Towards a better understanding of production system design and regulatory mechanisms to reduce greenhouse gas emissions: An integrated modelling approach

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Certificate of original authorship

I, Liang Yujie, declare that this thesis is submitted in fulfilment of the requirements for the award of Doctor of Philosophy, in the TD School at the University of Technology Sydney.

This thesis is wholly my own work unless otherwise referenced or acknowledged.

In addition, I certify that all information sources and literature used are indicated in the thesis.

This document has not been submitted for qualifications at any other academic institution.

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Abstract

This thesis explores how to identify and evaluate methods to reduce greenhouse gas (GHG) emissions via production system design and regulation. It theorises the design and application of an Environmentally Sustainable Residual Income Decomposition Modelling (ESRIDM) in an agricultural context. The aim is to enhance decision-making related to economic and environmental sustainability, particularly in the control and management of nitrogen emissions.

This thesis first explores how to identify and describe (categorise) alternative regulatory mechanisms for managing GHG emission reductions, deriving a comprehensive typology to categorise and describe elements of regulatory systems. Second, sets up the theoretical framework and builds an integrated modelling system to enable the identification and evaluation of alternative practices, technologies and other factors that may lead to more sustainable crop production through better nitrogen management. This research expands upon the Environmentally Sustainable Residual Income (ESRI) theory and integrates it with a Water and Economic Sustainability Performance Measurement (WESM) framework.

Third, the modelling is extended by integrating insights inspired by the co-production of knowledge theory, which highlights co-production's role in decision-making and emphasises component interactions between sustainable drivers and industry-level measurements. This modelling is further enhanced by integrating with a simulation model, Decision Support System for Agrotechnology Transfer (DSSAT). Together this modelling system has been designed to enable evaluations and outcomes of different regulatory options by using a combination of integrating economic modelling, accounting mechanism and agriculture system modelling.

The significance of this research can be articulated through three ways, this is one of first studies to model a practical linkage pertaining to the ESRI theory. Through a detailed case

study and comparative analysis of three nitrogen fertilizer options, the efficacy of the ESRIDM is demonstrated. This model aids in identifying key issues that require attention of industry stakeholders and offers an analysis of probable outcomes under various alternatives. Second, this study situates the simulation and evaluation of production system options, fundamentally based on science modelling, within an environment characterized by robust stakeholder linkages akin to the co-production of knowledge theory. It offers a novel approach to enhancing the efficiency and quality of interdisciplinary communication among diverse specializations that often arise in cross-disciplinary research. Third, this study creates an integrated modelling system designed to be adapted with a view to contributing to the evolution of more sustainable systems of production thereby furnishing new insights that potentially render the simulation and evaluation of future policies more time- and cost-efficient.

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Chapter 1 Introduction

1.1 Meta question and thesis objectives

The broad objective of this study is to develop an integrated model and method to enable the identification and evaluation of alternative ways to achieve GHG emission reductions through production system[1] design and regulation.

The overarching research question this thesis addresses is: *How can we identify and evaluate better ways to achieve GHG emission reductions through production system design and regulation?* The setting for the research is the Australian cotton production industry, specifically Australian primary producers and stakeholders within the production supply chain.

The meta question is based on two main challenges.

First, further exploration is warranted in the domain of systematic evaluation for environmental policies, particularly concerning environmental regulations with a focus on emissions (cf. Li et al., 2013; Bilotta, Milner & Boyd, 2014; Runhaar, Driessen & Uittenbroek, 2014; Bellou et al., 2017; Gao et al., 2018; Mardani et al., 2019). This avenue of research is essential for enhancing the robustness and comprehensiveness of regulatory frameworks. Existing research spans climate modelling (Lesk et al., 2016), current climate analysis (Zhou, 2015), environmental regulation improvement (Landis et al., 2019), and regulation evaluation methods (Finkelshtain & Kislev, 1997). However, until now, the emergent field of systematic evaluation theory for the effectiveness of different environmental regulations is incomplete, partly due to limited work

[1] "Production system" in this thesis is defined as the set of interdependent factors comprising a supply chain relating to the production of a good or service (c.f. Parnaby 1979; Holzworth et al., 2015; Yin, Stecke & Li, 2018). The focus in this thesis is related to the primary production of agricultural cropping industry, with a further emphasis on the cotton cultivation industry in Australia.

synthesising evaluation theory with economic and environment modelling and with related regulation options[2].

