kg ha^{-1} year^{-1}	IF (ABS(YEARLY__TIME-ORGANIC_FERTILIZER_DATE) < (DT/2)) THEN biochar_land_application_on_or_off*((nutrient_C_in_biochar)/DT) ELSE (0)	Carbon inflow from biochar into C-FSOM	–
F3	C_input_from_organic_fertilizer	kg ha^{-1} year^{-1}	IF (ABS(YEARLY__TIME-ORGANIC_FERTILIZER_DATE) < (DT/2)) THEN compost_and_manure_on_or_off*((C_in_compost+C_in_manure)/DT) ELSE (0)	Carbon inflow from organic fertilizer into C-FSOM	(Bossel, 2007)
F4	C_input_from_crop_ residues	kg ha^{-1} year^{-1}	C_in_crop_residues	Carbon inflow from crop residues into C-FSOM	(Bossel, 2007)
F5	decomposition_of_OM	kg ha^{-1} year^{-1}	DECOMPOSITION__RATE*decomposition_factor*carbon_in_FSOM*(1+(0.5*SIN(2*Pi_per_Year*(YEARLY_TIME-DELAY_TIME))))	Carbon outflow from C-FSOM due to decomposition of organic matter	(Bossel, 2007)
F6	C_loss_by_erosion	kg ha^{-1} year^{-1}	relative_erosion_loss*carbon_in__FSOM	Carbon outflow from C-FSOM due to erosion	(Bossel, 2007)
F7	C_input_to_SSOM		0.25*decomposition_of_OM	Carbon inflow from FSOM decomposition, 25% of which becomes SSOM (with the rest being lost)	(Bossel, 2007)

ID	Name	Unit	Equation	Description	Reference(s)
F8	C_losses_of_SSOM		carbon_in_SSOM*(0.2+relative_erosion_loss)	Carbon outflow from SSOM due to erosion of	(Bossel, 2007)
F9	C_input_from_biochar		IF (ABS(YEARLY__TIME-ORGANIC_FERTILIZER_DATE) < (DT/2)) THEN biochar__land_application_on_or_off*((C_in__biochar)/DT) ELSE (0)	Carbon inflow into C-SSOM due to biochar application	–
F10	N_input_from_biosolids	kg ha^{-1} $year^{-1}$	IF (ABS(YEARLY__TIME-ORGANIC_FERTILIZER_DATE)<DT/2) THEN biosolids__land_application_on_or_off*((C_in_biosolids/CN_IN_BIOSOLIDS)/DT) ELSE 0	Nitrogen inflow from biosolids into N-FSOM	–
F11	N_input_from_biochar	kg/ha/year	IF (ABS(YEARLY__TIME-ORGANIC_FERTILIZER_DATE)<DT/2) THEN biochar_land_application_on_or_off*((nutrient_C_in_biochar/CN_IN_BIOCHAR)/DT) ELSE 0	Nitrogen inflow from biochar into N-FSOM	–
F12	N_input_from_organic_fertilizer	kg ha^{-1} $year^{-1}$	IF (ABS(YEARLY__TIME-ORGANIC_FERTILIZER_DATE)<DT/2) THEN compost_and_manure__on_or_off*(((C_in_manure/CN_IN_MANURE)+(C_in_compost/CN_IN_COMPOST))/DT) ELSE 0	Nitrogen inflow from organic fertilizer into N-FSOM	(Bossel, 2007)
F13	N_input_from_crop_residues	kg ha^{-1} $year^{-1}$	N_in_crop_residues	Nitrogen inflow from crop residues into N-FSOM	(Bossel, 2007)

ID	Name	Unit	Equation	Description	Reference(s)
F14	N_input_by_transfer	kg ha^{-1} $year^{-1}$	N_transfer_function*PAN	Nitrogen inflow from transfer from PAN	(Bossel, 2007)
F15	N_loss_from_erosion	kg ha^{-1} $year^{-1}$	C_loss_by_erosion/CN_ratio	Nitrogen outflow from N-FSOM due to erosion	(Bossel, 2007)
F16	N_loss_from_OM_ decomposition	kg ha^{-1} $year^{-1}$	decomposition_of_OM/CN_ratio	Nitrogen outflow from N-FSOM due to OM decomposition	(Bossel, 2007)
F17	PAN_input_from_ biosolids	kg ha^{-1} $year^{-1}$	IF (ABS(YEARLY__TIME-ORGANIC_FERTILIZER_DATE) < (DT/2)) THEN biosolids__land_application_on_or_off*((PAN_ in_biosolids)/DT) ELSE (0)	Nitrogen inflow from biosolids into PAN	–
F18	PAN_input_from_ biochar	kg ha^{-1} $year^{-1}$	IF (ABS(YEARLY__TIME-ORGANIC_FERTILIZER_DATE) < (DT/2)) THEN biochar__land_application_on_or_off*((PAN__ in_biochar)/DT) ELSE (0)	Nitrogen inflow from biochar into PAN	–
F19	N_input_from_OM_ decomposition	kg ha^{-1} $year^{-1}$	N_loss_from_OM__decomposition	Nitrogen inflow from OM decomposition into PAN	(Bossel, 2007)
F20	N_input_from_ mineral_fertilizer	kg ha^{-1} $year^{-1}$	IF (ABS(YEARLY__TIME-MINERAL__FERTILIZER__DATE))<DT/2 THEN mineral__fertillizer_on_or_off*NITROGEN_F ERTILIZER_APPLIED/DT ELSE 0	Nitrogen inflow from mineral fertilizer into PAN	(Bossel, 2007)

ID	Name	Unit	Equation	Description	Reference(s)
F21	N_from_atmosphere	kg ha^{-1} $year^{-1}$	0.01*NITROGEN_INPUT_FROM_ATMOSPHERE	Nitrogen inflow from the atmosphere to PAN – changed as in The Bossel Model there was 25 kg/ha of PAN from the atmosphere at every time step (0.01)	(Bossel, 2007)
F22	N_uptake_by_plants	kg ha^{-1} $year^{-1}$	(growth_rate*(SPECIFIC__NITROGEN__UPTAKE-NITROGEN__FIXATION))	Nitrogen outflow from PAN due to plant uptake	(Bossel, 2007)
F23	N_loss_by_leaching	kg ha^{-1} $year^{-1}$	LEACHING__RATE*PAN*nitrogen_leaching	Nitrogen outflow from PAN due to leaching	(Bossel, 2007)
F24	N_loss_by_transfer_to_OM	kg ha^{-1} $year^{-1}$	N_input_by_transfer	Nitrogen outflow from PAN due to transfer to N-FSOM	(Bossel, 2007)
F25	relative_growth_rate	1 $year^{-1}$	IF ((YEARLY__TIME-INT(YEARLY__TIME) < BEGIN_GROWTH_PERIOD) OR (YEARLY__TIME-INT(YEARLY__TIME) > HARVEST__TIME)) THEN 0 ELSE (25*(20/52)/(HARVEST__TIME-BEGIN_GROWTH_PERIOD))*relative_plant_biomass*(1-relative_plant_biomass)*nitrogen_effect	Plant biomass inflow due to plant growth, only occurs between the beginning of the growth period and harvest time	(Bossel, 2007)
F26	harvest	1 $year^{-1}$	IF (ABS(YEARLY__TIME-HARVEST__TIME)<(DT/2)) THEN (relative_plant_biomass/DT) ELSE 0	Plant biomass outflow from relative plant biomass into the relative crop yield, due to crop harvest	(Bossel, 2007)

