Evaluation of New Zealand's Absolute Environmental Sustainability Performance: Development and Application of a Method to Assess the Climate Change Performance of New Zealand's Economic Sectors

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Executive Summary

Existing environmental sustainability assessment methods such as Life Cycle Assessment and environmental footprints quantify the environmental impacts of a system and compare it to a system that is similar to the nature or the function of the examined system. Hence, they are referred to as *relative environmental sustainability assessment* (RESA) methods. Although they provide useful information to improve the eco-efficiency of the system at a particular economic level, they generally fail to inform the environmental sustainability performance of a system against the so-called *absolute environmental boundaries*. Therefore, the significance of the contribution of an examined system to the overall environmental impacts of human activities is mostly overlooked. To address the limitations associated with RESA methods, researchers have suggested the development of *absolute environmental sustainability assessment* (AESA) methods, which guide how human societies can operate and develop within absolute environmental boundaries.

In this context, this research investigated the development of an innovative AESA framework called 'Absolute Sustainability-based Life Cycle Assessment' (ASLCA) based on the environmental indicators and absolute environmental boundaries proposed in three popular frameworks: Planetary Boundaries, Sustainable Development Goals and Life Cycle Assessment. The proposed framework was applied to assess the *production-based* climate change performance of New Zealand agrifood sector, particularly in terms of the two-degree Celsius (2°C) climate target. The results showed that the production-based greenhouse gas (GHG) emissions of New Zealand agri-food sector and its products exceeded the assigned shares of the 2°C global carbon budget. Similar results were observed when the *consumption-based* climate change performance of a typical New Zealand detached house was evaluated against the 2°C climate target.

The framework was then applied to address the consumption-based climate change performance of an economic system using environmentally-extended multi-regional input-output analysis. This framework was used to evaluate the consumption-based climate change performance of New Zealand's total economy (covering 16 sectors) in 2011 against the 2°C climate target, and the outcomes were compared with the production-based climate change performance in the given year. The consumption-based analysis showed that New

Zealand exceeded the assigned share of the 2°C global carbon budget; the consumption-based GHG emissions were 26% more than the assigned carbon budget share. However, the sector-level analysis indicated that three of the 16 sectors (financial and trade services, other services and miscellaneous) were within their assigned carbon budget shares. When the consumption-based GHG emissions were compared with the production-based GHG emissions, New Zealand was a net exporter of GHG emissions in 2011, and the dominating sectors were quite different. The results clearly imply that a significant reduction in GHG emissions associated with New Zealand's consumption and production activities are necessary to stay within the assigned shares of the 2°C global carbon budget.

Given that AESA methods (including ASLCA) are built upon multiple value and modelling choices, the outcomes of these studies may vary depending upon these choices. Therefore, the influence of different value and modelling choices on the outcomes of the ASLCA was investigated, particularly regarding the choice of GHG accounting method, the choice of climate threshold, the choice of approach to calculate the global carbon budget, and the choice of sharing principle to assign a share of the global carbon budget. The analysis showed that, for each GHG accounting method the largest uncertainty was associated with the choice of climate threshold, followed by the choice of sharing principle, and then the choice of calculation method for the global carbon budget.

Overall, the proposed ASLCA framework aims to address the question, "Are the environmental impacts of a system within the assigned share of the Earth's carrying capacity, and if not, what is the required reduction?" The outcomes of this research are useful to support policymakers in understanding the climate impacts of different economic sectors, goods and services, relative to global climate targets. The approach provides a basis for developing a range of environmental impact reduction targets that can potentially catalyse innovation and investment in the environmentally-transformative activities and technologies that are needed to enable human societies to operate and develop within the Earth's "safe operating space".

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Declaration

The thesis complies with the 'Guidelines for Doctoral Thesis by Publications' and with the requirements from the Handbook for Doctoral Study by the Doctoral Research Committee (DRC), Massey University. January 2011. Version 7.

Disclaimer

The opinions, figures and conclusions in the thesis are solely those of the author(s). Under no circumstances will the author(s) be responsible for any loss or damage of any kind resulted from the use of these methods and techniques presented in the thesis.

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Chapter 1: Introduction

1.1. Sustainable development

Global challenges such as climate change, resource depletion, air pollution, poverty, food insecurity and biodiversity loss have received increasing attention from multiple stakeholders, including academics, businesses, global agencies, governments and the general public (FAO et al. 2017; IUCN 2017; UN 2015; WWF 2017). In particular, the concept of *sustainable development* received much attention after the publication of the Brundtland report (WCED 1987). The concept focuses on how to meet the needs of today's human societies without compromising the requirements of future generations (UN 2015; WCED 1987). Achieving sustainable development is, however, not straightforward as it requires addressing the *triple bottom-line* (such as economic, social and environmental) and a number of quantitative and qualitative variables at multiple spatial and temporal dimensions (Banerjee 2003; Brandi 2015; Kim and Bosselmann 2015).

To make the concept more tangible and relevant for policy-makers, researchers attempted to relate it to neoclassical economic theory through the capital approach to sustainability and by proposing a set of indicators and targets related to economic systems. In economics, capital is defined as "a stock that yields a flow of valuable goods or services", which can be regarded as a physical condition for human welfare (McElroy and van Engelen 2012). Capital is generally classified into three categories: (i) *manufactured* or *reproducible capital* (e.g. roads, machinery, buildings); (ii) *human capital* (e.g. skills, education, health); and (iii) *natural capital* (e.g. land, forests, fossil fuels, fisheries) (e.g. Barbier and Burgess 2017; Ekins et al. 2003).

Since natural capital is not man-made and it provides necessary goods and services to the economy in terms of natural functioning and habitat, it needs to be treated differently from the other two types of capital (Barbier and Burgess 2017; Ekins et al. 2003). For example, an ecosystem can be considered as natural capital as it provides necessary services such as production of goods (e.g. food, timber), life support processes (e.g. pollination, air and water purification) and social conditions (e.g. beauty and serenity). The capital approach to sustainability also suggests that the value of the aggregate stock of all three types of capital (i.e. manufactured, human and natural) should be increased or maintained to ensure that overall welfare does not decline over time (Barbier and Burgess 2017). However, in this approach, there is a fundamental debate on whether to adopt *weak* or *strong sustainability*.

1.2. Weak versus strong sustainability

Weak sustainability does not differentiate the unique or essential roles of different types of capital and assumes that they are substitutable (Brekke 1997; Daly et al. 1994; Pezzey 1992). According to this conception, the value of the aggregate stock of different capitals should be increased or at least maintained over time, and there is a general expectation that technical solutions can compensate the environmental impacts that are related to the supply of manufactured and human capital (e.g. construction of a water treatment plant in place of a wetland to provide the service of water filtration). Effectively, this means that achieving socio-economic growth to supply manufactured and human capital at the expense of environmental degradation is acceptable (Daly et al. 1994; Jacobs and Stott 1992).

On the other hand, strong sustainability suggests that the different types of capital should not be considered substitutable and some natural capital that is crucial for Earth system functioning (referred to as "critical natural capital") should be maintained, for example, ecosystems, biodiversity and life-support functions (Chiesura and de Groot 2003; Ekins et al. 2003). The critical natural capital is subject to irreversible loss, has certain thresholds and cannot be substituted by either manufactured or human capital (Barbier and Burgess 2017; Ekins et al. 2003). For example, a building can be renovated or reconstructed if it is destroyed but the extinction of a species from an ecosystem is irreversible. Moreover, in many cases, production of manufactured capital relies on natural capital; for example, building construction largely requires raw materials that are extracted from the Earth or harvested from ecosystems (e.g. mineral ores such as iron, aluminium, and wood). Achieving strong sustainability therefore requires addressing all three types of capital in a complementary way while preserving the critical natural capital. There are parallels here with the *Māori worldview* (*te ao Māori*). According to the Māori worldview, there is a strong interconnection between people and nature, and all flora and fauna (a concept called *whakapapa*); therefore, people should be responsible for understanding and managing all the natural resources (the concept of *ki uta ki tai*) while preserving the highly valued natural resources (called *taonga*), and passing them to the next generation, in a caring and respectful manner (the concept of *taonga tuku iho*) (Harmsworth et al. 2016; Lyver et al. 2017).

In reality, many organisations (including governments and businesses) adopt weak sustainability, since their main objective is to achieve socio-economic growth on behalf of their stakeholders (Bjørn and Røpke 2018; Kim and Bosselmann 2015). As a result, environmental impacts environmental impacts have been increasing in recent years, and key Earth system boundaries, including climate change biogeochemical flows, land-system change and biodiversity loss, are being transgressed globally as well as locally (Rockström et al. 2009; Steffen et al. 2015). Human societies, therefore, need to urgently address two interrelated wicked problems on a global scale: (i) how to protect the entire Earth system and its subsystems (particularly critical natural capital); and (ii) how to operate and develop socio-economic systems within the Earth system boundaries. Hence, a strong sustainability conception is crucial for sustainable development.

1.3. Relative versus absolute environmental sustainability

Adoption of a strong sustainability conception in sustainable development also requires understanding the differences between *relative* and *absolute environmental sustainability*. Until recently, a relative environmental sustainability approach has been used to mitigate the environmental impacts of multiple systems at different economic levels (Bjørn and Hauschild 2015; Hauschild 2015; Kara et al. 2018). Here, a system can be either a product, process, company, sector, country or the entire Earth. For example, existing quantitative environmental sustainability assessment methods such as Life Cycle Assessment (LCA, ISO 2006; 2012) and environmental footprints (ISO 2013, 2014) quantify the environmental impacts of a system and compare it to a system which is similar to the nature or the function of the examined system (e.g. Coelho and McLaren 2013; Hauschild 2015; Moldan et al. 2012; Singh et al. 2009). Hence, they are referred to as *relative environmental sustainability assessment* (RESA) methods, and their indicators are referred to as *relative environmental sustainability indicators* (RESI) (Bjørn et al. 2016). Although these RESA methods provide useful information to improve the eco-efficiency of the system, they generally fail to inform the environmental sustainability performance of a system against the so-called *absolute environmental boundaries* (e.g. Bjørn et al. 2016; Chandrakumar and McLaren 2018a; Hauschild 2015). Thus, contributions of any of the examined systems to the overall environmental impacts of production and consumption activities of human societies are generally overlooked (Chandrakumar et al. 2019b; Hauschild 2015).

To address the limitation of the relative environmental sustainability approach, researchers have suggested adopting an *absolute environmental sustainability* approach, which addresses how human societies can operate and develop within absolute environmental boundaries (Bjørn and Hauschild 2015; Ryberg et al. 2018b). Nevertheless, operationalising absolute environmental sustainability is not straightforward; it involves addressing a number of challenges such as prioritising key environmental impacts; defining appropriate indicators and boundaries; evaluating impacts at multiple economic levels; and understanding the interaction between various types of environmental impacts (Chandrakumar and McLaren 2018a, b; Fang et al. 2015a; Häyhä et al. 2016; Ryberg et al. 2016). Therefore, the development of so-called *absolute environmental sustainability assessment* (AESA) methods is crucial in the pursuit of absolute environmental sustainability. An AESA method aims to answer the question, "Are the environmental impacts of a system within the assigned share of the Earth's carrying capacity[1], and if not, what is the required reduction?" (Bjørn et al. 2018).

1.4. Absolute environmental sustainability assessment methods

Rockström et al. (2009a, b) introduced the planetary boundaries (PBs) framework that informs how current human society is operating in terms of nine critical Earth system boundaries. Since then, researchers have been investigating how to address absolute environmental sustainability at sub-global levels (Clift et al. 2017; Muñoz and Sabag 2017): regional (e.g. Heijungs et al. (2014) for Europe), national (e.g. Nykvist (2013) for Sweden, Dao et al. (2018) for Switzerland, Cole et al. (2014) for South Africa), sector (e.g. Krabbe et al. (2015) for steel, SBT (2017) for power generation), and product levels (e.g. Brejnrod et al. (2017) for buildings, Sandin et al. (2015) for garments). Even though the PBs provide

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[1] The term 'carrying capacity' refers to "…the maximum sustained environmental intervention a natural system can withstand without experiencing negative changes in structure or functioning that are difficult or impossible to revert" (Bjørn and Hauschild 2015).

a scientific basis for addressing absolute environmental sustainability at multiple economic levels, there are a number of outstanding challenges in achieving absolute environmental sustainability, and in development and use of AESA methods in pursuit of this objective (Häyhä et al. 2016; Laurent and Owsianiak 2017; Ryberg et al. 2016). Regarding AESA methods, Chandrakumar and McLaren (2018b) advocated that an AESA should:

1. Quantitatively assess a comprehensive range of environmental impacts in absolute terms;
2. Inform the environmental impacts at an early stage in impact pathways; and
3. Be capable of use at multiple economic levels.

Regarding assessment at multiple economic levels, methods such as LCA and environmental footprints can be used to calculate the environmental impacts associated with a chosen system. However, research on developing absolute environmental boundaries is at an early stage. According to Häyhä et al. (2016), when developing boundaries at sub-global levels, three key dimensions should be considered. The first dimension is biophysical which characterises a particular environmental impact as a *systemic* process with global boundaries (e.g. climate change, ozone depletion) or an *aggregated* process with regional or local global boundaries (e.g. biogeochemical flows, land-system change). The second dimension is socio-economic that evaluates the interactions between socio-economic activities and environmental impacts (i.e. choice of socio-economic parameter [e.g. final consumption expenditure, gross value addition, employment] to investigate environmental impacts). The third dimension is ethical which addresses the challenge of downscaling global boundaries (particularly, systemic processes) to other sub-global levels (e.g. country, sector, company, product) and what ethical principles[2] (or rationales) should guide this process. Some researchers have already started addressing these dimensions in proposed methods to downscale the global boundaries of systemic processes to sub-global levels (e.g. Bjørn 2015; Sandin et al. 2015), while others are developing regional (or local) boundaries for aggregated processes (e.g. Dearing et al. 2014; Li et al. 2019; Teah et al. 2016).

Regarding the ethical dimension, several methods are proposed in the literature that guide downscaling the global boundaries in proportion to past environmental impacts,

[2] Also known as sharing principles (e.g. Bjørn et al. 2018; Ryberg et al. 2018a).

population, mass, economic value, and relative contribution to human well-being (e.g. Fang et al. 2015b; Liu and Bakshi 2018; Ryberg et al. 2018a). For a detailed discussion, see Bjørn (2015). Moreover, in the Science Based Targets (SBT 2017) initiative, methods like the Greenhouse Gas Emissions per unit of Value-Added (GEVA) and the Sectoral Decarbonization Approach (SDA) have been proposed to guide companies to set GHG emissions reduction targets. The GEVA recommends a carbon intensity reduction rate of 5% per year for all economic sectors and their companies, suggesting that it is sufficient to achieve a 50% reduction in GHG emissions globally in 2050. The method, however, fails to account for heterogeneities between regions, countries, sectors or companies (e.g. current environmental and economic performances, mitigation potentials, costs).

This limitation was later addressed through the SDA, which translated the sectoral emission pathways[3] of the International Energy Authority (IEA 2014) into sectoral intensity pathways[4] using physical (e.g. tonne of cement manufactured) or economic (e.g. value added) indicators, and then translated them into individual company intensity pathways. However, the SDA does not address all economic sectors, activities, or types of GHG emissions (SBT 2017). It focuses only on *homogenous sectors*[5] (e.g. power generation, iron and steel, cement and transport) and does not address *heterogeneous sectors*[6] such as agri-food and construction. One key reason is that the outputs of the heterogeneous sectors are highly diverse and their supply chains are relatively complex (Giesekam et al. 2018; Pero et al. 2017). The SDA only accounts for carbon dioxide (CO_2) emissions and omit other GHG emissions such as methane (CH_4) and nitrous oxide (N_2O), which are particularly relevant to agri-food sector (and its companies). Furthermore, proposed emissions pathways for the construction sector are generally based upon the IEA's 'other industry' sector, which also includes industries such as agriculture, food, beverage and tobacco processing, and fishing. As a result, predicting a credible future emissions scenario for the construction sector is difficult, and therefore it is challenging to set GHG emissions reduction targets. Hence,

[3] Emission pathways provide information on how emissions are likely to develop in future by accounting for the past emissions and future economic and technological developments.
[4] Intensity pathway= emission pathway/activity projections; here, an activity can be either physical or economic activity.
[5] A sector that can be represented using a single physical indicator due to the uniqueness of the characteristics of the sector and its outputs (SBT 2017).
[6] A sector that cannot be represented using a single physical indicator due to the difficulties in comparing the outputs (SBT 2017).

the SDA is best suited for companies in the homogenous sectors (Clift et al. 2017; Giesekam et al. 2018).

Furthermore, several environmental policies and/or initiatives are being developed and implemented at global as well as national levels to mitigate a range of environmental impacts; many of them are centred on climate change. For example, the Kyoto Protocol was adopted in 1997 to implement the objective of the United Nations Framework on Climate Change (UNFCCC 1998) to reduce global warming by reducing GHG emissions of individual countires. In 2016, the Paris Agreement was signed by 196 parties at the 21st Conference of the Parties of the UNFCCC with the long-term goal of limiting global average temperature increase to well below 2°C above pre-industrial levels and to limit the increase to 1.5°C since this would substantially reduce climate risks (IPCC 2018; UNFCCC 2015). Likewise, more recently, the New Zealand government introduced a new Zero Carbon Bill with the aim of reducing all emissions (except biogenic methane) to net zero by 2050 and reducing biogenic methane emissions within the range of 24-47% below 2017 levels by 2050[7] (Shaw 2019). While these global and national policies provide a basis to undertake AESAs, it should be noted that they generally represent a compromise between scientific knowledge and societal considerations (e.g. political feasibility, cost); hence they are mostly less strict than science-based absolute environmental boundaries such as the PBs (Acosta-Alba and Van der Werf 2011).

1.5. Environmental impacts of agri-food and construction sectors

Agri-food and construction are two of the primary sectors that significantly contribute to the increasing environmental impacts, both on a local and global level (e.g. Steffen et al. 2015; UNEP 2007; Willett et al. 2019). For example, agriculture occupies about 40% of global land (Foley et al. 2005; Springmann et al. 2018), and food production contributes up to 30% of global GHG emissions[8] (Vermeulen et al. 2012), and 70% of freshwater consumption (Steffen et al. 2015). Furthermore, the transformation of natural ecosystems to croplands and pastures is recognised as the largest factor resulting in loss of biodiversity (Tilman et al. 2017). Overuse (and misuse) of phosphorous and nitrogen cause

[7] This includes an additional target of reducing biogenic methane emissions to 10% below 2017 by 2030.
[8] This value included the GHG emissions associated with land use and land system change.

eutrophication in lakes and coastal areas (Carpenter and Bennett 2011; de Vries et al. 2013). The environmental impacts of the agri-food sector also include marine ecosystems. According to Klinger and Naylor (2012), almost 60% of world fish stocks are completely fished, more than 30% overfished, and catch by global marine fisheries has been rapidly decreasing. At the same time, the growing aquaculture industry can negatively impact freshwater, marine, and terrestrial ecosystems.

Similarly, the construction sector contributes to a number of environmental impacts including climate change (UNEP 2007), excessive consumption of global resources (Ding 2008), environmental (including water and air) pollution (Vyas et al. 2014; Yılmaz and Bakış 2015), and waste generation (Ajayi et al. 2016). The sector uses around 40% of energy and 40% of raw materials globally, while contributing to approximately 30% of global GHG emissions, 23% of air pollution, and 40% of solid wastes in cities (Sev 2009; UNEP 2007; Yılmaz and Bakış 2015).

On the other hand, the global demand for food and buildings is growing rapidly, which will obviously contribute to more environmental impacts in future. This, therefore, poses a challenge for policymakers and stakeholders i.e. How to operate and develop these sectors within the absolute environmental boundaries? In that context, researchers have already started investigating the development of absolute environmental boundaries (and AESA methods) for the agri-food and construction sectors. For example, Springmann et al. (2018) and Willett et al. (2019) developed absolute environmental boundaries (for six Earth system processes[9]) for the agri-food sector at the global level and benchmarked the impacts against those boundaries. Their study showed that the sector has already transgressed its boundaries; therefore, operating the sector within the proposed boundaries requires a synergistic combination of multiple measures including dietary changes towards healthier diets, improvements in technologies and management, and reductions in food loss and waste.

Similar efforts to evaluate the environmental impacts of buildings using an absolute environmental sustainability approach exist (e.g. Brejnrod et al. 2017; Hollberg et al. 2019;

[9] Six Earth systems processes include: greenhouse gas emissions, cropland use, water use, nitrogen application, phosphorous application and loss of biodiversity.

Russell-Smith et al. 2015). Russell-Smith et al. (2015) proposed the so-called Sustainable Target Value (STV) method to calculate GHG emissions targets for office buildings. The targets were based on the IPCC Fourth Assessment Report (AR4, IPCC 2007), which had modelled a 70-80% GHG emissions reduction below 1990 levels by 2050 for buildings. Likewise, Hollberg et al. (2019) suggested that the climate impact of a global citizen should be limited to 1 tonne carbon dioxide equivalent per capita per annum [$tCO_2eq \cdot cap^{-1} \cdot yr^{-1}$] by 2050 to stay within the 2 degree Celsius (°C) climate target, according to the *2000 Watt society vision* (Heeren et al. 2012). They subsequently set a climate target for a Swiss single-family house based on the relative contribution of the Swiss residential sector to the national GHG emissions[10]. However, these studies were limited in several aspects. In particular, while both studies have considered population growth when setting climate targets for buildings in 2050, none of them has modelled the growth in the number and size (i.e. floor area) of buildings nationally and/or globally (through to 2050). However, temporal aspects such as the growth in the number and size of buildings are critical in determining the available share of the carrying capacity for a building, and should be addressed when setting environmental impacts reduction targets for future buildings. Furthermore, the existing studies have proposed a single environmental target value for the whole life cycle of a building, and it would be challenging for building designers to use the proposed target as a guide in the design process given the lack of transparency regarding environmental hotspots.

Overall, following the introduction of the PBs framework, there is growing interest in developing AESA methods that can evaluate the environmental sustainability performance of different economic systems from an absolute environmental sustainability perspective. However, the existing AESA studies largely evaluate the environmental impact categories listed in the PBs, and they primarily focus on the homogeneous sectors. In particular, no study, to date, has investigated the environmental performance of agri-food systems at sub-global levels using an absolute environmental sustainability approach. Moreover,

[10] This was according to the *grandfathering* sharing principle (Chandrakumar et al. 2018), which assigned a share of the carbon budget to the residential sector based on its relative contribution to the national GHG emissions.

evaluation of the environmental performance of buildings in absolute terms requires further research.

1.6. Research aim and objectives

This research investigated the development of an innovative AESA framework called 'Absolute sustainability-based Life Cycle Assessment' (ASLCA) based on the environmental indicators and absolute environmental boundaries proposed in three popular frameworks: Planetary Boundaries (PBs), Sustainable Development Goals (SDGs) and Life Cycle Assessment (LCA). The specific objectives were to:

1. Investigate the interlinkages between the three frameworks: Planetary Boundaries, Sustainable Development Goals and Life Cycle Assessment;
2. Identify key environmental impact categories, environmental indicators and absolute environmental boundaries to underpin the development of ASLCA;
3. Investigate how the absolute environmental sustainability approach can be applied in the agri-food and construction sectors (at sub-global levels, e.g. New Zealand);
4. Understand the policy implications of the results of an AESA study, including the role of different methodological and value choices.

1.7. Structure of the thesis

The thesis comprises eight chapters. **Chapter 1** introduces the overall rationale, relevant literature, and research aim and objectives. Chapters 2-7 were developed sequentially to address each research objective individually or in combination.

The interlinkages between the Sustainable Development Goals (SDGs) and Planetary Boundaries (PBs) are investigated in **Chapter 2** by mapping the environmental indicators proposed in the two frameworks on to a Driver-Pressure-State-Impact-Response (DPSIR) causal framework. There is a substantial overlap between the SDGs and PBs; therefore, the absolute environmental boundaries proposed in the PBs can be used as a complementary set of environmental boundaries for the SDGs.

In **Chapter 3**, the Life Cycle Assessment (LCA) indicators and areas of protection are additionally mapped on to the DPSIR causal framework developed in Chapter 2. A set of

12 key environmental impact categories and an associated set of environmental indicators and absolute environmental boundaries are identified.

The development of the ASLCA framework is described in **Chapter 4** and the usefulness of the framework is illustrated by evaluating the production-based climate change performance of New Zealand agri-food systems relative to the two-degree Celsius (2°C) threshold. The study shows that these agri-food systems exceed the assigned shares of the 2°C global carbon budget, and also illustrate the scale of change necessary for agri-food systems to operate within their carbon budget shares. Note that the choice of the 2°C threshold was based on the availability of data at the time of this research, and it was the same as the climate threshold adopted in the Science Based Targets initiative (SBT 2017).

The ASLCA framework was adapted to calculate GHG emissions targets for future buildings, and used to calculate a GHG emissions target for the most common type of residential building in New Zealand i.e. detached house (**Chapter 5**). This study illustrates how future economic activities (including construction) can be planned and undertaken within the remaining share of the 2°C global carbon budget.

The proposed ASLCA framework was then applied to address the consumption-based climate change performance of an economic system. However, given the calculation of consumption-based climate change performance of a system requires more sophisticated methods and a significant amount of data, the environmentally extended multi-regional input-output (MRIO) analysis was introduced in **Chapter 6**. Using the methodology, the consumption-based GHG emissions of New Zealand's total economy (covering 16 sectors) for the year 2012 were calculated, and compared with the production-based GHG emissions. The results indicate that the country was a net carbon exporter in 2012, and the dominant contributing sectors to New Zealand's consumption- and production-based GHG emissions were very different. Thus, both approaches should be used in a complementary way when developing climate policies.

Acknowledging that AESA methods (including ASLCA) are built upon multiple value and modelling choices, the outcomes of these studies may vary depending upon these choices. Therefore, the influence of different value and modelling choices on the outcomes of the ASLCA was investigated in **Chapter 7**, particularly with regard to the choice of GHG accounting method, the choice of climate threshold, the choice of approach to calculate the global carbon budget, and the choice of sharing method to assign a share of the global carbon budget. It was found that, for each GHG accounting method, the largest uncertainty was associated with the choice of climate threshold, followed by the choice of sharing principle, and then the choice of calculation method for the global carbon budget. Policymakers should, therefore, be aware of these uncertainties, and take them into consideration when developing national climate policies.

Chapter 8 concludes with recommendations for future research.

1.8. List of publications/manuscripts

While **Chapter 2** is a published peer-reviewed book chapter, **Chapters 3-7** are published/submitted articles in different peer-reviewed scientific journals.

- **Chapter 2**: Chandrakumar C, McLaren SJ (2018a). Exploring the Linkages between the Environmental Sustainable Development Goals and Planetary Boundaries Using the DPSIR Impact Pathway Framework. In: Benetto E, Gericke K, Guiton M (eds) Designing Sustainable Technologies, Products and Policies: From Science to Innovation. Springer Cham, Switzerland, pp 413-423. doi:https://doi.org/10.1007/978-3-319-66981-6_46
- **Chapter 3**: Chandrakumar C, McLaren SJ (2018b). Towards a comprehensive absolute sustainability assessment method for effective Earth system governance: Defining key environmental indicators using an enhanced-DPSIR framework. Ecological Indicators 90:577-583. doi:https://doi.org/10.1016/j.ecolind.2018.03.063
- **Chapter 4**: Chandrakumar C, McLaren SJ, Jayamaha NP, Ramilan T (2019a). Absolute Sustainability-based Life Cycle Assessment (ASLCA): A Benchmarking Approach to Operate Agri-food Systems within the 2°C Global Carbon Budget. Journal of Industrial Ecology 23(4):906-917. doi:https://doi.org/10.1111/jiec.12830
- **Chapter 5**: Chandrakumar C, McLaren SJ, Dowdell D, Jaques R. An absolute sustainability approach for setting climate targets for buildings: the case of a New Zealand detached house. Submitted to Building and Environment

- **Chapter 6**: Chandrakumar C, McLaren SJ, Malik A, Ramilan T, Lenzen M (2019b). Understanding New Zealand's consumption-based greenhouse gas emissions: an application of multi-regional input-output analysis. Submitted to International Journal of Life Cycle Assessment. doi:https://doi.org/10.1007/s11367-019-01673-z
- **Chapter 7**: Chandrakumar C, Malik A, Mikołaj O, McLaren SJ, Ramilan T, Jayamaha NP, Lenzen M (under review). Assessing the climate change performance of New Zealand's economy using an absolute environmental sustainability assessment method: the influence of value and modeling choices. Submitted to Environmental Science & Technology

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Chapter 2: Exploring the linkages between the environmental Sustainable Development Goals and Planetary Boundaries using the Driver-Pressure-State-Impact-Response (DPSIR) impact pathway framework

Chanjief Chandrakumar and Sarah J McLaren (2018)

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Abstract

Most of the conventional environmental sustainability assessment methods, such as life cycle assessment and environmental footprints, evaluate economic goods and services in terms of the nature or the function of the studied systems. As such, these methods in general do not inform the variations in the overall magnitude of production and consumption patterns for the examined systems, specifically in terms of Earth system boundaries. As a result, the progress achieved in mitigating global environmental problems may be considered as slow. Hence this study explores the linkages between the Sustainable Development Goals (SDGs) and Planetary Boundaries (PBs) using the DPSIR (Drivers-Pressures-State of the Environment-Impacts-Responses) impact pathway framework- in support of developing absolute environmental sustainability assessment (AESA) method. The study demonstrates that there is a substantial overlap between the SDGs and PBs. The science-based thresholds listed in the PBs can therefore be adopted as a complementary set of environmental boundaries for the SDG indicators. Overall, the study lays the foundation for advancing an AESA method that can guide policy- and decision-makers to operationalize the SDGs effectively.

Keywords: absolute sustainability; absolute environmental sustainability assessment; planetary boundaries; life cycle assessment; sustainable development goals; DPSIR; causal network

2.1. Introduction

Planetary Boundaries (PBs) concept was introduced by Rockström and his associates (Rockström et al. 2009a, b), and was updated by Steffen et al. (2015). Rockström et al. (2009a) proposed nine critical Earth system processes and associated control variables and thresholds, claiming that transgressing any of the thresholds would potentially be devastating for human societies. Based on the nine PBs, a safe operating space for humanity was determined (Rockström et al. 2009a; Steffen et al. 2015). Here, the safe operating space refers to a relatively stable state called the *Holocene* epoch, in which human societies can continue to develop and thrive (Steffen et al. 2015). Today, both the scientific and political communities have agreed upon the notion that there are global limits for the Earth system and they should be respected. Consequently, studies adopting the PBs have started proliferating, and they can be classified into works that (i) define or refine the control variables and the associated thresholds (e.g. Rockström et al. 2009b; Steffen et al. 2015), (ii) downscale the global PBs to sub-global levels (e.g. Roos et al. 2016; Sandin et al. 2015), (iii) set impact reduction targets (e.g. Roos et al. 2016; Sandin et al. 2015), and (iv) devise policies and strategies (e.g. Nykvist 2013; Roos et al. 2016).

While the PBs concept reports global limits for environmental impacts to benchmark a system's performance globally, environmental sustainability assessment methods (ESAMs) such as life cycle assessment (LCA) and environmental footprints evaluate the environmental performance of a so-called product system (which is usually defined in terms of supplying a specified quantity of an economic product or service). Generally, the outcomes of an LCA or environmental footprint study guide decision-makers to improve the eco-efficiency of the chosen product system by identifying the environmental hotspots along its life cycle (ISO 2006). As a result, use of LCA and other related life cycle thinking approaches to support decision-making has become common within the academic and business communities (Bjørn and Hauschild 2015). However, although the outcomes of these studies underpin eco-efficiency improvements, the overall progress achieved in mitigating environmental problems can be considered as slow and insignificant (Bjørn and Hauschild 2013; Bjørn et al. 2016; Hauschild 2015). One contributing factor is that the conventional ESAMs like LCA do not benchmark the environmental sustainability performance of a system against a set of environmental boundaries (or standards). Instead, they rank a particular system in relative terms, by comparing it with a reference system that is relevant to the nature or the function of the system under investigation. As a result, the variations in the consumption and production patterns of the examined products and services are generally overlooked (Bjørn et al. 2016; Hauschild 2015). For example, Product A may be superior (or more sustainable) than Product B in terms of eco-efficiency, but neither could be sustainable on an absolute scale due to the predicted growth in global production and consumption volumes of the product (Bjørn and Hauschild 2013).

Therefore, recently, the scientific community began to focus on the so-called concept of *absolute environmental sustainability*. Absolute environmental sustainability is focused on how human societies can operate within the *carrying capacity* of the Earth system (Bjørn et al. 2016; Hauschild 2015). Here, the term "carrying capacity" refers to "the maximum sustained environmental interference a particular system can withstand without experiencing negative changes in structure or functioning that are difficult or impossible to revert" (Bjørn and Hauschild 2015, p. 1007). As a result of growing interest in absolute sustainability, scientists have started developing absolute environmental sustainability assessment (AESA) methods by supplementing the existing ESAMs with the Earth's carrying capacity (e.g. Bjørn and Hauschild 2015; Bjørn et al. 2016; Fang et al. 2015a, b; Hauschild 2015), for instance, supplementing the ecological footprint with the Earth's bio-capacity (available bio-productive area) (e.g. Borucke et al. 2013), LCA with PBs (e.g. Bjørn and Hauschild 2015; Sandin et al. 2015) and other environmental footprints with PBs (e.g. Nykvist 2013).

2.2. Operationalisation of sustainable development goals

The United Nations agreed on a set of Sustainable Development Goals (SDGs) in 2015 comprising 17 goals, 169 targets and 232 indicators (IAEG-SDGs 2017; UN 2015). The SDGs address a wide range of sustainable development problems (Brandi 2015; Wackernagel et al. 2017). Overall, the SDGs are intended to be universal with a shared common vision of progressing towards a safe, just and sustainable operating space for human societies (IAEG-SDGs 2017; Maier et al. 2016). However, the SDGs have been criticised as being difficult to implement due to having too many goals and targets, lacking clarity, and having overlapping objectives (Brandi 2015; Maier et al. 2016). Additionally, the SDG proposed for safeguarding the Earth system have been criticised as being neither sufficiently comprehensive nor ambitious (Brandi 2015; Wackernagel et al. 2017). For instance, many of the SDG indicators have been proposed without a relevant environmental boundary. Researchers, therefore, have begun exploring how to operationalize the SDGs within the Earth's carrying capacity, and specifically how to link them to the PBs and then to LCA (Brandi 2015; Dong and Hauschild 2017). Dong and Hauschild (2017) classified the indicators proposed in the SDGs, PBs and LCA using the DPSIR (Drivers-Pressures-State of the Environment-Impacts-Responses) impact pathway framework (see Song and Frostell (2012) for DPSIR impact pathway framework) and showed that all three approaches overlap in terms of seven impact categories (climate change, acidification, ozone depletion, eutrophication, chemical pollution, freshwater use and change in biosphere integrity). However, note that the study had only used the older version of the SDGs listed in UN (2015) and no studies have been identified that explore the interlinkages between the latest SDGs listed in IAEG-SDGs (2017), PBs and LCA. Additionally, until recently, the potential for operationalizing the SDGs using an AESA method has not been explored. To that end, this study identifies the SDG indicators that evaluate environmental problems, and then systematically explores the interlinkages with the PBs.

The rest of the chapter is organised as follows: Section 3 outlines the AESA framework presented in Chandrakumar et al. (2017), Section 4 establishes the linkages between the environmental SDGs and PBs, and Section 5 summarises how this work underpins the development of the proposed AESA.

2.3. Outline of the proposed approach

The aim of the proposed AESA is to operationalize the SDGs at sub-global levels (e.g. country, region, organisation, and product) by estimating environmental boundaries at these different levels (Chandrakumar et al. 2017). These boundaries can then be used to calculate distance-to-target measurements through benchmarking the system's (e.g. country, region, organisation, product) performance against the estimated boundaries. This involves, firstly, identifying the SDG indicators concerned with the conventional areas of protection (AoPs) used in LCA i.e. *human health*, *ecosystem quality*, *resource scarcity* and *man-made environment* (Huijbregts et al. 2016; ISO 2006).

According to Bjørn and Hauschild (2015) and Ryberg et al. (2016), many of the PB control variables differ from the indicators of the conventional ESAMs (including LCA), particularly with respect to the point of impact evaluation, although these indicators evaluate similar kinds of environmental impacts to those reported in the PBs. Hence, the chosen SDG indicators and the PB control variables are further classified into driver, pressure, state, impact and response indicators using the DPSIR framework explained by Song and Frostell (2012). Having classified them, the linkages between the SDGs and PBs are explored. This enables subsequent development of a complementary set of global boundaries for the SDG indicators using, where appropriate, the thresholds proposed for the control variables in the PBs. Afterwards, the global boundaries can be allocated to lower economic levels using a top-down approach. The method is operationalised by developing distance-to-target measurements based on the calculated environmental boundaries compared with the current environmental performance of the systems under analysis (calculated using conventional ESAMs like LCA and environmental footprint studies). These distance-to-target measurements could be positive or negative depending on whether the system is performing in line with the goals and targets reported in the SDGs.

2.4. Linkages between the sustainable development goals and planetary boundaries

This section details how the PBs can be employed as a complementary set of global boundaries for the SDGs by providing a systematic comparison between the SDG indicators and the PB control variables. Firstly, the SDG indicators concerned with the AoPs of *human health*, *ecosystem quality*, *resource scarcity* and *man-made environment* were chosen (a total of 73 indicators). This set of SDG indicators comprised all the SDG indicators under the SDGs for clean water and sanitation (SDG 6), responsible consumption and production (SDG 12), climate action (SDG 13), life below water (SDG 14) and life on land (SDG 15), plus CO_2 emission per unit of value added (SDG indicator 9.4.1), economic loss due to natural disasters (SDG indicator 1.5.2), levels of fine particulate matter in cities (SDG indicator 11.6.2), and proportion of land for sustainable agriculture (SDG indicator 2.4.1). Then, as outlined in Section 3, the chosen SDG indicators were mapped onto a network of cause-effect chains (developed based on the environmental problems addressed in the SDGs and PBs) along with the PB control variables and linked together wherever relevant (see Figure 2. 1). This mapping step further classified the SDG indicators into driver (0 SDG indicators), pressure (2 SDG indicators), state (19 SDG indicators), impact (14 SDG indicators) and response (38 SDG indicators) indicator categories.

As emphasised in Rockström et al. (2009a) and Steffen et al. (2015), human societies should be operating within the thresholds put forward in the PBs. Therefore, this section focuses on the PBs and discusses how each PB (shown in bold text) is related to different SDGs. Steffen et al. introduced a PB called **freshwater use** and two control variables to evaluate the challenges resulting from absolute water withdrawals (Steffen et al. 2015). The proposed control variables estimate the associated impacts at the global as well as the basin levels, and inform the impacts at the pressure and the state point of the DPSIR impact pathway, respectively (see Figure 2. 1). Meanwhile, SDG indicator 6.4.1 evaluates the water use efficiency at the pressure point in the impact pathway (in terms of water use efficiency); and at the state point, SDG indicator 6.4.2 accounts for the effects of excessive water withdrawals (i.e. the level of water stress), which are similar to the PB control variables (IAEG-SDGs 2017). However, the proposed SDG indicators do not include any absolute limits. We, therefore, recommend deploying the thresholds proposed for the freshwater use PB because the control variables and the SDG indicators largely overlap; and both inform the impacts at the same points in the impact pathway.

Increasing atmospheric CO_2 concentration and the associated CO_2 uptake by the oceans have resulted in ocean acidification problems. As a consequence, a PB called **ocean acidification** was introduced with a control variable (state point) and a threshold for carbonate ion concentration in terms of aragonite (Steffen et al. 2015). Meanwhile, the SDGs advanced an indicator (SDG indicator 14.3.1) that estimates the pH level of the oceans (IAEG-SDGs 2017). Although the control variable and the SDG indicator apply different units to track the ocean acidification effects, the objective and the point of assessment in the DPSIR framework are the same. However, SDG indicator 14.3.1 can be considered as relative since it does not include any relevant targets for ocean acidification impacts, whereas the ocean acidification PB control variable does. We, therefore, suggest using the PB control variable and its threshold for assessing ocean acidification impacts.