Second, the regulation of GHG emissions in agriculture is relatively silent with regard to the policymakers in general. As reported by IPCC (2014, p. 28), agriculture is the second largest sector contributing to global GHG emissions. The impact of agriculture on the environment is likely to increase, with challenges such as growing population and rising sea levels being key drivers (Kastner et al., 2012). Despite agricultural and land use being the second largest source of GHG emissions, there is little evidence that current attempts to regulate this sector are likely to achieve the emission reductions necessary to reverse the worst effects of climate change (IPCC, 2014).

Accordingly, the specific objectives of this thesis are to:

1. develop and explore a general model to identify and describe (categorise) alternative regulatory mechanisms for managing GHG emission reductions;
2. develop and explore an integrated model to understand and evaluate the economic and environmental performance of proposed GHG emission regulation options, in the context of agricultural production; and
3. develop and explore a method to identify, evaluate and evolve plausible production system options amenable to GHG emission regulation in the context of agriculture production.

In this thesis the terms 'regulatory mechanism', 'regulatory option' and 'production system option' have a meaning which relates to the concept of regulatory assemblages. Assemblage theory is a framework for analysing social complexity put forward by Gilles Deleuze and Félix Guattari (Jordan, 1995). Assemblage theory emphasises the fluidity, interchangeability and multi-function characteristics of components forming a complex system; the relationships of each component in complex systems are not immutable

[2] As detailed below, authors such as Liu, Mao, Tu & Jaccard (2014) go some way in developing systematic evaluation theory.

and relatively independent and replaceable (DeLanda et al., 2016). It is beyond the scope of this thesis to consolidate and reconcile different approaches to characterising assemblages, such as the systems vs. packages debate in management control systems assemblages (cf. Grabner & Moers, 2011). Accordingly, this research uses the term "regulatory assemblage" to refer to any group, package or system of regulation that contains multiple regulatory mechanisms. This is further explained and demonstrated in Chapter 2.

As previously elucidated, the term "regulation assemblage" refers to the amalgamation and synthesis of various regulatory mechanisms. Consequently, the phrase "regulatory option" is employed to delineate the specific regulation derived from such an assemblage which is under consideration. For example, in Chapter 3, the tree planting strategy is selected and posited as a regulation option, with its performance assessed in terms of both sustainability and economic viability.

Delving into more specific categorisations, a regulatory option is derived from consideration of a myriad of production system options. These options, encompassing factors such as technological choices, supplier decisions and on-farm strategies, significantly shape the sustainability and economic outcomes. However, not every element among these has a pivotal role in determining modifications to the regulation option. Despite this, the simulation and assessment of these production system alternatives are indispensable. They form a crucial component of the informational framework that guides adjustments in regulation options (Baron, 1988; Le Pira et al., 2017). In Chapter 4, this procedure is explored, commencing with the formulation of three varied nitrogen management scenarios and culminating in an in-depth simulation and analysis of results.

1.2 Motivations

There are three key motivations for this thesis.

Motivation One

Motivation One is the lack of consensus in the literature on how best (highest or significantly improved sustainable value in all plausible options) to regulate GHG emissions. Despite many years of public and academic scrutiny, there are still uncertainties and widespread debate on how best to regulate GHG emissions. Key issues include (i) how best to enable the optimisation to both reduce GHG emissions and enable economic prosperity (Goodstein et al., 2014; Lotjonen and Ollikainen, 2019); (ii) lack of uniformity, clarity and validity in accounting and other calculation mechanisms, as well as weak enforcement which may reduce regulation implementation effectiveness (Hashmi, 2008; Freestone and Streck, 2009; Foster et al., 2017); (iii) questions about the generalisability of emission reduction policy coverage and fairness across different industries and countries (D'autume & Schubert, 2016; Xie, Yuan & Huang, 2017; Wang et al., 2019); and (iv) the way regulation is implemented, affecting the outcomes from regulatory changes (Freestone and Streck, 2009; Bruvoll and Larsen, 2004; Goulder, 1995; Nilsson, 2009; Lotjonen & Ollikainen, 2019). Despite the emergence of systematic evaluation theory by authors such as Liu et al. (2014), it is not clear how best to regulate GHG emissions in many contexts.