ID	Name	Unit	Equation	Description	Reference(s)
F27	TCS_sorption_from_biosolids	kg ha-1 year-1	IF (ABS(YEARLY__TIME-ORGANIC_FERTILIZER_DATE) < (DT/2)) THEN biosolids__land_application_on_or_off*((TCS_in__biosolids)/DT) ELSE (0)	TCS inflow from biosolids due to biosolids application	–
F28	TCS_sorption_from_biochar	kg ha-1 year-1	IF (ABS(YEARLY__TIME-ORGANIC_FERTILIZER_DATE) < (DT/2)) THEN biochar__land_application_on_or_off*((TCS_in__biochar)/DT) ELSE (0)	TCS inflow from biochar due to biochar application	–
F29	TCS_photo_degradation	kg ha-1 year-1	TCS__in_soil*TCS_Photo_degradation_rate	TCS outflow from the soil due to photodegradation	–
F30	TCS_microbial_degradation	kg ha-1 year-1	TCS__in_soil*TCS_Microbial_degradation_rate	TCS outflow from the soil due to microbial degradation	–
F31	TCS_plant_uptake	kg ha-1 year-1	growth_rate*(SPECIFIC__TCS__UPTAKE)	TCS outflow from the soil due to plant uptake	–
F32	TCC_sorption_from_biosolids	kg ha-1 year-1	IF (ABS(YEARLY__TIME-ORGANIC_FERTILIZER_DATE) < (DT/2)) THEN biosolids__land_application_on_or_off*((TCC_in__biosolids)/DT) ELSE (0)	TCC inflow from biosolids due to biosolids application	–
F33	TCC_sorption_from_biochar	kg ha-1 year-1	IF (ABS(YEARLY__TIME-ORGANIC_FERTILIZER_DATE) < (DT/2)) THEN biochar__land_application_on_or_off*((TCC_in__biochar)/DT) ELSE (0)	TCC inflow from biochar due to biochar application	–

ID	Name	Unit	Equation	Description	Reference(s)
F34	TCC_microbial_ degradation	kg ha^{-1} year^{-1}	TCC_in_soil*TCC_Microbial_degradation_rate	TCC outflow from the soil due to microbial degradation	–
F35	TCC_plant_uptake	kg ha^{-1} year^{-1}	growth_rate*(SPECIFIC__TCC__UPTAKE)	TCC outflow from the soil due to plant uptake	–
F36	growth	people year^{-1}	population*pop__growth__rate	People inflow into the population as a result of growth (slow exponential increase)	–
F37	Biosolids_production	kg year^{-1}	IF (biosolids_production__on_or_off*(population *biosolids__per_capita))>=RTF__capacity THEN RTF__capacity ELSE (biosolids_production__on_or_off*(population *biosolids__per_capita))	Biosolids inflow into the biosolids storage as a result of biosolids production	–
F38	Biosolids_removed_ for_application	kg year^{-1}	IF (ABS(YEARLY__TIME-ORGANIC_FERTILIZER_DATE) < (DT/2)) THEN ((Demand_to_satisfy_BS_Application_Rate)/DT) ELSE (0)	Biosolids outflow from biosolids storage as a result of agricultural land application	–
F39	Excess_biosolids_ distrbiution	kg year^{-1}	IF (biosolids_storage/biosolids_storage_capacity) >1 THEN (biosolids_storage-biosolids_storage_capacity)*biosolids_distribution__rate ELSE 0	Biosolids outflow from biosolids storage as a result of the storage capacity being reached	–
F40	Biochar_production	kg year^{-1}	biochar_land_application_on_or_off*(biochar_ yield_per_biochar*biochar_storage)	Biochar inflow into the biochar storage as a result of biochar production	–

ID	Name	Unit	Equation	Description	Reference(s)
F41	biochar_removed_ for_application	kg year^{-1}	IF (ABS(YEARLY_TIME-ORGANIC_FERTILIZER_DATE) < (DT/2)) THEN ((Demand_to_satisfy_BC_Application_Rate)/ DT) ELSE (0)	Biochar outflow from biochar storage as a result of agricultural land application	–
F42	Excess_biochar_ distrbiution	kg year^{-1}	IF (biochar_storage/biochar_storage_capacity)>1 THEN (biochar_storage-biochar_storage_capacity)*biochar_distribution _rate ELSE 0	Biochar outflow from biochar storage as a result of the storage capacity being reached	–

Table 19. Converters in the model

ID	Name	Unit	Equation or Value	Description	Reference(s)
C1	C_in_biosolids	kg ha^{-1}	*(biosolids_agronomic_app_rate*C_per_OM*ODM_FRACTION_OF_BIOSOLIDS)*	Total carbon in the biosolids applied	–
C2	ODM_fraction_of_biosolids	–	0.575	Organic dry matter fraction of the biosolids	(Sullivan et al., 2015)
C3	CN_in_biosolids	–	*45*	C/N ratio in the biosolids	–
C4	PAN_in_biosolids	kg ha^{-1}	*biosolids_agronomic_app_rate*PAN_FRACTION_OF_BIOSOLIDS*	Total Plant Available Nitrogen in the biosolids	–
C5	PAN_fraction_of_biosolids	–	*0.0031*	Mass fraction of Plant Available Nitrogen in the biosolids	(Monteith, Sterne, & Dong, 2010)
C6	TCS_in_biosolids	–	biosolids_agronomic_app_rate*TCS_FRACTION_OF_BIOSOLIDS	Total Triclosan in the biosolids applied	–
C7	TCS_fraction_of_biosolids	–	0.00078725	Mass fraction of Triclosan in the biosolids	(La Guardia et al., 2003; Monteith, Sterne, et al., 2010)
C8	TCC_in_biosolids	–	biosolids_agronomic_app_rate*TCC_FRACTION_OF_BIOSOLIDS	Total Triclocarban in the biosolids applied	–
C9	TCC_fraction_of_biosolids	–	0. 000254667	Mass fraction of Triclocarban in the biosolids	(Monteith, Sterne, et al., 2010)

ID	Name	Unit	Equation or Value	Description	Reference(s)
C10	ODM_fraction_in_biochar	–	0.241	Organic dry matter fraction of the biochar	(Hossain et al., 2011)
C11	nutrient_C_in_biochar	kg ha^{-1}	(biochar__agronomic__app_rate*ODM_FRACTION_IN_biochar*C_per_OM)	Total carbon in the biochar applied	(Bossel, 2007)
C12	CN_in_biochar	–	*70.42*	C/N ratio in the biochar, average of highest-temperature biochars from Kloss et al. (2012)	(Kloss et al., 2012)
C13	PAN_in_biochar	kg ha^{-1}	*biochar_agronomic_app_rate*PAN_FRACTION_OF_biochar*	Total Plant Available Nitrogen in the biochar	–
C14	PAN_fraction_of_biochar	–	*0.000166*	Mass fraction of Plant Available Nitrogen in the biochar	(Hossain et al., 2011)
C15	TCS_in_biochar	–	biochar_agronomic_app_rate*TCS_FRACTION_OF_biochar	Total Triclosan in the biochar applied	–
C16	TCS_fraction_of_biochar	–	0	Mass fraction of Triclosan in the biochar	(Henderson, 2013; McNamara, Koch, Liu, & Zitomer, 2016)
C17	TCC_in_biochar	–	biochar_agronomic_app_rate*TCC_FRACTION_OF_biochar	Total Triclocarban in the biochar applied	–
C18	TCC_fraction_of_biochar	–	0	Mass fraction of Triclocarban in the biochar	(Monteith, Sterne, et al., 2010)
C19	compost_applied	kg ha^{-1}	5000	Amount of compost applied in the Baseline (halved from Bossel)	(Bossel, 2007)