The **changes in biosphere integrity** PB adopts two control variables to assess the two components of the biosphere: genetic and functional diversity (Rockström et al. 2009a; Steffen et al. 2015). The first component evaluates the extinction of species due to human pressures, whereas the second estimates the loss of biodiversity at different ecosystem levels. According to Figure 2. 1, the impacts pertaining to both components are expressed at the impact point of the DPSIR impact pathway. In this regard, the SDGs also propose a set of indicators for protecting terrestrial, marine and freshwater ecosystems (IAEG-SDGs 2017). SDG indicators 14.4.1 and 14.5.1 estimate the proportion of fish stocks existing within the biologically sustainable levels and the coverage of protected marine areas, respectively. Moreover, SDG indicator 6.6.1 tracks the changes occurring in both marine and freshwater ecosystems due to water quality degradation. Although these SDG indicators implicitly underpin the significance of operating within the Earth's carrying capacity, no relevant boundaries have been reported. However, given that the objectives of these control variables overlap with the SDG indicators, it makes sense to supplement the SDG indicators with the thresholds proposed for the "changes in biosphere integrity" PB to inform the environmental impacts in terms of genetic and functional diversities on an absolute scale.

Considering the intensive use of nutrients and the associated eutrophication effects in major ecosystems, Steffen et al. proposed the so-called **biogeochemical flows** PB and two associated control variables (Steffen et al. 2015). These control variables evaluate the eutrophication effects in oceanic, freshwater and terrestrial ecosystems. Given the major eutrophication problems arise from N and P fertiliser use and the control variables evaluate the impacts at the pressure point (as shown in Figure 2. 1), thresholds have been set for N and P fertiliser application (Steffen et al. 2015). Likewise, SDG indicator 14.1.1 evaluates the problem of marine eutrophication resulting from land-based activities, including nutrient pollution (IAEG-SDGs 2017). Since both SDG indicator 14.1.1 and the relevant control variables refer to the same problem of eutrophication, and particularly at the same point of the impact pathway (i.e. pressure), the PB thresholds can be used as they are complementary to SDG indicator 14.1.1.

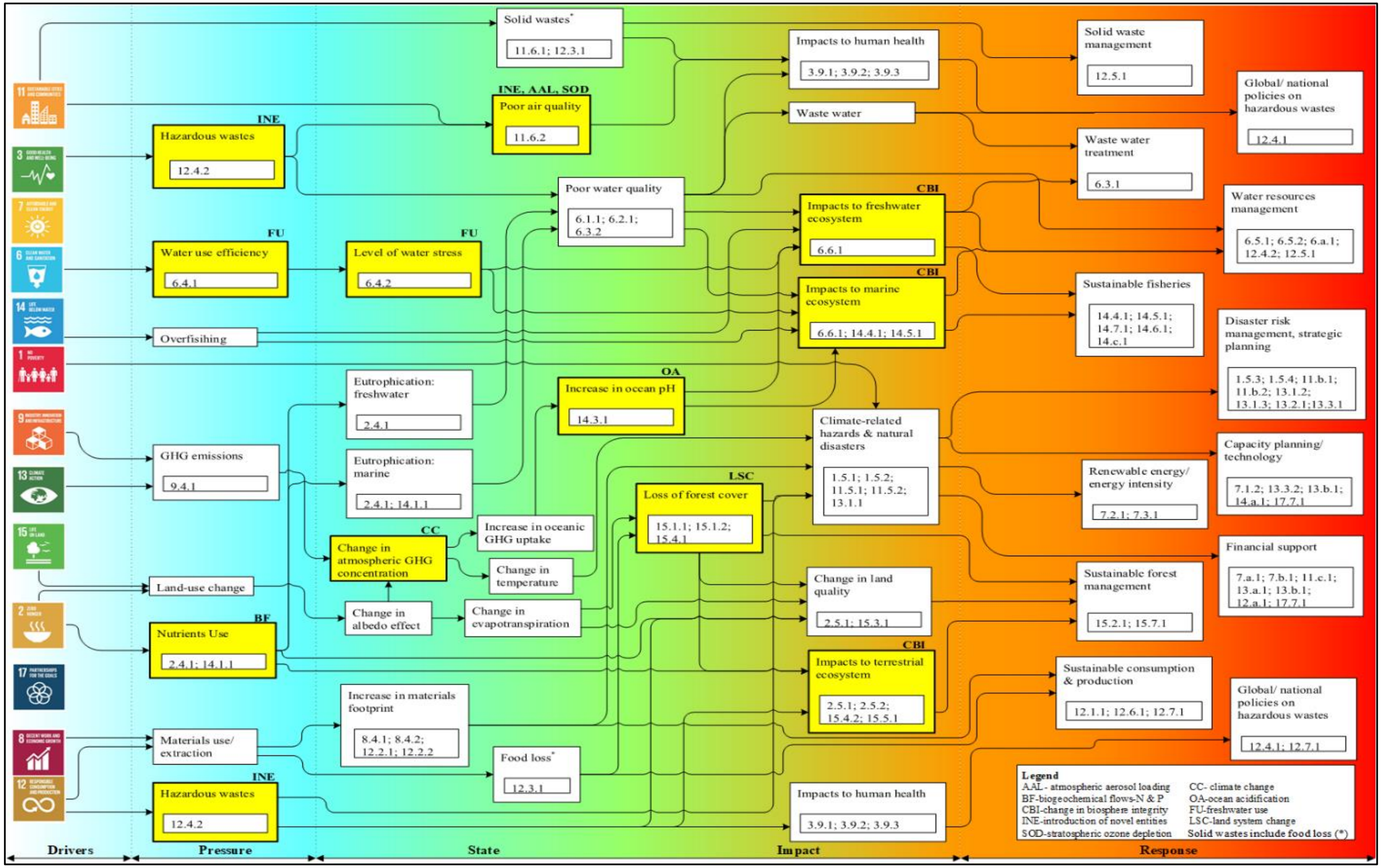

Figure 2. 1: The identified interlinkages between the SDGs and PBs (Chandrakumar and McLaren 2018). Each box in the figure refers to a specific environmental impact, while the numbers in the boxes refer to the relevant SDG indicator. The yellow boxes represent the environmental impacts addressed in the PBs: AAL = atmospheric aerosol loading; BF = biogeochemical flows; CBI = change in biosphere integrity (I = genetic diversity; II = functional diversity); CC = climate change (I = atmospheric CO_2 concentration; II = energy imbalance at top-of-atmosphere); FU = freshwater use (I = maximum amount of consumptive blue water use; II = blue water withdrawal as % of mean monthly river flow); INE = introduction of novel entities; LSC = land system change; OA = ocean acidification; SOD = stratospheric ozone depletion.

Changes in land use have significant effects on several biological and ecological systems, including climate and water. For example, changes in the area of boreal forests particularly affect the albedo of the land surface, and changes in the area of tropical forests specifically affect global evapotranspiration rates (Steffen et al. 2015). For this purpose, a PB called **land-system change** is suggested, which estimates the loss of forest cover at the state point of the impact pathway (see Figure 2. 1). Similarly, the SDGs report a set of indicators that evaluate the environmental problems resulting from forest cover loss as well as loss of other biomes (IAEG-SDGs 2017). For instance, SDG indicator 15.1.1 estimates the ratio between forest and total land area, SDG indicator 15.1.2 measures the proportion of protected areas for terrestrial, freshwater and mountain biodiversity, and, SDG indicator 15.4.1 estimates the land coverage allocated for mountain biodiversity. Nonetheless, none of these SDG indicators assesses the environmental degradation on an absolute scale. Rather, they merely report the proportions of the protected and degraded lands.

We, hence, recommend using SDG indicator 15.1.1 along with the threshold proposed for the land-system change PB for estimating the loss of forest cover on an absolute scale. However, since the PB focuses solely on the loss of forest cover, there remains a research gap in identifying relevant thresholds for other biomes addressed in SDG indicators 15.1.2 and 15.4.1.

The **climate change** PB and its associated control variables emphasise that the atmospheric CO_2 concentration and the radiative forcing from greenhouse gases (GHGs) should be reduced to 350 ppm CO_2 (350-450 ppm) and to 1 Wm^{-2}, respectively (at the state point in the DPSIR impact pathway) (Steffen et al. 2015). However, limiting the atmospheric concentration to 350 ppm CO_2 is unlikely as the current values of the control variables are 399 ppm CO_2 and 2.3 Wm^{-2} (Steffen et al. 2015), and the world population and economy are still growing (IPCC 2014). The IPCC, therefore, suggests that achieving a concentration of 450 ppm CO_2e is more likely (IPCC 2014). On the other hand, the SDGs present a set of SDG indicators concerned with mitigation of climate change problems (IAEG-SDGs 2017). At the response point, SDG indicator 13.1.1 estimates the fatalities and injuries due to climate change impacts, whereas SDG indicators 13.1.2, 13.1.3 and 13.2.1 focus on adopting policies and strategies to avert climate change impacts. SDG indicators 13.3.1, 13.3.2, 13.a.1 and 13.b.1 aim at strengthening institutional, systemic and individual capacity-building to implement adaptation, mitigation and technology transfer and development actions. In addition, SDG indicator 9.4.1 quantifies the carbon intensity of industries at the pressure point. In general, except SDG indicator 9.4.1, others focus only on averting the climate change impacts (as seen in Figure 2. 1), and none of them evaluates the climate change impacts on an absolute scale. Therefore, to our understanding, the PB thresholds (and the corresponding global carbon budget) can be used as a set of global boundaries for SDG indicator 9.4.1.

The **atmospheric aerosol loading** PB evaluates the impact resulting from the emissions of black and organic carbon from sources like cooking and heating with biofuels and diesel transportation, whereas the **introduction of novel entities** PB refers to the persistence, mobility and impacts of chemicals and other types of engineered materials or organisms produced by human activities (Steffen et al. 2015). Similarly, the **stratospheric ozone depletion** PB concentrates on the ozone concentration variations resulting due to synthetic chemicals release (Steffen et al. 2015). Interestingly, the associated control variables of these three PBs express the impacts at the state point of the DPSIR impact pathway. As far as we understand, some of the SDG indicators evaluate similar environmental impacts, but not explicitly. SDG indicator 11.6.2 monitors the levels of fine particulate matter in cities at the state point (with a focus on human health), whereas SDG indicators 11.6.1 and 12.4.2 quantify the solid and hazardous waste generated (pressure point). SDG indicator 12.4.2 also quantifies the amount of hazardous waste treated (response point) and SDG indicators 12.4.1 and 12.7.1 focus on the global initiatives taken to develop multinational agreements and policies on hazardous waste and other chemicals (response point). Although there are some overlaps between the above-listed three PBs and the SDG indicators, it would not be advisable to use them in a complementary way for the following reasons: (i) existence of an enormous number of hazardous substances (including chemicals); (ii) no SDG indicators directly assess the effects of ozone depletion; (iii) no control variables and thresholds have been proposed for the "introduction of novel entities" PB; (iv) the effects of some substances are still unknown, and some effects are not readily reversible; and (v) the PBs are located closer to the original activities that cause the environmental impacts, whereas the SDGs focus on waste management, and are generally located at the response point (Steffen et al. 2015). Further research is therefore needed to understand better the complementarities between these PBs and the SDGs.

In sum, the environmental SDG indicators mostly address the environmental problems reported in the PBs, which are primarily associated with the *ecosystem quality* AoP. However, in contrast to the PBs, the SDGs additionally concentrate on the other AoPs (*human health, resource scarcity* and *man-made environment*) through addressing the global challenges of unsustainable food and agriculture, soil quality degradation, impacts of ecosystem degradation on human health, direct human impacts on the ecosystem (wildlife trafficking and poaching, and overfishing) and lack of infrastructure for water quality and resources management by communities [12]. Regarding unsustainable food and agriculture, SDG indicator 2.5.1 estimates the number of plant and genetic resources secured for sustainable food and agriculture, whereas SDG indicator 2.5.2 estimates the local breeds under risk of extinction. Likewise, SDG indicator 2.4.1 evaluates the soil quality degradation. Nevertheless, these SDG indicators are suboptimal because they use a relative scale, and lack clarity; for instance, SDG target 2.4 (which includes SDG indicator 2.4.1) focuses on multiple environmental problems, including climate change and soil quality degradation (IAEG-SDGs 2017; Wackernagel et al. 2017).

On the other hand, a set of SDG indicators has been reported to evaluate the impacts of degradation of ecosystems on human health. SDG indicators 3.9.1, 3.9.2, 6.3.1 and 6.3.2 evaluate the human health problems resulting due to ambient air pollution, unsafe water, sanitation and hygiene, whereas SDG indicator 3.9.3 assesses the health problems resulting from unintentional poisoning (IAEG-SDGs 2017). Additionally, SDG indicators 2.4.1, 2.5.1 and 2.5.2 explicitly refer to the impacts on human health as a consequence of unsustainable food production and agricultural practices. Likewise, SDG indicators 15.7.1 and 15.c.1 estimate the impacts of wildlife trafficking and poaching, whereas SDG indicators 14.4.1 and 14.5.1 addresses the impacts associated with overfishing. Regarding lack of infrastructure for water quality and resources management by communities, SDG indicators 6.1.1, 6.2.1 and 6.a.1 concentrate on developing infrastructure for effective water resources management (IAEG-SDGs 2017; Maier et al. 2016).

2.5. Conclusions

The study underpins the development of an AESA framework by systematically exploring the interlinkages between the SDGs and PBs using the DPSIR impact pathway framework.

According to the analysis presented in Section 4, the two approaches demonstrate notable overlaps with regard to their indicators and control variables. Each of the PBs is linked to one (or more) SDG indicator(s), as shown in Figure 2. 1. In particular, the **freshwater use**, **ocean acidification**, **biogeochemical flows, land system change** and **change in biosphere integrity** PBs exhibit sound linkages with the SDG indicators. Interestingly, some of the control variables of these five PBs are located at the same point in the DPSIR impact pathway as the SDG indicators. But, in contrast, the **climate change** PB control variable is located at the state point of the impact pathway, whereas the SDG indicators associated with climate change are mostly located at the response point, except SDG indicator 9.4.1, which estimates the carbon intensity of industries at the pressure point. Moreover, no SDG indicators report an absolute limit for GHG emissions. The **introduction of novel entities** and **atmospheric aerosol loading** PBs show some overlaps with SDG indicators 11.6.1, 11.6.2 and 12.4.2, whereas no explicit linkages are found between the **stratospheric ozone depletion** PB and the SDG indicators. Furthermore, as discussed in Section 4, the SDGs additionally shed light on some other global challenges not explicitly addressed in the PBs such as unsustainable food and agriculture, soil quality degradation, impacts of ecosystem degradation on human health, direct human impacts on the ecosystem (wildlife trafficking and poaching, and overfishing), and lack of infrastructure for water quality and resources management by communities.

Overall, according to this study, it seems potentially feasible to adopt the science-based thresholds reported in the PBs as a complementary set of global boundaries for the SDG indicators. Moreover, advancing appropriate environmental boundaries for the additional challenges addressed in the SDGs, and using them alongside the PB thresholds, will provide a platform to benchmark the environmental sustainability performance of a system at a global level. However, further research is necessary to benchmark similar environmental impacts on a sub-global level, given that most of the impacts are, in fact, a result of the accumulated effects of discrete regional and local problems (Sandin et al. 2015; Steffen et al. 2015). Hence, some suggest allocating the global boundaries to sub-global levels using a variety of allocation principles (e.g. Roos et al. 2016; Sandin et al. 2015), while others propose developing appropriate independent boundaries at sub-global levels (e.g. Bjørn et al. 2016; Steffen et al. 2015). Finally, in order to develop an AESA framework, future studies should focus on linking the SDGs and PBs with the impact assessment phase of

LCA. Such an AESA has the potential to inform whether the chosen system is aligned with the environmental goals and targets listed in the SDGs as well as whether they are operating within the Earth's carrying capacity.

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Chapter 3: Towards a comprehensive absolute sustainability assessment method for effective Earth system governance: defining key environmental indicators using an enhanced-DPSIR framework

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Abstract

Reflecting the growing interest in the concept of absolute sustainability, this research defines an absolute environmental sustainability assessment (AESA) method with three key characteristics: (i) assessment of a comprehensive range of environmental impacts in absolute terms; (ii) evaluation of these impacts at an early stage in impact pathways; and (iii) the capacity to assess these impacts at multiple economic levels. To that end, using an enhanced Driver-Pressure-State-Impact-Response (eDPSIR) framework, this study systematically classified the environmental indicators reported in the Planetary Boundaries (PBs), Life Cycle Assessment (LCA) and Sustainable Development Goals (SDGs) by mapping them on to a network of cause-effect chains developed in previous work, and extended to include the areas of protection in LCA. It was found that twelve major environmental problems could be defined as key central nodes in this causal network, and that the PBs and LCA evaluated many of these environmental problems at an early stage in the causal network while the SDGs generally addressed similar problems at the latter end of the causal network. Six of these environmental problems were addressed in all three approaches (PBs, LCA and SDGs) and the others were addressed in one or two approaches. An associated (but incomplete) set of absolute environmental sustainability indicators were identified that are already available in one or more of the three approaches; some of these indicators require further methodological development in order to support the advancement of an AESA for effective Earth system governance.

Keywords: absolute sustainability; absolute sustainability assessment method; absolute environmental sustainability indicator; planetary boundaries; life cycle assessment; sustainable development goals; Enhanced DPSIR; causal network; earth system governance

3.1. Introduction

The concept of *sustainable development* has emerged out of a growing awareness of the global interrelationships between environmental impacts and the socio-economic dimensions of human activities (Hopwood et al. 2005). However, a fundamental debate regarding sustainable development is whether we should subscribe to a strong or a weak conception of sustainability. A weak conception of sustainability assumes that natural capital (e.g. mineral resources, clean air, fertile soil) and manufactured capital (aka physical capital, e.g. machines, buildings) are substitutable (Brekke 1997; Daly et al. 1994; Pezzey 1992a, b). According to this perspective, the aggregated stock of both types of capital should be increased or at least maintained for future generations, and there is a general expectation that technical solutions can compensate the environmental impacts that are (usually) associated with the supply of manufactured capital (e.g. construction of a water treatment plant in place of a wetland to deliver the service of water filtration) (Pezzey 1992a, b). Effectively this means that, from a weak sustainability perspective, achieving economic growth to supply manufactured capital at the cost of environmental degradation is acceptable (Daly et al. 1994; Jacobs and Stott 1992).

In contrast, a strong conception of sustainability suggests that the different types of capital are not substitutable i.e. economic growth should not be achieved at the cost of environmental degradation acceptable (Daly et al. 1994; Jacobs 1991; Jacobs and Stott 1992). The concept of *critical* natural capital is therefore fundamental in strong sustainability i.e. natural capital that is essential to the continued efficient functioning of the Earth system and human well-being. Examples of critical natural capital include the ozone layer and the global atmosphere whose functions cannot be substituted by other types of capital (Chiesura and de Groot 2003; Ekins et al. 2003). In contrast, manufactured capital is reproducible (Ekins et al. 2003; Turner 1993); for instance, built infrastructure can be reconstructed if it is destroyed but loss of species is irreversible. Moreover, in many cases, production of manufactured capital relies heavily on natural capital (Ekins et al.

2003); for example, building construction requires raw materials extracted from the Earth or harvested from ecosystems (e.g. mineral ores such as iron and aluminium, and wood). Furthermore, we have a limited understanding of the functioning of natural systems, and destruction of natural capital may have impacts on human well-being beyond those we can currently predict (Ekins et al. 2003; Patrício et al. 2016; Rockström et al. 2009).

In reality, a weak sustainability perspective has been adopted by governments in many countries as they focus on realising socio-economic benefits and minimise or ignore the associated environmental degradation (Kim and Bosselmann 2015; Muys 2013). As a result, many of the Earth system boundaries are already transgressed (ranging from global warming to air pollution, water quality degradation and loss of ecosystems). This has been highlighted in the last few years through the concept of Planetary Boundaries proposed by Rockström et al. (2009) and elaborated by Steffen et al. (2015). It means, that today, human societies need to urgently address two interrelated "wicked" problems on a global scale: (i) how to protect the entire Earth system and its subsystems (in particular, the stock of critical natural capital), and (ii) how to operate socio-economic systems within the Earth system boundaries. This, therefore, implies the adoption of a strong sustainability conception in order to achieve sustainable development.

In the context of addressing these two wicked problems, this article introduces a proposal for developing an absolute environmental sustainability assessment (AESA) method for effective Earth system governance. The term "Earth system governance" refers to the process of defining and developing socio-economic systems that will prevent drastic Earth system disruptions (Biermann et al. 2010). Section 3.3 provides an overview of existing environmental sustainability assessment methods and discusses their potential to address absolute environmental sustainability. Section 3.4 introduces the proposed AESA method and describes the systematic classification process for existing environmental indicators, and Section 3.5 presents the key findings of the classification. Section 3.6 concludes the article with a discussion on the contribution of this work in developing the proposed AESA, and reports the limitations of the study that require further exploration.

3.2. Current state of research on environmental sustainability assessment methods addressing absolute sustainability

3.2.1. Role of environmental sustainability assessment methods in Earth system governance

As outlined above, there are several complexities inherent in effective Earth system governance. Robust and comprehensive environmental sustainability assessment methods (ESAMs) are required to address these complexities for any given system under analysis and proposed interventions (e.g. Moldan et al. 2012; Ness et al. 2007; Singh et al. 2009). A comprehensive ESAM should address the following questions: (Q1) What is the environmental impact(s) of a chosen system? (Q2) What is the allocated biophysical limit(s) of the Earth system (aka the desired state) for the chosen system? and (Q3) How can proposed interventions in the system be measured with respect to their ability to bring the system within these biophysical limits? Here, the term "system" could be either a product, process, project, sector, nation or the entire Earth (Roos et al. 2016).

With regard to Q1, there exist a large number of ESAMs such as Environmental Impact Assessment, Life Cycle Assessment (LCA) and environmental footprints that quantify the environmental impact(s) of a system (Moldan et al. 2012; Ness et al. 2007; Singh et al. 2009). These ESAMs, in general, either implicitly or explicitly rank a particular system in relation to a reference system that is relevant to the nature (or the function) of the examined system and the objectives of the study. For example, they address issues such as "Is System A better than System B?" and "Which activity in the examined system is responsible for the most of the environmental impacts?" As a result, such ESAMs generally do not provide information on the environmental sustainability performance of the system with regard to the allocated biophysical limits of the Earth system, and are therefore classified as relative sustainability assessment methods using relative environmental sustainability indicators (Bjørn et al. 2016; Hauschild 2015).

On the contrary, ESAMs addressing Q2 and Q3 require the development of absolute environmental sustainability indicators (AESIs). AESIs are indicators that benchmark the actual environmental impact(s) of a system against a set of environmental targets or standards (Bjørn et al. 2016). These targets can be either policy targets or biophysical (science-based) thresholds. However, only a few such ESAMs (with AESIs) have been developed, for example, Tolerable Windows (Bruckner et al. 1999), Planetary Guardrails (WBGU 2011) and Planetary Boundaries (Rockström et al. 2009; Steffen et al. 2015). Tolerable Windows benchmarks climate change impacts against a set of pre-defined targets (Bruckner et al. 1999), whilst Planetary Guardrails does it for a list of environmental problems including climate change, soil degradation, biodiversity loss, and ocean acidification (WBGU 2011). Similarly, the concept of Planetary Boundaries (PBs) presents a set of control variables and thresholds for nine critical Earth system processes (see Rockström et al. 2009; Steffen et al. 2015).

3.2.2. Adaptation of existing environmental sustainability assessment methods to develop absolute environmental sustainability indicators

Recognising that only a few ESAMs address Q2 and Q3, and that there is potential for modifying LCA indicators into AESIs, some LCA researchers have used distance-to-target (DTT) methods at the Life Cycle Impact Assessment phase of an LCA (e.g. Bjørn and Hauschild 2015; Castellani et al. 2016; Seppälä and Hämäläinen 2001; Wang et al. 2011). These DTT methods derive weighting factors for LCA impact categories by comparing a system's actual environmental performance against existing environmental targets (Castellani et al. 2016). However, many of the studies published to date have adopted policy-based targets and benchmarked the sustainability performance of a particular system mostly at a regional or national level (e.g. Castellani et al. (2016) for Europe; Wang et al. (2011) for China; and Frischknecht and Büsser Knöpfel (2013) for Switzerland). The policy-based targets generally represent a compromise between scientific knowledge and societal considerations (political feasibility, cost), and hence they are (generally) less strict than science-based targets (Acosta-Alba and Van der Werf 2011). Acknowledging that, Bjørn and Hauschild (2015) explored how science-based targets (i.e. PBs in this context) can be adopted in the DTT methods to address absolute environmental sustainability at the global as well as regional (for Europe) levels. Following this study, other studies exploring the potential to use the PBs in combination with LCA indicators to benchmark the environmental sustainability performance of systems at different economic levels are emerging (e.g. Fang et al. 2015; Nykvist 2013; Roos et al. 2016; Sandin et al. 2015). For instance, Roos et al. (2016) and Sandin et al. (2015) benchmarked the sustainability performance of the Swedish apparel sector in terms of climate change, freshwater consumption and non-renewable energy resources, and calculated impact reduction targets at both sectoral and product levels. Similarly, Fang et al. (2015), at the national level, benchmarked the sustainability performance of 28 countries with regard to the climate change, land-use and freshwater use PBs.

Although such PBs-based LCA studies have begun to inform environmental sustainability performance of systems at different economic levels and in absolute terms, there are a number of outstanding challenges in undertaking these types of studies. These include: identifying and addressing the overlaps between the Earth system processes identified in the PBs; including spatial differentiation of control variables at sub-global levels; calculation of characterisation factors for the PB control variables; and defining methods for allocating the PBs at different economic levels (Bjørn and Hauschild 2015; Häyhä et al. 2016; Laurent and Owsianiak 2017; Ryberg et al. 2016).

3.3. Method for development of a comprehensive absolute sustainability assessment method

While some researchers have begun to address the limitations in the PBs-based LCA methodology, this approach may be inadequate in addressing absolute sustainability on a global level. One key reason is that not all environmental problems are addressed in the PBs (Chandrakumar and McLaren 2018a; Dong and Hauschild 2017; Ryberg et al. 2016). Moreover, the PBs are only concerned with impacts on the natural environment and do not address other impacts associated with either human health or the man-made environment (Dong and Hauschild 2017; Ryberg et al. 2016). In addition, further research is still necessary to estimate environmental boundaries for some PBs (e.g. introduction of novel entities and atmospheric aerosol loading) (Diamond et al. 2015; Sala and Goralczyk 2013; Steffen et al. 2015).

In order to (at least partially) address these challenges, we expanded the focus to include the Sustainable Development Goals (SDGs). In a previous study (Chandrakumar and McLaren 2018a), a network of cause-effect chains based on the environmental problems addressed in the SDGs and PBs was developed; this involved identifying the SDG indicators (73 indicators in total) concerned with the conventional areas of protection used in LCA (i.e. human health, ecosystem quality, resource scarcity and man-made environment), and mapping them onto a causal network along with the PB control variables. Then these indicators and control variables were classified using a Driver-Pressure-State-Impact-Response (DPSIR) framework (for DPSIR framework, see Niemeijer and de Groot 2006; Patrício et al. 2016). It was confirmed that the range of environmental problems addressed by the SDGs was much wider than those in the PBs; and the SDG indicators were mostly of the response type, whereas the PBs were generally of the pressure/state-type (Chandrakumar and McLaren 2018a).

This study introduces a proposal for advancing a comprehensive absolute environmental sustainability assessment (AESA) method called Absolute Sustainability-based Life Cycle Assessment (ASLCA) and proposes a set of key environmental problems with (some) associated absolute environmental sustainability indicators (AESIs) to be used in this method. An AESA, in this study, refers to a specific benchmarking framework comprising clusters of AESIs that can be used to assess a system's environmental sustainability in absolute terms. Based on the current identified research needs (as outlined in Section 3.3), the ASLCA is intended to have the following key characteristics:

(C1) the quantitative assessment of a comprehensive range of environmental impacts in absolute terms;

(C2) the quantitative assessment of the impacts at an early stage of an impact pathway (which reduces the complexity and inherent uncertainties in modelling through providing the closest link to the scientific knowledge basis (Hauschild et al. 2013)); and

(C3) the capacity to benchmark the impacts at different economic levels.

As a first step, we mapped the LCA indicators reported in the widely used ReCiPe2016 Impact Assessment method (Huijbregts et al. 2016) onto the causal network presented in Chandrakumar and McLaren (2018a). Then we linked all the environmental indicators reported in those three approaches (PBs, LCA and SDGs) to the relevant areas of protection (AoPs) and a new area of concern (AoC) (see Section 3.5 for a detailed description).

Finally, once all the indicators had been mapped and linked to the AoPs and AoC, the indicators were classified into Driver, Pressure, State, Impact and Response categories using the principles of an enhanced DPSIR (eDPSIR) framework (for theory, see Niemeijer and de Groot (2008)). According to Niemeijer and de Groot (2006, 2008), compared with a DPSIR framework, an eDPSIR framework more effectively explores the inter-linkages between different entities (which could be environmental problems, approaches or indicators) in a causal network, and enables identification of the key nodes in the causal network: root nodes, central nodes and end-of-chain nodes. Root nodes are nodes with many outgoing arcs; central nodes are nodes with multiple incoming and/or outgoing arcs; and end-of-chain nodes have many incoming arcs, which bring together multiple cause-effect chains. Although each of these nodes serves a specific purpose in environmental problem framing, the central nodes are generally considered the most useful nodes as they are more likely to represent a larger number of problems. Therefore, a set of central nodes (hereafter "key central nodes") were identified based on the incoming and/or outgoing arcs, as well as representation of distinct recognized environmental problems with assessment method(s) already developed (particularly in LCA methodology), and which together represent a comprehensive range of environmental problems.

3.4. Results of classification of environmental sustainability indicators

Figure 3. 1 presents the causal network linking the indicators in the PBs, LCA and SDGs, and extending from anthropogenic activities through to the areas of protection (AoPs) and area of concern (AoC). Note that to ensure comprehensive coverage of the environmental problems addressed in different LCA impact assessment methods, the ReCiPe2016 indicators were compared with those used in other LCA impact assessment methods (see Table S1. 1 in the Supplementary Material (SM) 1 for comparison). This comparison resulted in additionally adopting the terrestrial eutrophication indicator listed in the EDIP and IMPACT 2002+ methodologies, and the so-called AoP "man-made environment" proposed in the CML methodology.

During the mapping exercise, it was noticed that some of the SDG indicators are not directly related to assessing actual environmental impacts but are focused on human management of environmental risks. For example, SDG Target 1.5 states the need to "…build resilience of the poor and those in vulnerable situations and reduce their exposure and vulnerability to climate-related extreme events". SDG Target 2.4 and the associated indicator concentrate on the global challenge of sustainable food production by implementing resilient agricultural practices that "… strengthen capacity for adaptation to climate change, extreme weather, drought, flooding and other disasters". Likewise, SDG Target 11.b aims at substantially increasing cities adopting policies towards "…mitigation and adaptation to climate change, resilience to disasters"; and SDG Target 11.c focuses on supporting least developed countries for "building sustainable and resilient buildings" using locally available materials. SDG Target 13.1 emphasises the need to "strengthen resilience and adaptive capacity to climate-related hazards and natural disasters" globally, whereas SDG Target 13.2 attempts to integrate climate change measures into national policies to "…foster climate resilience". Therefore, in order to represent these risk management targets and indicators, an additional AoC ("environmental risks") was introduced into Figure 3. 1, which sits alongside the existing AoPs already used in LCA. The concept of an AoC, introduced by Ridoutt et al. (2016), refers to an environmental topic developed based on the concerns of stakeholders in society.

The results of the classification using the eDPSIR framework are also shown in Figure 3. 1. Twelve key central nodes were identified, and are indicated in red outlined boxes in Figure 3. 1; Table 3. 1 lists them together with their associated indicators in the PBs, LCA and SDGs. The environmental problems addressed in these twelve key central nodes are defined as the problems to be evaluated in the proposed ASLCA to support effective Earth system governance. For abiotic resource depletion/scarcity, it may be argued that this problem is only of relevance due to the instrumental value of resources to humans (Sonderegger et al. 2017), as opposed to Earth system. However, it is retained here as it is considered a legitimate focus of attention in the SDGs (IAEG-SDGs 2017) and in LCA (Sonderegger et al. 2017; Verones et al. 2017).

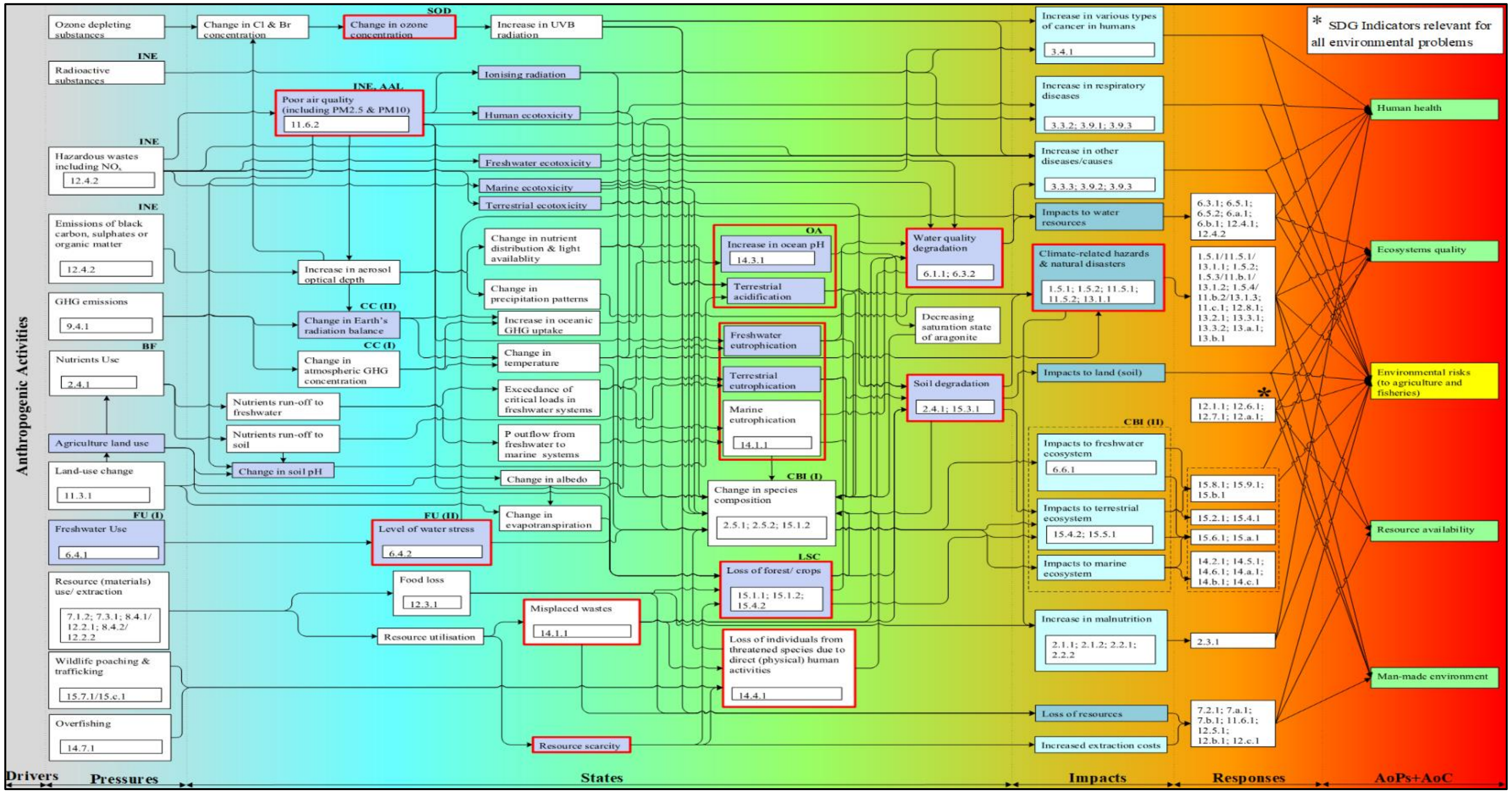

Figure 3. 1: The inter-linkages between the indicators in the PBs, LCA, and SDGs. Each box in the figure indicates a specific environmental issue, while the numbers in the boxes refer to the relevant SDG indicator (see IAEG-SDGs (2017) for a full list of SDG indicators) (Chandrakumar and McLaren 2018b). SDG Indicators 12.1.1; 12.6.1; 12.7.1; and 12.a.1 are relevant to all environmental problems at the response point. The red outlined boxes represent the 12 key central nodes identified in this study. Filled colour boxes represent: grey = LCA midpoint indicators; light blue = LCA endpoint indicators; dark blue = proposed additional endpoint indicators; green = AoPs used in LCA; yellow = proposed AoC. The text in bold (above the boxes) refers to the PBs: AAL = atmospheric aerosol loading; BF = biogeochemical flows; CBI = change in biosphere integrity (I = genetic diversity; II = functional diversity); CC = climate change (I = atmospheric CO_2 concentration; II = energy imbalance at top-of-atmosphere); FU = freshwater use (I = maximum amount of consumptive blue water use; II = blue water withdrawal as % of mean monthly river flow); INE = introduction of novel entities; LSC = land system change; OA = ocean acidification; SOD = stratospheric ozone depletion. The figure is based on Figure 2. 1 in Chapter 2.

Table 3. 1: Classification of environmental sustainability indicators reported in the planetary boundaries, life cycle assessment and sustainable development goals against identified environmental problems

No.	Environmental Problem	Indicators in Different Approaches		
		PBs	LCA	SDGs
1	Water scarcity (quantity)	*Maximum amount of consumptive blue water use (P); Blue water withdrawal as % of mean monthly river flow (S)*	Water use (P); water scarcity (S) (according to ISO (2006))	SDG Indicators 6.4.1 (P); 6.4.2 (S); 6.5.1 (R); 6.5.2 (R); 6.a.1 (R);6.b.1 (R)
2	Water quality degradation	-	Freshwater ecotoxicity (S); marine ecotoxicity (S)	SDG Indicators 12.4.2 (P); 6.1.1 (S); 6.3.2 (S); 6.3.1 (R); 6.5.1 (R); 6.5.2 (R); 6.a.1 (R); 6.b.1 (R); 12.4.2 (R)
3	Climate-related natural hazards	*Atmospheric CO_2 concentration (S); Energy imbalance at top-of-atmosphere (S)*	Infrared radiative forcing increase (S)	SDG Indicators 9.4.1 (P); 1.5.1/11.5.1/13.1.1 (R); 1.5.2 (R); 1.5.3/11.b.1/13.1.2 (R); 1.5.4/11.b.2/13.1.3 (R); 11.5.2 (R); 11.c.1 (R); 13.1.1 (R); 13.2.1 (R)
4	Terrestrial & aquatic eutrophication	*Phosphorous flow from freshwater systems into the ocean (P); Phosphorous flow from fertilizers to erodible soils (P); Industrial and intentional biological fixation of Nitrogen (P)*	Residence time of nutrients in freshwater or marine end compartment (S); Accumulated exceedance of critical loads of Nitrogen in terrestrial ecosystems (S)	SDG Indicator 14.1.1 (S)
5	Land-system change	Area of forested land as % of original forest cover (S); Area of forested land as % of potential forest (S)	Occupation and time-integrated land transformation (S)	SDG Indicators 11.3.1 (P); 15.1.1 (S); 15.1.2 (S); 15.4.2 (S); 15.2.1 (R); 15.4.1 (R); 15.a.1(R); 15.b.1 (R);
6	Ambient air pollution and other toxic effects	Aerosol Optical Depth (AOD) (S); AOD as a seasonal average over a region (S)	Tropospheric ozone increase (P); PM2.5 population intake (S); Risk increase of cancer and non-cancer disease incidence (I)	SDG Indicators 12.4.2 (P); 11.6.2 (S); *3.4.1 (I)*; 3.9.1 (I); 3.9.3 (I)
7	Terrestrial & ocean acidification	*Carbonate ion concentration, average global surface ocean saturation state with respect to aragonite (S)*	Proton increase in natural soils (S); changes in surface oceans pH and carbonate mineral saturation state (S)	SDG Indicator 14.3.1 (S)
8	Soil degradation and loss of soil	-	Soil organic matter (S)	SDG Indicators 2.4.1 (S); 15.3.1 (S)
9	Ozone depletion	*Stratospheric ozone concentration (S)*	Stratospheric ozone decrease (S)	-
10	Loss of individuals from threatened species due to direct (physical) human activities	-	-	SDG Indicators 14.7.1 (P); *15.7.1/15.c.1 (P)*; 14.4.1 (S); 14.2.1 (R); *14.5.1 (R)*; 14.6.1 (R); 14.b.1(R); 14.c.1 (R); 15.6.1 (R); 15.9.1(R)
11	Misplaced wastes	-	-	SDG Indicators 14.1.1 (S); 14.2.1 (R)
12	Abiotic resource depletion/ scarcity	-	Resource scarcity (S)	SDG Indicators 7.1.2 (P); 7.3.1 (P); 8.4.1/12.2.1 (P); 8.4.2/12.2.2 (P); 7.2.1 (R); 7.a.1 (R); 7.b.1 (R); 11.c.1 (R); 12.b.1 (R); 12.c.1 (R)

The letter shown in the parenthesis refers to the location of the impact assessment in the causal network: D=Driver; P=Pressure; S=State; I=Impact; and R=Response. The text in bold-italic indicates the already existing AESIs in the PBs, LCA and SDGs. Note that SDG Indicators 12.1.1 (R); 12.6.1 (R); 12.7.1 (R); and 12.a.1 (R) are relevant to all environmental problems; hence, not specifically represented in the table. See United Nations (2017) for a full list of SDG indicators. See Table S1. 2 in SM1 for more information.