ID	Name	Unit	Equation or Value	Description	Reference(s)
C20	C_in_compost	kg ha^{-1}	ODM_FRACTION_IN_COMPOST*C_per_OM*COMPOST__APPLIED	Total carbon in the compost applied	(Bossel, 2007)
C21	ODM_fraction_in_compost	–	0.423444	Organic dry matter fraction of the compost, average of 9 locally made composts	(Alba et al., 2019)
C22	CN_in_compost	–	16.69	C/N ratio in the compost, average of 9 locally made composts	(Alba et al., 2019)
C11	manure_applied	kg ha^{-1}	7500	Amount of compost applied in the Baseline (halved from Bossel)	(Bossel, 2007)
C12	C_in_manure	kg ha^{-1}	ODM_FRACTION_IN_MANURE*C_per_OM*MANURE_APPLIED	Total carbon in the manure applied	(Bossel, 2007)
C13	ODM_fraction_in_manure	–	0.247	Organic dry matter fraction of the manure, median value for solid beef manure	(Government of Manitoba, 2015)
C14	CN_in_manure	–	14.59	C/N ratio in the manure, median value for solid beef manure	(Government of Manitoba, 2015)
C15	C_in_crop_residues	kg ha^{-1} $year^{-1}$	IF ABS(YEARLY__TIME-HARVEST__TIME) < DT/2 THEN (C_per_OM*relative_plant_biomass*(STRAW__AND_LEAVES__REMAIN_IN_FIELD*AMOUNT_STRAW__AND_LEAVES+MAX_CROP__RESIDUES)/DT) ELSE 0	Total carbon in the crop residues remaining in the field after harvest	(Bossel, 2007)

<table>
<tr><th>ID</th><th>Name</th><th>Unit</th><th>Equation or Value</th><th>Description</th><th>Reference(s)</th></tr>
<tr><td>C16</td><td>decomposition_rate</td><td>1 year^{-1}</td><td>0.2</td><td>Rate of annual decomposition</td><td>(Bossel, 2007)</td></tr>
<tr><td>C17</td><td>decomposition_factor</td><td>–</td><td><table><tr><th>1/CN_ratio</th><th>Decomposition_factor</th></tr><tr><td>0</td><td>0</td></tr><tr><td>0.010</td><td>0.010</td></tr><tr><td>0.025</td><td>0.250</td></tr><tr><td>0.033</td><td>0.500</td></tr><tr><td>0.040</td><td>0.900</td></tr><tr><td>0.050</td><td>1.000</td></tr></table></td><td>Graphical function of the relationship between the inverse of the soil C/N ratio and the decomposition factor; as the inverse of the C/N ratio increases (i.e., the C/N ratio decreases), more decomposition occurs</td><td>(Bossel, 2007)</td></tr>
<tr><td>C18</td><td>fraction_of_organic_matter_in_soil</td><td>–</td><td>(carbon_in__FSOM+carbon_in_SSOM)/(DEPTH_OF__PLOW_LAYER*sqm__per_ha*ORGANIC_MATTER_SPECIFIC_WEIGHT*C_per_OM)</td><td>Fraction of organic matter (C-FSOM and C-SSOM) in the soil</td><td>(Bossel, 2007)</td></tr>
<tr><td>C19</td><td>initial_C_in_FSOM</td><td>kg ha^{-1}</td><td>ORGANIC__MATTER_IN_SOIL*(1-INITIAL_FRACTION_SSOM_IN_ORGANIC_MATTER)*DEPTH_OF__PLOW_LAYER*sqm__per_ha*ORGANIC_MATTER_SPECIFIC_WEIGHT*C_per_OM</td><td>Initial amount of carbon in the F-SOM</td><td>(Bossel, 2007)</td></tr>
<tr><td>C20</td><td>N_in_crop_residues</td><td>kg ha^{-1} year^{-1}</td><td>IF ABS(YEARLY__TIME-HARVEST__TIME) <DT/2
THEN
(C_per_OM*relative_plant_biomass*(STRAW__AND_LEAVES__REMAIN_IN_FIELD*AMOUNT_STRAW__AND_LEAVES/CN_RATIO__IN_STRAW+MAX_NITROGEN_IN__CROP_RESIDUES))/DT
ELSE 0</td><td>Total nitrogen in the crop residues remaining in the field after harvest</td><td>(Bossel, 2007)</td></tr>
</table>

<table>
<tr><th>ID</th><th>Name</th><th>Unit</th><th>Equation or Value</th><th>Description</th><th>Reference(s)</th></tr>
<tr><td>C21</td><td>N_transfer_function</td><td>1 year^{-1}</td><td><table><tr><th>1/CN_ratio</th><th>N_transfer_function</th></tr><tr><td>0</td><td>1.000</td></tr><tr><td>0.010</td><td>0.950</td></tr><tr><td>0.025</td><td>0.750</td></tr><tr><td>0.033</td><td>0.500</td></tr><tr><td>0.040</td><td>0.200</td></tr><tr><td>0.050</td><td>0</td></tr></table></td><td>Graphical function of the relationship between the inverse of the soil C/N ratio and the nitrogen transfer function; as the inverse of the C/N ratio increases (i.e., the C/N ratio decreases), less nitrogen is transferred</td><td>(Bossel, 2007)</td></tr>
<tr><td>C22</td><td>growth_rate</td><td>kg ha^{-1} year^{-1}</td><td>(relative_growth_rate*MAX__CROP__YIELD)</td><td>Growth rate of the crop</td><td>(Bossel, 2007)</td></tr>
<tr><td>C23</td><td>nitrogen_leaching</td><td>–</td><td><table><tr><th>Fraction_of_organic_matter_in_soil</th><th>Nitrogen_leaching</th></tr><tr><td>0.00</td><td>1.00</td></tr><tr><td>0.05</td><td>0.50</td></tr><tr><td>0.10</td><td>0.20</td></tr><tr><td>1.00</td><td>0.10</td></tr></table></td><td>Graphical function of the relationship between the fraction of organic matter in the soil and nitrogen leaching; as the fraction of organic matter increases, less nitrogen leaching occurs</td><td>(Bossel, 2007)</td></tr>
<tr><td>C24</td><td>leaching_rate</td><td>1 year^{-1}</td><td>1</td><td>Annual proportion of PAN leached</td><td>–</td></tr>
<tr><td>C25</td><td>relative_nitrogen_availability</td><td>–</td><td>PAN/(MAX__CROP__YIELD*SPECIFIC__NITROGEN__UPTAKE)</td><td>Relative availability of PAN to the crop</td><td>(Bossel, 2007)</td></tr>
</table>

ID	Name	Unit	Equation or Value	Description	Reference(s)
C26	begin_growth_period	years	0.25	Crop growth begins at 0.25 years (early April) in Victoria, based on the timing of the beginning of the growing season	(Canada, 2019a)
C27	harvest_time	years	0.7	Crop harvest occurs at 0.7 years (early September) in Victoria, based on the long growing season which extends into October	(Canada, 2019b)
C28	nitrogen_effect	–	*Relative_nitrogen_availability* / *Nitrogen_effect* 0.00 / 0.00 0.20 / 0.20 0.35 / 0.50 0.50 / 0.80 1.00 / 1.00 2.00 / 0.90 3.00 / 0.40 5.00 / 0.10 10.0 / 0.00	Graphical function of the relationship between the relative nitrogen availability and the nitrogen effect; the nitrogen effect peaks when relative nitrogen availability is 1	(Bossel, 2007)
C29	total_biomass	kg ha^{-1}	relative_plant_biomass*(MAX__CROP__YIELD+AMOUNT_STRAW__AND_LEAVES+MAX_CROP__RESIDUES)	Amount of crop biomass in the field	(Bossel, 2007)
C30	harvest_yield_green_weight	kg ha^{-1}	HARVEST__SPECIFIC_WEIGHT*relative_crop_yield*MAX__CROP__YIELD	Total weight of the harvested crop biomass	(Bossel, 2007)
C31	Specific_TCS_uptake	–	0.0000002687	Uptake of Triclosan by the crop (kg Triclosan/kg yield)	(Wu, Spongberg, Witter, Fang, & Czajkowski, 2010)