The list presented in Table 3. 1 is similar to the results reported by Dong and Hauschild (2017); however, the environmental problems of loss of individuals from threatened species due to direct (physical) human activities, and misplaced wastes were not considered in Dong and Hauschild (2017). Also, they used the older version of the SDG indicators (available in UN (2015)) along with the PBs and LCA indicators, and their classification was centred on the principles of a DPSIR framework. As previously discussed, compared to a DPSIR framework, an eDPSIR provides insights into the complex inter-linkages between multiple anthropogenic activities, the environment, and indicators by working with causal networks rather than causal chains, and identifies three types of key nodes with different purposes. Moreover, here we identify environmental risks as an additional aspect to be addressed in assessing environmental problems in the context of Earth system governance.

It can be observed from Table 3. 1 that most of the environmental problems are addressed in more than one approach. Water scarcity, climate-related hazards and natural disasters, terrestrial and aquatic eutrophication, land-system change, ocean acidification and ambient air quality and other toxic effects are addressed in all three approaches. However, only LCA and the SDGs assess the implications associated with water quality, soil quality, resource depletion/scarcity and human health. Likewise, only LCA and the PBs evaluate the implications of ozone depletion. Furthermore, some problems, in particular, loss of individuals from threatened species due to direct (physical) human activities (in particular, wildlife poaching and trafficking, and overfishing), and misplaced wastes (specifically, floating plastic debris in the oceans), are only addressed in the SDGs. As illustrated in Figure 3. 1, the indicators in the three approaches show some overlaps in terms of the location of the impact assessment along the causal network. The indicators reported in the PBs, LCA, and SDGs evaluate the water scarcity problem at the pressure and state points of the causal network. Similarly, all three approaches assess the implications of land system change, ambient air pollution, and ocean acidification at the state point (see Table 3. 1). LCA and the SDGs estimate the environmental problems of water quality degradation, soil degradation and resource depletion/scarcity, and LCA and the PBs evaluate ozone depletion at the state (or pressure) point. Nevertheless, for some other environmental problems, impact assessment occurs at different points in the causal network. LCA, for example, calculates the climate change impacts in terms of the increase in infrared radiative forcing (state point), and the PBs in terms of change in atmospheric CO_2 concentration and radiation balance (state point). Meanwhile, the SDGs focus on the same issue at the pressure point (i.e. SDG Indicator 9.4.1). LCA calculates terrestrial and aquatic (both in the freshwater and marine ecosystems) eutrophication and the SDGs estimate aquatic eutrophication (marine only) at the state point, while the PBs address aquatic eutrophication at the pressure point. However, as identified earlier, only the SDGs assess the loss of individuals from threatened species due to direct (physical) human activities, and misplaced wastes, and do so at the pressure, state and response points.

Additionally, these indicators have some overlaps in terms of the relative versus absolute nature of the impact assessment. As discussed in Section 3.3, all LCA indicators are relative environmental sustainability indicators. In contrast, by definition, PBs are AESIs as they include a set of environmental boundaries (Steffen et al. 2015). On the other hand, some SDG indicators are AESIs (SDG Indicators 3.4.1; 14.5.1; and 15.7.1/15.c.1) whilst the others are relative environmental sustainability indicators.

3.5. Discussion and conclusion

This study presents a simplified visualisation of complex cause-effect relationships between anthropogenic activities, the environment and society by mapping the environmental indicators reported in the PBs, LCA and SDGs onto a causal network, and systematically classifying these indicators using the eDPSIR framework.

Figure 3. 1 shows that a more comprehensive range of environmental problems are addressed in the SDGs than in the PBs and LCA, although some problems are omitted from the SDGs. Thus, the complementary use of the three approaches provides a basis for identifying a comprehensive range of environmental problems (as listed in Table 3. 1, and representing characteristic C1). However, characteristic C1 additionally requires the development of a list of AESIs pertaining to these environmental problems; only a few environmental problems currently have identified AESIs (in the PBs and SDGs) and, of them, some are not fully developed (see Table 3. 1 and discussion in Sections 3.4 and 3.5). Furthermore, all three AESIs in the SDGs adopt policy-based targets instead of science-based targets (see Section 3.5). With regard to characteristic C2 (the quantitative assessment

of environmental impacts at an early stage of an impact pathway), Table 3. 1 identifies the position(s) of the different indicators along the causal network: the PBs and LCA, in general, evaluate many of the environmental problems at an early position (pressure or state) on the causal network and the SDGs focus on similar problems at the latter end (impact or response) of the causal network. This was previously noted by Patrício et al. (2016) who commented that natural scientists (usually) are interested in the pressure/state point, whereas social scientists are more interested in the impact/response/driver point.

Use of the eDPSIR framework thus acts as a useful way of identifying key environmental problems through its ability to make clear the links between natural and socio-economic systems and reveal key central nodes. In this research, it has enabled identification of twelve environmental problems (listed in Table 3. 1) for assessment in the proposed ASLCA that meet characteristics C1 and C2. Moreover, further investigation is needed on various aspects. Firstly, performing absolute sustainability assessments on a sub-global level (i.e. characteristic C3) requires allocating the global level environmental thresholds for global problems (e.g. climate change) to sub-global levels using appropriate allocation procedures, and developing context-specific thresholds for regional or local problems (e.g. freshwater contamination, soil quality degradation and loss of soil) (Bjørn and Hauschild 2015; Häyhä et al. 2016). Secondly, future research should focus on developing environmental indicators and science-based thresholds for environmental problems such as marine pollution and introduction of novel entities (Diamond et al. 2015; Sala and Goralczyk 2013). Moreover, although the eDPSIR framework is recognised for its usefulness in supporting systematic investigation of the complexities of a causal network, it has limitations that require further investigation (a comprehensive review is available in Patrício et al. (2016)). For example, it is not always clear whether an indicator should be classified as a driver/pressure, pressure/state, or state/impact; the timescale over which the environmental impact occurs is generally overlooked; and many of the studies on the DPSIR framework and its derivatives like eDPSIR are conceptual. Therefore, more case studies centred on these kinds of conceptual models are necessary to understand the practical implications of advancing and implementing AESA frameworks for effective Earth system governance.

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Chapter 4: Absolute Sustainability-based Life Cycle Assessment (ASLCA): a benchmarking approach to operate agri-food systems within the 2°C global carbon budget

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Abstract

Given the increasing environmental impacts associated with global agri-food systems, operating and developing these systems within the so-called absolute environmental boundaries has become crucial, and hence the absolute environmental sustainability concept is particularly relevant. This study introduces an approach called Absolute Sustainability-based Life Cycle Assesment (ASLCA) that informs the climate impacts of an agri-food system (on any economic level such as global, national, sectoral and product) in absolute terms. Firstly, a global carbon budget was calculated which is sufficient to limit global warming to below 2°C. Next, a share of the carbon budget available to the global agri-food sector was determined, and then it was shared between agri-food systems on multiple economic levels using four alternative methods. Thirdly, the climate impacts of those systems were calculated using Life Cycle Assessment methodology, and were benchmarked against those carbon budget shares. This approach was used to assess a number of New Zealand agri-food systems (agri-food sector, horticulture industries and products) to investigate how these systems operated relative to their carbon budget shares. The results showed that, in 2013, the New Zealand agri-food systems were within their carbon budget shares for one of the four methods, and illustrate the scale of change required for agri-food systems to perform within their carbon budget shares. This method can potentially be extended to consider other environmental impacts with global boundaries; however, further development of the ASLCA is necessary to account for other

environmental impacts whose boundaries are only meaningful when defined at a regional or local level.

Keywords: absolute environmental sustainability; carbon budget; climate change; industrial ecology; life cycle assessment; agri-food; New Zealand

4.1. Introduction

Increasing global challenges such as climate change, water scarcity, loss of biodiversity, resource depletion and air pollution have received the attention of multiple stakeholders, including academics, industries, government agencies and civil societies. To mitigate these challenges, attention has been focused on enhancing the existing environmental sustainability performance of economic and product systems at multiple economic levels, ranging from product/process, company, sector and country to the entire Earth. As of now, a number of quantitative Environmental Sustainability Assessment Methods (ESAMs) have been proposed, such as Life Cycle Assessment (LCA) and environmental footprints, which provide useful information to support decision-making about the environmental sustainability performance of these systems (Chandrakumar et al. 2017; Coelho and McLaren 2013; Moldan et al. 2012; Singh et al. 2009). Nevertheless, overall progress in mitigating global environmental challenges is inadequate and many environmental problems are continuing to intensify (ICSU/ISSC 2015; IUCN 2017; WWF 2017). One general criticism of ESAMs is that they do not evaluate the sustainability performance of a system against the so-called *absolute environmental boundaries*; instead, they only rank the system relative to another system that is relevant to the nature (or the function) of the examined system (Bjørn and Hauschild 2013; Bjørn et al. 2016). As a result, the contributions of any examined system to the overall environmental impacts of production and consumption patterns of human societies are generally overlooked.

Scientists, therefore, have suggested adopting the concept of *absolute environmental sustainability* that focuses on how human societies can operate within the Earth's *carrying capacity* (Bjørn and Hauschild 2015; O'Neill et al. 2018). Achieving absolute environmental sustainability is, however, not straightforward as it involves addressing challenges such as prioritising key environmental impacts (hereinafter, impacts); defining appropriate indicators and boundaries; evaluating impacts at multiple economic levels (or scales) (from product/process to project, company, economic sector, country and the Earth); and understanding the interactions between various types of impacts (Chandrakumar and McLaren 2018b; Ryberg et al. 2016). Therefore, development of so-called *absolute environmental sustainability assessment* (AESA) methods is crucial in the pursuit of absolute environmental sustainability[11]. An AESA can be described as a benchmarking approach that classifies a system as absolutely sustainable by comparing its environmental impacts against a set of environmental boundaries (Bjørn et al. 2016).

Turning to the agri-food sector, today the sector is facing the huge challenge of meeting accelerating food demands due to growing population and changing dietary patterns (Garnett 2011; Godfray et al. 2010). Accelerating food demands are driving interventions like agricultural intensification and expansion, which have direct (and indirect) impacts on the environment, including in particular contributions to climate change, biodiversity loss, land-use change and water contamination (Chobtang et al. 2016, 2017; Ericksen et al. 2009; Smith et al. 2013). For example, it was calculated that the global agri-food sector contributes 19-29% of global climate impacts[12] (Vermeulen et al. 2012). In turn, these impacts threaten food security through changing patterns of agricultural production that affect food prices and food quality (Garnett et al. 2017). It is, therefore, crucial to investigate how current and future agri-food systems can operate and develop within absolute environmental boundaries including the ones proposed for climate change (Godfray et al. 2010; McLaren 2017; Repar et al. 2017; Smith et al. 2013; Wolff et al. 2017).

To that end, this study proposes the development of an innovative AESA method called Absolute Sustainability-based Life Cycle Assesment (ASLCA) to benchmark the climate impacts of agri-food systems against their shares of the 2°C global carbon budget at multiple economic levels, following a short literature review. Use of ASLCA is illustrated through a case study of the New Zealand agri-food sector, and the horticulture industry in particular. Finally, the results of the case study, limitations and future work required to develop ASLCA are discussed.

[11] In some literature, an AESA method is referred to as an absolute sustainability assessment method (ASAM) (Chandrakumar and McLaren 2018a, b).
[12] This includes GHG emissions related to land use and land system change.

4.2. A review of studies on absolute sustainability assessment methods

Rockström and colleagues (2009a, b) introduced the planetary boundaries (PBs) framework and illustrated how current human society is operating in terms of nine critical Earth system boundaries. Since then, scientists have been investigating how to apply the PBs to address absolute environmental sustainability on a sub-global level (Clift et al. 2017; Häyhä et al. 2016; Muñoz and Sabag 2017; Sim et al. 2016). A selection of AESA studies known to the authors of this article is presented in Table 4. 1, classified according to key research areas; the most relevant ones are discussed below.

Although the PBs provide a scientific basis for addressing absolute environmental sustainability, there are a number of outstanding challenges in achieving absolute environmental sustainability, and in development and use of AESA methods in pursuit of this objective (e.g. Laurent and Owsianiak 2017; Ryberg et al. 2016). Regarding AESA methods, Chandrakumar and McLaren (2018b) advocated that an AESA should: quantitatively assess a comprehensive range of environmental impacts in absolute terms; inform the impacts at an early stage in impact pathways; and be capable of benchmarking the impacts at multiple economic levels. They identified twelve key environmental impacts (including the ones listed in the PBs) for inclusion in an AESA.

Regarding assessment at multiple economic levels, methodologies such as LCA (ISO 2006a, b) and environmental footprints (ISO 2013, 2014) can be used to calculate the impacts associated with a chosen system. However, research on developing environmental boundaries has only emerged recently. According to Häyhä et al. (2016), when developing boundaries at multiple economic levels, three key dimensions should be considered. The first dimension is biophysical which characterises a particular environmental impact as a *systemic* process with global boundaries (e.g. climate change, ozone depletion) or an *aggregated* process with regional/local global boundaries (e.g. biogeochemical flows, land-system change). The second dimension is socio-economic that evaluates the interactions between socio-economic activities and environmental impacts (e.g. use of a production- or consumption-based accounting approach for impact assessment). The third dimension is ethical which addresses the challenge of downscaling global boundaries (particularly, systemic processes) to other sub-global levels (e.g. country, sector, company, product) and what ethical principles (or rationales) should guide this process. Some researchers have already started addressing these dimensions in proposed methods to downscale the global boundaries of systemic processes to sub-global levels (e.g. Bjørn et al. 2015; Meinshausen et al. 2009; Sandin et al. 2015), while others are developing regional (or local) boundaries for aggregated processes (e.g. Dearing et al. 2014; Teah et al. 2016).

Regarding the ethical dimension, several methods are available in the literature, which guide downscaling the global boundaries in proportion to past environmental impacts, population, mass, economic value, and relative contribution to human well-being (e.g. Liu and Bakshi 2019; Ryberg et al. 2018a). A detailed discussion is available in (Bjørn 2015).

Additionally, in the Science Based Targets (SBT 2017) initiative, methods like the Greenhouse Gas Emissions per unit of Value-Added (GEVA) and the Sectoral Decarbonization Approach (SDA) have been proposed to guide companies to set greenhouse gas (GHG) emissions reduction targets. The GEVA recommends a carbon intensity reduction rate of 5% per year for all economic sectors and their companies, stating that it is sufficient to achieve a 50% reduction in GHG emissions globally in 2050 (Randers 2012). The method, however, fails to account for heterogeneities between regions, countries, sectors or companies (e.g. current environmental and economic performances, mitigation potentials, costs). This limitation was later addressed through the SDA, which translated the sectoral emission pathways[13] of the International Energy Authority (IEA 2014) into sectoral intensity pathways[14] using physical (e.g. tonne of cement manufactured) or economic (e.g. value added) indicators, and then translated them into individual company intensity pathways (Krabbe et al. 2015). However, the SDA fails to address all economic sectors, activities, or types of GHG emissions. Hence, it is best suited for companies in the *homogeneous sectors* (e.g. power generation, iron and steel, cement), and further research is necessary to develop methods for companies belonging to *heterogeneous sectors*[15] (e.g. agriculture, forestry, construction) (Clift et al. 2017; Giesekam et al. 2018).

[13] Emission pathways provide information on how emissions are likely to develop in future by accounting for the past emissions and future economic and technological developments.
[14] Intensity pathway=emission pathway/activity projections; here, an activity can be either physical or economic activity.
[15] A sector that cannot be represented using a single physical indicator due to the uniqueness of the characteristics of the sector or the difficulties in comparing them (SBT 2017).

Table 4. 1: Key topics and related publications in absolute environmental sustainability research

Topics	References
Defining indicators and boundaries for environmental impacts	Carpenter and Bennett (2011)
	Clift et al. (2017)
	Cole et al. (2014)
	de Vries et al. (2013)
	Dearing et al. (2014)
	Gerten et al. (2013)
	Mace et al. (2014)
	Rockström et al. (2009a, b)
	Steffen et al. (2015)
	Teah et al. (2016)
Downscaling the global environmental boundaries into sub-global levels	Alcorn (2010)
	Bjørn et al. (2015, 2016)
	Brejnrod et al. (2017)
	Dao et al. (2015, 2018)
	Fanning and O'Neill (2016)
	Häyhä et al. (2016)
	Hoff et al. (2017)
	Krabbe et al. (2015)
	Nykvist (2013)
	O'Neill et al. (2018)
	Randers (2012)
	Ryberg et al. (2016, 2018a, 2018c)
	Sandin et al. (2015)
	SBT (2017)
Exploring the complementary linkages between ESAMs to develop AESA methods	Bjørn and Hauschild (2015)
	Bjørn et al. (2016)
	Chandrakumar and McLaren (2018a, b)
	Doka (2015, 2016)
	Dong and Hauschild (2017)
	Fang et al. (2015a)
	Laurent and Owsianiak (2017)
	Liu and Bakshi (2019)
	Muñoz and Sabag (2017)
	Repar et al. (2017)
	Ryberg et al. (2018b)
Supporting environmental sustainability-related policy- and decision-making	Barbier and Burgess (2017)
	Bendewald and Zhai (2013)
	Chandrakumar et al. (2018)
	Dao et al. (2015)
	Fang et al. (2015b)
	Galaz et al. (2012)
	Giesekam et al. (2018)
	Heijungs et al. (2014)
	Kara et al. (2018)
	Lucas and Wilting (2018)
	Nykvist (2013)
	Raworth (2012)
	Roos et al. (2016)
	Sandin et al. (2015)
	Sim et al. (2016)

Overall, then, the studies undertaken to date propose a number of methods (centred on various ethical principles) to benchmark the absolute environmental sustainability performance of countries, economic sectors and individual companies. However, they primarily focus on the homogeneous sectors and overlook heterogeneous sectors such as agriculture and forestry. In fact, to date, there is no specific study that has investigated the absolute environmental sustainability performance of agri-food systems at multiple economic levels.

As a step forward, this article presents an AESA method to downscale the environmental boundaries proposed for systemic processes in the context of agri-food systems (at multiple economic levels), and then to use these downscaled environmental boundaries (or budgets) to benchmark the environmental impacts of those agri-food systems. The method is elaborated for assessment of climate change as this is a systemic process where sufficient data are available to enable the calculations.

4.3. Methods

Benchmarking the climate impacts of any economic system requires the definition of a share of the carbon budget associated with a global climate boundary (Fanning and O'Neill 2016; Hoff et al. 2017). In the planetary boundaries (PBs), Steffen et al. (2015) proposed two global boundaries (i.e. a global average of CO_2 concentration of 350ppm CO_2 (or GHG concentration of 400ppm CO_2eq)[16] and a radiative forcing of 1 Wm^{-2}) based on the scientific evidence, which are sufficient to limit global warming to below 1.5°C. However, others have suggested that the global climate boundary "…should be in the middle of the PB range at 450 ppm CO_2eq…"(Clift et al. 2017), which would potentially limit global warming to below 2°C (Clift et al. 2017; Renaud and Matthews 2015)[17]. It should also be noted the 2°C target has already been ratified internationally and strategized within business activities (Clift et al. 2017; SBT 2017). This study, therefore, applied the 2°C target as the global climate boundary. Subsequently, using the method of Doka (2015, 2016)[18],

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[16] Note that the uncertainty zone for the CO_2 and GHG concentration are 350-450 ppm CO_2 (Steffen et al. 2015), 400-500 ppm CO_2eq (Clift et al. 2017), respectively.

[17] The 2°C target is also equivalent to a radiative forcing of 2.6 Wm^{-2} (Renaud and Matthews 2015).

[18] An annual carbon budget associated with a radiative forcing value can be calculated through dividing the radiative forcing (2.6 Wm^{-2} in this context) by the absolute global warming potential of CO_2eq (=8.69E-14 $(W{\cdot}yr)/(m^2{\cdot}kg\ GWP_{100yr}\ CO_2eq)$) (Doka 2015, 2016). Note that this annual carbon budget is independent of the assessment year.

an associated annual global carbon budget (CB_{Glo}) was calculated (= 29.9 $GtCO_2eq{\cdot}yr^{-1}$)[19]. Calculations are available in Table S2. 1 in Supplementary Material 2 (SM 2).

Regarding downscaling the 2°C carbon budget to sub-global levels, the share of the budget available for the global agri-food sector ($CB_{Glo,AgFd}$) was calculated based on the relative contribution of the global agri-food sector to global climate impacts, using the grandfathering principle (described below), as suggested by Alcorn (2010, p.147). This is the same as the approach used in the Sectoral Decarbonization Approach (SDA) method to calculate GHG emissions reduction targets on a (global) sector-level (Krabbe et al. 2015; SBT 2017)[20].

The GHG emissions from the global agri-food sector in 2013 were calculated based on the study by Vermeulen et al. (2012). Vermeulen et al. (2012) calculated the GHG emissions from the global agri-food sector in 2008 using the 100-year time horizon global warming potential (GWP_{100}) indicator, which included GHG emissions from a range of pre-production[21], production[22] and post-production[23] activities. These emissions were then adjusted to exclude catering, domestic food management and waste disposal, acknowledging the uncertainties associated with different practices undertaken in individual countries (Vermeulen et al. 2012). The adjusted climate impacts of the global agri-food sector were 13.1 $GtCO_2eq$, which was 23.6% of the global climate impacts in 2008 (see Table S2. 1 in SM2 for calculations). Assuming that this percentage contribution remained unchanged in 2013, $CB_{Glo,AgFd}$ was calculated as 7.1 $GtCO_2eq$ i.e. 23.6% of the CB_{Glo}.

[19] The distinction between different GHGs and their individual emission budgets have not been considered in this study due to two reasons. First, emission budgets are only available for carbon dioxide and methane, and research is still underway to estimate the budgets for other GHGs including nitrous oxide, hydrofluorocarbons and chlorofluorocarbons (Le Quéré et al. 2018). Second, the primary purpose of this study was to investigate the development of a SBT approach for the agri-food sector which is also aligned with the existing SBT approaches which consider all the GHGs together (proposed for sectors such as power generation, iron and steel, and cement, see Section 4.2).

[20] Note that the SDA method uses the data from the IEA 2°C scenario to break down the the carbon budget in the future (IEA 2014), whereas, an equal annual cabon budget is applied in this study.

[21] Pre-production: fertiliser manufacture, energy use in animal feed production, pesiticide production

[22] Production: direct and indirect emissions from agriculture

[23] Post-production: primary and secondary processing, storage, packaging and transport, refrigeration, retail activities, catering and domestic food management and waste disposal

Once the $CB_{Glo,AgFd}$ is calculated, it can be downscaled to sub-global levels and several methods are already available. However, considering the objective of the study (i.e. benchmarking the production-based climate impacts of different agri-food systems at multiple economic levels), four alternative methods were chosen for this particular study (descibed below). For each method, the $CB_{Glo,AgFd}$ was shared among the countries that produce agri-food products, and then among the agri-food industries of that country. Afterwards, the share of the carbon budget for each agri-food product was determined by dividing the carbon budget share of the agri-food industry by the total production volume of that particular agri-food product.

The first method is based on the grandfathering principle (hereafter called the **grandfathering method**) (Bjørn et al. 2016; Ryberg et al. 2018a). According to this method, the carbon budget is shared among emitters in terms of their current climate impact shares. The second method is centred on the idea that economic value can be considered a proxy for societal value creation (hereafter called the **economic method**) (Clift and Wright 2000), and sharing is based on the emitters' economic contribution to gross domestic product (GDP) (Bjørn et al. 2016).

The other two methods are (only) relevant to agri-food systems. The third method (hereafter called the **agri-land method**) represents the idea that agri-food systems require agricultural land (Smith et al. 2013); therefore, the carbon budget can be shared according to the territorial share of agricultural land of the emitter. A similar method has already been applied for downscaling the biogeochemical flows and land-system change PBs to sub-global levels (see Fanning and O'Neill 2016).

The final method is based on the calorie content of different foods (called the **calorific content method**). It uses calories as a proxy to represent the fact that the primary purpose of agri-food production is to feed people (Smith et al. 2013). Firstly, the total calories produced globally was calculated based on the production volume and calorific content (FAO 2018b; Roe et al. 2015) of a range of agri-food products listed in (FAO 2018b), for example, vegetables, fruits, nuts, livestock products and beverages (see Table S2.16 in SM2 for a complete list). Thereafter, the total calories available for human consumption was estimated by subtracting the loss of calories during agricultural production, livestock production (including animal feed) , handling, storage and transportation, and processing (see Tables S2.16 and S2. 18 in SM2). The $CB_{Glo,AgFd}$ was shared among the agri-food products, according to their calorific contirbution to the total calories available for human consumption. The carbon budget share for each agri-food system was then calculated with regard to the total production volume of that particular agri-food product.

4.4. An industry-level application of ASLCA

The agri-food sector is one of the largest economic sectors in New Zealand (NZ), making up more than 50% of the value of exported goods annually (Plant & FoodResearch 2016). In 2016, the horticulture industries contributed 10.3% of NZ's merchandise exports (equal to NZ$5 billion); of them, the exports from kiwifruit, apple and wine industries were dominant (Plant & FoodResearch 2016). Given the significance of NZ's "clean and green" image (OECD 2017; Seidel-Sterzik et al. 2018), it is interesting to investigate how NZ's dominant horticulture product categories perform relative to their 2°C carbon budget shares.

The following sub-sections discuss and present the calculation of (i) the climate impacts of the NZ agri-food sector, NZ horticulture industries (kiwifruit, apple and wine), and their products; (ii) the associated carbon budget shares; and (iii) the ASLCA results. Note that the study benchmarks the climate impacts in 2013 and, therefore, the data used for the analysis were based on that year unless specified otherwise.

4.4.1. Climate impacts of the NZ agri-food sector, horticulture industries and products

The climate impacts of the NZ agri-food sector were estimated by combining datasets from different sources (Fitzgerald et al. 2011a, b; MfE 2012; Stats NZ 2014). The direct and indirect impacts associated with NZ agricultural production activities were reported as 38.6 $MtCO_2eq$ in 2013 (FAO 2018a). The upstream and downstream impacts occurring within the "cradle-to-NZ port" system boundary were calculated using data in the national input-output tables for the year 2013 (Allan and Kerr 2016; Stats NZ 2014), and were 16.1 $MtCO_2eq$. These data included GHG emissions from a range of activities including fertiliser and pesticide production, electricity consumption, meat and meat products manufacturing, dairy product manufacturing, and local transportation (see Supplementary

Material (SM) 2 for a complete list). The data on GHGs emitted outside NZ during manufacture and onward transport of goods to NZ for use in agri-food systems, were derived from (Fitzgerald et al. 2011a, b), which were based on the fossil fuel consumption and maritime trade during the year 2007 (calculations are in SM2). They included the GHG emissions from international transportation activities associated with fertiliser and pesticide imports (=0.22 $MtCO_2$), animal feed imports (=0.16 $MtCO_2$) and NZ agri-food exports (=1.2 $MtCO_2$). The total climate impacts from these different activities, representing NZ agri-food sector's life cycle-based climate impacts, were determined as 56.3 $MtCO_2eq$.

The climate impacts of the NZ horticulture industries and products were based on the data collected during the 2007-2009 plantation season, as these were the most recent LCA studies available. On a product-level, Mithraratne et al. (2010) calculated the climate impacts of NZ kiwifruit produced in 2008. The climate impacts of a kilogram of green kiwifruit for the "cradle-to-retail gate" was taken from this study, and the sea freight life cycle stage was adjusted to represent a weighted distance from NZ to all NZ kiwifruit export destinations (see Table S2. 4 in SM2 for calculation and for details of export destinations, distances and export volumes). Based on these adjustments, the average climate impacts of one kilogram NZ kiwifruit (hereinafter "NZ kiwifruit") was calculated as 1,143 gCO_2eq.

Since there was no dataset explicitly reporting the climate impacts associated with the entire NZ kiwifruit industry, the product-level impacts reported by Mithraratne et al. (2010) were scaled up to the total production of NZ kiwifruit (a total of 399,947 tonnes in 2013, according to FAO (2018b)). Hence, the estimated climate impacts from the NZ kiwifruit industry were 457 $ktCO_2eq$ (see Table S2. 6 in SM2).

The same procedure was used to calculate the "cradle-to-retail gate" climate impacts of apple and wine produced in NZ and consumed in any part of the world (hereinafter "NZ apple" and "NZ wine"); the climate impact analyses of McLaren et al. (2009) and Barry (2011) were used for apple and wine, respectively. The adjusted climate impacts of one kg NZ apple and wine were 1,062 gCO_2eq (in 2007) and 1,640 gCO_2eq (in 2009), respectively. When scaled up to the industry-level, the climate impacts from the NZ apple and wine industries were 489 $ktCO_2eq$ and 408 $ktCO_2eq$, respectively (see Tables S 2. 10 and S2. 11 in SM2).

4.4.2. Carbon Budget Shares for NZ agri-food sector, horticulture industries and products

To calculate carbon budget shares at multiple economic levels, the available carbon budget share for the global agri-food sector ($CB_{Glo,AgFd}$, 7.1 $GtCO_2eq$) was downscaled using the four methods described above. According to the **grandfathering method**, given the NZ agri-food sector's share of the global agri-food sector's climate impacts in 2013 was 0.49%[24], the carbon budget share for the NZ agri-food sector was 35 $MtCO_2eq$. In the same year, of the NZ agri-food sector's climate impacts, the shares of the NZ kiwifruit, apple and wine industries were 0.81%, 0.87% and 0.72%, respectively. Therefore, the carbon budget shares for the NZ kiwifruit, apple and wine industries were 284 $ktCO_2eq$, 303 $ktCO_2eq$ and 253 $ktCO_2eq$, individually. Subsequently, downscaling the industry-level carbon budget shares to product-level (based on the production volume) resulted in individual carbon budget shares of 709 gCO_2eq/kg kiwifruit, 659 gCO_2eq/kg apple and 1,017 gCO_2eq/kg wine for the three horticulture products (for calculations, see SI1).

Using the **economic method**, the share of the $CB_{Glo,AgFd}$ for the NZ agri-food sector was calculated based on its relative contribution (US$21.8 billion, FAO 2018d) to the global agri-food sector's GDP (US$1.34 trillion, FAO 2018d). Therefore, the carbon budget share of the NZ agri-food sector was 115.1 $MtCO_2eq$. This share was further downscaled to industry- and product-levels based on the same principle. Given the relative contribution of the NZ kiwifruit, apple and wine industries to the NZ agri-food sector's GDP were 1.4%, 0. 63% and 2.8%, respectively, the corresponding carbon budget shares were 1,633 $ktCO_2eq$, 723 $ktCO_2eq$ and 3,194 $ktCO_2eq$. The product-level carbon budget shares were then calculated based on the production volumes of each industry as 4,084 gCO_2eq/kg kiwifruit, 1,571 gCO_2eq/kg apple, and 12,858 gCO_2eq/kg wine.

In the **agri-land method**, carbon budget shares for the different agri-food systems were calculated based on the agricultural land used by these systems. According to FAO (2018c), of the globally available agricultural land (=48,843,738,300 ha), 11,106,000 ha is located in NZ. Therefore, the share of the carbon budget assigned to the NZ agri-food sector was

[24] It is the ratio between the climate impact of the New Zealand agri-food sector (=56.3 $MtCO_2eq$) and the global agri-food sector (=13.1 $GtCO_2eq$).

1.6 $MtCO_2eq$. A share of this NZ budget was then assigned to each of the three horticulture industries based on their agricultural land use; the carbon budget shares were 1.8 $ktCO_2eq$, 1.3 $ktCO_2eq$, and 5.2 $ktCO_2eq$ for the NZ kiwifruit, apple and wine industries, respectively. Based on the production volumes of these industries (FAO 2018b), the product-level carbon budget shares were calculated as 4.5 gCO_2eq/kg kiwifruit, 2.7 gCO_2eq/kg apple, and 21 gCO_2eq/kg wine.

Finally, using the **calorific content method**, carbon budget shares were calculated according to the calorific contribution of different agri-food systems to the total calories produced by the global agri-food sector for human consumption. Of the total calories produced globally in 2013 (=7.3×10^{15} Cal), 8.2×10^{12} Cal were produced in NZ (see Table S2. 18 in SM2). The carbon budget share of the NZ agri-food sector was therefore 7.9 $MtCO_2eq$. This was further downscaled to the industry- and product-levels based on their associated calorific contributions. According to that, the carbon budget shares for the NZ kiwifruit, apple and wine industries were 17 $ktCO_2eq$, 123 $ktCO_2eq$ and 49 $ktCO_2eq$, respectively. According to the production volumes (available for human consumption) of these industries, the product-level carbon budget shares were calculated as 48 gCO_2eq/kg kiwifruit, 421 gCO_2eq/kg apple, and 646 gCO_2eq/kg wine. See SM2 for detailed calculations.

4.4.3. The ASLCA results

Figure 4. 1 graphically represents how different agri-food systems in NZ operate in terms of their 2°C carbon budget shares calculated using the four alternative methods (for the year 2013). Given the calculated climate impacts of the NZ agri-food sector were 56.3 $MtCO_2eq$, the sector operated within its carbon budget share in the given year only for the **economic method**.

The results in Figure 4. 1(a) inform that the carbon budget shares of the NZ agri-food sector calculated using the four methods varied significantly; the highest (115 $MtCO_2eq$) and the lowest (1.6 $MtCO_2eq$) shares were obtained using the **economic** and **agri-land methods**, respectively. Similar results were obtained for the studied NZ horticultural industries and their products (see Figures 4. 1(b) and 4. 1(c)).

The carbon budget results obtained for the sector can be explained by the specific characteristics of that sector. A high proportion of agricultural land in NZ is used for extensive grazing of livestock (Parliamentary Commissioner for the Environment 2016); as a result, NZ has significantly contributed to the total global production of dairy and meat

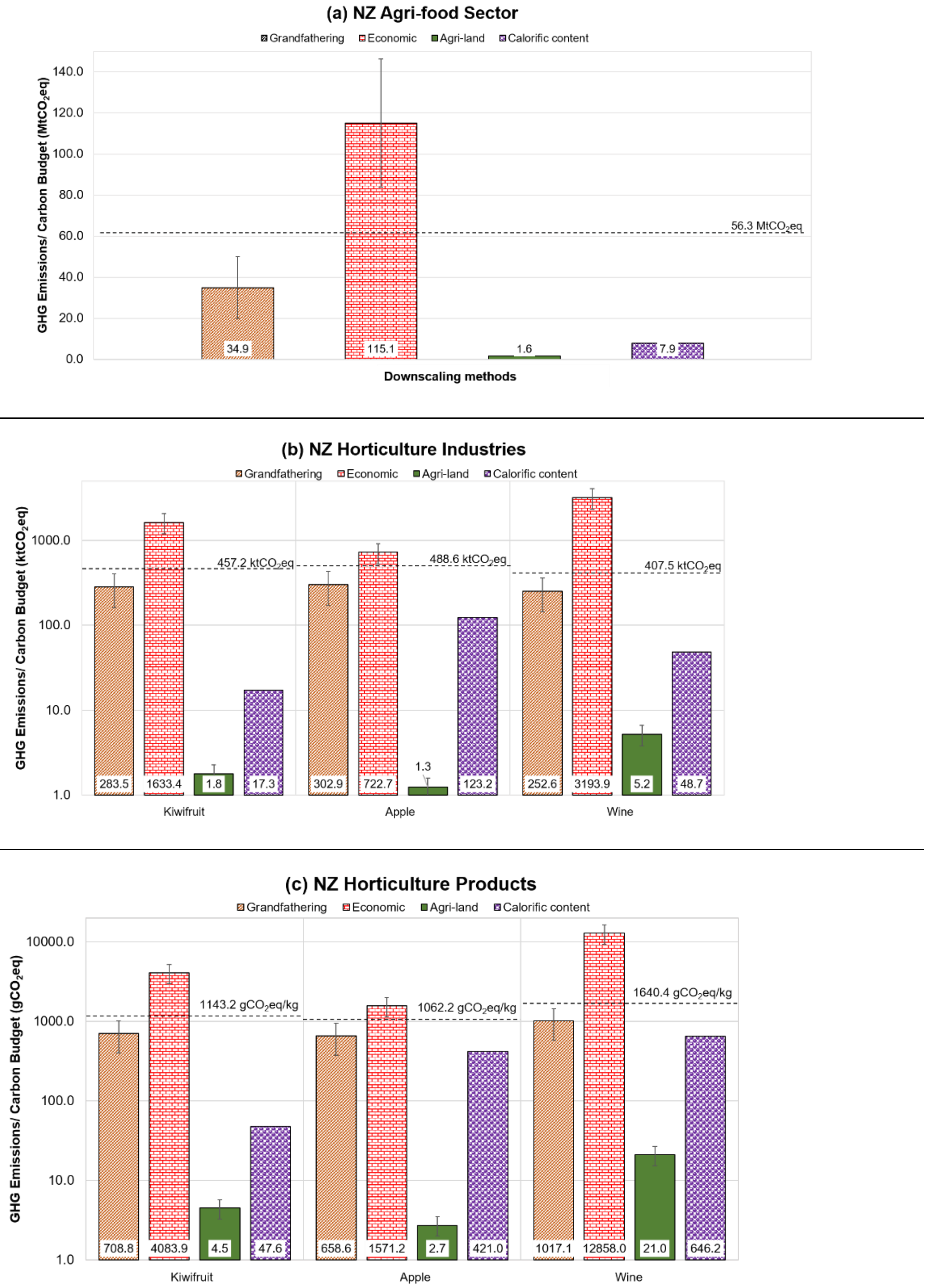

Figure 4. 1: Benchmarking the climate impacts of NZ agri-food systems: (a) NZ agri-food sector; (b) NZ horticultural industries; and (c) NZ horticulture products. The dashed line represents the climate impacts of the agri-food systems at those economic levels. Error bars represent the uncertainty associated with the data/results (the uncertainty data related to the calorific content method was not available) (Chandrakumar et al. 2019). Refer to SM 2 for calculations.

products (by mass, see Tables S2. 16 and S2. 18). These are all products with relatively high carbon footprints (see, for example, Chobtang et al. 2016, 2017), economic values, and calories compared with many other agri-food products on a mass basis (e.g. many cereals, fruits and vegetables); however, the yield per hectare is low compared with many other major agri-food products (e.g. maize, potatoes, sugar cane). Therefore the carbon budget for the NZ agri-food sector is relatively high when using the **grandfathering**, **economic** and **calorific content methods**, and low when using the **agri-land method**. The particularly high carbon budget using the **economic method** is explained by the fact that many NZ agri-food sector products achieve price premiums in international markets (Plant & Food Research 2016).