ID	Name	Unit	Equation or Value	Description	Reference(s)
C32	TCS_microbial_ degradation_rate	1 $year^{-1}$	13.3248425119	Triclosan microbial degradation rate, based on a 19-day half-life	(Henderson, 2013; Ying, Yu, & Kookana, 2007)
C33	TCS_photo_ degradation_rate	1 $year^{-1}$	14.8924710415	Triclosan photodegradation rate, based on a 17-day half-life	(Henderson, 2013)
C34	Specific_TCC_uptak e	–	0.0000000576	Uptake of Triclocarban by the crop (kg Triclocarban/kg yield)	(Wu et al., 2010)
C35	TCC_microbial_ degradation_rate	1 $year^{-1}$	2.59663597646	Triclocarban microbial degradation rate, based on a 97.5-day half life	(Wu et al., 2010)
C36	Pop_growth_rate	1 $year^{-1}$	0.009500451	Population growth rate, based on an exponential fit to the P.E.O.P.L.E. projection	(BC Stats, 2018)
C37	Biosolids_per_capita	kg $person^{-1}$ $year^{-1}$	29.157895	Amount of biosolids produced per person per year	(UN-HABITAT, 2008)
C38	RTF_capacity	kg $year^{-1}$	14000000	Estimate of the annual treatment capacity of the RTF	(CRD Wastewater Treatment Project, 2017, December)
C39	Demand_to_satisfy_ BS_application_rate	kg $year^{-1}$	biosolids_agronomic_app_rate*AGRICULTUR AL__LAND_AREA	Required amount of biosolids to apply to meet the agronomic application rate in the CRD	–
C40	Biosolids_agronomic _ app_rate	kg ha^{-1}	10000, 15000, 20000	Biosolids Low, Medium and High application rates	–

ID	Name	Unit	Equation or Value	Description	Reference(s)
C41	Biosolids_storage_capacity	kg	1000000	Capacity for biosolids storage, based on similarly sized facilities	–
C42	Biosolids_distribution_rate	–	1	Proportion of biosolids distributed when distribution occurs	–
C43	RTF_capacity	kg $year^{-1}$	14000000	Estimate of the annual treatment capacity of the RTF	(CRD Wastewater Treatment Project, 2017, December)
C44	Demand_to_satisfy_BC_application_rate	kg $year^{-1}$	biochar_agronomic_app_rate*AGRICULTURAL__LAND_AREA	Required amount of biochar to apply to meet the agronomic application rate in the CRD	–
C45	Biochar_agronomic_app_rate	kg ha^{-1}	5000, 10000, 15000	Biochar Low, Medium and High application rates	–
C46	Biochar_storage_capacity	kg	500000	Capacity for biochar storage	–
C47	Biochar_distribution_rate	–	1	Proportion of biochar distributed when distribution occurs	–
C48	Biosolids_produced_in_wet_tonnes	tonnes $year^{-1}$	biosolids_production/kg_per_ton	Amount of biosolids produced, measured in wet tonnes	–
C49	Kg_per_ton	kg $tonne^{-1}$	1000	Kilograms per tonne	–

ID	Name	Unit	Equation or Value	Description	Reference(s)
C50	Bio_gas_produced	kg year^{-1}	biosolids_production*bio_gas_yield_per_unit_biosolids	Amount of biogas generated during biosolids production	–
C51	Bio_gas_yield_per_unit_biosolids	–	0.11	Yield of biogas per unit of biosolids	(McNamara et al., 2016)
C52	Py_gas_produced		biochar_production*py_gas_yield_per_unit_biochar	Amount of py-gas generated during biochar production	–
C53	Py_gas_yield_per_unit_biochar		0.15	Yield of py-gas per unit of biochar	(McNamara et al., 2016)
C54	Py_oil_produced	–	biochar_production*py_oil_yield_per_unit_biochar	Amount of py-oil generated during pyrolysis	–
C55	Py_oil_yield_per_unit_biochar	–	0.45	Yield of py-oil per unit of biochar	(McNamara et al., 2016)
C56	Initial_CN_in_FSOM	–	25	Initial C/N ratio in the FSOM, based on accepted optimal level	–
C57	Organic_matter_in_soil	–	0.05	Proportion of organic matter in the soil, typical value for a mineral soil	–
C58	Initial_fraction_SSOM_in_organic_matter	–	0.75	Initial fraction of SSOM in the organic matter	(Bossel, 2007)
C59	Depth_of_plow_layer	m	0.2	Depth of the plow layer	(Bossel, 2007)

ID	Name	Unit	Equation or Value	Description	Reference(s)
C60	Soil_loss_by_erosion	kg ha^{-1} $year^{-1}$	5000	Loss of soil due to erosion; tolerable erosion rates range from 2-11 kg/ha/year in the US	(FAO & ITPS, 2015)
C61	Initial_available_ nitrogen	kg ha^{-1}	50	Initial amount of nitrogen in the soil	(Bossel, 2007)
C62	Soil_specific_weight	kg/m^3	1510	Soil bulk density (mass per unit volume), mean across 33 BC sites	(Y. Zhao et al., 2008)
C63	Ng_per_kg	–	1000000000	Nanograms per kilogram	–
C64	Mineral_fertilizer_dat e	years	0.22	Mineral fertilizer date; assumed to be in mid-March	–
C65	Nitrogen_fertilizer_ applied	kg ha^{-1}	78	Half the amount of mineral fertilizer recommended to be applied yearly to a Humo-Ferric Podzol growing corn/alfalfa	–
C66	Organic_fertilizer_ date	years	0.18	Organic fertilizer date; assumed to be in early March	–
C67	Organic_matter_ specific_weight	kg/m^3	1500	Organic matter specific weight (mass per unit volume)	–
C68	Agricultural_land_are a	ha $year^{-1}$	4464	Total area of agricultural land for crop production in the CRD in 2016	(B.C. Ministry of Agriculture, 2016)

ID	Name	Unit	Equation or Value	Description	Reference(s)
C69	Sqm_per_ha	m^2 ha^{-1}	10000	Square meters per hectare	–
C70	C_per_OM	–	0.58	Proportion of carbon in organic matter (OM = organic C * 1.724)	–
C71	Pi_yer_year	1 $year^{-1}$	π (3.14159)	pi	(Bossel, 2007)
C72	Yearly_time	years	TIME MOD 1	Produces a value between 0 and 1 for each one-year period; for instance, at t = 0.6 and 1.6 the function equals 0.6	–
C73	Crop_biyearly_time	years	TIME MOD 2	Produces a value between 0 and 2 for each two-year period; for instance, at t = 0.4 and 2.4 the function equals 0.4	–
C74	Max_crop_yield	kg ha^{-1}	IF CROP__BIYEARLY__TIME >= 1.25 OR CROP__BIYEARLY__TIME < 0.25 THEN 7000 ELSE 7800	Maximum crop yield varies with the crop; for corn, the value is 7800 while for alfalfa it is 7000	(Bossel, 2007)
C75	Specific_nitrogen_ uptake	–	IF CROP__BIYEARLY__TIME >= 1.25 OR CROP__BIYEARLY__TIME < 0.25 THEN 0.029 ELSE 0.032	Specific nitrogen uptake (kg N / kg yield) varies with the crop; for corn, the value is 0.032 while for alfalfa it is 0.029	(Bossel, 2007)

ID	Name	Unit	Equation or Value	Description	Reference(s)
C76	Harvest_specific_ weight	–	IF CROP__BIYEARLY__TIME >= 1.25 OR CROP__BIYEARLY__TIME < 0.25 THEN 1.14 ELSE 1.15	Harvest yield specific weight (kg fresh weight / kg dry matter) varies with the crop; for corn, the value is 1.15 while for alfalfa it is 1.14	(Bossel, 2007)
C77	Max_crop_residues	kg ha^{-1}	IF CROP__BIYEARLY__TIME >= 1.25 OR CROP__BIYEARLY__TIME < 0.25 THEN 4000 ELSE 2000	The maximum amount of crop residues varies with the crop; for corn, the value is 2,000 kg/ha while for alfalfa it is 4,000 kg/ha	(Bossel, 2007)
C78	Amount_straw_and_ leaves	kg ha^{-1}	IF CROP__BIYEARLY__TIME >= 1.25 OR CROP__BIYEARLY__TIME < 0.25 THEN 6900 ELSE 8600	The amount of straw and leaves; for corn, the value is 8,600 kg/ha while for alfalfa it is 6,000 kg/ha	(Bossel, 2007)
C79	CN_ratio_in_straw	–	IF CROP__BIYEARLY__TIME >= 1.25 OR CROP__BIYEARLY__TIME < 0.25 THEN 13 ELSE 57	The C/N ratio in straw varies with the crop; for corn, the value is 57 while for alfalfa it is 13	(USDA Natural Resources Conservation Service, 2011)
C80	Max_nitrogen_in_ crop_residues	kg ha^{-1}	IF CROP__BIYEARLY__TIME >= 1.25 OR CROP__BIYEARLY__TIME < 0.25 THEN 126 ELSE 39	The maximum amount of nitrogen in crop residues varies with the crop; for corn, the value is 39 kg/ha while for alfalfa it is 126 kg/ha	(Torma et al., 2017)
C81	Nitrogen_fixation	–	IF CROP__BIYEARLY__TIME >= 1.25 OR CROP__BIYEARLY__TIME < 0.25 THEN 0.025 ELSE 0	Nitrogen fixation depends on the crop; for corn, the value is 0 while for alfalfa it is 0.025	(Bossel, 2007)
C82	Straw_and_leaves_ remain_in_field	–	0	Proportion of straw and leaves that remain in the field	(Bossel, 2007)

ID	Name	Unit	Equation or Value	Description	Reference(s)
C83	Compost_and_manure _on_or_off	–	0 or 1	'Switch' used to turn compost/manure application on or off	–
C84	Mineral_fertilizer_ on_or_off	–	0 or 1	'Switch' used to turn mineral fertilizer application on or off	–
C85	Biosolids_production _on_or_off	–	0 or 1	'Switch' used to turn biosolids production on or off	–
C86	Biosolids_land_ application_ on_or_off	–	0 or 1	'Switch' used to turn biosolids application on or off	–
C87	Biochar_land_ application_ on_or_off	–	0 or 1	'Switch' used to turn biochar production on or off	–
C88	Relative_erosion_loss	1 $year^{-1}$	(SOIL_LOSS_BY_EROSION/(DEPTH_OF__PLOW_LAYER*sqm__per_ha*SOIL__SPECIFIC_WEIGHT))	Relative amount of soil lost due to erosion	(Bossel, 2007)

Appendix B

System Dynamics Model

In order to clean up the model interface, I used *ghost elements*. A ghost element acts as a copy of that model element, which can be place elsewhere in the model and inform other model elements. *Stella* will not allow any ghost element to have arrows connect 'to' it (can only have arrows going 'from' it). Note the *italics* label and dashed icon outline for the ghost element.

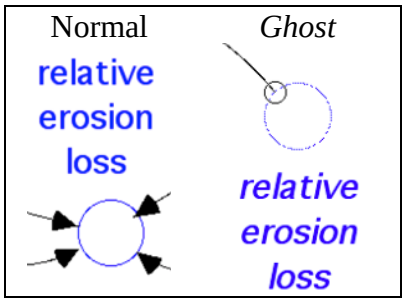

Figure 42. The difference between regular elements and ghost elements – in this case, converters

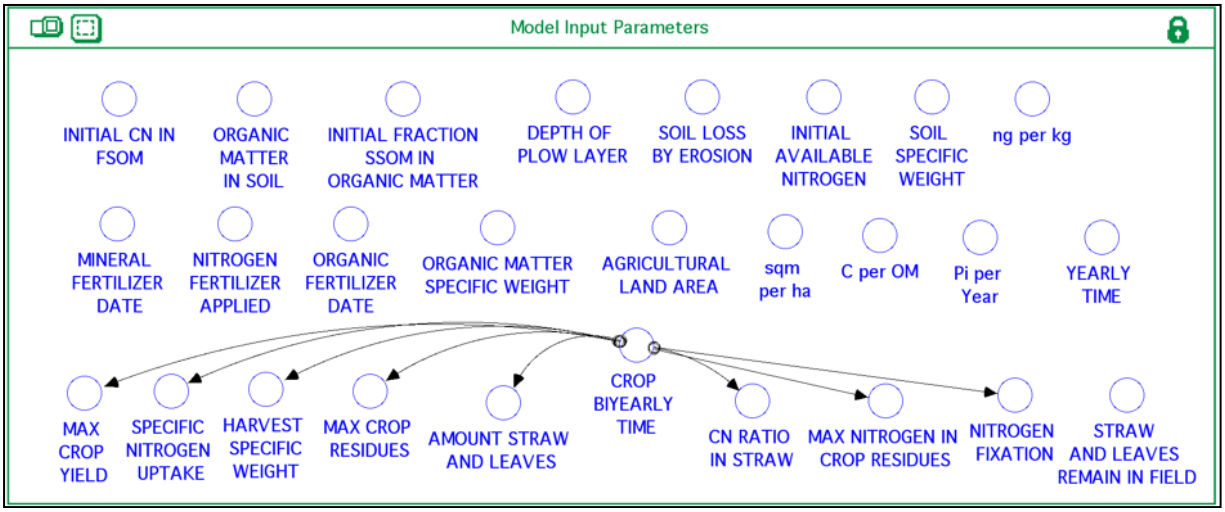

Figure 43. The model input parameters sector.

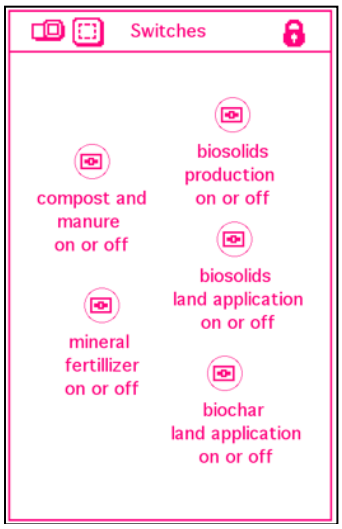

Figure 44. The switches sector.

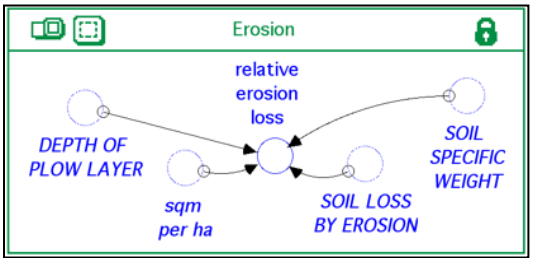

Figure 45. The erosion sector.

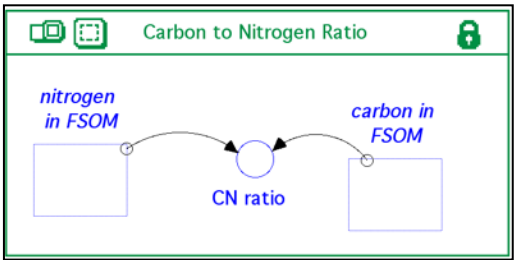

Figure 46. The C/N ratio – a ratio of carbon to nitrogen in the readily decomposable OM.