According to Figure 4. 1(b), the NZ kiwifruit, apple and wine industries generally have a greater difference in carbon budget shares between the **economic** and the **other methods**, compared with the NZ agri-food sector results. This is due to the premium prices achieved for these products, their relatively higher yields per hectare (at least for kiwifruit and apples), and lower calorific content per kg, compared with other major agri-food products produced in NZ (such as dairy and meat products). As seen in Figure 4. 1(b), the carbon budget share calculated using the **economic method** for the NZ wine industry (3,194 $ktCO_2eq$) is between two and four times larger than the carbon budget shares of the NZ kiwifruit (1,633 $ktCO_2eq$) and apple (723 $ktCO_2eq$) industries; this is despite the fact that the wine industry produces roughly 40% less product (by mass) than the other two industries. These results provide an example of how an agri-food industry (i.e. the wine industry in this case) receives a relatively higher carbon budget share (using the **economic method**) as it adds more value to its agri-food supply chain.

The carbon budget shares calculated for the NZ agri-food products present similar patterns of variation between results for the different methods as for the NZ agri-food industries. However, it is worth noting that, using the **grandfathering method**, the carbon budget share assigned to the NZ wine is the highest (1,017 gCO_2eq/kg wine) while the NZ apple is the lowest (659 gCO_2eq/kg apple), and this is opposite to the ranking at industry level where the carbon budget share assigned to the NZ apple industry (303 $ktCO_2eq$) is higher than the share assigned to the wine industry (253 $ktCO_2eq$). This is explained by the higher climate impacts of NZ wine (per kg) but lower production volume of the wine industry, compared with the apple industry. Moreover, according to the **calorific content method**, at product level, the carbon budget of the NZ wine is the highest (646 gCO_2eq/kg wine); however, at industry level, the carbon budget of the NZ apple industry is the highest (123 $ktCO_2eq$). These differences are explained by the different production volumes and calorie contents of these agri-food products.

4.5. Discussion

This study demonstrates that the ASLCA method can provide a basis for setting GHG emissions reduction targets to operate agri-food systems within carbon budget shares. Setting a sector-level target for the NZ agri-food sector may influence its supply chain stakeholders (e.g. industries, companies, farmers) to align their business activities by indicating the level of GHG emissions reductions (in absolute terms) at industry-, company- and farm-levels. This target-setting process will provide information about inefficient producers and their poor practices, which may incentivise the development of a range of technical and managerial initiatives. Examples include adaptation of land management practices, improved pest and disease management practices, development of more sustainable supply chains, and uptake of innovative technologies (Garnett 2011; Smith et al. 2013; Vermeulen et al. 2012). At the same time, ASLCA can be used to inform consumers about the climate impacts of a range of agri-food products and services.

However, the case study results illustrate the significance of the choice of method for downscaling the global carbon budget to sub-global levels. Since each of the four alternative methods (i.e. **grandfathering**, **economic**, **agri-land**, and **calorific content**) effectively rewards (via higher carbon budget shares) some countries, industries and/or producers over others, it is, therefore, important to reflect on the implications of choice of any one method. The **grandfathering method** effectively rewards the larger emitters with larger carbon budget shares, leaving small emitters (who may have lower emission intensities) with smaller, or potentially even zero, carbon budget shares. This method, therefore, provides little incentive to countries (and/or industries and/or producers) that are operating agricultural systems inefficiently to become more efficient, and fails to recognize producers who are already operating agricultural systems efficiently. In a similar way, the **agri-land method** effectively rewards emitters (via higher carbon budgets) who have more agricultural land available, and fails to reward (via higher carbon budgets) those countries, industries and/or producers with high production efficiencies. Also, this method is based on an intrinsic assumption that agricultural land will inevitably be used for agri-food production (Smith et al. 2013). The **economic method** assigns higher carbon budget shares to those countries (and/or industries and/or producers) producing high-value agri-food products per unit of GHG emission (such as some alcoholic beverages, vegetable oils, nuts, sweet treats) but there is no recognition of the role of agri-food systems in meeting the nutritional requirements of the world's human population. The **calorific-content method** overcomes this shortcoming of the economic method by assigning carbon budget shares for different agri-food systems based on the calorie contribution of those systems; however, it fails to account for the range of nutrients provided by different agri-food products. Future research should, therefore, focus on developing a greater understanding of the role of different carbon accounting methods in incentivising desired changes in agri-food systems.

Regarding other modelling aspects, the selection of global climate boundary and method to calculate the carbon budget has different implications for decision-making. In this study, the 2°C target was used as the global boundary and an associated annual carbon budget was calculated (=29.9 $GtCO_2eq$) using the method of Doka (2015, 2016). Although the 2°C target has been ratified globally and strategized within different business activities (e.g. Clift et al. 2017; SBT 2017), it is recognised that the target is partly political as it does not address all types of climate impacts (Bjørn and Hauschild 2015). Future benchmarking studies should, therefore, consider adopting the global boundaries defined in the planetary boundaries, which are solely based on scientific analysis (Steffen et al. 2015).

There are alternative methods for calculating an annual carbon budget which give different values (e.g. Alcorn 2010; Bjørn and Hauschild 2015; Dao et al. 2018). For example, applying the Absolute Global Temperature change Potential (AGTP) metrics for GHG emissions, Bjørn and Hauschild (2015) estimated an annual carbon budget of 6.8 $GtCO_2eq$ for the 2°C target. This is approximately one-fifth of the carbon budget applied in this work. If this carbon budget is used instead, all the agri-food systems considered in this study would have transgressed their carbon budget shares in all methods.

Moreover, the metric of Global Warming Potential for a 100-year time horizon (GWP_{100}) was used to calculate the climate impacts of agri-food systems. This metric has been criticized for assigning equal weights to GHG emissions regardless of emission time; relying on the integrative radiative forcing; and failing to address the importance of short-lived GHG emissions (e.g. methane, hydrofluorocarbons, sulphates) (Jørgensen et al. 2014; Reisinger et al. 2013; Reisinger et al. 2017). Researchers have started developing alternative metrics, approaches and time horizons, such as the Global Temperature change Potential (GTP) and Climate Tipping Potential (CTP) (Jørgensen et al. 2014; Owsianiak et al. 2018; Reisinger et al. 2017). Although these alternative metrics and time horizons are relevant to this study, and particularly in relation to the assessment of short-lived GHG emissions from agri-food systems, this aspect is not addressed in this study and is recommended for future research.

4.6. Conclusions

This article contributes to the emerging field of absolute environmental sustainability by introducing an AESA method (called ASLCA), which informs climate impacts of different agri-food systems in absolute terms. The study used a novel approach to benchmark the climate impacts of agri-food systems, and in particular the NZ agri-food sector, NZ horticulture industries and their products. The proposed method provides a basis for calculating GHG emissions reduction targets for agri-food systems at multiple economic levels. Moreover, it has the potential to provide information about inefficient producers and their poor practices, which could incentivise the development of a range of technical and managerial initiatives. At the same time, it can provide information to consumers about the climate impacts (in absolute terms) of a range of agri-food products and services. However, as described in the previous section, a number of aspects of the proposed method require further consideration and development in order to utilise the ASLCA as a comprehensive AESA framework as envisaged by Chandrakumar and McLaren (2018b).

Overall, then, the study provides some insights into the complex topic of how to achieve food security within the 2°C global carbon budget. Of course, other environmental, social and economic aspects associated with agri-food systems must be considered, alongside climate impacts, in the quest for sustainable agri-food systems (Biermann et al. 2016; McLaren 2017; Sala et al. 2017). In addition, an important methodological aspect concerns how to account for environmental impacts whose boundaries are only meaningful when defined at a regional or local level (e.g. eutrophication, acidification). However, use of absolute environmental sustainability-based budgets at multiple economic levels provides a mechanism for supporting global governance capable of delivering a sustainable future for humanity.

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Chapter 5: An absolute sustainability approach for setting climate targets for buildings: the case of a New Zealand detached house

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Abstract

Climate change mitigation requires the construction of low-carbon buildings, and this is a challenge for designers. The use of Life Cycle Assessment (LCA) provides useful information to support eco-efficiency improvements and therefore, to reduce the climate impact of buildings. However, it does not ascertain whether a proposed design aligns with achieving the global climate target of limiting global warming to below 1.5°C or 2°C. This study, therefore, introduces an absolute sustainability-based approach for setting climate targets for individual buildings using a whole-of-life cycle perspective. It involves assigning a share of the 2°C global carbon budget for 2018-2050 to a country, its construction sector, and finally to each life cycle stage of a building. A stock model is used to account for the projected growth in the number of buildings and associated climate impact in a country up to 2050. The approach was applied to define a climate target for a New Zealand new-built detached house, the most common residential building type in the country. The climate impact of the New Zealand new-built detached house was compared with the defined climate target to understand the scale of change required for operating this type of building within the 2°C climate target. The results showed that the new-built detached house exceeded its climate target by a factor of five. When the climate impact was compared with the climate targets at each life cycle stage, exceedances were a factor three to five higher across the different life cycle stages. This modelling approach has potential to guide designers and other interested stakeholders in development of building designs, to enable the construction sector to operate within a selected global climate target (such as the 1.5°C or 2°C climate target).

Keywords: absolute environmental sustainability, climate change, residential building, life cycle assessment, New Zealand

5.1. Introduction

The construction sector addresses several human needs (e.g. provision of housing, hospitals, schools and transport infrastructure) but at the cost of a range of environmental impacts including climate change (IPCC 2014; UNEP 2007). For example, the construction sector[25] uses an estimated 40% of global energy and therefore contributes around 30% of global greenhouse gas (GHG) emissions annually (UNEP 2007). At the same time, due to growing populations and economies around the world, there is high demand for construction, and this is expected to cause more climate impacts in future. It is, therefore, critical to consider the issue of climate change mitigation for the construction sector (Giesekam et al. 2018).

Efforts to mitigate the climate impact of this sector in the past have tended to focus on the use stage of buildings. However, as the operational energy of buildings declines due to the use of high-efficient appliances, windows and insulation in walls, ceilings and floors, and there is greater uptake of renewable energy, researchers are becoming more interested in opportunities to reduce the so-called "embodied GHG emissions" (Ghose et al. 2019; Ibn-Mohammed et al. 2013). These are the emissions associated with the manufacturing of construction materials, and the construction, maintenance and end-of-life of buildings (Chastas et al. 2018; Hollberg et al. 2019). This requires analysis of the climate impact associated with buildings throughout their complete life cycles (Hoxha et al. 2016; Zimmermann et al. 2005). Life Cycle Assessment (LCA, ISO 2006a, b) accounts for all inputs and outputs in the complete life cycle of a building and can be used for this type of analysis. Evaluating the climate impact of a building using LCA is, however, not sufficient to mitigate climate change globally (Brejnrod et al. 2017; Hollberg et al. 2019), as LCA only provides information about the climate impact of a building relative to another building and does not provide information about the building's performance in terms of any global climate target (or threshold) (Bjørn and Hauschild 2015; Chandrakumar and McLaren 2018a, b; Hauschild 2015). For example, building X may be considered better than building

[25] Includes the operational use stages of buildings.

Y if it emits less GHG emissions over its lifetime; however, it may be that neither of them can be considered sustainable if their GHG emissions are more than their assigned shares of the global carbon budget. This insight has led researchers to focus on the development of benchmarks using a top-down approach (Chandrakumar et al. Accepted; Hollberg et al. 2019; Hoxha et al. 2016; Lützkendorf et al. 2012; Russell-Smith et al. 2015; Zimmermann et al. 2005). A top-down benchmark, in general, aims to cascade global climate targets down to sub-global levels, enabling the quantification of individual building target values (Hoxha et al. 2016).

Some researchers have already calculated top-down benchmarks for buildings. For example, Zimmermann et al. (2005) suggested that the GHG emissions of a global citizen should be limited to 1 tonne carbon dioxide equivalent per capita per year [$tCO_2eq \cdot cap^{-1} \cdot yr^{-1}$] by 2050 to stay within the 2 degree Celsius (°C) climate target, according to the "2000 Watt society vision"[26]. They subsequently set a climate target for a Swiss single-family house based on the relative share of household expenditure for housing in Switzerland, applying the sharing principle of *final consumption expenditure*[27] (Brejnrod et al. 2017). Following this method, the climate target of a Swiss single-family house was 370 kilogram carbon dioxide equivalent per capita per year [$kgCO_2eq \cdot cap^{-1} \cdot yr^{-1}$]. Another similar top-down approach, also based on the 2000 Watt society vision (Heeren et al. 2012), was recently developed for Switzerland (Hollberg et al. 2019). When assigning a share of the carbon budget (of 1 $tCO_2eq \cdot cap^{-1} \cdot yr^{-1}$ in 2050) to a residential building, Hollberg et al. (2019) used the *grandfathering* sharing principle, which assigned a share of the carbon budget to the residential sector based on its relative contribution to national GHG emissions. According to this approach, the climate target of a Swiss single-family house was 360 $kgCO_2eq \cdot cap^{-1} \cdot yr^{-1}$.

[26] The vision suggests a global per capita energy use of 2000 Watt [W] in 2050, which is considered sufficient for all societies to develop and achieve an appropriate level of prosperity. If global warming is to be stabilised and natural resources are conserved, only 500 W can be generated by fossils, and the balance would be met by renewables. Hence, the vision complies with the Intergovernmental Panel for Climate Change (IPCC) target of approximately 10 $GtCO_2eq$ in 2050 for the 2°C temperature increase (IPCC 2001). If the global population is assumed to be 10 billion by 2050, the per capita carbon budget would be 1 $tCO_2eq \cdot cap^{-1} \cdot yr^{-1}$ in 2050 (Heeren et al. 2012; Hollberg et al. 2019; Zimmermann et al. 2005).

[27] This principle suggests that a share of the per-capita carbon budget should be assigned to housing based on the expense of housing, relative to a person's total expenses (Brejnrod et al. 2017).

In another study, Brejnrod et al. (2017) defined climate targets for a single-family house in Denmark (for the year 2010). They calculated the carbon budget available for a global citizen in 2010 for both i) 2°C global climate target (985 $kgCO_2eq{\cdot}cap^{-1}{\cdot}yr^{-1}$) and ii) 1 Watt per square meter [Wm^{-2}] climate change planetary boundary[28] (522 $kgCO_2eq{\cdot}cap^{-1}{\cdot}yr^{-1}$) targets. They used the sharing principle of final consumption expenditure, as was previously used by Zimmermann et al. (2005). Following this method, the climate targets of a Danish single-family house in 2010 were equal to 110 $kgCO_2eq{\cdot}cap^{-1}{\cdot}yr^{-1}$ for 2°C and 58 $kgCO_2eq{\cdot}cap^{-1}{\cdot}yr^{-1}$ for 1 Wm^{-2}. However, given the aim of the study was only to calculate GHG emission reduction targets for existing buildings (in the year 2010), no climate targets for future buildings were recommended.

Similar efforts to calculate climate targets for commercial buildings exist (Hoxha et al. 2016; Russell-Smith et al. 2015). For example, Russell-Smith et al. estimated a target of 2.29 $tCO_2eq{\cdot}m^{-2}$ for the whole life cycle of a commercial building in the USA, considering a 50-year lifetime (Russell-Smith et al. 2015). The target was based on the GHG emissions projections in the IPCC Fourth Assessment Report (IPCC 2007), which recommended a 70-80% GHG emissions reduction below 1990 levels by 2050 in order for buildings to operate within the 2°C climate target. Likewise, using a similar approach of Zimmermann et al. (2005), Hoxha et al. (2016) proposed climate targets for a set of commercial buildings in 2050, including offices (14 $kgCO_2eq$ per square metre floor area per annum [$kgCO_2eq{\cdot}m^{-2}{\cdot}yr^{-1}$]), restaurants (20.3 $kgCO_2eq{\cdot}m^{-2}{\cdot}yr^{-1}$), food stores (19.8 $kgCO_2eq{\cdot}m^{-2}{\cdot}yr^{-1}$) and hotels (11.7 $kgCO_2eq{\cdot}m^{-2}{\cdot}yr^{-1}$).

Although studies defining climate targets using a top-down approach for both residential and commercial buildings in different countries exist, no similar study is available for New Zealand. The climate targets proposed in other studies are not transferrable to New Zealand given the large variations in the construction materials, climate conditions and energy mix in different parts of the world. Moreover, the existing studies are limited in several aspects. In particular, while all the existing studies have considered population growth when setting

[28] Two climate change thresholds were proposed in the planetary boundaries framework: a global average atmospheric carbon dioxide (CO_2) concentration of 350 parts per million (ppm) CO_2 and a radiative forcing of 1 Wm^{-2}. These thresholds are sufficient to limit the atmospheric global average temperature to below 1.5 degree Celsius (°C) above pre-industrial levels (Steffen et al. 2015).

climate targets for buildings in 2050, none of them has modelled the growth in the number and size (i.e. floor area) of buildings nationally and/or globally through to 2050. However, temporal aspects such as the growth in the number and size of buildings are critical in determining the available share of the global carbon budget of a building, and should be addressed when setting climate targets for future buildings. Also, many of the studies have proposed a single climate target value for the whole life cycle of a building, making it more challenging for building designers to use the proposed target as a guide in the design process, given the lack of transparency regarding environmental hotspots at the different life cycle stages of a building.

In that context, this study developed an LCA-based top-down approach to calculate the climate target for both existing and future buildings in any country over a specified time period, and which provides a breakdown of this climate target into individual life cycle stages. To illustrate use of this approach, it was applied to define a climate target for a new-built detached house in New Zealand of the type commonly built in the country: detached houses make up almost 80% of residential buildings in New Zealand (Johnstone 2001a). Subsequently, the LCA climate impact of the New Zealand new-built detached house was calculated and compared against the proposed climate target in order to address the question, "Are New Zealand new-built detached houses aligned with achievement of the 2°C global climate target?"

5.2. Methods

Answering the question "Are New Zealand new-built detached houses aligned with achievement of the 2°C global climate target?" requires calculation of both the climate impact and the climate target of the New Zealand new-built detached house. To that end, the following sub-sections explain the procedure to calculate the climate impact (Sections 5.2.1 and 5.2.2) and climate target (Section 5.2.3) of the New Zealand new-built detached house.

5.2.1. Climate impact of New Zealand detached houses

The climate impact of a typical New Zealand detached house, built recently to meet the latest building regulations, was first calculated using LCA methodology (following the EN

15978:2011 standard (CEN 2011)). The typical New Zealand detached house (hereafter, referred to 'pre-existing detached house') is a single-storey building with an average gross floor area[29] of 166 m^2. It has an entrance directly to a living-room area, three bedrooms, 1.5 bathrooms, a kitchen, and a garage. Table 5.1 provides a summary of the characteristics of this typical detached house.

The climate impact of the new-built detached house (built between 2018 and 2050) was assumed to be the same as the typical detached house but scaled up to account for a larger gross floor area (207 m^2), except for the operational energy use[30] and operational water use[31] stages. The functional unit is the 'construction and occupation of a detached house over its lifetime'. For this study, an average lifetime of 90 years was considered, as previously estimated in (Johnstone 2001a, b).

Table 5. 1: Construction elements in a typical New Zealand detached house

Element	Characteristics/details
Structure	Softwood timber frame
Floor(s)	100 mm thick concrete slab with expanded polystyrene (25mm) around the perimeter
External walls	Weatherboard on a 90mm timber frame with fibreglass batt insulation and plasterboard lining
Internal walls	90mm wall no insulation and plasterboard lining
Windows	Aluminium double glazed (thermally unbroken)
Roof	Concrete tile with fibreglass batt insulation and timber frame trusses

Inventory data were categorised according to a modular structure into the following life cycle stages: product (A1-A3), construction process (A4-A5), maintenance (B2) and replacement (B4), operational energy use (B6), operational water use (B7), end-of-life (C1-C4), and benefits and loads beyond the system boundary (D).

For the *product stage*, the embodied GHG emissions of all types and quantities of materials used in the detached house were considered[32]. For the *construction process stage*,

[29] The gross floor area, in general, includes habitable areas (called net floor area) as well as non-habitable areas (e.g. garages or car ports).
[30] Electricity for interior lighting, heating, cooling and fans depends on the net floor area of the detached house, whereas electricity for plug loads and hot water depends on the household size. Hence, the electricity for interior lighting, heating, cooling and fans was scaled up to account for the change in gross floor area,
[31] Electricity for getting water in/out of the building (pumping and treatment) depends on the household size; thus, the electricity for pumping and treatment was assumed to be the same as for the pre-existing detached house.
[32] Note that the current LCA model does not include the following in the product stage: flashings, spouting, fitted kitchen units, cooker, dishwater, refrigerator, fitted bathroom units, bath, electrical (including wiring,

the GHG emissions associated with activities such as transportation, assembly and energy for the construction machinery[33] were estimated. Similarly, for the *maintenance and replacement* stages, the GHG emissions associated with activities such as painting and replacement were considered.

The GHG emissions related to the energy consumption (appliances, cooking, lighting, water heating and space conditioning) were calculated at the *operational energy use* stage[34]. Likewise, for the *operational water use stage*, the GHG emissions associated with energy consumption for getting water in/out of the building (pumping and treatment)[35] were considered. New Zealand detached houses utilise electricity as their main energy source but also use wood, natural gas and liquefied petroleum gas [28]. However, factors such as government policy on climate change, energy efficiency regulations, housing policy and building code requirements on insulation, and environmentally conscious consumers are likely to lead to greater uptake of electricity to meet the operational energy demand of New Zealand houses in future [29]. Therefore, for this study, the GHG emissions related to the operational energy use and operational water use stages were quantified assuming 100% electricity as the energy source, and using an annual New Zealand grid GHG intensity calculated based on MBIE's Mixed Renewables Scenario [30] for each year during the period 2018-2050[36]. The uncertainty associated with this modelling assumption is analysed in Supplementary Material (SM) 3.

For the *end-of-life stage*, the GHG emissions of the demolition activities of the building were considered. Finally, the GHG emissions (or benefits) associated with reuse, recovery

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switches, plug points, fuse box and meter), plumbing (including pipes and taps) and fixings (such as nails, screws and nail plates).

[33] Although the annual New Zealand grid GHG intensity reduces over the period 2018-2050, this aspect was not considered when calculating the GHG emissions related to the energy used for construction machinery, given their low contribution to the total climate impacts of the detached house over its lifetime. However, note that the annual change in GHG intensity was taken into consideration in the operational energy and water use stages.

[34] The operational energy demand was based on the simulated energy use in the detached house, so that the detached house is operated to maintain an internal temperature of between 18 and 25°C (for health reasons) as well as to provide hot water, lighting and plug loads (e.g. refrigerators, televisions, computers etc.). This simulation provides energy demand that is the same annually but its climate impact changes due to variations in the sources of electricity in future.

[35] In New Zealand, the operational water use energy demand is solely met by electricity.

[36] The annual New Zealand grid GHG intensity after 2050 was assumed to be the same as the intensity of the year 2050.

or recycling of construction materials which substitute for primary production were considered in *benefits and loads beyond the system boundary*.

A complete description of activities considered in this study is available in SM 3. The calculation of climate impact was undertaken in LCAQuick v3.0 (BRANZ 2018) using characterisation factors set out in EN 15804: 2012 + A1 (CEN 2013).

5.2.2. Climate impact of New Zealand detached housing sector

To estimate the climate impact of the New Zealand detached housing sector[37], a stock projection developed by BRANZ was used, which was based on several assumptions including socio-economic growth in different regions of New Zealand, gross floor area of a new-built detached house, and demolition rate (see SM 3). The model consisted of two components: one projected the growth in the number and total gross floor area[38] of detached houses up to 2050, and the other calculated the associated climate impact.

First, the total number and the total gross floor area of detached houses that existed at the end of 2017 ('pre-existing detached houses') were modelled, and projected up to 2050 based on their ages and assuming a 90-year lifetime. Next, the total number and total gross floor area of new-built detached houses for 2018-2050 were projected based on the long-term trend in building consents. Finally, the climate impact of the New Zealand detached housing sector for each year during 2018-2050 was estimated based on the calculated climate impact of the pre-existing and new-built detached houses, and the projected numbers of pre-existing and new-built detached houses for each year from 2018 to 2050.

5.2.3. Climate target of New Zealand new-built detached house

5.2.3.1. Overview of the top-down approach

The procedure for calculating the climate target for a building was:

[37] The term 'detached housing sector' refers to the total number of detached houses (including pre-existing and new-built) in New Zealand.

[38] Here, the term "total gross floor area" refers to the total gross floor area of detached houses (both pre-exist and new-built) in a particular year minus the total gross floor area (of detached houses) demolished in the given year.

a. Determine the maximum acceptable amount of GHG emissions that can be emitted globally while respecting the chosen global climate target during a particular time period (referred to as the global carbon budget) (Section 5.2.3.2).
b. Assign a share of the global carbon budget to a country based on population projections (Section 5.2.3.3).
c. Assign a share of the country's carbon budget to the country's construction sector[39] based on the relative contribution of the sector to the country's total climate impact in a chosen reference year (or period) (Section 5.2.3.4).
d. Calculate the climate target for different building categories by assigning the construction sector's carbon budget to the different building types based on the LCA climate impact of each building type and the projected number of those buildings, both pre-existing and new-built stock, in the chosen time period. Note that this means that, for example, buildings constructed in 2030 will only include 20 years of utilisation if the chosen time period extends to 2050 (Section 5.2.3.5).

The following sub-sections describe the proposed top-down approach in detail, applied to the New Zealand detached housing sector (see Figure 5. 1).

5.2.3.2. Global climate target and carbon budget

In this study, 2°C was chosen as the global climate target i.e. the maximum amount of GHG emissions that can be emitted and still limit average global warming to below 2°C above pre-industrial levels. The chosen global climate target was subsequently translated into a global carbon budget of 1110 GtCO_2eq for the period of 2018-2050 ($CB_{Glo,2018\text{-}2050}$), using the approach proposed by Rogelj et al. (2015). The year 2018 was chosen as the starting point due to the accessibility of better quality data developed as part of BRANZ's New Zealand whole-building whole-of-life framework research (BRANZ 2018). Data were modelled up to 2050, the year chosen for the target year of many on-going climate change negotiations, including the proposed New Zealand Climate Change Response (Zero Carbon) Amendment Bill (Shaw 2019).

[39] Note that the term includes the operation and end-of-life of buildings already constructed, and the construction and operation of buildings in the future.

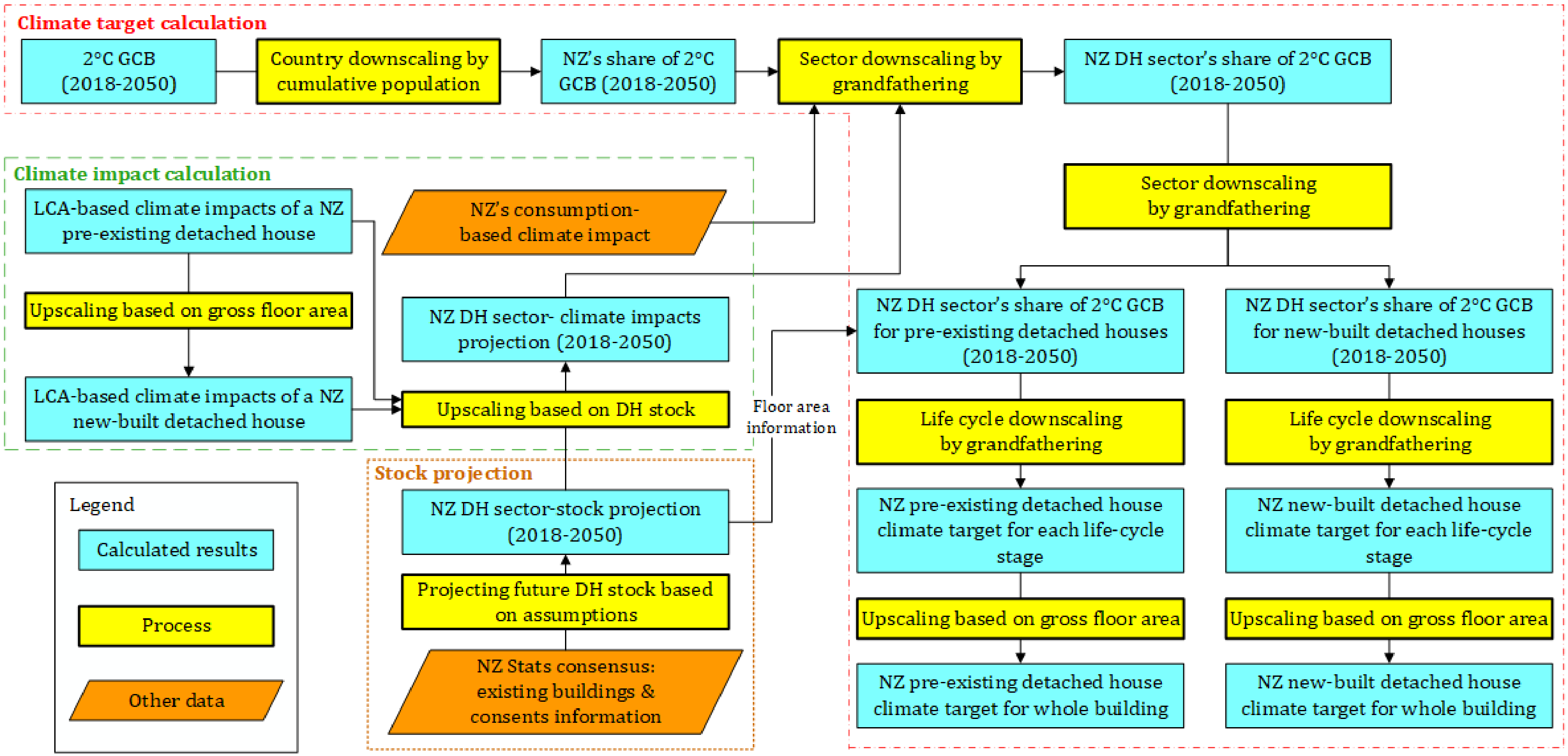

Figure 5. 1: Proposed top-down approach to calculate a climate target for a detached house in New Zealand (NZ). GCB= global carbon budget; and DH= detached house.

5.2.3.3. *Carbon budget of New Zealand*

To assign a share of the 2°C global carbon budget to New Zealand, the so-called sharing principle of *cumulative impacts per capita* was applied. This principle focuses on achieving equality in terms of the cumulative climate impact of different populations (Sandin et al. 2015; Yu et al. 2011). This means that, if the people of New Zealand emit more GHG emissions today than the global average per capita, future people of New Zealand should be restricted to emit a smaller proportion of GHG emissions in future based on the global carbon budget. Therefore, people in less-developed regions who may emit less GHG emissions today than the global average per capita, will be entitled to emit a higher proportion of GHG emissions in future based on the global carbon budget. The cumulative carbon budget available for New Zealand for 2018-2050 was calculated as follows:

$$CB_{NZ,2018\text{-}2050} = \frac{POP_{NZ,2018\text{-}2050}}{POP_{Glo,2018\text{-}2050}} \times CB_{Glo,2018\text{-}2050} \qquad (5.1)$$

where:

$CB_{NZ,2018\text{-}2050}$ - the share of the global carbon budget available for New Zealand for 2018-2050

$POP_{NZ,2018\text{-}2050}$ - the cumulative population of New Zealand for 2018-2050

$POP_{Glo,2018\text{-}2050}$ - the cumulative population of the world for 2018-2050

$CB_{Glo,2018\text{-}2050}$ - the global carbon budget for 2018-2050.

5.2.3.4. *Carbon budget of New Zealand detached housing sector*

The *grandfathering* sharing principle was used to assign a share of New Zealand's carbon budget to the New Zealand detached housing sector (as applied by Hollberg et al. (2019)). The grandfathering principle (Chandrakumar et al. 2018, 2019a) assigns a carbon budget share to the chosen sector based on its relative contribution to New Zealand's consumption-based climate impact in a reference year, as previously calculated by Chandrakumar et al. (2019b), represented in Eq. 5.2. Ideally, this year should have been 2017, which is the year prior to the period under analysis. However, due to data limitations[40], the year 2012 was

[40] At the time of undertaking this study, Chandrakumar et al. (2019) was the only peer-reviewed study that had estimated the consumption-based climate impact of New Zealand and it was for the year 2012.

selected, effectively assuming that the relative contribution of the detached housing sector to New Zealand's consumption-based climate impacts remained unchanged during the period from 2012 to 2050.

$$CB_{NZ,DH,2018\text{-}2050} = \frac{GHG_{NZ,DH,2012}}{GHG_{NZ,2012}} \times CB_{NZ,2018\text{-}2050} \tag{5.2}$$

where:

$CB_{NZ,DH,2018\text{-}2050}$ - the share of the global carbon budget available for the New Zealand detached housing sector for 2018-2050

$GHG_{NZ,DH,2012}$ - the GHG emissions of the New Zealand detached housing sector in 2012

$GHG_{NZ,2012}$ - the consumption-based GHG emissions of New Zealand in 2012

$CB_{NZ,2018\text{-}2050}$ - the share of the global carbon budget available for New Zealand for 2018-2050.

5.2.3.5. Climate target of New Zealand detached house

The share of the carbon budget available for the New Zealand detached housing sector for 2018-2050 ($CB_{NZ,DH,2018\text{-}2050}$) was divided between the pre-existing ($CB_{NZ,DH\text{-}PRE,2018\text{-}2050}$) and new-built ($CB_{NZ,DH\text{-}NEW,2018\text{-}2050}$) stocks, using the grandfathering principle i.e. based on their relative contributions to the climate impact of the New Zealand detached housing sector in 2018-2050.

Using the same grandfathering principle, a share of $CB_{NZ,DH\text{-}PRE,2018\text{-}2050}$ was then assigned to each life cycle stage based on its relative contribution to the climate impact of the pre-existing New Zealand detached housing sector. The same approach was applied to assign a share of $CB_{NZ,DH\text{-}NEW,2018\text{-}2050}$ to each life cycle stage of the new-built detached housing sector. Subsequently, the climate targets for 1 m^2 floor area for individual life cycle stages were determined (separately for pre-existing and new-built) by dividing the available carbon budget for each life cycle stage by the associated total gross floor area[41] of the pre-existing or new-built detached housing sector.

[41] See Section 5.2.2 for description about the calculation of total gross floor area.

Finally, the climate targets for individual pre-existing and new-built detached houses were derived by multiplying the climate targets for individual life cycle stages with the respective gross floor areas (166 m^2 and 207 m^2 for pre-existing and new-built, respectively) at each of those stages. The results were summed to give the total climate targets for individual pre-existing and new-built detached houses.

5.3. Results

5.3.1. Climate impact of New Zealand new-built detached house

Figure 5. 2 presents the climate impact of the new-built detached house over a 90-year lifetime and a breakdown of this impact in terms of individual life cycle stages. The climate impact of the new-built detached house over its lifetime is 277,170 $kgCO_2eq$, which is equivalent to 15 $kgCO_2eq{\cdot}m^{-2}{\cdot}yr^{-1}$. Note that the calculated climate impact of the new-built detached house does not account for the biogenic carbon in the building (particularly from the product stage, and replacement module in the use stage) and the avoided burden due to the reuse, recovery, recycling of construction materials. As observed from Figure 2, the embodied[42] and operational GHG emissions of the detached house over its lifetime are 104,967 $kgCO_2eq$ and 172,203 $kgCO_2eq$ respectively. The largest contributor of the climate impact of the new-built detached house is the operational energy use (55%), followed by maintenance and replacement (17%), product (13%), operational water use (7%), end-of-life (4.8%), and construction process stages (2.7%).

When the detached house was credited for the biogenic carbon in the building (a description of the method used in this study to calculate biogenic emissions is available in SM3) and the avoided burden due to the reuse, recovery, recycling of construction materials, the climate impact of the new-built detached house reduced by 16% (to 231,476 $kgCO_2eq$).

The climate impact of the New Zealand new-built detached house calculated in this study is comparable to the climate impact of residential buildings in other countries. The

[42] These are the emissions associated with manufacturing of construction materials (i.e. product stage), and the construction, maintenance and end-of-life of buildings.

climate impact of the New Zealand detached house (at 15 $kgCO_2eq \cdot m^{-2} \cdot yr^{-1}$) is at the lower end of the GHG emissions ranges presented in a recent review on the climate impact of residential buildings throughout the world (10-90 $kgCO_2eq \cdot m^{-2} \cdot yr^{-1}$, according to Chastas et al. (2018)). However, note that although the review considered 95 case study residential buildings around the world, none of them was from New Zealand.

5.3.2. Climate impact of New Zealand detached housing sector

The climate impact of the New Zealand detached housing sector in 2018-2050 is estimated as 140,945 $ktCO_2eq$ by scaling up the climate impact of both pre-existing and new-built detached houses based on the proposed stock projection (see Table 5. 2). According to Table 2, 67% of the climate impact is related to the pre-existing detached houses, and the remaining (33%) is related to the new-built[43] detached houses. The largest contributor to the sector's climate impact during the period is the operational energy use (59%), followed by the maintenance and replacement (16%), product (14%), operational water use (8%), construction process (2.8%), and end-of-life stages (0.4%)[44].

These results can be partly explained by the relative numbers of pre-existing and new-built detached houses that exist in 2018-2050. A small number of new-built detached houses are constructed in the given period (= 527,609) compared with the pre-existing stock, and this means that the total climate impact of the construction and product stages is relatively low. Also, the small number of demolished detached houses (= 50,820) during the given period means that the end-of-life stage contributes a small proportion of the total climate impact compared with other life cycle stages. At the same time, the large proportion of climate impact associated with the operational use stage at the sector level can be explained by the fact that this stage contributes the highest proportion of the climate impact of the detached house over its lifetime (see Figure 5. 2).

[43] That is, those built post 2018.
[44] Note that the benefits and loads beyond the system boundary were not considered here. For example, the impacts associated with product stage does not account for the biogenic emissions.

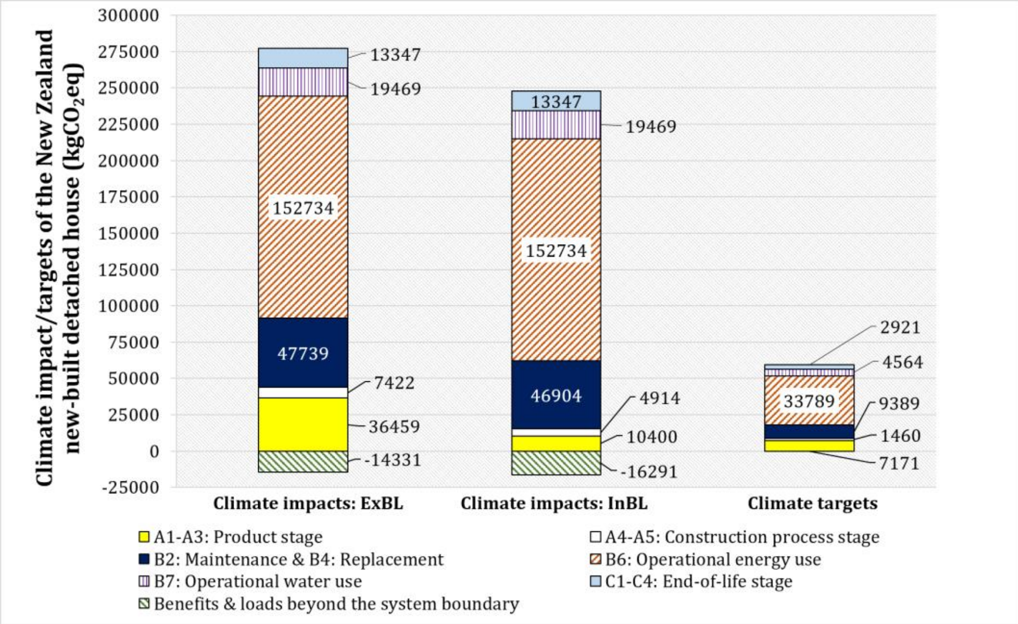

Figure 5. 2: LCA climate impacts of a typical New Zealand detached house over a 90-year lifetime. The first and second columns represent climate impact of the new-built detached house, excluding (ExBL scenario) and including (InBL scenario) the biogenic carbon in the building and the avoided burden due to the reuse, recovery, recycling of construction materials, respectively, whereas the third column represents the climate target of the new-built detached house for each life cycle stage. Note that no climate target is proposed for the life cycle stage "benefits and loads beyond the system boundary".