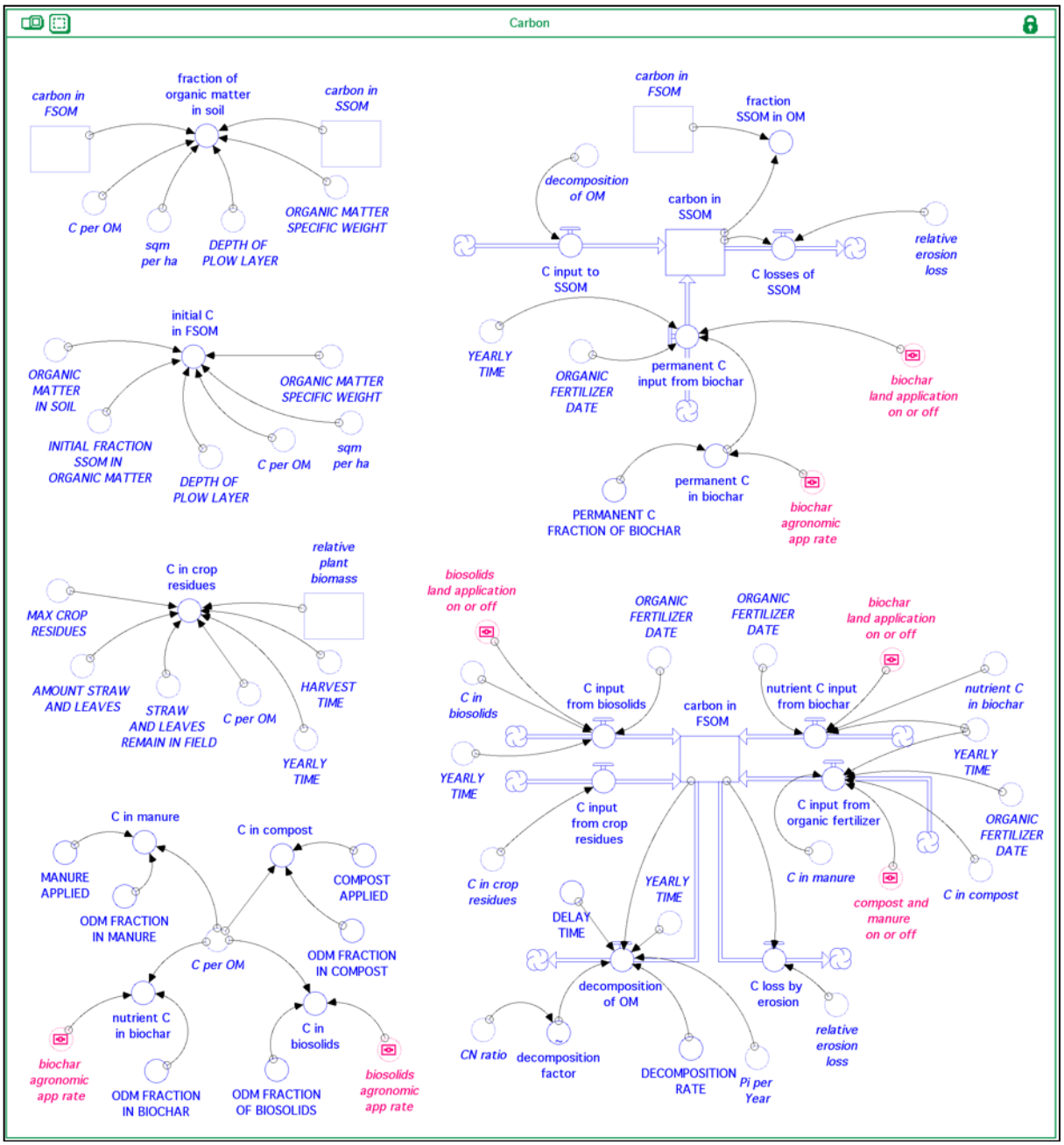

Figure 47. The soil carbon sector.

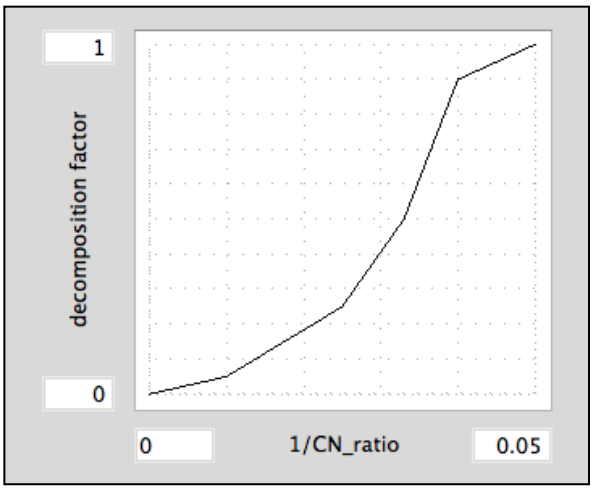

Figure 48. Decomposition factor as a function of the inverse of the C/N ratio. As the C/N ratio decreases, the decomposition factor increases (and vice versa).

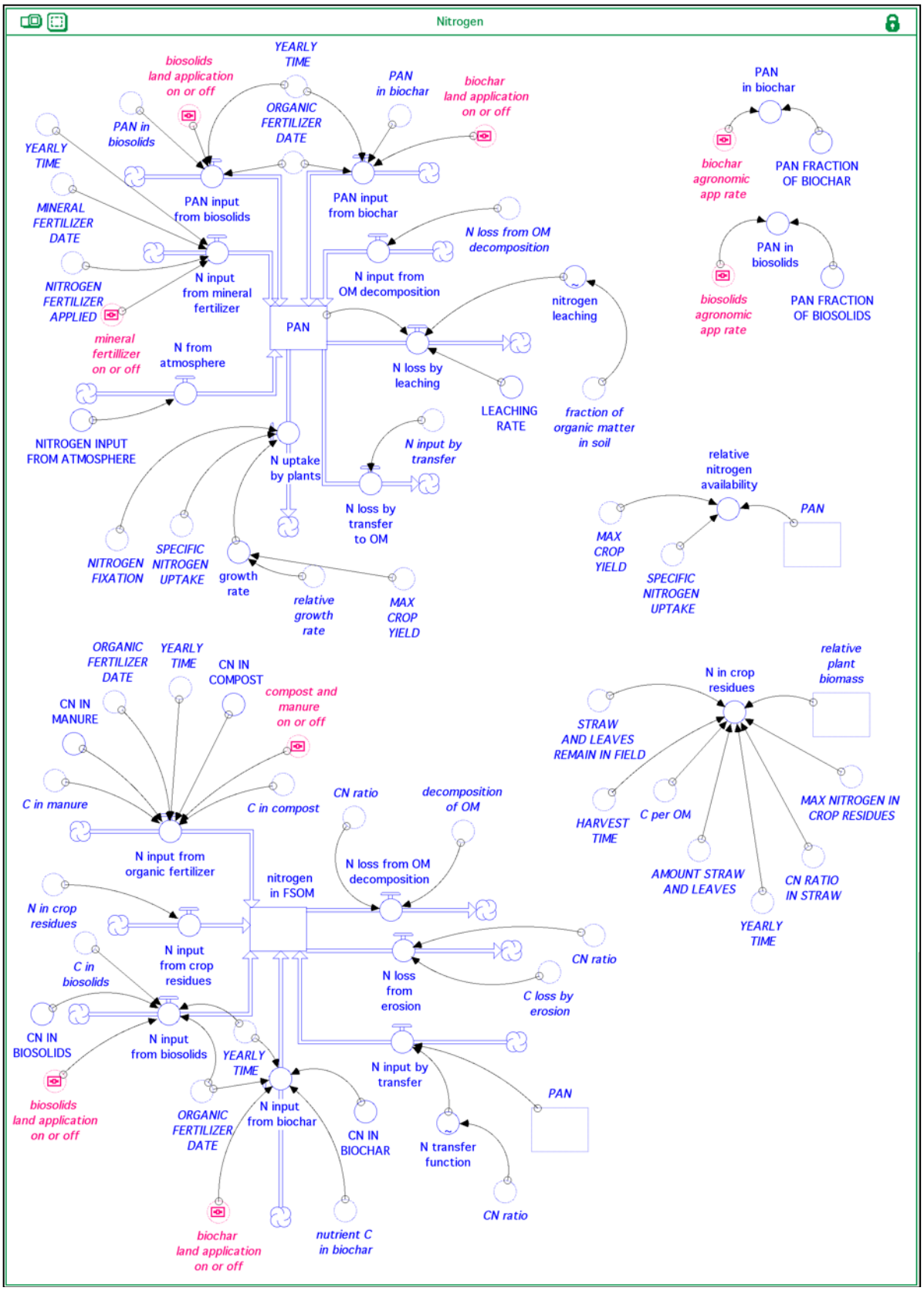

Figure 49. The soil nitrogen sector.