5.3.3. Climate taregt of New Zealand detached house

The climate targets for a detached house were first calculated in terms of individual life cycle stages. As presented in Table 5. 2, the climate targets were calculated as total $kgCO_2eq{\cdot}m^{-2}$ for the life cycle stages occurring at one point in time, and as total $kgCO_2eq{\cdot}m^{-2}{\cdot}yr^{-1}$ for the activities occurring repeatedly over a number of years. In the former category, the climate targets for the product (i.e. materials), construction process, and end-of-life stages are 35, 7 and 14 $kgCO_2eq{\cdot}m^{-2}$ respectively[45]. In the latter category, the climate targets for the maintenance and replacement, operational energy use, and operational water use stages are 0.51, 1.82 and 0.25 $kgCO_2eq{\cdot}m^{-2}{\cdot}yr^{-1}$, respectively.

Using the targets proposed for individual life cycle stages, the climate target for the whole life cycle of a New Zealand new-built detached house over a 90-year lifetime was calculated as 59,293 $kgCO_2eq$ (as shown in Figure 5. 2). This is equivalent to a climate target of

[45] The targets for the product and construction stages are only applicable to a new-built detached house, whereas the target for the end-of-life stages is only applicable to a pre-existing detached house.

244 $kgCO_2eq \cdot cap^{-1} \cdot yr^{-1}$, given the average New Zealand household size is 2.7 people (Stats NZ 2014). The calculated climate target for the New Zealand new-built detached house is smaller than the targets for the Swiss detached house (360-370 $kgCO_2eq \cdot cap^{-1} \cdot yr^{-1}$, according to Hollberg et al. (2019) and Zimmermann et al. (2005)) but greater than the target for the Danish detached house (110 $kgCO_2eq \cdot cap^{-1} \cdot yr^{-1}$, according to Brejnrod et al. (2017)). However, comparisons should be made with caution given the significant differences in methodologies and assumptions applied in previous studies (as already described in Section 5.1).

5.3.4. Climate performance of New Zealand new-built detached house and detached housing sector

When the climate impact of the new-built detached house was compared with the proposed climate target, it was found that the new-built detached house exceeded its climate target by a factor of 4.7 (see Figure 5. 3).

At the same time, when the climate impact was compared with the climate targets of individual life cycle stages, the largest exceedances were observed in the product (A1-A3) and construction process stages (A4-A5); they exceeded their climate targets by a factor of 5.1 (see ExBL scenario[46] in Figure 5.3). The third largest exceedance was associated with the end-of-life (C1-C4) stage (a factor of 4.6). The exceedances in the operational water use (B7), maintenance and replacement (B2-B4), operational energy use (B6) stages were 2.9, 2.8 and 2.8 respectively.

When the new-built detached house was credited for the biogenic carbon in the materials and the avoided burden due to their reuse, recovery, and recycling, the exceedances in the product (A1-A3) and construction process (A4-A5) stages reduced from 5.1 to 1.5 and 3.4, respectively, and the exceedance in the maintenance and replacement (B2-B4) stage reduced from 2.8 to 2.7 (see InBL scenario[47] in Figure 5. 3).

[46] ExBL scenario does not account for the environmental benefits and loads beyond the life cycle of the detached house.

[47] InBL scenario does account for the environmental benefits and loads beyond the life cycle of the detached house.

Table 5. 2: Climate impacts and total gross floor area of New Zealand detached housing (DH) sector, and the proposed climate target (CT) for the reference New Zealand detached house

Life cycle stage	Climate impact- DH sector for 2018-2050 [a] ($ktCO_2eq$)			Carbon budget - DH sector for 2018-2050 ($ktCO_2eq$)			Total gross floor area- DH sector for 2018-2050 (m^2)			CT per unit gross floor area ($kgCO_2eq \cdot m^{-2}$)	CT: new-built detached house over 90-year lifetime [a] ($kgCO_2eq$)
	Pre-existing	New-built	Total	Pre-existing	New-built	Total	Pre-existing	Future	Total		
Product	-	19,236	19,236 (14%)		3,783	3,783 (14%)	-	108,987,284	108,987,284	34.71	7,171 (12%)
Construction process (A4-A5)	-	3,916	3,916 (2.8%)		770	770 (2.8%)	-	108,987,284	108,987,284	7.07	1,460 (2.5%)
Maintenance & replacement (B2&B4)	17,974	5,085	23,060 (16%)	3,535	1,000	4,535 (16%)	6,999,735,240 [b]	1,980,409,867 [b]	8,980,145,107 [b]	0.51 [c]	9,389 (16%)
Operational energy use (B6)	66,711	16,270	82,981 (59%)	12,722	3,599	16,321 (59%)	6,999,735,240 [b]	1,980,409,867 [b]	8,980,145,107 [b]	1.82 [c]	33,789 (57%)
Operational water use (B7)	9,134	2,074	11,208 (8%)	1,718	486	2,204 (7.8%)	6,999,735,240 [b]	1,980,409,867 [b]	8,980,145,107 [b]	0.25 [c]	4,564 (7.7%)
End-of-life (C1-C4)	544	-	554 (0.4%)	107		107 (0.4%)	7,572,180	-	7,572,180	14.14	2,921 (4.9%)
Whole life cycle	**94,364 (67%)** [d]	**49,581 (33%)** [d]	**140,945 (100%)**	**18,560 (67%)** [e]	**9,162 (33%)** [e]	**27,721 (100%)**	-	-	-	-	**59,293 (100%)**

[a] The climate impact values of pre-existing and new-built detached houses were the same for the product, construction process, and end-of-life stages, since it was assumed that the future building would have similar construction materials and activities. However, the projected average floor area of a new-built detached house was modelled as 206.57 m^2, whereas the average floor area of a pre-existing detached was modelled as 165.78 m^2.

[b] Since the net floor area of this life cycle stage is a function of area and time the unit is square meter annum ($m^2 \cdot a$). For example, the number approved consents for the construction of detached houses in 2018 were 21,009 (representing 4,339,793 m^2 of total gross floor area), with a 32.5-year utilisation period (by the end of 2050). The total gross floor area and the utilisation period were first multiplied. This procedure was repeated for subsequent years and summed.

[c] The unit is $kgCO_2eq \cdot m^{-2} \cdot yr^{-1}$.

[d] The climate impact shown as percentages of the total climate impact of the New Zealand detached housing sector during 2018-2050.

[e] The carbon budgets shown as percentages of the total carbon budget share available for the New Zealand detached housing sector during 2018-2050.

Furthermore, when the climate impact (140,945 ktCO_2eq) of the New Zealand detached housing sector was compared with the assigned share of the global carbon budget (27,721 ktCO_2eq) in 2018-2050, the sector exceeded its climate target by a factor of 5.1.

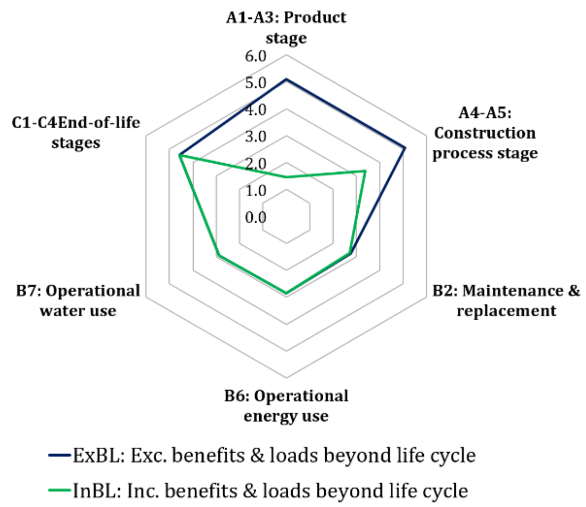

Figure 5. 3: Climate impact of the New Zealand new-built detached house over a 90-year lifetime compared with the proposed climate targets for individual life cycle stages. The ExBL scenario does not account for the biogenic carbon in the building and the avoided burden due to the reuse, recovery, recycling of construction materials, whereas the InBL scenario considers those emissions. The values shown in the diagram indicate the exceedances (i.e. ratios between climate impact and targets); for example, the value five (in ExBL scenario for A1-A3) indicates that the climate impact of this life cycle stage were five times greater than the proposed climate target. Note that no climate target is proposed for the life cycle stage "benefits and loads beyond the system boundary".

5.4. Discussion

The climate impact of the New Zealand new-built detached house calculated in this study was noticeably lower than the climate impact of detached houses in most other countries. These large variations in climate impact between houses in different countries arise from the significant differences in the construction materials, climate conditions and sources of energy supplying grid electricity. In New Zealand, timber is a common construction material for detached houses, whereas, in many other countries, bricks and concrete are commonly used (Cuéllar-Franca and Azapagic 2012; Ortiz et al. 2009). Furthermore, compared with other countries, approximately 40% of New Zealand's primary energy and around 85% of New Zealand's electricity are supplied from renewable energy sources (Madrigal 2015; MBIE 2016).

Although similar research to define climate targets for residential buildings exists (Brejnrod et al. 2017; Hollberg et al. 2019; Zimmermann et al. 2005), direct comparisons were not possible for three major reasons. Firstly, no study to date defined a climate target for a New Zealand detached house (or any other type of New Zealand residential building). Secondly, as described in Section 5.1, the existing studies have failed to account for any projected future growth in the number (and size) of buildings when calculating climate targets for buildings. Thirdly, all the existing studies propose a single climate target for the whole life cycle of a building, and not in terms of individual life cycle stages.

When the climate impact of the New Zealand new-built detached house was compared against the proposed climate targets, the results showed that the new-built detached house exceeded its climate target by a factor of five. At the same time, when the climate impact was compared with the climate targets for individual life cycle stages, exceedances were a factor three (in operational water use, maintenance and replacement, operational energy use) to five (in product and construction process stages) higher across the different life cycle stages. These results, therefore, suggest that substantial efforts (i.e. a reduction of approximately 80%) are necessary to align the climate performance of the New Zealand new-built detached house with achieving the 2°C global climate target.

At the sector level, the New Zealand detached housing sector exceeded its climate target by a factor of five (see Table 2). Out of the total climate impact of this sector during 2018-2050, 67% was associated with the pre-existing detached houses, and the remaining (33%) was related to the new-built detached houses. The largest contributors to the new-built detached housing sector during this period were the product (41%) and operational energy use (35%) life cycle stages. Most of the climate mitigation efforts should focus on mitigating climate impact associated with these two life cycle stages given they together contribute 76% of the climate impact of the new-built detached housing sector. Likewise, for the pre-existing detached housing sector, mitigation efforts should mainly focus on the operational energy use stage as it contributes 71% of the climate impact of the pre-existing detached housing sector during 2018-2050.

To that end, the use of timber from sustainable forestry management practices can make a contribution by lowering the embodied emissions of the New Zealand new-built detached housing sector (by up to 11%, see SM 3). On the other hand, achieving a similar reduction in the operational use stage requires technological and systemic changes. For example, the New Zealand government is currently investigating the potential to transition from 85% renewable electricity (at present, according to MBIE (2016)) to 100% renewable electricity by 2035 (MfE and Stats NZ 2019). Other opportunities exist around improving the energy efficiency of new-built buildings and reducing house sizes. Similarly, the redesign, retrofitting and refurbishment of pre-built detached houses, in particular to reduce operational energy use, should be an important focus for New Zealand's pre-existing detached housing sector.

Although this study provides key insights to support national climate policymaking by proposing a top-down approach and climate targets for detached houses, there are limitations in the proposed methodology. For example, when assigning a share of the total New Zealand carbon budget to the detached housing sector, the grandfathering principle was applied but other sharing principles could have been used instead (such as the final consumption expenditure principle). Ideally, the effects of the choice of alternative sharing principles on the outcomes should be investigated.

The calculated GHG emissions associated with the operational energy use and operational water use stages of the pre-existing New Zealand detached houses were based on the assumption that the energy requirement was completely met by electricity. However, in reality, some pre-existing detached houses use other energy sources such as wood, natural gas and liquefied petroleum gas in addition to electricity (as discussed in Section 5.2.1). To test the influence of this modelling assumption, the GHG emissions associated with the annual operational energy use and operational water use stages were calculated for two alternative scenarios representing an older and newer house, and using a mix of energy sources. It was found that the annual GHG emissions for these two life cycle stages were 12% lower (new house) or 8% higher (older house) than the baseline scenario (see calculations in SM 3). Therefore the annual GHG emissions for the operational energy use and operational water use stages calculated based on 100% electricity are likely to be fairly representative of the average NZ pre-existing detached house.

Similarly, when estimating the climate impact of the New Zealand detached housing sector, the stock projection model proposed by BRANZ (based on several assumptions) was used. Assumptions in this model include: the building materials used for new construction, the rates of socio-economic growth in different regions of New Zealand, the gross floor area of a newly built house (207 m^2), and a 0.1% rate of demolition of the total pre-existing detached house stock. These assumptions are, of course, associated with a significant amount of uncertainty (whose quantification was however outside the scope of this study). But they all are best estimates, made by experts in the field.

5.5. Conclusion

This research investigated the importance of accounting for the temporal aspects when setting climate targets for future buildings by proposing a new top-down approach. The approach, for the first time, includes a national stock projection that accounts for the forecasted growth in the number (and size) of buildings and their associated climate impact up to the year 2050. When the approach was applied to a New Zealand new-built detached house, it was found that aligning the climate impact of this building with the 2°C global climate target would require at least a factor five reduction in GHG emissions. In particular, based on the impacts at each life cycle stage, designers and other interested stakeholders should focus on developing and facilitating the uptake of disruptive technologies for low embodied carbon building materials and low-energy using houses, to leapfrog the gap between the performance of the current new-built detached house and its climate target. As new-built detached houses contribute 33% of the projected climate impact of the New Zealand detached housing sector for 2018-2050, this is an appropriate focus. However, as pre-existing detached houses contribute 67% of the projected climate impact of this sector over the same time period, and as almost three-quarters of that impact is due to operational energy use, then redesign, retrofitting and refurbishment of these pre-existing houses should be the highest priority.

The proposed top-down approach can also be applied to other residential building types (e.g. townhouse and apartment) as well as non-residential buildings (e.g. offices, schools and hospitals). Indeed, it is a generic method for calculating climate targets that account for the building requirements of future as well as current generations, and thus is aligned with the principle of sustainable development. The integration of these climate targets into LCA tools that are used by designers and other interested stakeholders (including architects, civil engineers, scientists and investors) has potential to catalyse innovation on a more ambitious scale than has been common to date, and which is required to enable countries to live within the planetary boundary for climate change.

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Chapter 6: Understanding New Zealand's consumption-based greenhouse gas emissions: an application of multi-regional input-output analysis

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Abstract

Consumption- and production-based accounting approaches for national greenhouse gas (GHG) emissions provide different insights to support climate policymaking. However, no study has yet comprehensively assessed the consumption-based GHG emissions of the entire New Zealand's economy. This research quantified New Zealand's GHG emissions using both consumption- and production-based accounting approaches, and considered the policy implications for adopting a consumption-based approach over a production-based approach. A global multi-regional input-output (MRIO) analysis was undertaken to calculate the consumption- and production-based GHG emissions of New Zealand for the year 2012. The MRIO analysis was based on the Eora database, which accounts for 14839 industry sectors from 189 countries. Given the sectoral classification of each country is quite different and in order to ease interpretation of the results, the industry sectors of each country were classified and aggregated into 16 key sectors, and GHG emissions were calculated for those key sectors. The MRIO analysis showed that New Zealand's consumption- and production-based GHG emissions in 2012 were 61,850 and 81,667 ktCO_2eq, respectively, indicating that the country was a net exporter of GHG emissions in 2012. The dominant contributors to the consumption-based GHG emissions were the other services and construction key sectors (each representing 16% of consumption-based emissions), followed by food and beverages (14%), transport and equipment (12%), and financial and trade services (11%), whereas, the dominant contributor to the production-based GHG emissions was the agriculture key sector (representing 52% of production-based GHG emissions). The results of the study provided two key insights to support climate mitigation activities and policymaking. First, the consumption- and production-based accounting approaches results have different rankings for the most dominant economic sectors contributing to New Zealand's GHG emissions. Second, only the consumption-based accounting approach enables the quantification of the embodied emissions in New Zealand's trade activities, and it indicated that a large proportion of GHG emissions are embodied in New Zealand's trade activities. This has important implications for future policies that could positively influence the consumption patterns of New Zealand citizens, and the production structure and efficiency of New Zealand's trade partners. As the consumption- and production-based accounting approaches provide different insights, both approaches should be used in a complementary way when developing climate policies. However, implementation of a consumption-based accounting to support development and implementation of climate policies and instruments requires further consideration.

Keywords: consumption-based accounting; production-based accounting · multi-regional input-output · Eora · greenhouse gas · climate change · New Zealand

6.1. Introduction

One of the most urgent environmental challenges of our time is climate change. The international community's response includes provisions for countries to pursue greenhouse gas (GHG) mitigation (UNFCCC 2015). Alongside many other countries, New Zealand is committed to mitigating climate change, and transitioning towards a low-carbon and climate-resilient economy (MfE 2017; OECD 2017).

New Zealand met its emissions reduction target under the first commitment period (2008-2012) of the Kyoto Protocol with the help of the New Zealand Emissions Trading Scheme (MfE 2013, 2017; OECD 2017). New Zealand, however, did not commit to meeting targets in the second commitment period (2013-2020); instead, the country adopted an unconditional target to reduce its emissions under the United Nations Framework on Climate Change (UNFCCC) (MfE 2017). This decision was at least partially influenced by New Zealand's unusual emissions profile compared with many other developed countries; in particular, New Zealand has a large proportion of biological GHG emissions (methane and nitrous oxide) in its emissions profile which are challenging to mitigate with existing technologies and farm management practices. According to MfE (2017), the agri-food sector contributes nearly half of New Zealand's GHG emissions annually, and more than 90% of the agri-food sector's emissions are biological emissions (MfE 2016; OECD 2017).

Under the Kyoto Protocol and its successor, the Paris Agreement, responsibility for GHG emissions is assigned to the country (or region) in which the GHG emissions are released (Caro et al. 2015; Steininger et al. 2018). This approach is known as production-based accounting (PBA). According to the PBA approach, New Zealand is only responsible for the GHG emissions released in the production of goods and services, including agri-food products and services. Although the approach is straightforward and widely used in many countries, it is often criticized for its limitations (e.g. Andrew and Forgie 2008; Franzen and Mader 2018; Steininger et al. 2018). For example, the PBA approach fails to consider the emissions associated with the imported goods and services that are used in a county (or region). Furthermore, it does not account for the emissions associated with international air and maritime transportation; hence, attribution of those emissions to specific countries is not straightforward (Afionis et al. 2017). A perverse outcome of the approach is that emission-intensive industries in countries with strict regulations (and/or taxes) may benefit financially from relocating to countries with weak regulations in order to avoid the need to reduce their GHG emissions (Arroyo-Currás et al. 2015; Franzen and Mader 2018). This effect is known as 'carbon leakage' and has been extensively documented in the climate change policy literature although it remains politically contentious and analytically difficult to quantify (Arroyo-Currás et al. 2015; Franzen and Mader 2018; Ward et al. 2015). Given these limitations, researchers have raised the question of whether we should adopt an alternative accounting approach for determining responsibilities (e.g. Afionis et al. 2017; Jackson 2011, 2016).

Consumption-based accounting (CBA) has so far emerged as the most prominent alternative; it attributes responsibility for GHG emissions to the final consumers of goods and services (e.g. Afionis et al. 2017; Le Quéré et al. 2018; Malik et al. 2018; Wiedmann 2009). The main difference between the two approaches is that the CBA approach provides a basis "…to cede responsibility for GHG emissions associated with a country's export production and accept responsibility for the embodied emissions of the imported goods and services" (Afionis et al. 2017). The CBA approach requires more sophisticated methods and a significant amount of data (Malik et al. 2018; Wiedmann 2009), resulting in a limited number of applications in the past. However, the development of multi-regional input-output (MRIO) analysis and several large-scale MRIO databases (e.g. Eora, EXIOBASE, GTAP, WIOD) with increasing global coverage, resolution, and accuracy of input-output data, have enabled more useful and accurate CBA studies in recent years (Malik et al. 2018; Wiedmann and Lenzen 2018; Wiedmann et al. 2011).

There is a growing interest in quantifying the consumption-based GHG emissions of countries (Le Quéré et al. 2018; Meinshausen et al. 2017). For example, using the WIOD database, Caro et al. (2015) calculated the consumption-based GHG emissions of Luxembourg for the period 1995-2009, and compared them with the production-based GHG emissions[48] for the same period. The study showed that Luxembourg was a net importer of GHG emissions during this period (largely because it was highly dependent on the imports of goods and services). A similar study was conducted for Austria (for 1997-2011) but using the GTAP-based MRIO analysis (Steininger et al. 2018). According to this study, Austria was also a net importer of GHG emissions, and more than 60% of Austria's consumption-based emissions occurred outside its national borders. These studies concluded that the CBA and PBA approaches provide different insights to support climate policymaking; hence, mitigation efforts need to use both approaches in a complementary way. Similar studies exist for other countries including Canada (Dolter and Victor 2016), Italy (Ali et al. 2018), Sweden (Dawkins et al. 2019) and the UK (Wood et al. 2010).

There are, however, a limited number of studies on calculation of New Zealand's GHG emissions using a CBA approach. As observed from Table 6.1, these include one journal article (Andrew and Forgie 2008), two conference papers (Andrew et al. 2008; Chandrakumar et al. 2018) and three technical reports (Allan and Kerr 2016; Allan et al. 2015; Vickers et al. 2018); note that only the first three were peer-reviewed.

[48] Note that the production-based GHG emissions of Luxembourg were based on the guidelines proposed by the Intergovernmental Panel on Climate Change (IPCC 2006).

Table 6. 1: Studies on calculation of New Zealand's greenhouse gas emissions using a consumption-based accounting approach

Study	Type	Peer-reviewed	Accounting approach[a]	Year of analysis	Method of analysis	Scope of analysis
Andrew and Forgie (2008)	Journal article	YES	CBA and PBA	2001	Domestic emissions (MfE 2006); embodied emissions in imports were not considered	Agriculture and forestry sectors
Andrew et al. (2008)	Conference paper	YES	CBA and PBA	2001	GTAP-based MRIO analysis	Entire economy
Allan and Kerr (2016)	Technical report	NO	CBA	2006, 2012	National input-output analysis (Stats NZ 2014)	Final household consumption
Allan et al. (2015)	Technical report	NO	CBA	2006, 2012	National input-output analysis (Stats NZ 2014)	Final household consumption
Chandrakumar et al. (2018)	Conference paper	YES	CBA and PBA	2012	Eora-based MRIO analysis	Agri-food sector
Vickers et al. (2018)	Technical report	NO	CBA and PBA	2015	Domestic emissions (MfE 2017a); Embodied emissions using a US-based MRIO analysis	Entire economy

[a] CBA=consumption-based accounting, PBA= production-based accounting

As a first attempt, Andrew and Forgie (2008) estimated the consumption-based GHG emissions of New Zealand for the year 2001 and compared the results to the production-based emissions. The analysis was based on New Zealand's official National Inventory Report published by the Ministry for the Environment (MfE 2006). Nevertheless, when calculating the consumption-based emissions, the embodied emissions in imports were omitted; moreover, the scope of the study was limited to the agriculture and forestry sectors. Andrew et al. (2008) subsequently attempted to quantify the consumption-based GHG emissions of the entire New Zealand economy for the year 2001, using the GTAP-based MRIO analysis for the first time. However, the authors indicated that the primary purpose of the work was to present the methodology and available data at that time, and clearly stated that the results of the study need to be treated cautiously due to the limitations in the GTAP database.

Later, Allan and colleagues (2015; 2016) investigated whether New Zealand households are becoming greener consumers by comparing the consumption-based GHG emissions of households in 2006 and 2012. For this study, they used the national input-output tables with detailed information on household consumption in the given years. Although the study accounted for the embodied emissions in imports, the focus was only on households; the consumption activities of other final demand actors (private and public entities) were not assessed. For the first time in New Zealand, Chandrakumar et al. (2018) used the Eora-based MRIO analysis to quantify the consumption- and production-based GHG emissions (in the year 2012); however, the study was only limited to the agri-food sector. At the same time, Vickers et al. (2018) estimated the GHG emissions of New Zealand for the year 2015 using both CBA and PBA approaches. While the study covered the entire economy of the country, the embodied emissions in imports and exports were estimated based on an environmental input-output based life cycle assessment model developed for the USA.

Given the limitations of the existing work on this topic, this study undertook an MRIO analysis using the Eora database (Lenzen et al. 2012, 2013b) to investigate the policy implications of using a CBA versus PBA approach to calculate New Zealand's GHG emissions. The study also aimed to identify the sectors that account for the largest share of consumption- and production-based GHG emissions.

6.2. Methodology

6.2.1. Multi-regional input-output analysis

Multi-regional input-output (MRIO) analysis is used to examine the economic interdependency of industry sectors and countries, and can also be used to calculate the magnitude of anthropogenic environmental pressures such as GHG emissions (e.g. Franzen and Mader 2018; Malik and Lan 2016; Steininger et al. 2018), freshwater use (e.g. Lenzen et al. 2013a; Yu et al. 2010), nitrogen pollution (e.g. Oita et al. 2016), phosphorous pollution (e.g. Li et al. 2019) and land use change (e.g. Yu et al. 2016). One of the benefits of undertaking a MRIO analysis is that it captures both direct and indirect environmental pressures associated with a country's final consumption demand by tracing the whole production and supply chain of traded goods (including services) and associated materials back to the original source of primary extraction (Lenzen et al. 2013b; Malik and Lan 2016; Tukker et al. 2009; Wiedmann et al. 2011). The other benefit is that it encompasses entire global supply chains and, therefore, overcomes the system boundary issues that typically arise in Life Cycle Assessment studies (Nakamura and Nansai 2016; Suh and Huppes 2005; Suh et al. 2004).

Countries are linked through international trade in the MRIO analysis. The production coefficient matrix A is calculated by $a_{ij}^{pq}=z_{ij}^{pq}/x_j^q$, which represents the monetary input from sector i in country p to sector j in country q to produce one unit of total output of sector j in country q; x_j^q is the total output of sector j in country q. y is a final demand matrix consisting of y^{pq}, which refers to a vector of each sector's output produced in country p consumed by the final consumer in country q. x is a vector of sectoral outputs of all countries.

$$\mathbf{A}=\begin{bmatrix} A^{11} & A^{12} & \cdots & A^{1n} \\ A^{21} & A^{22} & \cdots & A^{2n} \\ \vdots & \vdots & \ddots & \vdots \\ A^{n1} & A^{n2} & \cdots & A^{nn} \end{bmatrix}; \quad \mathbf{y}=\begin{bmatrix} y^{11} & y^{12} & \cdots & y^{1n} \\ y^{21} & y^{22} & \cdots & y^{2n} \\ \vdots & \vdots & \ddots & \vdots \\ y^{n1} & y^{n2} & \cdots & y^{nn} \end{bmatrix}; \quad \mathbf{x}=\begin{bmatrix} x^{1} \\ x^{2} \\ \vdots \\ x^{n} \end{bmatrix}$$

The MRIO model can, therefore, be represented as Eq. (6. 1):

$$\mathbf{x} = \mathbf{Ax} + \mathbf{y} \qquad (6.\ 1)$$

To solve ***x***, Eq. (6. 2) is obtained:

$$\mathbf{x} = (\mathbf{I} - \mathbf{A})^{-1}\mathbf{y} = \mathbf{Ly} \qquad (6.\ 2)$$

where $L=(I - A)^{-1}$ is the Leontief inverse matrix, which captures both direct and indirect inputs to satisfy one unit of final demand in monetary values; I is the identity matrix.

To calculate the embodied GHG emissions in goods and services, the MRIO table was extended with the GHG emissions coefficients, as expressed in Eq. (6. 3):

$$\mathbf{Q} = \mathbf{qx} = \mathbf{q}(\mathbf{I} - \mathbf{A})^{-1}\mathbf{y} = \mathbf{qLy} \qquad (6.\ 3)$$

where Q is a satellite account containing GHG emissions Q_i of sector i (carbon dioxide, methane, nitrous oxide, etc.) associated with final consumption, q is a matrix of the direct GHG emissions intensity q_i of sector i (i.e. different types of GHG emissions per unit of economic output), qx is the production-side breakdown of total GHG emissions Q, and qLy is the consumption-side breakdown of Q.

6.2.2. Data sources

In this study, the Eora database was applied for the year 2012; this was the latest year available on the database at the time of the analysis[49]. The Eora database provides a time series of input-output and trade tables with matching environmental and social satellite accounts for 14,839 industry sectors from 189 countries (Lenzen et al. 2012; 2013b). For the New Zealand economy, there are 127 industry sectors and 210 commodities (goods and services). Given the sectoral classification of each country is quite different, and in order to ease interpretation of the results, the authors classified and aggregated all the industry sectors of each country into 16 key sectors (see Supplementary Material (SM) 4).

6.3. Results and discussion

6.3.1. Greenhouse gas emissions of New Zealand

According to the Eora-based MRIO analysis, New Zealand's GHG emissions using the CBA and PBA approaches (for 2012) were 61,850 $ktCO_2eq$ and 81,667 $ktCO_2eq$, respectively (see SM 4). This is equivalent to 14 tonnes (t) CO_2eq per capita and 18 tCO_2eq per capita, respectively[50]. Given the production-based GHG emissions of New Zealand were approximately 20,000 $ktCO_2eq$ higher than the consumption-based emissions, the country was a net exporter of GHG emissions in 2012.

Figure 6. 1 shows New Zealand's GHG emissions calculated using the CBA and PBA approaches at the sector level. From a consumption perspective (Figure 6. 1(a)), the dominant contributors were the other services and construction key sectors (each representing 16% of consumption-based emissions), followed by food and beverages (14%), transport and equipment (12%), and financial and trade services (11%). Consumption activities related to electricity, gas and water, electrical and machinery, textiles and wearing apparel, and agriculture each contributed between 4% and 7% of New Zealand's consumption-based GHG emissions. From a production perspective (Figure 6. 1(b)), the dominant contributor was the agriculture key sector (representing 52% of production-

[49] Note that development of MRIO databases requires a significant amount of time as it involves a long procedure, including collection of source (survey) data, imputation and balancing, allocation, assuming proportionality and homogeneity, aggregation, and multipliers (Wiedmann 2009). There is therefore always a lag in the latest year available in any MRIO database including Eora.
[50] Note that the population of New Zealand in 2012 was 4,467,743 (FAO 2018b).

based GHG emissions). The contributions of the other key sectors were small when compared with agriculture: the next largest contributing key sectors were transport and equipment (17%), electricity, gas and water (14%), and petroleum, chemical and non-metallic mineral products (4%).

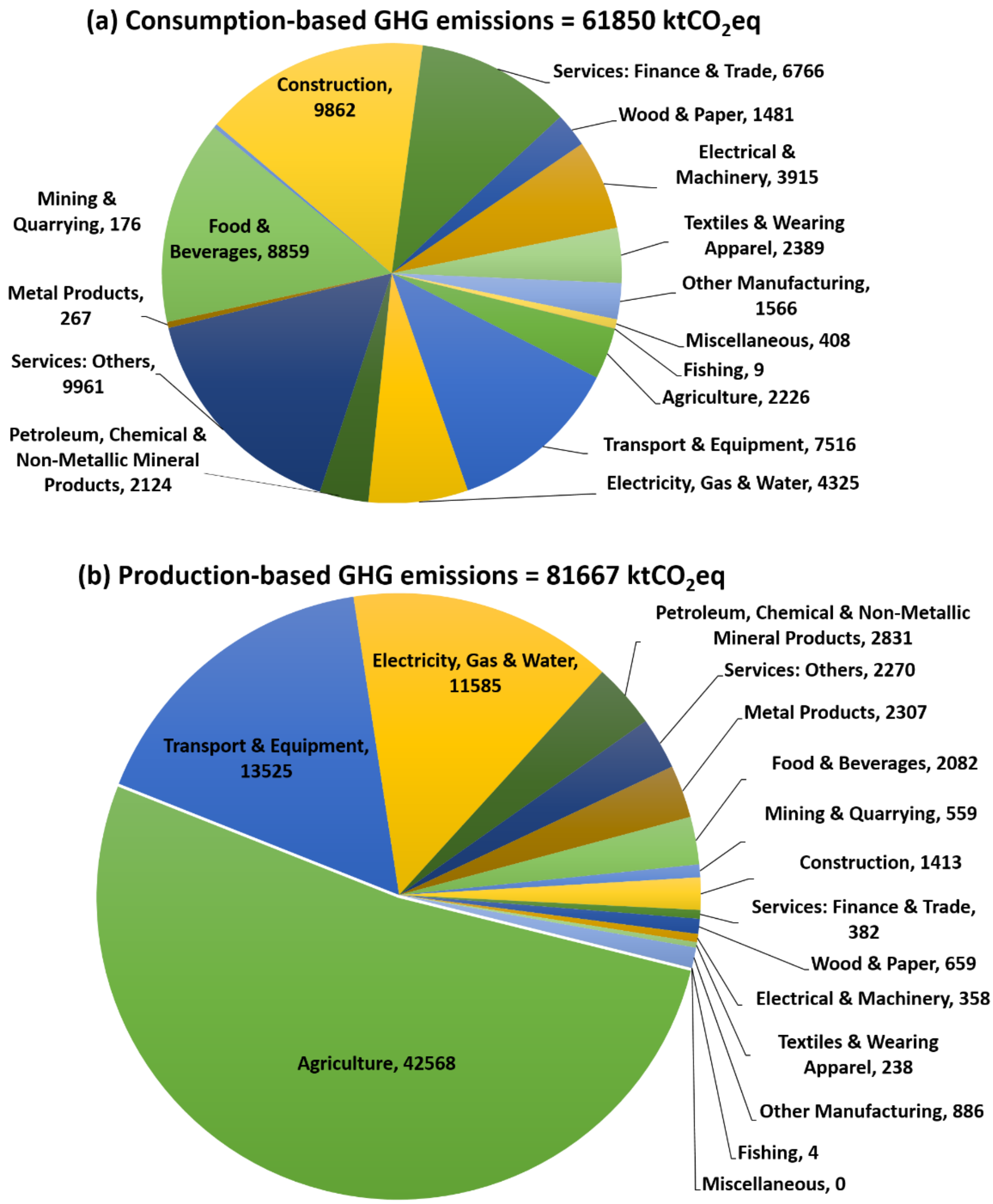

Figure 6. 1: Contribution of the 16 key sectors to New Zealand's (a) consumption- and (b) production-based GHG emissions, respectively, for the year 2012. The radius of each pie chart is representative of the total GHG emissions. Consumption- and production-based GHG emissions calculations are available in SM 4.

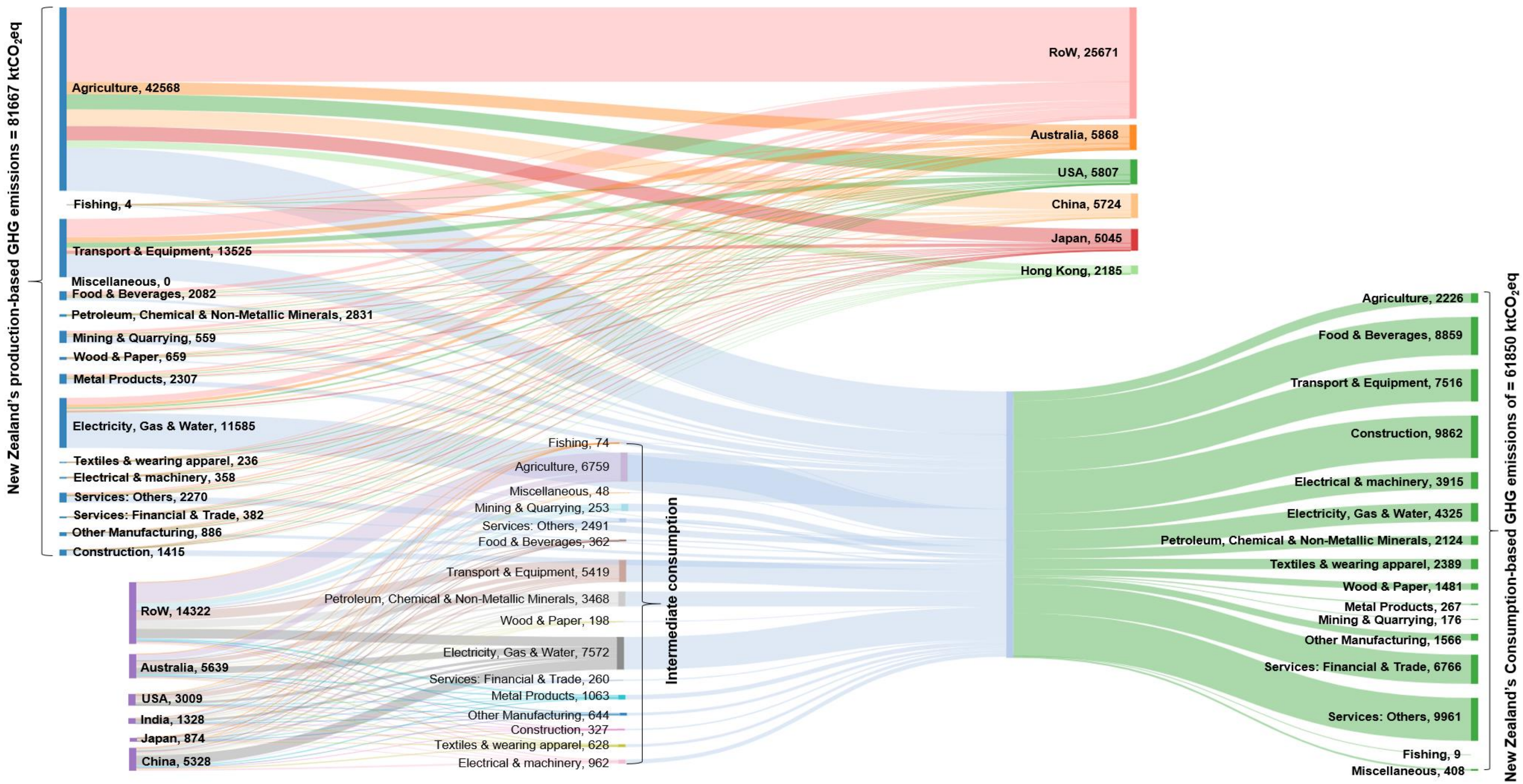

Figure 6. 2: GHG emissions embodied in New Zealand's trade in the year 2012. The texts in bold represent the exporting and importing countries, and 16 key sectors, whereas the normal texts refer to the intermediate consumption of the New Zealand economy in terms of the 16 key sectors. RoW=rest of the world

6.3.2. Embodied emissions of New Zealand's trade

Figure 6. 2 shows the embodied emissions of New Zealand's trade activities for the 16 key sectors and New Zealand's primary trade partners. Embodied emissions in imports (EEI) to New Zealand were 30,499 $ktCO_2eq$, representing 49% of New Zealand's consumption-based GHG emissions; thus, New Zealand's total consumption-based emissions comprised approximately equal contributions from activities based in New Zealand versus other countries. The rest of the world (RoW) region contributed the largest EEI (14,322 $ktCO_2eq$, representing 47% of total EEI). Australia was the second largest contributor of EEI (5,639 $ktCO_2eq$, 9%), followed by China (5,328 $ktCO_2eq$, 9%), USA (3,009 $ktCO_2eq$, 5%), India (1,328 $ktCO_2eq$, 2%) and Japan (874 $ktCO_2eq$, 1%). On the other hand, embodied emissions in exports (EEE) from New Zealand were 50,299 $ktCO_2eq$, representing 62% of New Zealand's production-based GHG emissions. The EEE to the RoW was the largest (25,671 $ktCO_2eq$ representing 51% of total EEE). The second largest EEE was to Australia (5,868 $ktCO_2eq$, 12%), followed by the USA (5,807 $ktCO_2eq$, 12%), China (5,724 $ktCO_2eq$, 11%), Japan (5,045 $ktCO_2eq$, 10%) and Hong Kong (2,185 $ktCO_2eq$, 4%).

6.3.3. Comparisons with the existing studies

Direct comparisons between the results of this study and previous work were not possible for three reasons. First, no previous study quantified the GHG emissions for the year 2012 using a CBA. Second, as discussed earlier, the scope of the existing studies was limited to a particular economic sector (e.g. agriculture and forestry, Andrew and Forgie (2008)) or category of final demand actor (e.g. households, Allan et al. (2015); Allan and Kerr (2016)). Third, although some studies have assessed the entire economy of the country (Andrew et al. 2008; Vickers et al. 2018), as noted in the Introduction, the data and assumptions used for those analyses mean that a direct comparison with the results of this study was not possible. Nevertheless, the results of this study can be compared with the work by Chandrakumar et al. (under review)[51], in which they also used the Eora-based MRIO

[51] This study is the Chapter 7 of the thesis.

analysis[52] to calculate the consumption-based GHG emissions of New Zealand, but for the year 2011 (see Figure 6. 3).

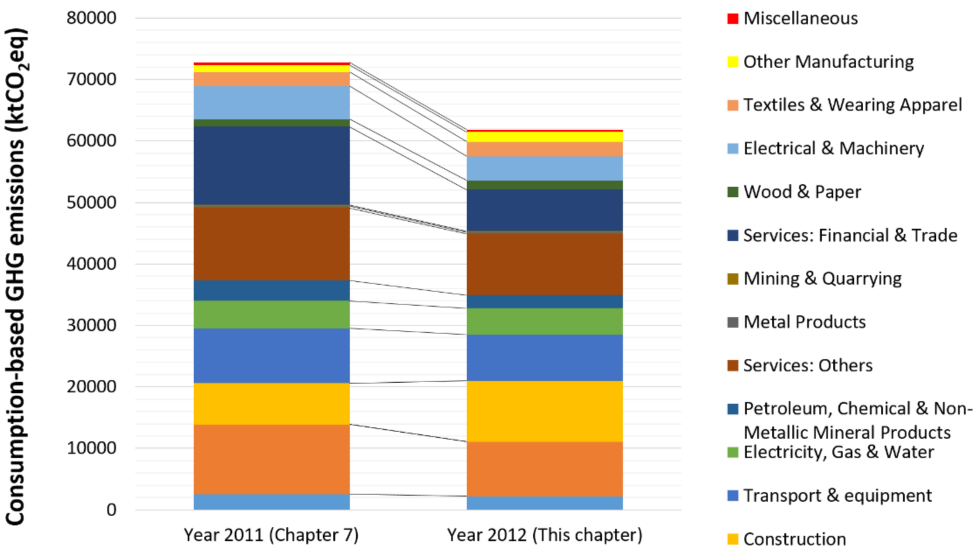

Figure 6. 3: Comparison of the results of this study to the work by Chandrakumar et al. (under review). Note that Chandrakumar et al. (under review) and this study focus on 2011 and 2012, respectively. Calculations are available in SM 4.

According to Chandrakumar et al. (under review), New Zealand's consumption-based GHG emissions in 2011 were 72,773 $ktCO_2eq$; this value is 10,923 $ktCO_2eq$ higher than the consumption-based emissions in 2012. The difference is mainly due to the differences in the GHG emissions of the financial and trade-related services, food and beverages, and transportation and equipment key sectors. As seen in Figure 6. 3, the consumption-based GHG emissions of the financial and trade services approximately halved from 2011 to 2012. Likewise, the emissions of the food and beverages in 2012 were approximately three-quarters of the emissions in 2011. While the GHG emissions of many key sectors reduced in 2012, the emissions of some sectors increased in 2012. In particular, the construction sector's emissions were approximately 50% higher than the emissions in 2011.This difference in the consumption-based GHG emissions for the construction sector can be explained by the earthquake that struck the Canterbury region in New Zealand in February 2011 (Breetzke et al. 2018; Doherty 2011; Potter et al. 2015). The earthquake-related total economic loss was estimated to be NZ$ 20 billion, including damage to buildings and

[52] Note that the global MRIO database Eora26 was used by Chandrakumar et al. (under review); this database also provides a time series of input-output and trade tables with matching environmental and social satellite accounts for 189 countries but in terms of 26 (aggregated) economic sectors (see Chapter 7 for more details).

infrastructure, loss of employment, relocations, and impacts on tourism (Doherty 2011; Potter et al. 2015). As a result, the New Zealand government initiated a number of post-disaster relocation and reconstruction activities (Francis et al. 2018) which resulted in investing more on construction, and contributed to the increase in consumption-based emissions for the construction sector in 2012.

6.3.4. Discussion of results

The magnitude of consumption-based emissions of New Zealand is comparable (on a per-capita basis) to many other developed countries, including Australia and the Netherlands (Hertwich and Peters 2009). However, a similar comparison for the production-based emissions is not meaningful because, as noted earlier, New Zealand has an unusual GHG emissions profile compared with many other developed countries (OECD 2017; Parliamentary Commissioner for the Environment 2016). Production-based GHG emissions of developed countries are generally dominated by energy production and/or energy-intensive sectors, and transport, while the contribution of agriculture is relatively very small (e.g. Barrett et al. 2013; Hertwich and Peters 2009); however, New Zealand has a large agri-food sector whose GHG emissions predominate in its GHG emissions profile.

The results indicate that the dominant contributing sectors to New Zealand's GHG emissions are quite different for the CBA and PBA approaches. Using a consumption perspective, other services, construction, and food and beverages, ranked highest in terms of GHG emissions. Using a production perspective, agriculture, transport and equipment, and electricity, gas and water, were the highest-ranking sectors. The differences in the dominant sectors suggest that national GHG mitigation measures that are informed only by production-based emission profiles are unlikely to be sufficient to mitigate climate change globally. This point can be illustrated with two examples. Firstly, New Zealand's agriculture is recognised as relatively GHG-efficient compared with many other countries (OECD 2017; Parliamentary Commissioner for the Environment 2016); costly mitigation measures to reduce this sector's emissions may make New Zealand's exported agri-food products less competitive in international markets, and they could potentially be displaced by products from other countries with less GHG-efficient agricultural practices. Indeed, such an approach may even lead to a global increase in GHG emissions given the carbon leakage effect (Afionis et al. 2017; Barrett et al. 2013; Ward et al. 2015). As a second example, a PBA approach is relevant for informing mitigation options to improve the GHG emissions profile of New Zealand electricity (with associated benefits for goods and services produced using New Zealand electricity). However, this approach does not account for the GHG emissions from Chinese electricity that are embodied in many of the goods and services imported to New Zealand. These Chinese electricity GHG emissions comprised nearly 4% of New Zealand's consumption-based emissions in 2012; in contrast, New Zealand electricity GHG emissions comprised nearly 7% of New Zealand's consumption-based emissions in the same year (see Figure 6. 2). Given that the Chinese emissions were more than half of the New Zealand emissions, this suggests that it is relevant to focus on Chinese as well as New Zealand electricity GHG emissions in New Zealand climate mitigation activities.

In terms of policy, then, if national GHG mitigation measures in countries around the world are also informed by consumption-based emission profiles, then carbon leakage is less likely to be an issue (Afionis et al. 2017; Barrett et al. 2013). Others have already suggested that international policies should primarily focus on the emissions intensity of imports and ensuring that they are produced using the best available technologies (e.g. Meinshausen et al. 2017; Peters 2008; Steininger et al. 2018). In turn, this will drive exporting countries to invest strategically in state-of-the-art manufacturing facilities, and catalyse the development of appropriate financial investment schemes and technology transfer programs (e.g. Afionis et al. 2017; Barrett et al. 2013; Dolter and Victor 2016; Wiedmann 2009). A CBA approach is also appropriate to support the development of consumer-oriented policies that can inform a shift towards low-GHG consumption patterns (e.g. Bjørn et al. 2018; Girod et al. 2013; 2014). For example, such policies could support prioritised development of sector-based voluntary environmental performance certification and labelling schemes that provide additional information to consumers about environmentally (in this context, climate change) preferable goods and services. These schemes have been shown to shift average industry practices over time towards goods and services with improved environmental performances (Girod et al. 2014; Steininger et al. 2018; Wiedmann 2009). Alternatively, mandatory environmental standards (e.g. energy efficiency standards and building codes) can be used for the same purpose (Barrett et al. 2013; Wiedmann 2009).

6.3.5. Limitations and outlook

In this study, two GHG emissions accounting (CBA and PBA) approaches were used to quantify New Zealand's GHG emissions. As described in the Introduction, both approaches have advantages and disadvantages. There is a third approach, called 'shared producer and consumer responsibility' (Lenzen et al. 2007), which shares GHG emissions mitigation responsibilities among different actors in the value chain through to the consumers according to the benefit obtained by each actor, using value-added as a proxy. Potential future research could be to apply this approach to the entire New Zealand economy, and compare the outcomes with the results of this work.

The consumption- and production-based emission results calculated in this study could be potentially used to answer the question "How far New Zealanders are away from achieving the 2°C climate target?" For example, a similar method as in Chapter 5 (i.e. Absolute Sustainability-based Life Cycle Assessment [ASLCA]) can be applied to assign a share of the 2°C global carbon budget to New Zealand using CBA and PBA approaches, and compare the consumption- and production-based emissions against the assigned carbon budget shares. This piece of work is however beyond the scope of this chapter, and has been undertaken in Chapter 7.

Furthermore, there are uncertainties associated with different MRIO databases as they are based on several assumptions and philosophies (e.g. Afionis et al. 2017; Malik et al. 2018; Owen 2017; Wiedmann et al. 2011). For example, in this study, the Eora database was used, which provides a detailed sectoral disaggregation (14839 industry sectors for 189 countries). However, some sectors (including agriculture) are still aggregated for most countries (Lenzen et al. 2013b). For example, the Swiss agriculture sector includes products from both the forestry and fishing industries. Hence, additional effort is necessary when developing climate policies to mitigate GHG emissions from these aggregated sectors. Although the GTAP database provides a relatively higher disaggregation for sectors like agriculture, its geographical coverage (140 countries) is limited when compared with Eora (189 countries) (Li et al. 2019; Stadler et al. 2018). Therefore, it would be interesting to undertake a similar MRIO analysis for New Zealand but using an alternative database, as already done for Sweden (Dawkins et al. 2019). There are additional uncertainties related to the MRIO methodology, which are extensively discussed in the literature (Karstensen et al. 2015; Owen 2017; Peters et al. 2012; Steininger et al. 2018). For a detailed discussion, see Karstensen et al. (2015) and Owen (2017).

6.4. Conclusions

The study highlighted two key insights when considering the application of CBA versus PBA to support climate mitigation activities and policymaking. Firstly, the CBA and PBA approaches provide different information about the dominant contributing sectors to New Zealand's GHG emissions. Therefore, both approaches should be used in a complementary way when developing national climate policies in order to focus mitigation efforts on the most significant activities contributing to national GHG emissions (Barrett et al. 2013; Steininger et al. 2018). Secondly, only the CBA approach enables quantification of the embodied emissions in international trade, and the study showed that a significant proportion of GHG emissions are embodied in New Zealand's trade activities (both imports and exports). An implication is that future international trade policies fostering climate action could have a significant impact on the New Zealand economy. These two insights suggest that future research should focus on how to develop and implement effective consumption-based policies and instruments given that PBA rather than CBA has been the dominant approach to date.

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Chapter 7: Assessing the climate change performance of New Zealand's economy using an absolute environmental sustainability assessment method: the influence of value and modelling choices

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Abstract

Climate change mitigation requires informing the climate change performance of countries in relation to the absolute climate thresholds, such as the critical level of radiative forcing increase or maximum level of temperature increase below pre-industrial times. This is, however, influenced by several value and modelling choices, such as the choice of absolute climate threshold, the choice of approach for estimating the global carbon budget, and the choice of approach to assigning shares of the carbon budget to economic sectors within a country. This study assessed the influence of these choices on the overall climate change performance of a country by taking New Zealand (for the year 2011) as an exemplar of a relatively complex, developed economy. The production- and consumption-based carbon footprints were calculated using the Eora-based multi-regional input-output analysis. New Zealand's production- and consumption- based footprints were similar; however, there were large differences between these two footprints for some sectors. Both the New Zealand footprints exceeded their carbon budget shares irrespective of the value and modelling choices. When the carbon budget was assigned to sectors using the economic sharing principle, a few sectors performed within their limits but most exceeded them (by up to 55 times). The largest source of uncertainty was the choice of climate threshold, followed by the choice of sharing principle, and the choice of approach for calculating carbon budget. Overall, the study stressed the significance of addressing value and modelling choices when assessing the climate change performance of a country in relation to absolute climate thresholds.

Keywords: Absolute environmental sustainability, value choice, modelling choice, carbon budget, benchmarking, climate change, multi-regional input-output analysis, life cycle assessment

7.1. Introduction

Anthropogenic greenhouse gas emissions increase the risk of transgressing absolute climate boundaries, referred to as tipping points for the climate system (Lenton et al. 2008; Levermann et al. 2012). Current indicators of climate change, including Global Warming Potential (GWP, Allen et al. 2016) and Global Temperature change Potential (GTP, Shine et al. 2005) do not directly provide information about the climate change performance of a country or sector relative to these *absolute climate boundaries* (also referred to as *climate thresholds*); hence, they can be regarded as providing insufficient information to mitigate climate change globally (Akenji et al. 2016; Bjørn et al. 2016)

Acknowledging the existence of absolute environmental boundaries, researchers have started developing so-called *absolute environmental sustainability assessment* (AESA) methods (Bjørn et al. 2015, 2018b; Chandrakumar and McLaren 2018a, b), which can be used to assess the environmental performance of an economic system in relation to a given absolute boundary (Bjørn et al. 2018a; Chandrakumar et al. 2019; Girod et al. 2013; Jørgensen et al. 2014). In these methods, relative indicators (such as those based on GWP) are expressed in the same unit as the boundary. However, there are additional value and modelling choices that need to be considered when carrying out an AESA, which are potentially important sources of uncertainty. These include selection of:

(i) A greenhouse gas (GHG) accounting method to calculate the carbon footprint of an economic system.
(ii) An absolute climate threshold at the global level.
(iii) An estimated global carbon budget for the chosen absolute climate threshold.
(iv) A sharing principle for partitioning the global carbon budget amongst different parts of an economy.

Regarding **selection of a GHG accounting method**, these methods can be either production-based (also referred to as territorial) or consumption-based

(Hertwich and Peters 2009; Lenzen et al. 2007; Wiedmann 2009). In production-based methods, emissions of the production of goods and services are associated with the producer; these methods do not account for the emissions associated with imported goods and services that may be used by the producer (Lenzen et al. 2013a; Malik and Lan 2016; Malik et al. 2016). In consumption-based methods, emissions of the production of goods and services are associated with the consumer; these methods do not account for the emissions associated with additional goods and services that are exported (Lenzen et al. 2013a; Malik and Lan 2016; Malik et al. 2016). As each of these methods addresses only a subset of activities associated with economic systems, it is argued that applying one of the two methods is not sufficient to mitigate climate change globally (Afionis et al. 2017; Barrett et al. 2013; Steininger et al. 2018). Hence, both methods should be applied in a complementary way.

Once the carbon footprint is calculated, the footprint of the chosen system needs to be benchmarked against an assigned share of the global carbon budget, which requires choosing an appropriate climate threshold. There are several climate thresholds proposed in the literature. For example, in the Planetary Boundaries (PBs) framework, Steffen et al. (2015) defined two **climate thresholds**: a global average atmospheric carbon dioxide (CO_2) concentration of 350 parts per million (ppm) CO_2 and a radiative forcing of 1 watt per square metre (Wm^{-2}). These PB thresholds are sufficient to limit the atmospheric global average temperature to below 1.5 degree Celsius (°C) above pre-industrial levels (Clift et al. 2017; Steffen et al. 2015). These thresholds are based on scientific evidence such as the intensity, frequency, and duration of extreme climate events (Steffen et al. 2015) and are, therefore, considered to be *scientific thresholds*. At the same time, many governments and companies are debating whether to set the climate threshold at 450 ppm CO_2eq, which would be sufficient to limit the atmospheric global average temperature to below 2°C above pre-industrial levels (Butz et al. 2018; Clift et al. 2017; Stoknes and Rockström 2018). Given the threshold is primarily based on political consensus, it should be considered a *political threshold* (Bjørn and Hauschild 2015; Jørgensen et al. 2014). Although the difference in the two boundaries might seem small (=0.5°C), it is sufficient to make a significant difference to the climate system, particularly in terms of increases in mean temperature in many land and ocean regions, hot extremes in several regions, and the probability of drought and precipitation deficits in some regions (IPCC 2018).

The chosen climate threshold then needs to be translated into a measurable budget called **global annual carbon budget**; different approaches already exist for this purpose. For example, Bjørn and Hauschild (2015) applied the Absolute Global Temperature change Potential (AGTP) for GHG emissions: 3.6 gigatonnes carbon dioxide equivalent ($GtCO_2eq$) for the 1 Wm^{-2} PB threshold and 6.8 $GtCO_2eq$ for the 2°C threshold. Using the method of Doka (2016) for the 2°C threshold, Chandrakumar et al. (2019) calculated a global annual carbon budget (hereafter, global carbon budget) of 29.9 $GtCO_2eq$. These two carbon budgets are constant over time; however, other methods propose values for carbon budgets based on future projections for GHG emissions. For example, Bjørn et al. (2018a) estimated a carbon budget for the 2°C climate threshold (=19 $GtCO_2eq$), based on the median global GHG emissions of the RCP 2.6 pathway. Details of different approaches to estimate global carbon budget are available in the Supplementary Material (SM) 5.

Finally, to **assign a share** of the global carbon budget to a chosen economy or a sector within the economy, there are several methods centred on different sharing principles (e.g. historical responsibility (aka grandfathering), population, economic output (or value addition), final consumption expenditure, territorial area) (Fang et al. 2015; Liu and Bakshi 2018; Ryberg et al. 2018a). Each sharing principle has its own benefits and limitations. For example, historical responsibility-based sharing method effectively rewards the larger emitters with larger shares of carbon budget, while leaving smaller emitters with smaller (or even zero) carbon budget shares (Ryberg et al. 2018a; Sandin et al. 2015). On the contrary, population-based sharing method equally shares the global carbon budget between all the individuals in the world; however, it assigns smaller shares for countries with lower population (Sandin et al. 2015). Given each sharing principle effectively rewards some countries and/or sectors, the choice of method to assign a share of the global carbon budget is generally considered normative and non-scientific (Chandrakumar et al. 2018; Ryberg et al. 2018a)

It is, therefore, evident that value choices and modelling choices need to be made when assessing the climate change performance of an economy from an absolute sustainability perspective. To that end, some studies have already assessed the influence of the choice of methods to assign a share of the global carbon budget (Chandrakumar et al. 2018;

Ryberg et al. 2018a). However, the additional influence of the choice of GHG accounting methods, choice of climate thresholds, and approaches for calculating the global carbon budget, has not been assessed together in a single study.

This study, therefore, for the first time, assessed the influence of all of the above-mentioned value and modelling choices on the overall climate change performance of a country by taking New Zealand in the reference year 2011[53] as an example. New Zealand is a relatively complex, developed economy (OECD 2017), which is currently debating the implementation of a Zero Carbon Bill, with an aim of bringing net GHG emissions to zero by 2050 (Shaw 2019). However, achieving this ambitious goal requires identifying sector-based priorities for the development of innovative climate policies. In this context, using both production and consumption accounting methods, the carbon footprint of the New Zealand economy was calculated and benchmarked against a set of 12 shares of the global carbon budgets developed using the above-mentioned value and modelling choices.

7.2. Methods

This section provides an overview of the assessment methodology. It then describes the procedure for calculating production- and consumption-based carbon footprints, and the selected climate thresholds, global carbon budgets, and methods for assigning shares of the global carbon budget to the economic sectors of New Zealand.

7.2.1. Overall assessment method

The assigned share of the global carbon budget (SoGCB) for each economic sector was calculated using Eq. 7. 1.

$$\mathbf{SoGCB_{X,SP}=GCB\times aS_{SP}} \qquad (7.\ 1)$$

Where $SoGCB_{X,SP}$ is the SoGCB assigned to sector X, using the selected sharing principle (SP), GCB is the global carbon budget related to a given climate threshold, and aS_{SP} [%] is

[53] The latest year available in the Eora database is the year 2012. Given the climate impact of New Zealand's economy has already been calculated for the year 2012 in Chapter 6, the year 2011 was chosen in this study.

the percentage share assigned to sector X, using the chosen sharing principle (SP). The absolute climate change performance of sector X is given by Eq. 7. 2.

$$\mathbf{occSoGCB_{X,SP}} = \frac{CF_X}{SoGCB_{X,SP}} \times 100\% \qquad (7.\ 2)$$

Where $occSoGCB_{X,SP}$ is the SoGCB occupied by sector X using the chosen sharing principle. CF_X is the carbon footprint of sector X. $SoGCB_{X,SP}$ is the SoGCB assigned to sector X. If $occSoGCB_{X,SP}$ is equal to or less than 100%, sector X can be considered as sustainable in absolute terms for the climate change impact category.

7.2.2. Calculation of carbon footprints at sector level

The global MRIO database Eora26 was used to calculate the production- and consumption-based carbon footprints of New Zealand (Lan et al. 2016; Lenzen et al. 2013b). Eora26 provides a time series of input-output and trade tables with matching environmental (including climate change) and social satellite accounts for 26 economic sectors of 189 countries. For the methodology of the Eora-based MRIO analysis, see Lan et al. (2016) and Malik and Lan (2016). Some of the 26 sectors are similar in terms of their activities and so they were combined together. For example, transport, and transport equipment were combined and named 'transport and equipment'. Likewise, service-providing sectors were aggregated as (i) 'financial and trade-related services'; and (ii) 'other services'. Finally, the 'others' and 'private households' sectors were aggregated and named 'miscellaneous'. A detailed description is available in Supplementary Material (SM) 5.

7.2.3. Selection of climate thresholds and global carbon budget

Two climate thresholds were selected for this study: (i) a scientific threshold, a radiative forcing of 1 Wm^{-2} as defined in the PBs; and (ii) a political threshold, the 2°C threshold. To calculate the global carbon budget for each of the selected climate thresholds, the method proposed by Bjørn and Hauschild (2015) was used for the scientific and political thresholds. The methods proposed by Bjørn et al. (2018a) and Chandrakumar et al. (2019) (based on Doka (2016)) were used for the political threshold only. Details of the three calculation methods are available in SM5 (see Table S5. 1).

7.2.4. Assigning a share of the global carbon budget to each sector

Two commonly used sharing principles were chosen to assign a share of the global carbon budget to different New Zealand sectors: grandfathering and economic value. For the former, a share was calculated based on the relative contribution of each sector to the global GHG emissions that occurred in a chosen reference year (Bjørn et al. 2016; Ryberg et al. 2018a). For the latter, the economic indicators gross value added (GVA) and final consumption expenditure (FCE) were applied for production- and consumption-based methods respectively (Chandrakumar et al. 2018), indicating that economic value can be considered a proxy for societal value creation (Clift and Wright 2000; Ryberg et al. 2018a).

Table 7. 1: Overview of sharing principles for calculating the share of the operating space assigned to each economic sector

GHG Accounting method	Sharing principle	Equation
Production	Grandfathering	$aS_{SP} = \frac{PBF_X}{CF_{World}}$ Where aS_{SP} is the share of the operating space assigned to the studied sector X of a country. PBF_X is the production-based carbon footprint of sector X, whereas CF_{World} is the global GHG emissions.
	Gross value added (GVA)	$aS_{SP} = \frac{GVA_X}{GVA_{World}}$ Where GVA_X is gross value added by sector X, and GVA_{World} is the total gross value added in the world.
Consumption	Grandfathering	$aS_{SP} = \frac{CBF_X}{CF_{World}}$ Where CBF_X is the consumption-based carbon footprint of sector X, whereas CF_{World} is the global GHG emissions.
	Final consumption expenditure (FCE)	$aS_{SP} = \frac{FCE_X}{FCE_{World}}$ Where FCE_X is the money spent by consumers on sector X, and FCE_{World} is the total final expenditure of the world.

Variants of these methods have already been applied in other studies (e.g. Bjørn et al. 2016; Brejnrod et al. 2017; Chandrakumar et al. 2018; Ryberg et al. 2018a). The GVA method is used when assigning a share of the global carbon budget for the production-based carbon footprint of a system based on the relative contribution to the gross value-added globally (Bjørn et al. 2016; Chandrakumar et al. 2018). The method associates responsibility for GHG emissions from production activities with the financial benefits obtained by different actors in the supply chain (Andrew and Forgie 2008; Chandrakumar et al. 2019). On the other hand, the FCE method is appropriate for assigning a share of the global carbon budget for a system's consumption-based carbon footprint, reflecting consumer preferences for different products and services that are driving the increase in GHG emissions globally (Ryberg et al. 2018a). Table 7. 1 summarises these methods.

Table 7. 2: Details of the scenarios

GHG accounting method	**Scenarios**[a]	**Value choices**		
		Threshold	**Global carbon budget**[b] **($GtCO_2eq$)**	**Sharing principle**[c]
Production	S1(2DC,Cha,GF)	2°C	29.9, Chandrakumar et al. (2019)	Grandfathering
	S2(2DC,Bjø,GF)	2°C	19.0, Bjørn et al. (2018a)	Grandfathering
	S3(1RF,BH,GF)	1 Wm^{-2}	3.6, Bjørn and Hauschild (2015)	Grandfathering
	S4(2DC,Cha,GVA)	2°C	29.9, Chandrakumar et al. (2019)	Gross value-added
	S5(2DC,Bjø,GVA)	2°C	19.0, Bjørn et al. (2018a)	Gross value-added
	S6(1RF,BH,GVA)	1 Wm^{-2}	3.6, Bjørn and Hauschild (2015)	Gross value-added
Consumption	S7(2DC,Cha,GF)	2°C	29.9, Chandrakumar et al. (2019)	Grandfathering
	S8(2DC,Bjø,GF)	2°C	19.0, Bjørn et al. (2018a)	Grandfathering
	S9(1RF,BH,GF)	1 Wm^{-2}	3.6, Bjørn and Hauschild (2015)	Grandfathering
	S10(2DC,Cha,FCE)	2°C	29.9, Chandrakumar et al. (2019)	Final consumption expenditure
	S11(2DC,Bjø,FCE)	2°C	19.0, Bjørn et al. (2018a)	Final consumption expenditure
	S12(1RF,BH,FCE)	1 Wm^{-2}	3.6, Bjørn and Hauschild (2015)	Final consumption expenditure

[a]2DC=2°C climate threshold; 1RF=1 Wm^{-2} radiative forcing climate threshold. Bjø= Bjørn et al. (2018a); BH=Bjørn and Hauschild (2015); Cha= Chandrakumar et al. (2019) An economic indicator can be either gross value-added (GVA) or final consumption expenditure (FCE).

[b]Carbon budget calculation methods: (i) using the method proposed by Doka (2016), Chandrakumar et al. (2019) calculated a 2°C annual global carbon budget by dividing the radiative forcing value of 2.6 Wm^{-2} (equivalent to the 2°C climate threshold) by the absolute GWP metrics for different GHG emissions; (ii) Bjørn and Hauschild (2015) estimated a 2°C annual global carbon budget by applying the absolute global temperature change potential metrics for different GHG emissions; and (iii) Bjørn et al. (2018a) estimated 2°C annual global carbon budget based on the GHG emissions projections developed under the RCP 2.6 scenario, based on the work by van Vuuren et al. (2011).

[c]GVA and FCE were used for benchmarking production- and consumption-based carbon footprints, respectively.

7.2.5. Scenarios

A set of 12 scenarios was developed by combining the two GHG accounting methods (production and consumption) with different value and modelling choices: two climate thresholds, three global carbon budgets, and two different sharing principles. Details of the scenarios are available in Table 7. 2.

7.3. Results and discussion

7.3.1. New Zealand's carbon footprints

The total production-and consumption-based footprints of New Zealand in 2011 were calculated as 78,600 kilotonnes (kt) CO_2eq and 72,800 ktCO_2eq, respectively (calculations are available in SM 5). The production- and consumption-based footprints were similar; however, the country was a net carbon exporter in the given year because the production-based footprint was 5,800 ktCO_2eq higher than the consumption-based footprint.

Figure 7. 1 shows New Zealand's production- and consumption-based footprints at sector level. The figure shows that nearly half of New Zealand's production-based GHG emissions were from agriculture; the contributions of other sectors were small when compared to agriculture, comprising transport and equipment (13,600 ktCO_2eq, 17%), electricity, gas and water (10,700 ktCO_2eq, 14%), petroleum, chemical and non-metallic mineral products (3,300 ktCO_2eq, 4%), and services-others (3,300 ktCO_2eq, 4%). These results clearly demonstrate that New Zealand has an unusual climate change profile compared with most other developed countries (Andrew and Forgie 2008; Bjørn et al. 2018a; Caro et al. 2015; Franzen and Mader 2018). Generally, energy production and/or energy-intensive sectors, and transport dominate the production-based footprint of developed countries, while agriculture makes a relatively small contribution (Barrett et al. 2013; Hertwich and Peters 2009).

On the other hand, from a consumption perspective, as seen in Figure 7. 1(b), financial and trade-related services were the largest contributor (12,700 ktCO_2eq), representing 17% of New Zealand's consumption-based footprint, followed by other services (11,800 ktCO_2eq, 16%), food and beverage (11,300 ktCO_2eq, 16%), transport and equipment (8,900 ktCO_2eq, 12%), and construction (6,700 ktCO_2eq, 9%). These five sectors together contributed approximately 70% of New Zealand's consumption-based footprint. Furthermore, consumption activities associated with electrical and machinery, electricity, gas and water, and petroleum, chemical and non-metallic mineral products each contributed between 4% and 8% of the consumption-based footprint.

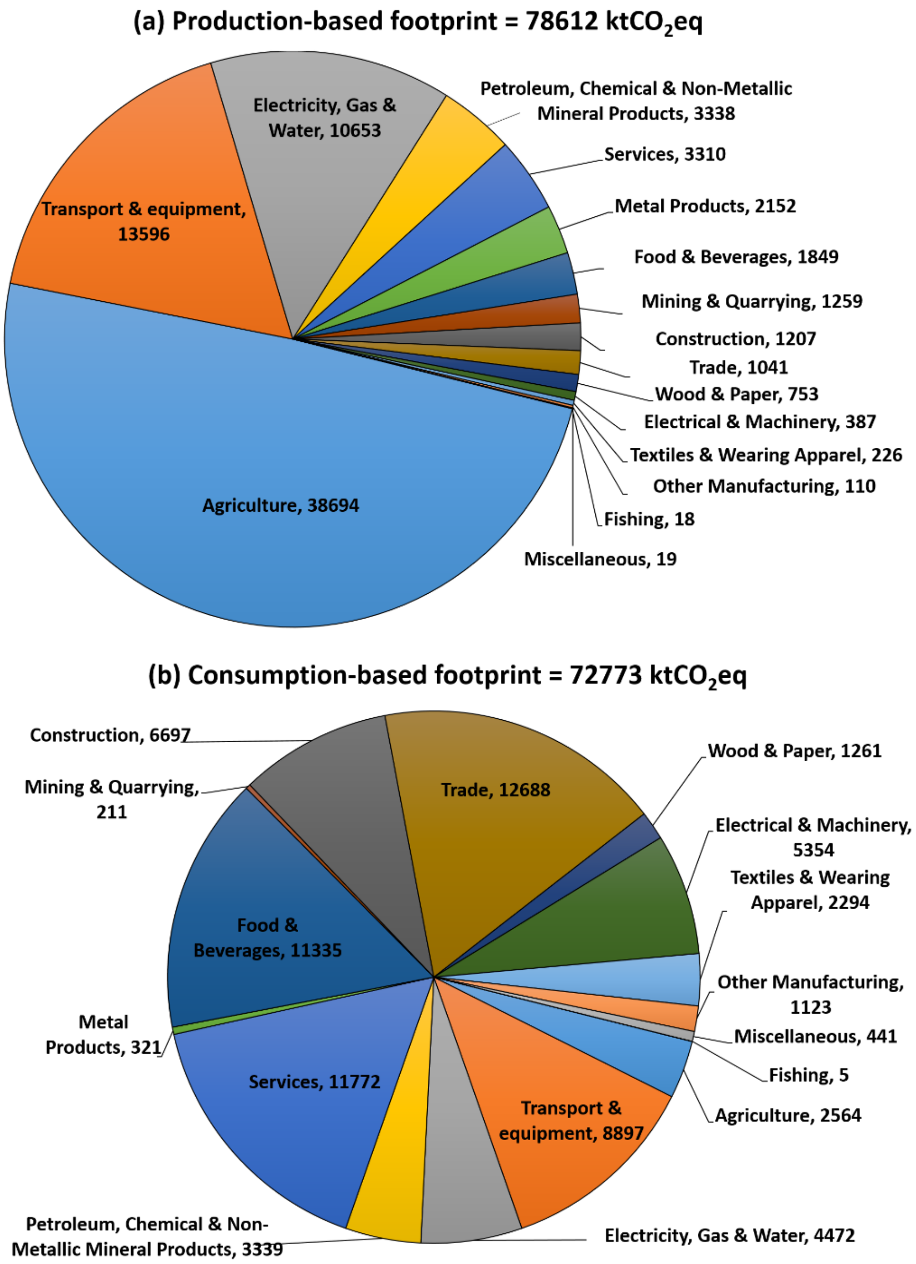

Figure 7. 1: Contribution of 16 aggregated sectors to New Zealand's production-based footprint (a) and consumption-based footprint (b), respectively, for the year 2011. The radius of each pie chart is representative of the total GHG emissions.

Although New Zealand's production-based footprint results were found to be unusual for a developed country, the consumption-based footprint results were comparable (in terms of percentage contributions) to some other developed countries, for example Australia and the Netherlands (Hertwich and Peters 2009).

7.3.2. Comparisons with previous work

Although no study to date has assessed the climate change performance of the entire economy of a country (at sector level) from an absolute sustainability perspective, two studies have estimated New Zealand's carbon footprint (MfE 2013; Vickers et al. 2018). According to the Ministry for the Environment (MfE, 2013), New Zealand's production-based footprint was 72,800 $ktCO_2eq$ in 2011, which is slightly lower than the value reported in this study (=78,600 $ktCO_2eq$). This is because the MRIO analysis accounts for the upstream GHG emissions associated with the production activities of a chosen economy (Caro et al. 2015; Wiedmann et al. 2011). The other study was for the year 2015 (Vickers et al. 2018); it estimated that the consumption-based footprint of New Zealand was 59,900 $ktCO_2eq$ in 2015, which is approximately 12% less than the value calculated in this study. However, direct comparisons between these results were not possible because the year of analysis was different and the methodologies (and assumptions) applied were not the same. In particular, Vickers et al. (2018) estimated the GHG emissions associated with imports and exports based on an environmental input-output based LCA model developed for the United States and assumed that demand for construction materials was met locally. Therefore, the differences in the carbon footprint results can be (at least partially) explained by the variations in the methodologies and assumptions used in those studies.

7.3.3. New Zealand's carbon footprints in absolute terms

Table 7. 3 presents the influence of the value and modelling choices in terms of occupied shares of the global carbon budget for all the economic sectors. As seen in Table 7. 3, New Zealand as a whole exceeded its shares of the global carbon budget in all 12 scenarios. The exceedance values varied by a factor of 1.3 to 15 across the production- and consumption-based footprints.

The sector-level analysis provided four key insights. Firstly, all the economic sectors exceeded their carbon budget shares in the grandfathering-based scenarios. According to Table 7. 3, the exceedances were the same for all the sectors, irrespective of the selection of the production- or consumption-based accounting method (180% for S1-2DC,Cha,GF and S7-2DC,Cha,GF; 283% for S2-2DC,Bjø,GF and S8-2DC,Bjø,GF; and 1,500% for S3-1RF,BH,GF and S9-1RF,BH,GF). This is because when using the grandfathering sharing principle, the share of the global carbon budget of a sector is calculated based on the sector's relative contribution to the global GHG emissions in a chosen year. Subsequently, the sector's occupied share of the global carbon budget is calculated by dividing its GHG emissions by its share of the global carbon budget; as this approach is followed for all sectors, they all have the same occupied share of the global carbon budget in the given year.

Secondly, although all 16 sectors exceeded their carbon budget shares in the grandfathering-based scenarios, some sectors were within their carbon budget shares for economic indicators-based scenarios. As seen in Table 7. 3, ten sectors were within their shares in the production-based scenarios, while three sectors were within their shares in the consumption-based scenarios. However, it should be noted that the exceedances in the economic indicators-based scenarios varied significantly; they were in the range of factor 1.4 to 55, and 1.1 to 42, for the production- and consumption-based scenarios, respectively. These significant variations are due to the widely different GHG emissions and economic contributions (and expenditures) of these sectors. For example, in 2011, the production-based footprint of the agriculture sector was 38,700 $ktCO_2eq$, which was approximately 37 times the footprint of the financial and trade services sector. On the other hand, the economic contribution of the agriculture sector was equivalent to US$14 billion, which was approximately a quarter of the economic contribution of the financial and trade services sector (see Table 7. 3). As a result, the services sector was effectively assigned a four times larger carbon budget share compared with the agriculture sector, and consequently, the agriculture sector was unsustainable for all the economic indicators-based scenarios.

Thirdly, the largest exceedances were observed for agriculture in both the production- and consumption-based scenarios (factors 55 and 42 respectively) followed by electricity, gas and water (factors 53 and 40 respectively). Transport and equipment (factor 29) showed the third largest exceedance from a production perspective, whereas food and beverages (factor 22) was the third from a consumption perspective. This is also explained by the widely different GHG emissions and economic contributions (and expenditures) of those economic sectors.

Fourthly, for all 16 sectors, the largest exceedances were observed in scenarios based on the 1 Wm^{-2} PB threshold: S6-1RF,BH,GVA or S3-1RF,BH,GF (from a production perspective), and S12-1RF,BH,FCE or S9-1RF,BH,GF (from a consumption perspective). The smallest exceedances, for most of the sectors, were observed in scenarios based on the 2°C global carbon budget of Chandrakumar et al. (2019): S1-2DC,Cha,GF (from a production perspective) or S7-2DC,Cha,GF (from a consumption perspective). The large differences between the largest and smallest exceedance values observed for each sector can be explained in terms of the choice of climate thresholds for calculating the global carbon budget: the former four scenarios were based on the 1 Wm^{-2} PB threshold (scientific) and the latter two were based on the 2°C threshold (political). As presented in Table 7. 2, the global carbon budgets associated with the 2°C threshold are at least five times greater than the carbon budgets associated with the 1 Wm^{-2} PB threshold. It is also noticeable that, although some scenarios were based on the same climate threshold and sharing principle, the difference in exceedance values was still considerable. For example, the agriculture sector's exceedance in S2-2DC,Bjø,GF (scenario based on the 2°C global carbon budget of Bjørn et al. (2018a) and the grandfathering principle) was 57% higher than the exceedance in S1-2DC,Cha,GF (scenario based 2°C global carbon budget of Chandrakumar et al. (2019) and the grandfathering principle) even though both scenarios were based on the 2°C threshold and the grandfathering principle. This is because the modelling approaches used for calculating the global carbon budgets were quite different (Bjørn et al. 2018a; Chandrakumar et al. 2019).

Overall, the largest source of uncertainty was the choice of climate threshold; scenario results where only this parameter was changed varied by a factor of 5.3 to 8.3. This was followed by the choice of sharing principle (varying by a factor of 0.02 to 3.7), and then the choice of the calculation method for the global carbon budget (a factor of 1.6) (see Table ES5.2.2 of SM 5 for calculations).

7.3.4. Research implications- carbon footprint results

The carbon footprint results showed that the economic sectors making the largest contributions to New Zealand's production- and consumption-based footprints were different. This implies that climate change mitigation efforts informed by just one of the two perspectives are likely to be insufficient to mitigate climate change globally. Efforts informed by just the production-based footprints will omit consideration of imported goods and services, whilst those informed by just the consumption-based footprints will omit consideration of domestic economic activities where the goods and services are exported. Therefore, when developing climate policies, both (production and consumption) perspectives should be applied in a complementary way (Afionis et al. 2017; Barrett et al. 2013; Steininger et al. 2018).