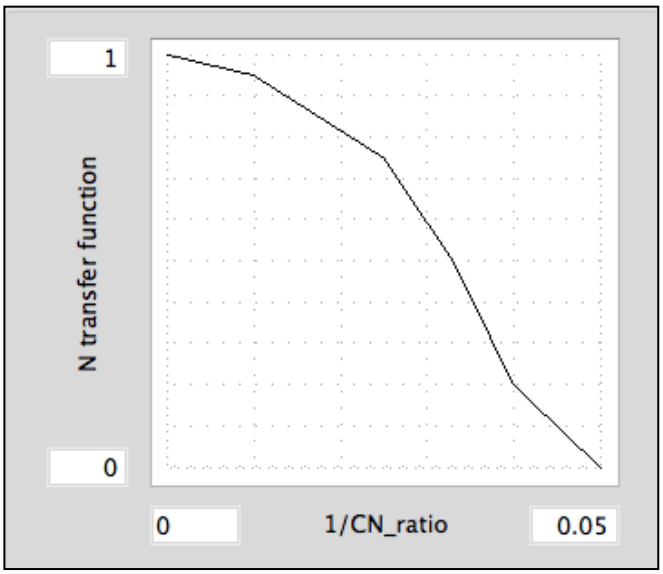

Figure 50. N transfer function as a function of the inverse of the C/N ratio. As the C/N ratio increases, the N transfer function decreases (and vice versa).

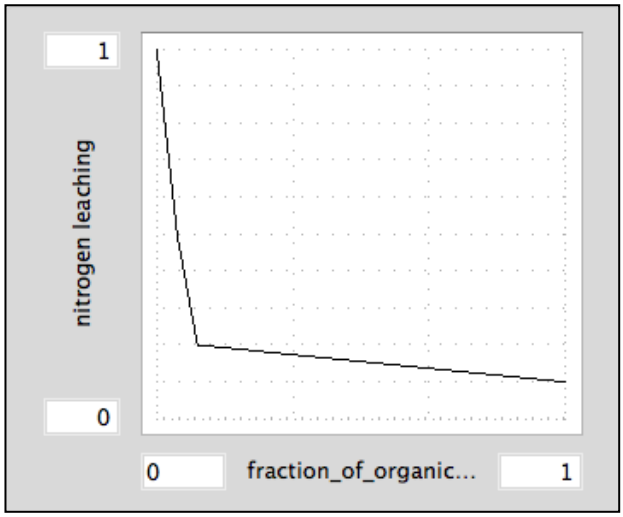

Figure 51. Nitrogen leaching as a function of the fraction of OM in the soil. With no OM in the soil, all N is leached, but this sharply declines to 20% N leaching at 10% OM and declines to 10% N leaching at 100% OM.

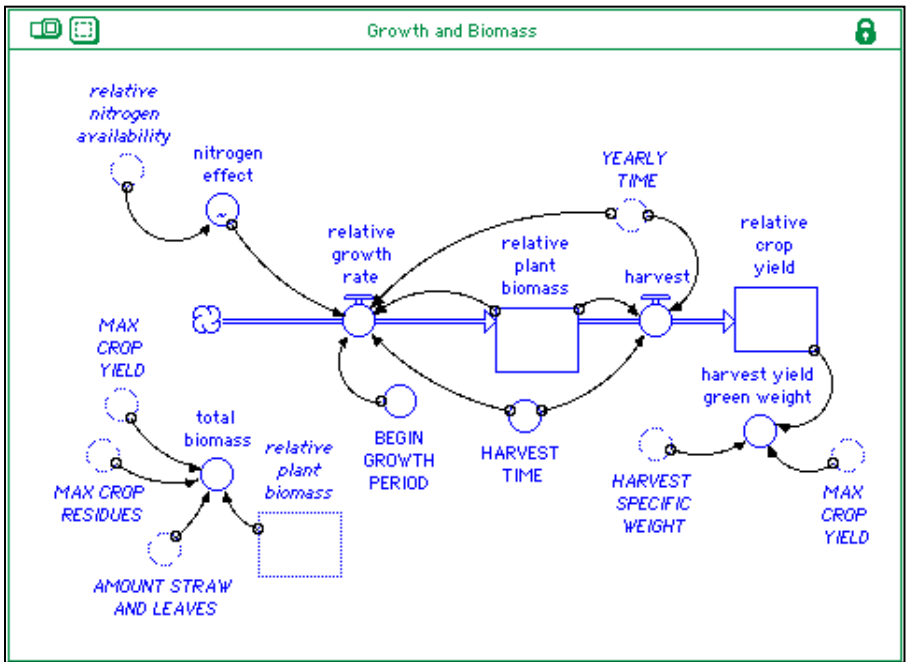

Figure 52. Crop growth, biomass and yield sector. Main stocks are relative plant biomass and relative crop yield. Relative nitrogen availability (RNA) drives the relative growth rate, which determines the relative plant biomass (RPB). RPB is harvested and accumulated in a stock of total relative crop yield (RCY).

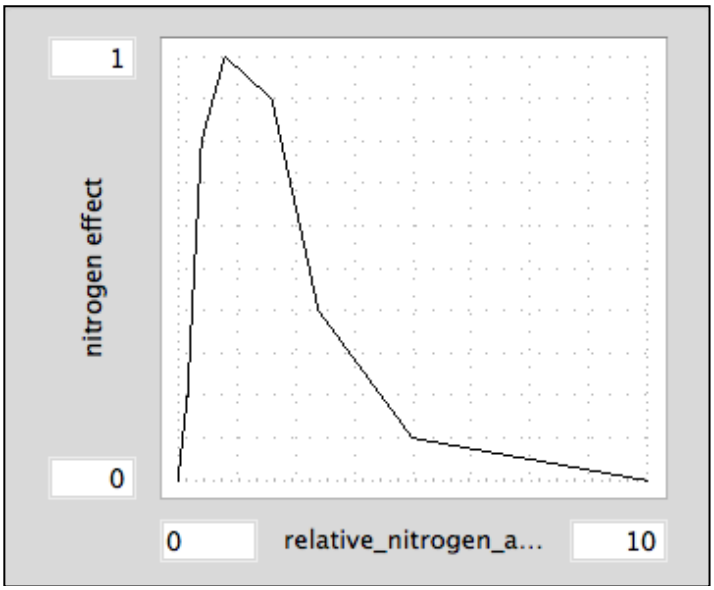

Figure 53. Nitrogen effect (NE) as a function of relative nitrogen availability (RNA). Greatest NE occurs at RNA of 1.