From a production perspective, it may seem obvious that the priority sectors should be those with the largest carbon footprints because they make the largest contribution to New Zealand's production-based footprint. However, it may also be argued that the carbon efficiency (i.e. the carbon footprint per unit of gross value added) of each of those sectors needs to be considered when selecting priority sectors for climate change mitigation. The rationale is that a particular economic sector in a country may already be relatively carbon-efficient compared with that same sector in other countries; hence, a further reduction in GHG emissions from this sector (using existing technologies and management practices) may economically disadvantage it compared with other countries. For example, New Zealand's agriculture is recognized as relatively carbon-efficient compared with many other countries (OECD 2017; Reisinger et al. 2017); further costly mitigation efforts could shift agricultural production to other countries with less carbon-efficient practices, which may even lead to a global increase in GHG emissions. This effect is called as 'carbon leakage' in the climate policy literature (Arroyo-Currás et al. 2015; Franzen and Mader 2018). As well as the size of the carbon footprint and carbon efficiency, other aspects to be considered in prioritising economic sectors for climate change mitigation activities include ease and cost of implementation of mitigation technologies and management practices, the direct consequences of mitigation activities, and stakeholder attitudes (Bjørn et al. 2018a; Ganzenmüller et al. 2019).

From a consumption perspective, likewise, it may be argued that the priority sectors should be those with the largest carbon footprints but that the carbon intensity (i.e. the carbon footprints per unit of final consumption expenditure) of the different consumption activities should also be considered. The rationale here is that New Zealand's consumption-based footprint would reduce if New Zealanders adopted low-carbon consumption patterns i.e. spending discretionary income on goods and services with low carbon intensities. Other aspects to be considered in prioritising sectors for climate change mitigation activities include consumer preferences (e.g. comfort, cleanliness, and convenience), initial capital expenditure requirements, and the degree of social acceptance of alternative consumption (Barrett et al. 2013; Bjørn et al. 2018a; Girod et al. 2014).

7.3.5. Research implications- absolute sustainability results

For the absolute sustainability results, the ranking of top three economic sectors by exceedance value, from a **production perspective**, was the same as for the carbon footprint ranking. From a **consumption perspective**, only the food and beverages sector ranked in the top three economic sectors for both the exceedance value and the carbon footprint. However, the absolute sustainability results are an inappropriate basis for prioritising different sectors from a national policymaking perspective, because a sector with high exceedance value may nevertheless make an insignificant contribution to a country's overall carbon footprint if it is a very small sector. For New Zealand, examples of these sectors are the other manufacturing, and textiles and wearing apparel, sectors from a production perspective, and the mining and quarrying, and metal products, sectors from a consumption perspective.

The absolute sustainability results, rather than providing a basis for prioritising different sectors, instead provide a systematic methodological basis for setting GHG emissions reduction targets across different economic sectors (or indeed at product level, Chandrakumar et al. 2019), and this has potential to catalyse innovation and support investment in low-emissions activities and technologies (Barrett et al. 2013; Bjørn et al. 2018a; Chandrakumar et al. 2019; Girod et al. 2014).

As demonstrated in this study, however, the current methods used for absolute sustainability assessment involve a large amount of uncertainty, particularly related to different value and modelling choices. Therefore, when developing climate policies, policymakers should be aware of, and take into consideration, these uncertainties. There is also a need to develop an international consensus on the preferred approach.

7.3.6. Research limitations

In this work, commonly used climate thresholds, global carbon budgets, and sharing principles were tested. There are other alternatives for climate thresholds, global carbon budgets and sharing principles (Dao et al. 2018; Fang et al. 2015; Ryberg et al. 2018b), whose influence on the final results could be investigated in further analyses. Also, there are additional uncertainties associated with the MRIO methodology, as well as the choice of the MRIO database that are extensively addressed in the literature (Karstensen et al. 2015; Peters et al. 2012; Steininger et al. 2018). For a detailed discussion, see Karstensen et al. (2015) and Peters et al. (2012). When calculating New Zealand's production- and consumption-based carbon footprints, this study applied the 100-year time horizon global warming potential (GWP_{100}). This metric, however, does not address the arguably greater significance of short-lived GHG emissions, such as methane and hydrofluorocarbons (Owsianiak et al. 2018; Reisinger et al. 2013). The choice of GHG metric and time horizon are particularly relevant for agriculture countries like New Zealand whose production-based carbon footprints are largely dominated by the biological emissions (including methane and nitrous oxide) from agriculture (predominantly, dairy and cattle farming) (Reisinger et al. 2013; 2017), for example, 90% of New Zealand agriculture sector's GHG emissions are biological emissions (MfE 2017; OECD 2017). Hence, a potential future research could be to extend this study by including the choice of GHG metric and time horizon as additional value choice.

Table 7. 3: Overview of the carbon footprints and economic contributions of the New Zealand economy, and the occupied share of the global carbon budget (SoGCB) by each sector for six scenarios (in %), from both production and consumption perspectives. If $occSoGCB_{X,SP}$ is greater than 100%, sector X is unsustainable in absolute terms for the climate change impact category. The values in the table are rounded off to the first decimal point. See SI for the actual values. PBF=production-based carbon footprint; CBF=consumption-based carbon footprint; GVA=gross value added; FCE=final consumption expenditure; GF=grandfathering; EI=economic indicator; $occSoGCB_{X,SP}$=occupied share of the global carbon budget by a sector for a chosen sharing principle (SP). Description of the scenarios: S1/S7-2DC,Cha,GF =2°C, 29.9 $GtCO_2eq$, GF; S2/S8-2DC,Bjø,GF =2°C, 19.0 $GtCO_2eq$, GF; S3/S9-1RF,BH,GF =1 Wm^{-2}, 3.6 $GtCO_2eq$, GF; S4/S10-2DC,Cha,GVA or FCE =2°C, 29.9 $GtCO_2eq$, GVA or FCE; S5/S11-2DC,Bjø,GVA =2°C, 19.0 $GtCO_2eq$, GVA or FCE; and S6/S12-1RF,BH,GVA or FCE =1 Wm^{-2}, 3.6 $GtCO_2eq$, GVA or FCE. Colors indicate combinations where the occupied SoGCB is less than 100% (green), >100% (yellow), >1000% (orange) and >4000% (red).

Sector	Production-based accounting method									Consumption-based accounting method									
	PBF ($ktCO_2eq$)	GVA (million US$)	GHG efficiency (gCO_2eq/US$)	$occSoGCB_{X,SP}$ (%)						CBF ($ktCO_2eq$)	FCE (million US$)	GHG intensity (gCO_2eq/US$)	$occSoGCB_{X,SP}$ (%)						
				S1-2DC,Cha,GF	S2-2DC,Bjø,GF	S3-1RF,BH,GF	S4-2DC,Cha,GVA	S5-2DC,Bjø,GVA	S6-1RF,BH,GVA				S7-2DC,Cha,GF	S8-2DC,Bjø,GF	S9-1RF,BH,GF	S10-2DC,Cha,FCE	S11-2DC,Bjø,FCE	S12-1RF,BH,FCE
Agriculture	38,694	14,295,000	2707	180	283	1,494	665	1,047	5,524	2,564	1,365,343	1,878	180	283	1,494	509	800	4,225
Transport & equipment	13,596	9,498,600	1431	180	283	1,494	352	553	2,921	8,897	12,614,298	705	180	283	1,494	191	301	1,587
Electricity, Gas & Water	10,653	4,086,100	2607	180	283	1,494	641	1008	5,320	4,472	2,506,447	1,784	180	283	1,494	483	761	4,014
Petroleum, Chemical & Non-Metallic Mineral Products	3,338	3,581,100	932	180	283	1,494	229	360	1,902	3,339	4,473,072	747	180	283	1,494	202	318	1,680
Services: Others	3,310	30,492,160	109	180	283	1,494	27	42	222	11,772	44,636,098	264	180	283	1,494	71	112	593
Metal Products	2,152	2,133,900	1008	180	283	1,494	248	390	2,058	321	436,961	734	180	283	1,494	199	313	1,651
Food & Beverages	1,849	5,152,000	359	180	283	1,494	88	139	732	11,335	11,406,966	994	180	283	1,494	269	424	2,236
Mining & Quarrying	1,259	1,837,800	685	180	283	1,494	168	265	1,398	211	262,621	804	180	283	1,494	218	343	1,808
Construction	1,207	6,949,500	174	180	283	1,494	43	67	354	6,697	16,944,539	395	180	283	1,494	107	168	889
Services: Financial & Trade	1,041	53,917,177	19	180	283	1,494	5	7	39	12,688	42,177,924	301	180	283	1,494	81	128	677
Wood & Paper	753	4,563,500	165	180	283	1,494	41	64	337	1,261	2,017,834	625	180	283	1,494	169	266	1,406
Electrical & Machinery	387	2,172,800	178	180	283	1,494	44	69	363	5,354	10,620,564	504	180	283	1,494	137	215	1,134
Textiles & Wearing Apparel	226	1,438,500	157	180	283	1,494	39	61	321	2,294	3,189,208	719	180	283	1,494	195	307	1,619
Other Manufacturing	110	735,980	149	180	283	1,494	37	58	305	1,123	2,085,443	539	180	283	1,494	146	230	1,212
Miscellaneous	19	1,177,087	16	180	283	1,494	4	6	33	441	1,749,190	252	180	283	1,494	62	97	514
Fishing	18	530,570	34	180	283	1,494	8	13	69	5	7,502	626	180	283	1,494	170	267	1,409
Total-New Zealand	78,612	14,2561,774	551	180	283	1,494	135	213	1,125	72,773	156,494,011	465	180	283	1,494	126	198	1,046

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Chapter 8: General conclusions and future work

The aim of the research was to investigate the development of an absolute environmental sustainability assessment (AESA) framework called 'Absolute sustainability-based Life Cycle Assessment' (ASLCA) based on the environmental indicators and absolute environmental boundaries proposed in three popular frameworks: Planetary Boundaries (PBs), Sustainable Development Goals (SDGs) and Life Cycle Assessment (LCA). In this context, the study specifically addressed four research objectives (see Section 1.6 of Chapter 1). The key findings associated with each objective are discussed in the following subsections.

8.1. Key findings

8.1.1. Investigation of interlinkages between Planetary Boundaries, Sustainable Development Goals and Life Cycle Assessment (Objective 1)

For the first time, the interlinkages between PBs, SDGs and LCA were investigated by mapping the environmental indicators listed in those three frameworks on to a Driver-Pressure-State-Impact-Response (DPSIR) causal network (Chapters 2 and 3). It was found that there are substantial overlaps between the three frameworks, particularly in terms of the environmental impacts that they address. Although the number of environmental impacts addressed in each of the three frameworks varies, all three frameworks evaluate six common environmental impacts (water scarcity, climate-related natural disasters, terrestrial and aquatic eutrophication, land-system change, ambient air pollution and other toxic effects, and terrestrial and ocean acidification). There is also some overlap in terms of the location of the impact assessment of the above-mentioned environmental impacts; all three frameworks address those impacts at the pressure or state points of the causal network.

Furthermore, the research showed that some environmental impacts, in particular, loss of individuals from threatened species due to direct (physical) human activities, and misplaced wastes, are only addressed in the SDGs. It was also found that the SDGs address environmental risks that are not addressed in LCA, given that the LCA is not a risk assessment tool. Therefore, to represent the environmental risks, an additional area of concern called "environmental risks" was introduced (in Chapter 3).

8.1.2. Identification of key environmental impact categories, environmental indicators and absolute environmental boundaries for development of ASLCA (Objective 2)

In order to achieve Objective 2, the central nodes in the DPSIR causal network (developed in Chapter 3) were initially identified based on the number of incoming and/or outgoing arcs, given the central nodes are most likely to represent a large number of environmental impacts when compared with other node types (such as root and end-of-chain nodes). Following that, the nodes representing distinct environmental impacts with assessment method(s) already developed (particularly in LCA) were identified in Chapter 3. This resulted in a set of 12 central nodes representing a comprehensive range of environmental impacts: water scarcity, water quality degradation, climate-related natural hazards, terrestrial and aquatic eutrophication, land-system change, ambient air pollution and other toxic effects, terrestrial and ocean acidification, soil degradation and loss of soil, ozone depletion, loss of individuals from threatened species due to direct (physical) human activities, misplaced wastes, and abiotic resource depletion/scarcity. These environmental impacts were subsequently considered as the key environmental impacts that were appropriate for development of the ASLCA framework.

Regarding the absolute environmental boundaries, it was found that the PBs and SDGs provide some absolute environmental boundaries for these key environmental impacts. However, no absolute environmental boundaries were available for water quality degradation, ambient air pollution and other toxic effects, soil degradation and loss of soil, misplaced waste, and abiotic resource depletion/scarcity. Furthermore, it was noticed that the absolute environmental boundaries in the PBs are proposed at the early stage of the causal network, while the boundaries in the SDGs are generally proposed at the latter end of the causal network.

8.1.3. Application of absolute environmental sustainability approach to the agri-food and construction sectors at sub-global levels (Objective 3)

A method for applying the ASLCA framework was developed to investigate whether the climate change performance of the agri-food (Chapter 4) and construction (Chapter 5) sectors were aligned with achieving the 2°C global climate target.

To that end, an annual global carbon budget related to the 2°C climate target was initially calculated, which was constant over time. A share of the annual global carbon budget was assigned to the global agri-food sector, and subsequently to different agri-food systems in New Zealand (agri-food sector, horticulture industries and products) using four different sharing methods: grandfathering, economic value, agricultural land and calorific content (see Section 4.3 for details). Next, the LCA climate impacts of these agri-food systems were calculated and benchmarked against the assigned carbon budget shares. The research showed that although the global agri-food sector exceeded its carbon budget share in the year 2013, the New Zealand agri-food systems were within their carbon budget shares for one of the four methods (i.e. economic value method). This can be explained by the fact that many New Zealand agri-food sector products achieve price premiums in international markets.

Buildings have different characteristics when compared with agri-food systems, since they are long-lived. Thus, addressing absolute environmental sustainability in this sector requires accounting for temporal aspects such as growth in population and number (and size) of buildings in a given time period. In that context, the study in Chapter 5 introduced the so-called "top-down" approach to propose a climate target for the whole life cycle of a building in any country. The novelty of this research was two-fold. First, it included a stock projection that accounted for the projected growth in the number and size of buildings, and the associated climate impacts in a country up to 2050. Second, it provided a breakdown of the climate target into individual life cycle stages. The proposed top-down approach was then applied to define a climate target for a detached house in New Zealand, which is the most common type of residential building in the country, representing 80% of residential buildings. The climate impacts were calculated using LCA and compared against the defined target to address the question, "Are New Zealand new-built detached houses aligned with achieving the 2°C global climate target?" The research showed that the new-built detached house exceeded its climate targets in all life cycle stages; therefore, substantial effort is necessary to align the climate change performance of the New Zealand detached house (and therefore, the sector) with achieving the 2°C global climate target.

8.1.4. Understanding the policy implications of the results and the role of different methodological and value choices of an AESA study (Objective 4)

In Chapter 5, greenhouse gas (GHG) emission reduction targets (aligned with the 2°C global climate target) were calculated for New Zealand agri-food systems, including the agri-food sector, horticulture industries (such as kiwifruit, apple and wine) and their products. At the same time, in Chapter 6, a climate target for a New Zealand new-built detached house was proposed. The former study was based on a production-based accounting approach, whereas the latter was based on a consumption-based accounting approach.

Both (i.e. production- and consumption-based accounting) approaches were then applied to evaluate the climate impacts of the total economy of New Zealand in 2011 (Chapter 7) and 2012 (Chapter 6). For this purpose, the Eora-based multi-regional input-output (MRIO) analysis was applied. Note that this was the first study to apply MRIO analysis to calculate the climate (or any other environmental) impacts of New Zealand. The results showed that New Zealand was a net exporter of GHG emissions in 2011 and 2012. Furthermore, the research provided two key insights to support climate change mitigation activities and policymaking. First, the production- and consumption-based accounting approaches provided different rankings for the most dominant economic sectors contributing to New Zealand's GHG emissions. Second, only the consumption-based accounting approach enabled the quantification of the embodied emissions in New Zealand's trade activities, and it was found that around 50% of consumption-based GHG emissions are embodied in New Zealand's trade activities. Since the production- and consumption-based accounting approaches provide different insights, both approaches should be used in a complementary manner when developing climate policies.

Finally, although the uncertainty associated with different value choices in AESA studies were discussed in the literature, no study has previously investigated the influence of value and/or modelling choices on the outcomes of an AESA. To that end, this study investigated the influence of different value and modelling choices on the outcomes of an AESA by applying the ASLCA to the total economy of New Zealand (Chapter 7). The research showed that, for each GHG emission accounting approach, the largest uncertainty was associated with the choice of climate threshold, followed by the choice of sharing principle, and then the choice of calculation method for the global carbon budget. Therefore, when setting climate targets based on the outcomes of an AESA, these uncertainties should be taken into consideration.

8.2. Limitations and future work

To utilise the proposed ASLCA framework as a comprehensive AESA framework as envisaged in Chapter 3, further research is still required. Evaluating environmental impacts such as freshwater contamination, eutrophication (nitrogen and phosphorus), acidification and soil quality degradation requires the development of appropriate absolute environmental boundaries on a regional/local scale (Häyhä et al. 2016; Li et al. 2019; Willett et al. 2019). While research on developing absolute environmental boundaries on a regional/local scale is still emerging, existing environmental standards proposed in the national policies can be used as a proxy[54]. For example, for a country like New Zealand, national water quality standards can be utilised to develop nitrogen and phosphorus boundaries at a catchment level based. Likewise, ambient air quality guidelines proposed by the World Health Organization (WHO 2006) can be used when setting absolute environmental boundaries for atmospheric aerosol loading and stratospheric ozone depletion.

Furthermore, when the ASLCA framework becomes mature and capable of evaluating multiple environmental impacts on an absolute scale), some researchers may advocate the use of a single index to aggregate the results across different impact categories using numerical factors based on value choices (called weighting), as currently being practiced within the LCA community (Bengtsson and Steen 2000; Pizzol et al. 2017). Although weighting facilitates decision-making, it is not recommended to use a single index as it results in burden shifting.

The uncertainty associated with the results of the ASLCA framework was quantified in terms of several value and modelling choices such as the choice of absolute climate threshold, the choice of approach for estimating the global carbon budget, and the choice of downscaling to assign a share of the global carbon budget to economic systems at sub-

[54] A group of LCA researchers globally (called LCAbsolute) is currently investigating the development of AESA methods to evaluate a number of environmental impact categories on an absolute scale. Meanwhile, an emerging interest to undertake AESA studies is observed within the environmentally-extended input-output (EEIO) analysis community.

global levels (ranging from country to sector, industry and product) (Chapter 7). However, the uncertainty associated with the choice of climate metric and the choice of time horizon was not quantified (as discussed in Chapter 4). Since these aspects are particularly relevant to the assessment of short-lived GHG emissions (Owsianiak et al. 2018; Reisinger et al. 2017), a potential future study could be to investigate the uncertainty related to all of these value and modelling choices.

Likewise, in Chapter 5, when estimating the climate impacts of the New Zealand detached house building sector, the stock projection developed by BRANZ was used (R Jaques, personal communication, Dec 21, 2018). This projection was based on several assumptions including socio-economic growth in different regions of New Zealand, gross floor area of a new-built detached house, and demolition rate. Furthermore, the GHG emissions associated with the operational energy use stages of the pre-existing New Zealand detached houses were based on the assumption that the energy requirement was completely met by electricity. However, in reality, there exist some pre-existing detached houses which still use other energy sources such as wood, natural gas and liquefied petroleum gas in addition to electricity (as previously discussed in Section 5.2.1). The uncertainty associated with these assumptions was not comprehensively evaluated in this research. Hence, future research should focus on understanding the influence of these underlying assumptions on the outcomes of the proposed top-down approach and climate target by undertaking scenario analyses on potential New Zealand energy mixes and building materials.

There are additional limitations associated with the methodologies and data used in this research. The DPSIR framework used (in Chapters 2 and 3) to classify the environmental indicators (listed in the PBS, SDGs and LCA) has some limitations (Patrício et al. 2016). For example, it is not always clear whether an indicator should be classified as a driver/pressure, pressure/state, or state/impact; the timescale over which the environmental impact occurs is generally overlooked; and many of the studies on the DPSIR framework and its derivatives like enhanced-DPSIR are conceptual. Therefore, more case studies investigating the application of these kinds of conceptual models are necessary.

Similarly, there are uncertainties related to the MRIO methodology and databases, which are extensively discussed in the literature (Karstensen et al. 2015; Malik et al. 2018;

Owen 2017; Peters et al. 2012; Steininger et al. 2018). For example, in the research presented in Chapters 6 and 7, the Eora database (Lenzen et al. 2013) was used, which provides a detailed sectoral disaggregation (14839 industry sectors for 189 countries). However, some sectors (including agriculture) are still aggregated for most countries (Lenzen et al. 2013). For example, the Swiss agriculture sector includes products from both the forestry and fishing industries. Hence, additional effort is necessary when developing climate policies to mitigate GHG emissions from these aggregated sectors. Although the GTAP database provides a relatively higher disaggregation for sectors like agriculture, its geographical coverage (140 countries) is limited when compared with Eora (189 countries) (Li et al. 2019; Stadler et al. 2018). Therefore, it would be interesting to undertake a similar MRIO analysis for New Zealand but using an alternative database (e.g. GTAP, EXIOBASE, WIOD), as previously done for Sweden (Dawkins et al. 2019).

8.3. Conclusions

Overall, the proposed ASLCA framework developed in this research addresses the question, "Are the environmental impacts of a system within the assigned share of the Earth's carrying capacity, and if not, what is the required reduction?" The outcomes of this research are useful to support policymakers in understanding the climate impacts of different economic sectors, goods and services, relative to global climate targets. However, further research is necessary to evaluate the other environmental impacts identified in this study, particularly in absolute terms. Nevertheless, this approach provides a basis for developing a range of environmental impact reduction targets that can potentially catalyse innovation and investment in the environmentally-transformative activities and technologies that are needed to enable human societies to operate and develop within the Earth's "safe operating space".

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Supplementary Material 1

a) Comparison of the impacts coverage of different Life Cycle Assessment (LCA) impact assessment methods
b) Classification of the environmental indicators proposed in the Planetary Boundaries, Sustainable Development Goals and Life Cycle Assessment, using the principles of DPSIR (Driver-Pressure-State-Impact-Response) framework.

Table S1. 1: Impact coverage of different LCA impact assessment methods (European Commission- Joint Research Centre 2010)

Environmental impact	ReCiPe2016	CML	Eco-indicator 99	EDIP	IMPACT 2002+
Water scarcity (quantity)	✔	-	-	-	-
Water quality degradation	✔ (freshwater & marine ecotoxicity)	✔ (freshwater & marine ecotoxicity)	✔	✔ (freshwater & marine ecotoxicity)	✔ (aquatic ecotoxicity)
Climate-related hazards	✔	✔	✔	✔	✔
Terrestrial & aquatic eutrophication	✔ (aquatic only)	✔	-	✔	✔
Land-system change	✔ (agricultural purpose)	✔	✔	✔	✔
Ambient air pollution and other toxic effects	✔ (PMs, photochemical oxidants)	✔ (photochemical oxidants)	✔ (PMs, VOCs)	✔ (PMs)	✔ (PMs, photochemical oxidants)
Terrestrial & ocean acidification	✔	✔	✔	✔	✔
Soil degradation and loss of soil	✔		✔	✔	✔
Ozone depletion	✔ (stratospheric)	✔ (tropospheric & stratospheric)	✔	✔	✔
Loss of individuals from threatened species due to direct (physical) human activities	-	-	-	-	-
Misplaced wastes	-	-	-	-	-
Abiotic resource depletion/ scarcity	✔	✔	✔	✔	✔
AoPs	ReCiPe2016	CML	Eco-indicator 99	EDIP	IMPACT 2002+
AoPs **Human health**	✔	✔	✔	✔	✔
Ecosystem Quality	✔	✔	✔	✔	✔ , plus life support system (climate change)
Resource availability	✔	✔	✔	✔	✔
Man-made environment	-	✔	-	-	-

Table S1. 2: Classification of environmental sustainability indicators reported in three different ESAMs against identified environmental problems

No.	Environmental Problem	Point of Evaluation	Indicators in Different Approaches		
			PBs	LCA	SDG Indicators
1	Water scarcity (quantity)	P	***Maximum amount of consumptive blue water use***	Water use	6.4.1 Change in water-use efficiency over time
		S	***Blue water withdrawal as % of mean monthly river flow***	Water scarcity	6.4.2 Level of water stress: freshwater withdrawal as a proportion of available freshwater resources
		I	-	-	-
		R	-	-	6.5.1 Degree of integrated water resources management implementation (0-100); 6.5.2 Proportion of transboundary basin area with an operational arrangement for water cooperation; 6.a.1 Amount of water- and sanitation-related official development assistance that is part of a government-coordinated spending plan; 6.b.1 Proportion of local administrative units with established and operational policies and procedures for participation of local communities in water and sanitation management
2	Water quality degradation	P	-	-	12.4.2 Hazardous waste generated per capita and proportion of hazardous waste treated, by type of treatment
		S	-	Freshwater ecotoxicity; Marine ecotoxicity	6.1.1 Proportion of population using safely managed drinking water services; 6.3.2 Proportion of bodies of water with good ambient water quality
		I	-	-	-
		R	-	-	6.3.1 Proportion of wastewater safely treated; 6.5.1 Degree of integrated water resources management implementation (0-100); 6.5.2 Proportion of transboundary basin area with an operational arrangement for water cooperation; 6.a.1 Amount of water- and sanitation-related official development assistance that is part of a government-coordinated spending plan;

					6.b.1 Proportion of local administrative units with established and operational policies and procedures for participation of local communities in water and sanitation management; 12.4.2 Hazardous waste generated per capita and proportion of hazardous waste treated, by type of treatment
3	Climate-related natural hazards	P	-		9.4.1 CO2 emission per unit of value added
		S	***Atmospheric CO_2 concentration; Energy imbalance at top-of-atmosphere***	Infrared radiative forcing increase	-
		I	-	-	-
		R	-	-	1.5.1/11.5.1/13.1.1 Number of deaths, missing persons and directly affected persons attributed to disasters per 100,000 population; 1.5.2 Direct economic loss attributed to disasters in relation to global gross domestic product (GDP); 1.5.3/11.b.1/13.1.2 Number of countries that adopt and implement national disaster risk reduction strategies in line with the Sendai Framework for Disaster Risk Reduction 2015–2030 1.5.4/11.b.2/13.1.3 Proportion of local governments that adopt and implement local disaster risk reduction strategies in line with national disaster risk reduction strategies 11.5.2 Direct economic loss in relation to global GDP, damage to critical infrastructure and number of disruptions to basic services, attributed to disasters; 11.c.1 Proportion of financial support to the least developed countries that is allocated to the construction and retrofitting of sustainable, resilient and resource efficient buildings utilizing local materials 13.1.1 Number of deaths, missing persons and directly affected persons attributed to disasters per 100,000 population 13.2.1 Number of countries that have communicated the establishment or operationalization of an integrated policy/strategy/plan which increases their ability to adapt to the adverse impacts of climate change, and foster climate resilience and low greenhouse gas emissions development in a manner that does not threaten food production (including a national adaptation plan, nationally determined contribution, national communication, biennial update report or other)

4	Terrestrial & aquatic eutrophication	P	***Phosphorous flow from freshwater systems into the ocean;*** ***Phosphorous flow from fertilizers to erodible soils;*** ***Industrial and intentional biological fixation of Nitrogen***	-	-
		S	-	Residence time of nutrients in freshwater or marine end compartment; Accumulated exceedance of critical loads of Nitrogen in terrestrial ecosystems	14.1.1 Index of coastal eutrophication and floating plastic debris density
		I	-	-	-
		R	-	-	-
5	Land-system change	P	-	-	11.3.1 Ratio of land consumption rate to population growth rate
		S	**Area of forested land as % of original forest cover;** **Area of forested land as % of potential forest**	Occupation and time-integrated land transformation	15.1.1 Forest area as a proportion of total land area; 15.1.2 Proportion of important sites for terrestrial and freshwater biodiversity that are covered by protected areas, by ecosystem type ; 15.4.2 Mountain Green Cover Index
		I	-	-	-

		R	-	-	15.2.1 Progress towards sustainable forest management; 15.4.1 Coverage by protected areas of important sites for mountain biodiversity; 15.a.1 Official development assistance and public expenditure on conservation and sustainable use of biodiversity and ecosystems; 15.b.1 Official development assistance and public expenditure on conservation and sustainable use of biodiversity and ecosystems
6	Ambient air pollution and other toxic effects	P	-	Tropospheric ozone increase;	SDG Indicators 12.4.2 (P);
		S	Aerosol Optical Depth (AOD); AOD as a seasonal average over a region	PM2.5 population intake;	11.6.2 Annual mean levels of fine particulate matter (e.g. PM2.5 and PM10) in cities (population weighted)
		I	-	Risk increase of cancer and non-cancer disease incidence	***3.4.1 Mortality rate attributed to cardiovascular disease, cancer, diabetes or chronic respiratory disease***; 3.9.1 Mortality rate attributed to household and ambient air pollution; 3.9.3 Mortality rate attributed to unintentional poisoning
		R	-	-	-
7	Terrestrial & ocean acidification	P	-	-	-
		S	***Carbonate ion concentration, average global surface ocean saturation state with respect to aragonite***	Proton increase in natural soils; Changes in surface oceans pH and carbonate mineral saturation state	14.3.1 Average marine acidity (pH) measured at agreed suite of representative sampling stations
		I	-	-	-
		R	-	-	-
8	Soil degradation and loss of soil	P	-	-	2.4.1 Proportion of agricultural area under productive and sustainable agriculture
		S	-	Soil organic matter	-
		I	-	-	15.3.1 Proportion of land that is degraded over total land area
		R	-	-	-
9	Ozone depletion	P		-	-

		S	***Stratospheric ozone concentration***	Stratospheric ozone decrease	-
		I	-	-	-
		R	-	-	-
10	Loss of individuals from threatened species due to direct (physical) human activities	P	-	-	14.7.1 Sustainable fisheries as a proportion of GDP in small island developing States, least developed countries and all countries; ***15.7.1/15.c.1 Proportion of traded wildlife that was poached or illicitly trafficked***
		S	-	-	14.1.1 Index of coastal eutrophication and floating plastic debris density ; 14.4.1 Proportion of fish stocks within biologically sustainable levels
		I	-	-	-
		R	-	-	14.2.1 Proportion of national exclusive economic zones managed using ecosystem-based approaches; ***14.5.1 Coverage of protected areas in relation to marine areas***; 14.6.1 Progress by countries in the degree of implementation of international instruments aiming to combat illegal, unreported and unregulated fishing; 14.b.1 Progress by countries in the degree of application of a legal/regulatory/policy/institutional framework which recognizes and protects access rights for small-scale fisheries; 14.c.1 Number of countries making progress in ratifying, accepting and implementing through legal, policy and institutional frameworks, ocean-related instruments that implement international law, as reflected in the United Nation Convention on the Law of the Sea, for the conservation and sustainable use of the oceans and their resources; 15.6.1 Number of countries that have adopted legislative, administrative and policy frameworks to ensure fair and equitable sharing of benefits; 15.9.1 Progress towards national targets established in accordance with Aichi Biodiversity Target 2 of the Strategic Plan for Biodiversity 2011-2020
11	Misplaced wastes	P	-	-	14.1.1 Index of coastal eutrophication and floating plastic debris density
		S	-	-	-
		I	-	-	-
		R	-	-	14.2.1 Proportion of national exclusive economic zones managed using ecosystem-based approaches;

12	Abiotic resource depletion/ scarcity	P	-	Mineral resource scarcity	7.1.2 Proportion of population with primary reliance on clean fuels and technology; 7.3.1 Energy intensity measured in terms of primary energy and GDP; 8.4.1/12.2.1 Material footprint, material footprint per capita, and material footprint per GDP; 8.4.2/12.2.2 Domestic material consumption, domestic material consumption per capita, and domestic material consumption per GDP;
		S	-	-	-
		I	-	-	-
		R	-	-	7.2.1 Renewable energy share in the total final energy consumption; 7.a.1 International financial flows to developing countries in support of clean energy research and development and renewable energy production, including in hybrid systems; 7.b.1 Investments in energy efficiency as a proportion of GDP and the amount of foreign direct investment in financial transfer for infrastructure and technology to sustainable development services; 11.c.1 Proportion of financial support to the least developed countries that is allocated to the construction and retrofitting of sustainable, resilient and resource-efficient buildings utilizing local materials; 12.b.1 Number of sustainable tourism strategies or policies and implemented action plans with agreed monitoring and evaluation tools; 12.c.1 Amount of fossil-fuel subsidies per unit of GDP (production and consumption) and as a proportion of total national expenditure on fossil fuels

Note that since most of the anthropogenic activities act as the drivers of these environmental problems, they have not specifically shown in this table. SDG Indicators 12.1.1 (Number of countries with sustainable consumption and production (SCP) national action plans or SCP mainstreamed as a priority or a target into national policies); 12.6.1 (Number of companies publishing sustainability reports); 12.7.1 (Number of countries implementing sustainable public procurement policies and action plans); and 12.a.1 (Amount of support to developing countries on research and development for sustainable consumption and production and environmentally sound technologies) are relevant to all environmental problems at the response point; hence, not specifically represented in the table. The letter shown in the parenthesis refers to the position of impact assessment along the causal network: D-Driver; P-Pressure; S-State; I-Impact; and R-Response. Moreover, the text in bold-italic indicates the already existing AESIs in the PBs, LCA and SDGs. For a complete description of PBs, see Steffen et al. (2015), whereas for SDGs, see United Nations (2017). Moreover, for LCA indicators see Huijbregts et al. (2016) and ISO (2006).

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Supplementary Material 2

a) Greenhouse gas (GHG) emissions of different economic systems including the entire Earth, global agri-food sector, New Zealand agri-food sector, New Zealand horticulture industries, and their products (kiwifruit, apple and wine)
b) Details on production volume, economic value, agriculture land use, and calorific contribution of the above-mentioned systems
c) Calculated carbon budget shares for each system
d) Carbon budget downscaling methods: grandfathering, economic and agri-land (see Supplementary Material (SM) 2. d, available online)
e) Carbon budget downscaling method: calorific content (SM 2. e, available online)

Table S2. 1: Summary of data used in the study

Economic level	Parameter	Year	Unit	Value	Uncertainty (+/- in %)	Sources
Global	GHG Glo	2013	$GtCO_2eq$	48.3	9.2	CAIT (2017)
	CB Glo	Annual	$GtCO_2eq$	29.9		Calculated from (Doka 2016)
Global-Agrifood Sector	GHG %Glo,AgFd	Calculated for 2008 and applied it to 2013, assuming that the GHG emissions contribution was same in both years	%	23.6	27.1	Calculated from (CAIT 2017; Vermeulen et al. 2012)
	GHG Glo,AgFd	2013	$GtCO_2eq$	11.4	36.2	Calculated from (Vermeulen et al. 2012)
	Land Glo,Ag	2013	ha	48843738300		FAO (2018a,b)
	EC Glo, AgFd	2013	1000 US$	1340037594		FAO (2018c)
Sectoral-New Zealand AgFd	GHG NZ,AgFd	2013	$MtCO_2eq$	56.3	16.0	Calculated from (Allan 2016; Fitzgerald et al. 2011a, 2011b; FAO 2018a,b)
	Land NZ,AgFd	2013	ha	11106000		FAO (2018a,b)
	EC NZ,AgFd	2013	1000 US$	21846243		FAO (2018c)
Industry-New Zealand Kiwifruit	GHG NZ,AgFd,Ikiwi	Calculated based on the data collected during 2008	$ktCO_2eq$	457.2		Calculated from (Mithraratne et al. 2010)
	EC NZ,AgFd,Ikiwi	2013	M NZ$	965		Calculated from (Stats NZ 2014)
	Land NZ,AgFd,Ikiwi	2013	ha	12375		(FAO 2018a)
Industry-New Zealand Apple	GHG NZ,AgFd,Iapple	Calculated based on the data collected during 2008/2009	$ktCO_2eq$	488.6		Calculated from (Mithraratne et al. 2010)
	EC NZ,AgFd,Iapple	2013	M NZ$	427		Calculated from (Stats NZ 2014)
	Land NZ,AgFd,Iapple	2013	ha	8685		(FAO 2018a)

Industry-New Zealand Wine	GHG NZ,AgFd,Iwine	Calculated based on the data collected during 2009/2010	$ktCO_2eq$	407.5	Calculated from (McLaren et al. 2009)
	EC NZ,AgFd,Iwine	2013	M NZ$	1887	Calculated from (Stats NZ 2014)
	Land NZ,AgFd,Iwine	2013	ha	36060	(FAO 2018a)
Product-kiwifruit	GHG NZ,AgFd,Ikiwi	Calculated based on the data collected during 2008	$ktCO_2eq/kg$	1143.2	Calculated from (Mithraratne et al. 2010)
Product- apple	GHG NZ,AgFd,Iapple	Calculated based on the data collected during 2008/2009	$ktCO_2eq/kg$	1062.2	Calculated from (McLaren et al. 2009)
Product- wine	GHG NZ,AgFd,Iwine	Calculated based on the data collected during 2009/2010	$ktCO_2eq/kg$	1640.4	Calculated from (Barry 2011)

Table S2. 2: Global-level results

Parameter	Unit	Value		Source
1. Global-level				
GHG emissions in 2013	$GtCO_2eq$	48.3		CAIT (2017)
GHG target at the global level				
Radiative forcing	Wm^{-2}	2.6		Renaud and Matthews (2015)
GWP of CO2	$W{\cdot}a/(m^2kgGWP_{100}CO_2eq)$	8.69×10^{-14}		Doka (2016)
Annual carbon budget	$GtCO_2eq/a$	29.9		
2. Global agrifood sector-level		**Lower limit**	**Upper limit**	
GHG emissions from the global agrifood sector	$MtCO_2eq$	9800	16900	Vermeulen et al. (2012)
In percentage	%	19	29	Vermeulen et al. (2012)
GHG emissions from catering and domestic food management	$MtCO_2eq$	232	232	Vermeulen et al. (2012)
Adjusted GHG emissions from the global agrifood sector	$MtCO_2eq$	9568	16668	Calculated from (Vermeulen et al. 2012)
Adjusted % GHG emissions from the global agrifood sector	%	18.6	28.6	Calculated from (Vermeulen et al. 2012)
Adjusted % GHG emissions from the global agrifood sector (Average)	%	23.6		Calculated from (Vermeulen et al. 2012)
Uncertainty	%	27.06		
Allocated carbon budget for the agrifood sector	$GtCO_2eq/a$	7.06		It was assumed that the GHG emissions contribution of the global agri-food sector (in %) remained the same in 2008 and 2013.
Uncertainty	%	36.2		

Table S2. 3: National (New Zealand)-level results

Parameter	Year	Unit	Value	Sources
National-level				
GHG emissions in NZ	2013	$MtCO_2eq$	66.7	MfE (2015)
NZ Agri-food sector emissions				
Direct and indirect emissions from agricultural production activities	2013	$MtCO_2eq$	38.6	FAO (2018a)
Upstream and downstream emissions within the cradle-to-NZ port system boundary	2013	$MtCO_2eq$	16.1	Calculated from (Allan 2016; Stats NZ 2014)
Export related shipping emissions for agri-foods	Reported for the year 2007; assumed to be same in 2013	$MtCO_2eq$	1.2	Fitzgerald et al. (2011a, 2011b)
Fertiliser/pesticide imports related emissions	Reported for the year 2007; assumed to be same in 2013	$MtCO_2eq$	0.22	Fitzgerald et al. (2011a, 2011b)
Animal Feedstock import	Reported for the year 2007; assumed to be same in 2013	$MtCO_2eq$	0.16	Fitzgerald et al. (2011a, 2011b)
Total agri-food sector emissions	2013	$MtCO_2eq$	56.3	
National Agri-Food sector-level				
EC NZ,AgFd	2013	MNZ$	68010	Stats NZ (2014)
EC NZ	2013	MNZ$	485135	Stats NZ (2014)
GHG NZ,AgFd	2013	$MtCO_2eq$	56.3	Calculated above
Global Ratio (GR)1	2013		0.004944934	
EC NZ,AgFd	2013	1000US$	21846243	FAO (2018c)
EC Glo,AgFd	2013	1000US$	1340037594	FAO (2018c)
GR2	2013		0.016302709	
Land NZ,Ag	2013	ha	11106000	FAO (2018b)
Land Glo,Ag	2013	ha	48843738300	FAO (2018b)
GR3	2013		0.000227378	
Carbon budget calculation				
CB NZ,AgFd1		$MtCO_2eq$	34.9	
CB NZ,AgFd2		$MtCO_2eq$	115.1	
CB NZ,AgFd3		$MtCO_2eq$	1.6	
CB NZ,AgFd4		$MtCO_2eq$	7.9	

Table S2. 4: New Zealand kiwifruit export information (volume and value) (Fresh Facts-New Zealand Horticulture 2013)

No.	Country/region	Volume	Value (NZ$)	% of total volume	Accumulated %
1	European Union	162,832	305,343,361	42.41	42.41
2	Japan	72,035	238,474,945	18.76	61.17
3	China	40,909	123,005,867	10.65	71.82
4	Taiwan	25,979	69,368,492	6.77	78.59
5	USA	9,686	21,472,024	2.52	81.11
6	Republic of Korea	17,329	44,300,422	4.51	85.62
7	Australia	16,444	33,392,197	4.28	89.90
8	Hong Kong	7,072	21,462,466	1.84	91.75
9	Malaysia	4,579	11,864,023	1.19	92.94
10	Singapore	3,541	9,944,022	0.92	93.86
11	India	2,376	5,207,129	0.62	94.48
12	Indonesia	2,435	6,802,360	0.63	95.11
13	Canada	1,868	3,655,019	0.49	95.60
14	Thailand	2,025	4,617,399	0.53	96.13
15	Vietnam	1,003	2,657,942	0.26	96.39
16	United Arab Emirates	2,118	4,424,000	0.55	96.94
17	Brazil	1,639	3,191,843	0.43	97.37
18	South Africa	1,730	3,644,347	0.45	97.82
19	Saudi Arabia	713	1,718,583	0.19	98.00
20	Philippines	755	1,538,432	0.20	98.20
21	Mexico	1,484	3,184,578	0.39	98.59
22	Kuwait	816	1,730,517	0.21	98.80
23	Russia	1,552	2,855,989	0.40	99.20
24	Israel	401	902,590	0.10	99.31
25	Mauritius	571	1,250,870	0.15	99.46
26	Bahrain	358	735,689	0.09	99.55
27	Reunion	381	813,638	0.10	99.65
28	Qatar	0	0	0.00	99.65
29	Lebanon	286	548,408	0.07	99.72
30	Costa Rica	190	394,771	0.05	99.77
31	New Caledonia	284	844,047	0.07	99.85
32	French Polynesia	149	489,275	0.04	99.88
33	El Salvador	120	249,963	0.03	99.92
34	Guatemala	143	283,463	0.04	99.95
35	Fiji	31	86,585	0.01	99.96
36	Argentina	0	0	0.00	99.96
37	Cambodia	0	0	0.00	99.96
38	Pacific Islands	13	42,809	0.00	99.96
39	Myanmar	0	0	0.00	99.96
40	Papua New Guinea	3	9,804	0.00	99.97

41	Egypt	0	0	0.00	99.97
42	Chile	0	0	0.00	99.97
43	Uruguay	0	0	0.00	99.97
44	Syria	0	0	0.00	99.97
45	Oman	0	0	0.00	99.97
46	Jordan	124	172,097	0.03	100.00
47	Bermuda	0	0	0.00	100.00
48	Sri Lanka	9	15,948	0.00	100.00

Table S2. 5: Calculation of weighted distances between New Zealand and kiwifruit export destinations

Country/region	Major port	% of total volume	Distance (nautical miles)*	% X Distance
European Union	Bruges	42.41	20,822	883,061.0
Japan	Yokohama	18.76	11,272	211,462.7
China	Shanghai	10.65	5,029	53,558.9
Taiwan	Keelung	6.77	5,358	36,273.7
USA	Portsmouth	2.52	5,070	12,776.4
Republic of Korea	Busan	4.51	8,312	37,487.1
Australia	Melbourne	4.28	5,301	22,688.3
Hong Kong	Hong Kong	1.84	1,490	2,741.6
Malaysia	Kuala Baran	1.19	5,266	6,266.5
Singapore	Singapore	0.92	4,998	4,598.2
India	Calcutta	0.62	4,998	3,098.8
Indonesia	Jakarta	0.63	6,658	4,194.5
Total % X Distance		**95.1**	4,748	1,278,207.7

Table S2. 6: New Zealand kiwifruit industry results

Variable	Year	Unit	Value	Source
NZ kiwifruit industry-level				
Carbon footprint of one kg of kiwifruit	Calculated based on the data collected in 2008	$kgCO_2e/kg$	1.143	Mithraratne et al. (2010)
Total production in 2013 for human consumption	2013	t	399,947	Table S2.5 in SI2
GHG NZ,AgFd, Ikiwi	Calculated based on the data collected in 2008	$ktCO_2eq$	457.2	Calculated from (Mithraratne et al. 2010)
GHGAgFd, Ikiwi	Calculated based on the data collected in 2008	$MtCO_2eq$	0.457235431	Calculated from (Mithraratne et al. 2010)
EC AgFd, Ikiwi		MNZ$	965	Stats NZ (2014)
Land AgFd, Ikiwi		ha	12375	FAO (2018a)
National Ratio (NR)1-GHG			0.0081	
NR2-EC			0.0142	
NR3-Land			0.0011	
CB NZ,AgFd1,Ikiwi		$MtCO_2eq$	0.2835	
CB NZ,AgFd2,Ikiwi		$MtCO_2eq$	1.6334	
CB NZ,AgFd3,Ikiwi		$MtCO_2eq$	0.0018	

Table S2. 7: New Zealand kiwifruit (product) level results

Parameter	Units	Value
Weighted shipping distance	km	24,892.1
CO2 intensity of international shipping (sea freight)	$kgCO_2e/(t.km)$	0.02
Waste percentage	%	10
Carbon footprint associated with international shipping	$kgCO_2e/tray$	2.0
Carbon footprint of NZ kiwifruit consumed in any part of the world	$kgCO_2e/tray$	3.8
Carbon footprint of NZ kiwifruit consumed in any part of the world	$kgCO_2e/kg$	1.1432
CB NZ,AgFd1,Pkiwi	$kgCO_2e/kg$	0.7088
CB NZ,AgFd2,Pkiwi	$kgCO_2e/kg$	4.084
CB NZ,AgFd3,Pkiwi	$kgCO_2e/kg$	0.00447

Table S2. 8: New Zealand apple export information (volume and value) (Fresh Facts-New Zealand Horticulture 2013)

No.	Country/region	Volume	% of total volume	Accumulated %
1	European Union	92,432	28.69	28.69
2	UK	43,923	13.63	42.33
3	USA	38,788	12.04	54.37
4	Thailand	27,078	8.41	62.78
5	UAE	18,096	5.62	68.39
6	India	15,048	4.67	73.06
7	Hong Kong	11,395	3.54	76.60
8	china	9,856	3.06	79.66
9	Taiwan	8,858	2.75	82.41
10	Singapore	8,148	2.53	84.94
11	Canada	7,918	2.46	87.40
12	Malaysia	7,213	2.24	89.64
13	Russia	6,422	1.99	91.63
14	Indonesia	3,860	1.20	92.83
15	Vietnam	3,794	1.18	94.01
16	Japan	2,362	0.73	94.74
17	Fiji	2,204	0.68	95.42
18	Sri Lanka	2,194	0.68	96.11
19	New Caledonia	1700	0.53	96.63
20	All other countries	10846	3.37	100.00

Table S2. 9: Calculation of weighted distances between New Zealand and apple export destinations

Country/region	Major port	% of total volume	Distance (nautical miles)*	% X Distance
European Union	Bruges	28.69	11,272	598,999.4
UK	Dover	13.63	11,200	282,821.9
USA	New Jersey (strait of Magellan)	12.04	8,312	185,355.7
Thailand	Bangkok	8.41	5,985	93,171.7
UAE	Dubai	5.62	7,909	82,282.5
India	Calcutta	4.67	6,658	57,600.4
Hong Kong	Hong Kong	3.54	5,266	34,498.3
china	Shanghai	3.06	5,358	30,360.3
Taiwan	Keelung	2.75	5,070	25,819.4
Singapore	Singapore	2.53	4,998	23,412.6
Canada	Toronto (via panama canal)	2.46	10,012	45,576.3
Malaysia	Kuala Baran	2.24	4,998	20,726.0
Russia	Novorossiysk (via Suez canal)	1.99	10,442	38,552.9
Indonesia	Jakarta	1.20	4,748	10,536.6
Vietnam	Ho Chi Minh	1.18	5,510	12,018.5
Japan	Yokohama	0.73	5,029	6,829.1
Fiji	Lautoka	0.68	1,554	1,969.1
Total % X Distance		95.42		1,550,530.9

Table S2. 10: New Zealand apple industry results

Parameter	Year	Unit	Value	Source
NZ applefruit industry-level				
Carbon footprint of one kg of applefruit	Based on the data collected in 2008/2009	$kgCO_2e/kg$	1.1	Barry (2011)
Total production in 2013	2013	t	460000.0	Table S2.4 in SI2
GHG NZ,AgFd, Iapple	Based on the data collected in 2008/2009	$ktCO_2eq$	488.6	
GHG AgFd,Iapple		$MtCO_2eq$	0.4886	
EC AgFd,Iapple	2013	MNZ$	427	Stats NZ (2014)
Land AgFd, Iapple	2013	ha	8685	FAO (2018a)
NR1			0.0087	
NR2			0.0063	
NR3			0.0008	
CB NZ,AgFd1,Iapple		$MtCO_2eq$	0.303	
CB NZ,AgFd2,Iapple		$MtCO_2eq$	0.723	
CB NZ,AgFd3,Iapple		$MtCO_2eq$	0.001	

Table S2. 11: New Zealand apple (product) level results

Parameter	Unit	Value
Weighted shipping distance	km	30092.8
CO2 intensity of international shipping (sea freight)	$kgCO_2e/(t.km)$	0.022
Waste percentage	%	10
Carbon footprint associated with international shipping	$kgCO_2e/kg$	0.74
Carbon footprint of NZ apple consumed in any part of the world	$kgCO_2e/kg$	1.0622
CB NZ,AgFd1,Papple	$kgCO_2e/kg$	0.6586
CB NZ,AgFd2,Papple	$kgCO_2e/kg$	1.57
CB NZ,AgFd3,Papple	$kgCO_2e/kg$	0.0027

Table S2. 12: New Zealand wine export information (Fresh Facts-New Zealand Horticulture 2013)

No.	Country/region	% of total value	Accumulated %
1	Australia	31.00	31.00
2	UK	23.00	54.00
3	USA	24.00	78.00
4	Canada	6.00	84.00
5	Netherlands	2.00	86.00
6	China	2.00	88.00
7	Ireland	1.00	89.00
8	Singapore	1.00	90.00
9	Hong Kong	1.00	91.00
10	All other countries	9.00	100.00

Table S2. 13: Calculation of weighted distances between New Zealand and wine export destinations

Country/region	Major port	% of total volume	Distance (nautical miles)*	% X Distance
Australia	Melbourne	31.00	1490	46190
UK	Dover	29.00	11200	324800
USA	New Jersey (strait of Magellan)	20.00	8312	166240
Canada	Toronto (via panama canal)	6.00	10012	60072
Netherlands	Amsterdam (via panama)	2.00	11360	22720
china	Shanghai	2.00	5358	10716
Ireland	Dublin (via panama)	2.00	10992	21984
Singapore	Singapore	1.00	4998	4998
Hong Kong	Hong Kong7	1.00	5266	5266
Total % X Distance		94		662986

Table S2. 14: New Zealand wine industry results

Parameter	Year	Unit	Value	Source
NZ wine industry-level				
Carbon footprint of one kg of wine	Based on the data collected in2009/2010	$kgCO_2e/kg$	1.64	Barry (2011)
Total production in 2013		t	248400.0	Table S2.4 in SI2
GHG NZ,AgFd, Iwine	Based on the data collected in 2009/2010	$ktCO_2eq$	407.5	Calculated from (Barry 2011)
GHG Iwine,AgFd		$MtCO_2eq$	0.407483872	
EC Iwine,AgFd	2013	MNZ$	1887	Stats NZ (2014)
Land Iwine, Ag	2013	ha	36060	FAO (2018a)
National Ratio (NR)1			0.0072	
NR2			0.0277	
NR3			0.0032	
CB NZ,AgFd1,Iwine		$MtCO_2eq$	0.253	
CB NZ,AgFd2,Iwine		$MtCO_2eq$	3.194	
CB NZ,AgFd3,Iwine		$MtCO_2eq$	0.005	

Table S2. 15: New Zealand wine (product) level results

Parameter	Unit	Value
Weighted shipping distance	km	13062.23481
CO2 intensity of international shipping (sea freight)	$kgCO_2e/(t.km)$	0.0223
Waste percentage	%	10
Carbon footprint associated with international shipping	$kgCO_2e/bottle$	0.390325701
Carbon footprint of NZ wine consumed in any part of the world	$kgCO_2e/bottle$	1.230325701
Carbon footprint of NZ wine consumed in any part of the world	$kgCO_2e/kg$	1.640434267
CB NZ,AgFd1,Pwine	$kgCO_2e/kg$	1.0171
CB NZ,AgFd2,Pwine	$kgCO_2e/kg$	12.858
CB NZ,AgFd3,Pwine	$kgCO_2e/kg$	0.02099

Table S2. 16: Global agri-food production in terms of volume and calories

Item	Food supply quantity (kg/capita/yr) (FAO 2018a)	Food supply (Cal/capita/day) (FAO 2018a)	Total production available for human consumption (t)	Total production available for human consumption ($x10^6$ Cal)	Calorific content (Cal/t)	Remarks
Apples and products	10.1	12	70,672,993	30,648,288	433,663	
Aquatic Animals, Others	0.18	0	1,259,519	-	-	Not considered in this study
Aquatic Plants	2.04	2	14,274,545	5,108,048	357,843	Not considered in this study
Bananas	12.28	21	85,927,163	53,634,504	624,186	
Barley and products	0.97	7	6,787,406	17,878,168	2,634,021	
Beans	2.49	23	17,423,342	58,742,552	3,371,486	
Beer	26.73	33	187,038,524	84,282,792	450,617	
Beverages, Alcoholic	3.04	24	21,271,871	61,296,576	2,881,579	
Beverages, Fermented	4.2	5	29,388,769	12,770,120	434,524	
Bovine Meat	9.32	40	65,215,078	102,160,960	1,566,524	
Butter, Ghee	1.39	30	9,726,283	76,620,720	7,877,698	
Cassava and products	14.38	37	100,621,548	94,498,888	939,152	
Cephalopods	0.51	1	3,568,636	2,554,024	715,686	Not considered in this study
Cereals, Other	0.76	6	5,317,968	15,324,144	2,881,579	
Citrus, Other	1.62	1	11,335,668	2,554,024	225,309	
Cloves	0.01	0	69,973	-	-	
Cocoa Beans and products	0.68	5	4,758,182	12,770,120	2,683,824	
Coconut Oil	0.29	7	2,029,225	17,878,168	8,810,345	
Coconuts - Incl Copra	3.18	12	22,251,497	30,648,288	1,377,358	
Coffee and products	1.16	1	8,116,898	2,554,024	314,655	
Cottonseed	0	0	-	-	-	
Cottonseed Oil	0.56	13	3,918,503	33,202,312	8,473,214	
Cream	0.43	2	3,008,850	5,108,048	1,697,674	

Crustaceans	1.79	2	12,525,214	5,108,048	407,821	Not considered in this study
Dates	0.93	4	6,507,513	10,216,096	1,569,892	
Demersal Fish	2.89	6	20,222,272	15,324,144	757,785	Not considered in this study
Eggs	9.19	36	64,305,426	91,944,864	1,429,815	
Fats, Animals, Raw	1.49	29	10,426,016	74,066,696	7,104,027	
Fish, Body Oil	0.01	0	69,973	-	-	Not considered in this study
Fish, Liver Oil	0	0	-	-	-	Not considered in this study
Freshwater Fish	6.95	13	48,631,416	33,202,312	682,734	Not considered in this study
Fruits, Other (except kiwifruit)			-	-	#DIV/0!	Calculations are shown below
Grapefruit and products	1.18	1	8,256,845	2,554,024	309,322	
Grapes and products (excluding wine)	4.59	7	32,117,726	17,878,168	556,645	
Groundnut Oil	0.55	13	3,848,529	33,202,312	8,627,273	
Groundnuts (Shelled equivalent)	1.75	25	12,245,321	63,850,600	5,214,286	
Honey	0.24	2	1,679,358	5,108,048	3,041,667	
Infant food	0.11	1	769,706	2,554,024	3,318,182	
Kiwifruit			-	-	-	See Table S 2.17
Lemons, Limes and products	2.03	1	14,204,572	2,554,024	179,803	
Maize and products	17.89	147	125,182,162	375,441,527	2,999,162	
Maize Germ Oil	0.31	8	2,169,171	20,432,192	9,419,355	
Marine Fish, Other	1.25	2	8,746,658	5,108,048	584,000	Not considered in this study
Meat, Aquatic Mammals	0	0	-	-	-	Not considered in this study

Meat, Other	0.98	3	6,857,379	7,662,072	1,117,347	
Milk - Excluding Butter	90	138	629,759,340	352,455,311	559,667	
Millet and products	3.29	27	23,021,203	68,958,648	2,995,441	
Molluscs, Other	2.5	1	17,493,315	2,554,024	146,000	Not considered in this study
Mutton & Goat Meat	1.91	11	13,364,893	28,094,264	2,102,094	
Nuts and products	2.34	16	16,373,743	40,864,384	2,495,726	
Oats	0.56	3	3,918,503	7,662,072	1,955,357	
Offal, Edible	2.24	7	15,674,010	17,878,168	1,140,625	
Oil crops Oil, Other	0.25	6	1,749,332	15,324,144	8,760,000	
Oil crops, Other	0.14	1	979,626	2,554,024	2,607,143	
Olive Oil	0.4	10	2,798,930	25,540,240	9,125,000	
Olives (including preserved)	0.39	1	2,728,957	2,554,024	935,897	
Onions	10.97	11	76,760,666	28,094,264	365,998	
Oranges, Mandarins	12.39	9	86,696,869	22,986,216	265,133	
Palm kernels	0	0	-	-	-	
Palm Oil	2.17	52	15,184,197	132,809,247	8,746,544	
Palm kernel Oil	0.28	7	1,959,251	17,878,168	9,125,000	
Peas	0.83	8	5,807,781	20,432,192	3,518,072	
Pelagic Fish	3.1	8	21,691,711	20,432,192	941,935	Not considered in this study
Pepper	0.06	0	419,840	-	-	
Pig meat	16.02	124	112,097,163	316,698,975	2,825,218	
Pimento	0.45	4	3,148,797	10,216,096	3,244,444	
Pineapples and products	2.97	3	20,782,058	7,662,072	368,687	
Plantains	3.27	8	22,881,256	20,432,192	892,966	
Potatoes and products	34.17	64	239,098,629	163,457,535	683,641	
Poultry Meat	14.99	59	104,889,917	150,687,415	1,436,624	
Pulses, Other and products	3.89	37	27,219,598	94,498,888	3,471,722	
Rape and Mustard Oil	1.4	34	9,796,256	86,836,816	8,864,286	

Rape and Mustard seed	0	2	-	5,108,048	-
Rice (Milled Equivalent)	53.92	541	377,295,818	1,381,726,979	3,662,185
Rice bran Oil	0.12	3	839,679	7,662,072	9,125,000
Roots, Other	2.01	6	14,064,625	15,324,144	1,089,552
Rye and products	0.79	6	5,527,888	15,324,144	2,772,152
Sesame seed	0.21	3	1,469,438	7,662,072	5,214,286
Sesame seed Oil	0.1	2	699,733	5,108,048	7,300,000
Sorghum and products	3.45	28	24,140,775	71,512,672	2,962,319
Soybean Oil	3.48	82	24,350,694	209,429,967	8,600,575
Soybeans	1.52	14	10,635,936	35,756,336	3,361,842
Spices, Other	0.73	7	5,108,048	17,878,168	3,500,000
Sugar (Raw Equivalent)	20.53	200	143,655,103	510,804,798	3,555,772
Sugar beet	0.01	0	69,973	-	-
Sugar cane	4.6	4	32,187,700	10,216,096	317,391
Sugar non-centrifugal	0.97	9	6,787,406	22,986,216	3,386,598
Sunflower seed	0.08	1	559,786	2,554,024	4,562,500
Sunflower seed Oil	1.46	35	10,216,096	89,390,840	8,750,000
Sweet potatoes	8.15	22	57,028,207	56,188,528	985,276
Sweeteners, Other	2.98	24	20,852,031	61,296,576	2,939,597
Tea (including mate)	0.85	1	5,947,727	2,554,024	429,412
Tomatoes and products	20.59	11	144,074,942	28,094,264	194,998
Vegetables, Other	108.92	74	762,148,748	188,997,775	247,980
Wheat and products	65.43	527	457,835,040	1,345,970,643	2,939,859
Wine	3.29	6	23,021,203	15,324,144	665,653
Yams	4.65	13	32,537,566	33,202,312	1,020,430
Total			4,690,097,698	7,284,076,419	1,553,076
Total-considered for this study			4,541,614,440	7,194,685,580	

Table S2. 17: Global kiwifruit production in terms of volume and calories

Item	Food supply quantity (kg/capita/yr) (FAO 2018a)	Food supply (Cal/capita/day) (FAO 2018a)	Total production available for human consumption (t)	Total production available for human consumption (Cal)	Calorific content (Cal/t)	Remarks
Fruits, Other	26.51	31	1.85499E+11	79,174,744	426.8	
Kiwifruit			3464394	169,755	49000.0	Calorific content from Roe et al. (2015); production volume from FAO (2018b)
Fruits, Other (except kiwifruit)			1.85496E+11	79,004,988	425.9	

Table S2. 18: New Zealand agri-food production in terms of volume and calories

Item	Production quantity (t) (FAO 2018a)	Domestic supply quantity (t) (FAO 2018a)	Food supply quantity (t) (FAO 2018a)	Imports	Calorific content (Cal/t)	Total NZ production available for human consumption globally (t)	Total NZ production available for human consumption globally (x10^{6} Cal)	Remarks
Apples and products	460,000	116,521	74,108	42,043	433,663	292,563	126,874	Calorific content values for NZ agri-food products are assumed to be the same as the global calorific values
Aquatic Animals, Others	914	1,026	1,026	201	-	914	-	Not considered in this study
Aquatic Plants	1,185	1,919	-	961	357,843	-	-	Not considered in this study
Bananas	-	62,730	56,251	62,750	624,186	-	-	
Barley and products	416,478	393,399	1,804	18,750	2,634,021	1,910	5,031	
Beans	-	7,538	7,386	7,617	3,371,486	-	-	

Beer	288,811	297,134	297,134	34,143	450,617	288,811	130,143	
Beverages, Alcoholic	13,020	32,644	16,644	24,076	2,881,579	6,638	19,129	
Beverages, Fermented	2,310	602	602	1,339	434,524	2,310	1,004	
Bovine Meat	627,174	101,318	101,318	14,801	1,566,524	627,174	982,483	
Butter, Ghee	390,000	41,684	41,684	830	7,877,698	390,000	3,072,302	
Cassava and products	-	28,216	3,284	28,236	939,152	-	-	
Cephalopods	24,820	1,485	1,485	2,746	715,686	24,820	17,763	Not considered in this study
Cereals, Other	12,500	17,504	17,205	5,593	2,881,579	12,286	35,404	
Citrus, Other	8,329	12,512	12,092	4,463	225,309	8,049	1,814	
Cloves	-	22	22	22	-	-	-	
Cocoa Beans and products	-	12,940	334	21,992	2,683,824	-	-	
Coconut Oil	-	4,228	-	5,354	8,810,345	-	-	
Coconuts - including Copra	-	9,543	9,543	9,746	1,377,358	-	-	
Coffee and products	-	18,260	18,260	21,591	314,655	-	-	
Cottonseed	-	-		-	-	-	-	
Cottonseed Oil	-	763	687	764	8,473,214	-	-	
Cream	31,000	846	846	1,103	1,697,674	31,000	52,628	
Crustaceans	3,862	10,430	10,430	12,086	407,821	3,862	1,575	Not considered in this study
Dates	-	1,122	1,110	1,156	1,569,892	-	-	
Demersal Fish	318,582	88,985	42,265	15,744	757,785	151,316	114,665	Not considered in this study
Eggs	57,200	54,485	44,643	1,006	1,429,815	46,868	67,012	
Fats, Animals, Raw	186,492	72,642	20,283	13,499	7,104,027	52,072	369,921	
Fish, Body Oil	2,700	1,282	-	1,237	-	-	-	Not considered in this study
Fish, Liver Oil	620	326	-	330	-	-	-	Not considered in this study
Freshwater Fish	13,044	9,362	9,362	4,338	682,734	13,044	8,906	Not considered in this study

Fruits, Other (except kiwifruit)	95,594				18,416	8,703,207	160,274	See Table S 2.19
Grapefruit and products	661	2,445	2,336	1,914	309,322	632	195	
Grapes and products (excluding wine)	345,000	386,420	41,423	41,796	556,645	36,983	20,586	
Groundnut Oil	-	211	211	215	8,627,273	-	-	
Groundnuts (Shelled Eq)	-	9,798	9,798	9,915	5,214,286	-	-	
Honey	17,823	9,123	9,123	57	3,041,667	17,823	54,212	
Infant food	-	919	919	919	3,318,182	-	-	
Kiwifruit	399,947				49,000	364,126	17,842	Calculations are shown below
Lemons, Limes and products	5,498	5,631	5,293	1,264	179,803	5,168	929	
Maize and products	201,659	189,394	19,982	4,037	2,999,162	21,276	63,810	
Maize Germ Oil	1,116	1,245	1,245	151	9,419,355	1,116	10,512	
Marine Fish, Other	31	52	36	21	584,000	21	13	Not considered in this study
Meat, Aquatic Mammals	-		618,457	-	-	-	-	Not considered in this study
Meat, Other	29,572	12,327	24,950	498	1,117,347	59,854	66,878	
Milk - Excluding Butter	19,469,000	2,594,516	220,731	97,896	559,667	1,656,344	927,001	
Millet and products	-	1,072		1,107	2,995,441	-	-	
Molluscs, Other	89,856	8,678	23,349	2,546	146,000	241,766	35,298	Not considered in this study
Mutton & Goat Meat	481,859	86,278	40,051	2,970	2,102,094	223,683	470,203	
Nuts and products	1,800		27,250	26,643	2,495,726	-	-	
Oats	28,225	58,034	14,842	9,901	1,955,357	7,218	14,115	
Offal, Edible	200,254	135,187		1,322	1,140,625	-	-	
Oil crops Oil, Other	1,803	17,594	1,306	16,481	8,760,000	134	1,172	
Oil crops, Other	2,400	11,223	186	10,125	2,607,143	40	104	
Olive Oil	-	4,959	4,959	5,001	9,125,000	-	-	
Olives (including preserved)	-	1,429	1,429	1,452	935,897	-	-	

Onions	-				365,998	-	-	
Oranges, Mandarins	16,322	58,424	55,481	47,508	265,133	15,500	4,110	
Palm kernels	-	-	2,503	-	-	-	-	
Palm Oil	-	24,829	87	25,071	8,746,544	-	-	
Palm kernel Oil	-	860		868	9,125,000	-	-	
Peas	23,000	9,338	3,474	708	3,518,072	8,557	30,103	
Pelagic Fish	88,462	40,051		17,505	941,935	-	-	Not considered in this study
Pepper	-	511	511	526	-	-	-	
Pig meat	47,159	100,437	100,437	54,404	2,825,218	47,159	133,234	
Pimento	-	376	376	378	3,244,444	-	-	
Pineapples and products	-	15,120	14,751	16,435	368,687	-	-	
Plantains	-	17,665	17,665	17,665	892,966	-	-	
Potatoes and products	560,000	524,673	243,010	67,637	683,641	259,372	177,317	
Poultry Meat	169,645	157,631	157,631	845	1,436,624	169,645	243,716	
Pulses, Other and products	3,200	5,506	5,619	2,515	3,471,722	3,266	11,338	
Rape and Mustard Oil	-	37,491	9,701	37,554	8,864,286	-	-	
Rape and Mustard seed	2,800	2,239	1,116	1,475	-	1,396	-	
Rice (Milled Equivalent)	-	41,291	41,288	44,014	3,662,185	-	-	
Rice bran Oil	-	-	-	-	9,125,000	-	-	
Roots, Other	-	6,625	4,617	6,673	1,089,552	-	-	
Rye and products	-	1	1	84	2,772,152	-	-	
Sesame seed	-	715	715	747	5,214,286	-	-	
Sesame seed Oil	-	303	303	306	7,300,000	-	-	
Sorghum and products	-	75,389	-	75,389	2,962,319	-	-	
Soybean Oil	-	16,561	11,775	13,688	8,600,575	-	-	
Soybeans	-	1,992	1,992	2,007	3,361,842	-	-	
Spices, Other	-	2,489	2,496	2,622	3,500,000	-	-	
Sugar (Raw Equivalent)	-	240,231	227,955	261,725	3,555,772	-	-	
Sugar beet	-	288	288	288	-	-	-	
Sugar cane	-	-	-	-	317,391	-	-	
Sugar non-centrifugal	-	-	-	-	3,386,598	-	-	

Sunflower seed	-	1,336	1,336	1,353	4,562,500	-	-	
Sunflower seed Oil	-	11,236	5,064	12,343	8,750,000	-	-	
Sweet potatoes	16,500	16,672	15,004	183	985,276	14,849	14,631	
Sweeteners, Other	16,294	32,417	29,550	71,894	2,939,597	14,853	43,662	
Tea (including mate)	-	2,363	2,363	2,541	429,412	-	-	
Tomatoes and products	94,102	124,912	115,482	38,588	194,998	86,998	16,964	
Vegetables, Other	903,665	538,071	468,094	60,687	247,980	786,142	194,948	
Wheat and products	447,800	772,035	346,544	504,169	2,939,859	201,004	590,925	
Wine	248,400	112,604	34,159	39,970	665,653	75,353	50,159	
Yams	-	207	206	207	1,020,430	-	-	
Total						14,977,123	8,360,903	
Total-considered for this study						14,541,379	8,182,684	

Table S2. 19: New Zealand kiwifruit production in terms of volume and calories

Item	**Production quantity (t) (FAO 2018a)**	**Domestic supply quantity (t) (FAO 2018a)**	**Food supply quantity (t) (FAO 2018a)**	**Imports**	**Calorific content (Cal/t)**	**Total NZ production available for human consumption globally (t)**	**Total NZ production available for human consumption globally (x10^6 Cal)**	**Remarks**
Fruits, Other	495,541	181,945	165,649	40,362	394,799	451,158	178,116	
Kiwifruit	399,947				49,000	364,126	17,842	Calorific content from Roe et al. (2015); production volume from FAO (2018b)
Fruits, Other (except kiwifruit)	95594				18415.5	8703207.291	160,274	

Table S2. 20: Comparison of global and New Zealand calorie contribution

Variable	Value	Source
NZ's contribution in global calories (%)	0.11	
CB Glo,AgFd (GtCO2-eq)	7.1	
CB NZ,AgFd4 (MtCO2-eq)	7.9	
Calorie Production NZ Ikiwi (Cal)	17842151252.0	Table S2. 19
Calorie Production NZ IApple (Cal)	126873656990.1	Table S2. 18
Calorie Production NZ Iwine (Cal)	50159262063	Table S2. 18

Table S2. 21: Carbon budgets for New Zealand agri-food systems using calorific method

Industry	Calorie contribution to NZ total calorie production (ratio)	Industry-level carbon budget (ktCO2e)	Produced for human consumption NZ (t)	Product-level carbon budget (gCO2e)
NZ Ikiwi	0.002180477	17.3	364125.5	47.6
NZ Iapple	0.01550514	123.2	292562.5	421.0
NZ Iwine	0.006129928	48.7	75353.4	646.2

Table S2. 22: Sensitivity of the results to the global carbon budget values

Parameter	Unit	Grandfathering	Economic	Agri-land	Calorific content
CB NZ,AgFd	$MtCO_2eq$	7.9	26.2	0.4	1.8
CB NZ,AgFd,Ikiwi	$ktCO_2eq$	64.5	371.5	0.4	3.9
CB NZ,AgFd,Iapple	$ktCO_2eq$	68.9	164.4	0.3	28.0
CB NZ,AgFd,Iwine	$ktCO_2eq$	57.5	726.4	1.2	11.1
CB NZ,AgFd,Pkiwi	gCO_2eq	161.2	928.8	1.0	10.8
CB NZ,AgFd,Papple	gCO_2eq	149.8	357.3	0.6	95.7
CB NZ,AgFd,Pwine	gCO_2eq	231.3	2924.2	4.8	147.0

The alternative global carbon budget value of 6.8 $GtCO_2eq$ was used for comparison (Bjørn and Hauschild 2015)

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Supplementary Material 3

a) Description of the method used to calculate the biogenic carbon in wood
b) The stock model developed by BRANZ (R Jaques, personal communication, Dec 21, 2018) (see Supplementary Material (SM) 3. b, available online)
c) Climate impacts associated with the energy consumption of the detached house building sector (see SM 3. b, available online)
d) Climate impacts of a typical New Zealand detached house (D Dowdell, personal communication, Dec 18, 2018) (see SM 3. b, available online)

Modelling of Biogenic Carbon Storage in Wood Products

It is recognised that there are several methods for quantifying biogenic carbon storage in Life Cycle Assessment; however, there is not yet a consensus on a preferred method (Brandão et al. 2019; De Rosa et al. 2018). Different methods have been reviewed recently by Brandão et al. (2019), who used three example case studies to illustrate the influence of the choice method to quantify biogenic carbon storage.

Stated that, this note describes the approach used in Chapter 5 to quantify the carbon footprint of wood used for the construction of a detached house.

Wood from sustainably managed forests

Principle

All wood that is harvested and used in products receives a credit for the amount of time that it is stored. There is no credit for the standing forest as it at a "status quo" with regards to carbon storage i.e. the amount of carbon stored in the trees on the specific land remains constant when averaged over 100 years.

Calculation

The carbon footprint value is equivalent to the climate change impact of releasing that carbon but is pro-rated to 100 years. In other words, if wood contains 5 kg of carbon and this wood is used for 50 years and subsequently burned, then the avoided carbon footprint is 5 * (44/12) * (50/100) = (-9.2) kgCO_2eq. If the wood is stored for more than 100 years, it gets the maximum carbon footprint credit which is 5 * (44/12) * (100/100) = (-18.3) kgCO_2eq.

This follows the method recommended by the European Committee for Standardisation (CEN 2014).

Representation in Building/Construction LCAs

The full credit for anticipated carbon storage in the wood is given in Module A1; in our example, it would be (-18.3) kgCO_2eq. Note that this value is calculated for the gross wood that is processed after harvesting from the forest. Any losses in processing are calculated as carbon footprint in other modules. For example, if wood containing 5 kg of carbon is harvested, and 20% of this wood becomes waste and is burned during processing (i.e. the final product contains 4 kg of carbon), then the carbon footprint value in Module A1 is (-18.3) kgCO_2eq, and the 1 kg of carbon which is burned is considered in Module A3 as +3.6 kgCO_2eq.

If the wood is not actually stored for at least 100 years, then the reduction in benefit from the originally anticipated storage considered in Module A1 is represented in Module C4. For example, if the wood is used for 50 years and then burned, the associated carbon footprint representing in Module C4 is +2.5 kgCO_2eq.

On the other hand, if the wood is sent to landfill, then there are additional methane emissions as the wood degrades. The carbon footprint of methane emissions are accounted for in Module C4 regardless of the time when they happen. That is if the landfilled wood continues to produce methane emissions in 200 years, these are still relevant for assessment. For carbon dioxide produced as a result of wood degradation in the landfill, again these emissions are relevant for assessment regardless of the point in time at which they are emitted.

At the same time, if the wood is recycled into another product at end-of-life of the current product, then the “unused” portion of the carbon storage that can potentially be credited to this future product is represented as a carbon emission from the system under study in Module D. This means that this carbon storage benefit can then be credited to the future product.

References of Supplementary Material 3

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Supplementary Material 4

Consumption- and Production-based greenhouse gas emissions of New Zealand for the year 2012, in terms of 16 economic sectors

a) For consumption-based results, see Supplementary Material (SM) 4. a

b) For production-based results, see SM 4. b

Supplementary Material 5

a) Details of different climate thresholds and methods to calculate the associated global carbon budgets.
b) Production- and consumption-based greenhouse gas emissions of New Zealand for the year 2011, in terms of 16 economic sectors (see Supplementary Material (SM) 5. b, available online)
c) Absolute climate performance of New Zealand (SM 5.c, available online)

Table S5. 1: Overview of thresholds and related assessment methods employed in the study

Threshold	Origin	Method where used	Application in practice	Interpretation of impact score
Scientific thresholds				
350 ppm CO_2 **1 Wm^{-2}**	Thresholds are based on the precautionary principle, as used in the PBs framework (Rockström et al. 2009; Steffen et al. 2015)	Planetary Boundary-based Life Cycle Impact Assessment (PB-LCIA) method (Ryberg et al. 2018b), where impact indicators express average change in atmospheric CO_2 concentrations resulting from a change in continuous annual emissions of CO_2 or its precursors (for the 350 ppm threshold) or change in radiative forcing per unit change in continuous greenhouse gas (GHG) emission (for the 1 Wm^{-2} threshold) .	Impact score is calculated by multiplying elementary flow, like emission of a GHG (in mass per time unit) by GHG-specific impact indicator calculated by Ryberg et al. (2018b). Resulting impact score is compared with a SoGCB assigned to the studied system. Full (safe) operating space (72 ppm CO_2 or 1 Wm^{-2}) is used to estimate the SoGCB assigned to the studied system according to chosen sharing principle (Ryberg et al. 2018a).	As the climate change PBs have already been exceeded, safe operating space is defined as the operating space corresponding to the difference between levels of the PB and natural (preindustrial) levels (278 ppm CO_2 or 0 Wm^{-2}) and that of.(Ryberg et al. 2018a; Ryberg et al. 2018b) Given activity or sector can still be considered absolutely sustainable. However, because it would be possible to reduce and maintain pressures associated with anthropogenic activities within the safe operating space if all other activities reduce their emissions to a level that does not exceed their assigned SoGCBs (Ryberg et al. 2018a).
Political threshold				
2 °C	Threshold reflects long-term goal based on a political agreement to limit global warming to below 2°C above pre-industrial levels.	Carrying capacity-based normalization in LCA, where normalization reference expresses the carrying capacity of a region in terms of its relative contribution to the global population. Here, the carrying capacity refers to the operating space of GHG emissions that avoid the 2°C global warming.	Impact score is calculated by multiplying elementary flow, like GHG emission (in mass) by a specific global warming potential (GWP_{100}). The impact score can be related to the operating space based on the normalization reference and compared with traditional emission-based normalization reference. Operating space (aka carbon budget) related to the 2°C threshold is calculated using different methods. One of them is projecting the allowable GHG emissions associated with the RCP2.6 emissions pathway and share it equally during the remaining time period, as was done by	This threshold is equivalent to an atmospheric CO_2 concentration of 450 ppm CO_2 and it has not been exceeded. The operating space corresponds to the cumulative emissions that drive the 2°C compared to the natural (preindustrial) level (2°C). Given activity can be considered absolutely sustainable as no emissions associated with an activity or sector will lead to exceeding the 2°C threshold in the long term (2 million years)

	Doka (2015, 2016). The other method is applying the Absolute Global Temperature change Potential (AGTP) for GHG emissions (Bjørn and Hauschild 2015). Since recently, carbon budgets reducing overtime are being calculated, which are on the GHG emissions pathways projected by IPCC or IIASA (Bjørn et al. 2018). These carbon budgets shrink rapidly throughout the 21st century.	

References of Supplementary Material 5

Bjørn A, Hauschild MZ (2015) Introducing carrying capacity-based normalisation in LCA: framework and development of references at midpoint level. International Journal of Life Cycle Assessment 20:1005-1018. doi:https://doi.org/10.1007/s11367-015-0899-2

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