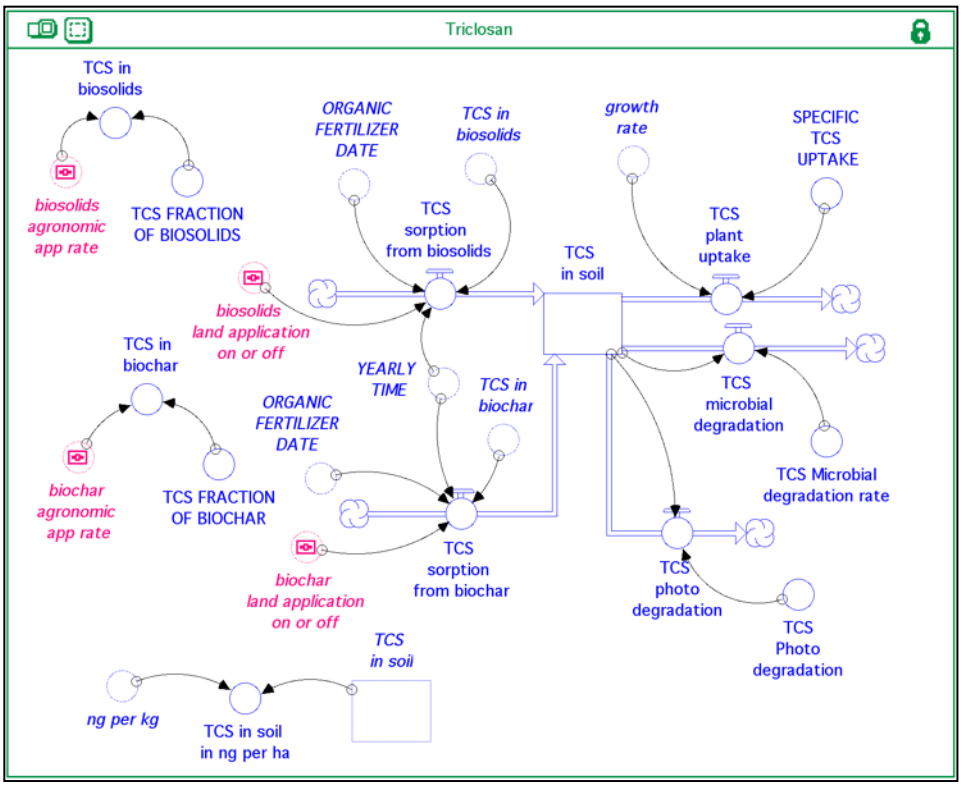

Figure 54. The Triclosan sector.

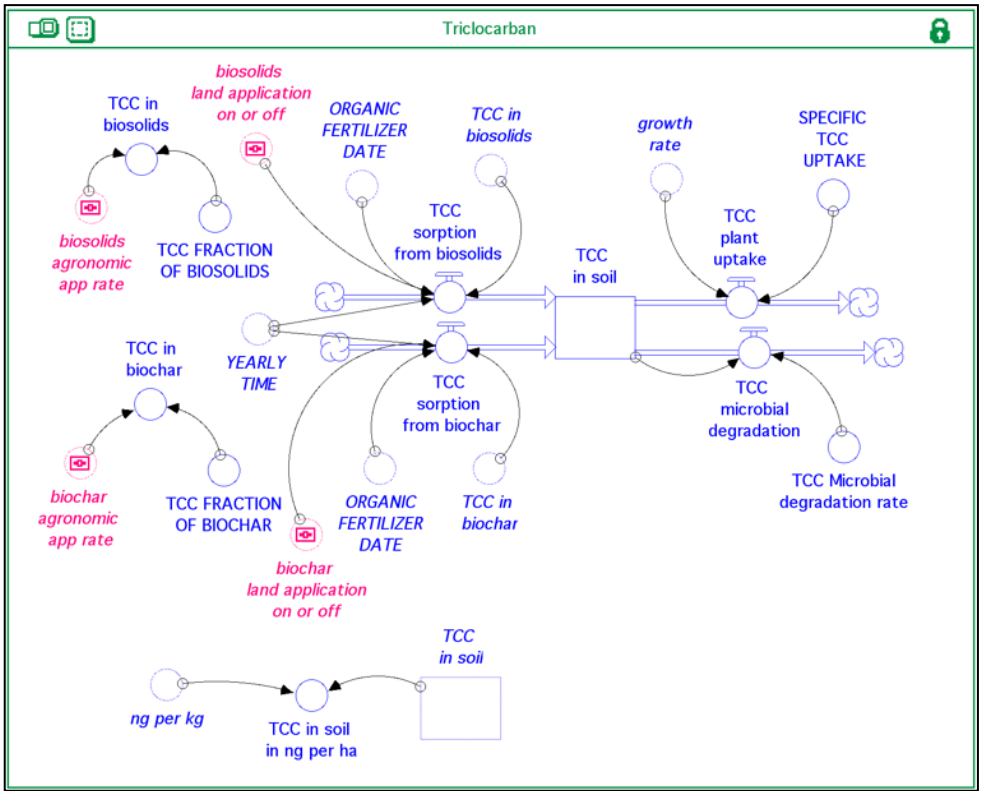

Figure 55. The Triclocarban sector.

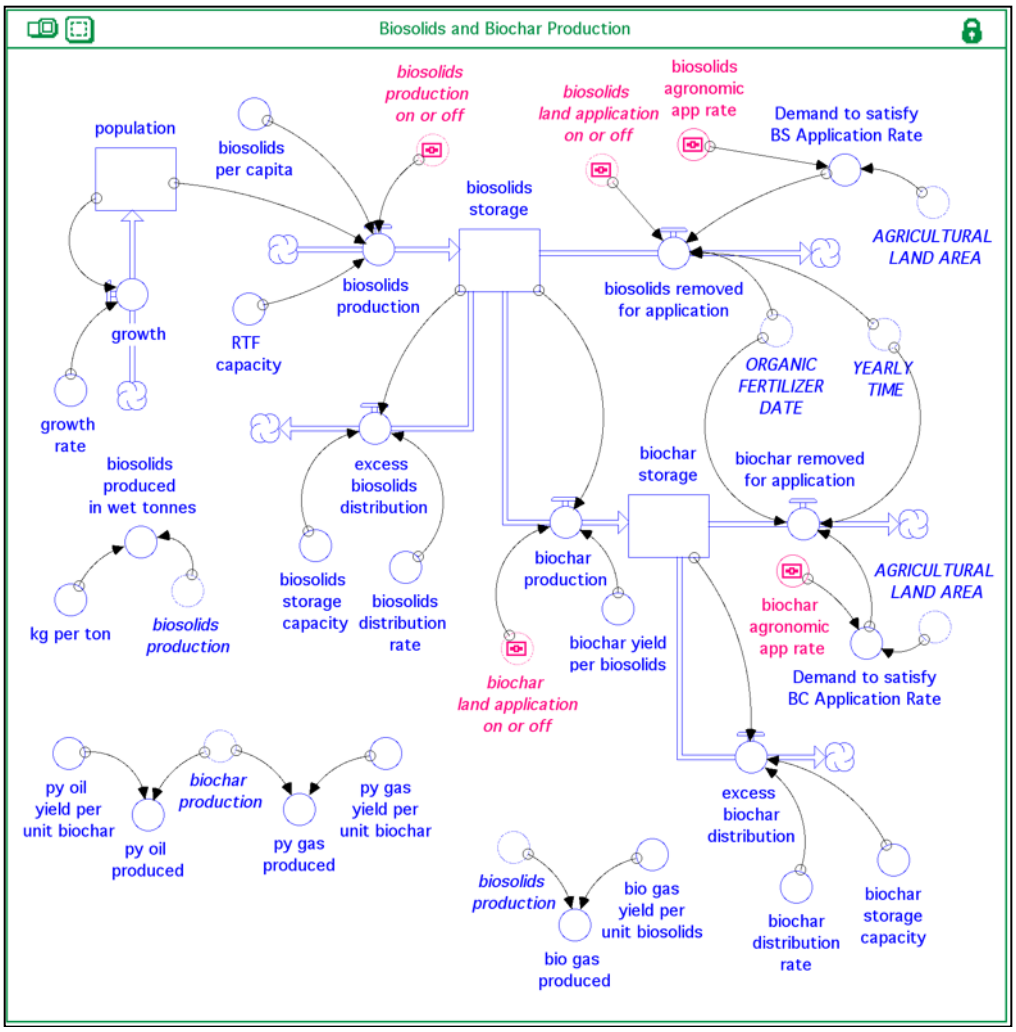

Figure 56. The biosolids and biochar production sector.