Motivation Two

The second motivation is the lack of theoretically sound and valid models that integrate knowledge and stakeholder domains to enable the evaluation of regulation alternatives in the context of GHG emissions. One of the key challenges of policy assessment is related to disciplinary framing, in that the factors selected in regulation evaluation are often limited, which may lead to the absence or neglect of other key factors in the evaluation. For example, economic vs. social vs. environment[3]; scientific vs. economic

[3] The *Garnaut Climate Change Review* is criticised by corporations such as the Australian Chamber of Commerce and Industry (2008) for its economic impact, while multiple environmental organisations criticise it for being too weak on the reduction level (Lawson, 2012; Australian Conservation Foundation, 2008).

factors[4]; macro vs. microeconomic considerations[5]; policy maker paradigms; and political party or faction preferences[6]. While many of the factors and ways of evaluating them may be valid in certain circumstances, there is limited development and adoption of models that integrate relevant knowledge domains in a comprehensive evaluation (cf. Ahmed & Ozturk, 2016; D'autume & Schubert, 2016; Stiglitz, 2019). For example, even large policy projects, such as environmental tree planting projects, are limited by not considering the difficulty of natural ecosystem reproduction and harm related to afforestation, such as the vulnerability facing pests and diseases, higher risk of forest fire and loss of soil nutrients and natural run-off and possible encroachment upon arable farmland or other value due to afforestation initiatives (Farley et al., 2005; Brockerhoff et al., 2008; Cao et al., 2010; Freer-Smith et al., 2019; Wang, Pedersen & Svenning, 2023).

One way to address some of these challenges is by integrating different knowledge and stakeholder domains in the measurement and calculation of cost-benefit for different regulatory options such as the rationality of public policy discussed by Kuruvilla and Dorstewitz (2009). In addition, several examples from the literature which demonstrate the plausibility of this approach, albeit in adjacent settings. For example, Hadjimichael et al. (2016) built up an integrated modelling system, comparing four measures of the urban wastewater system with a 'no action baseline' to discuss the reduction of the overall environmental impact of the Eindhoven urban wastewater system (UWS). They

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[4] Though many scientific publishers tried to solve agriculture emissions from chemical and biological perspectives (cf. Lotjonen & Ollikainen, 2019; Hasler et al., 2019), there are limited applications in public policy (ACIL Allen Consulting, 2019).

[5] Mardones and Lipski (2020) analyse agricultural emission reduction and its impact on the macroeconomy and put forward the view that only regulation adjustments at the farm level may not have a large impact on overall emission reduction but would need cooperation in other related fields.

[6] Kousser and Tranter (2018) suggest that political differences have created obstacles to solving the problem of global warming; Australian voters are responsive to political leaders' stances on climate change policy. When political leaders diverge in opinions, the polarisation of voters increases and vice versa; when consensus can be reached by party leaders, it is usually considered to be beneficial in overcoming the stalemate in climate policy.

illustrate how taking an integrated modelling approach enabled a more valid, reliable and comprehensive evaluation of a physical system. Pham et al. (2020) designed and tested a Water and Economic Sustainability Performance Measurement (WESM) model that combines simulation modelling, accounting and other data to solve the problem of the Environmental Performance Measurement System (EPMS) framework, applied in agricultural settings to support management decisions. Their integrated model explains how management decisions influence economic and environmental sustainability, and they provide a reliable quantitative reference for decision-making in social problems. They provide an example of multidisciplinary approaches to the measurement and evaluation of alternative decision options for agriculture practices, which may be adapted to evaluate different regulatory options. Accordingly, there is an opportunity to develop and evaluate integrated models to identify more optimal ways to regulate, in order to achieve GHG emission reductions.

Motivation Three

While the challenges presented by the regulation of GHG emissions apply to a range of settings, one setting is particularly salient to the research objective – the regulation of GHG emissions in agriculture. Agriculture provides food and fibre to the world and plays an important role in economics (FAO, 2017). However, it also is a key contributor to climate change, via direct and indirect GHG emissions and land clearing (IPCC, 2014). Lots of work has been done to better understand this sector and how to regulate it (including theoretically); but despite these attempts, uncertainties remain about how best to regulate it in this context (Ivec & Braithwaite, 2015; Olesen & Bindi, 2002; Snyder et al., 2009). One of the most fruitful endeavours in agricultural research has been the building of integrated simulation models, such as the Decision Support System for Agrotechnology Transfer (DSSAT)[7] which consolidates diverse knowledge

[7] The DSSAT model was designed to allow the users to simulate crop growth using a climate database that already stored or allowed the input of firsthand data of water and soil data sets (Jones et al., 2003).

about crop production. Despite being able to evaluate the effect of key decisions on farm performance (cf. Pham et al., 2020), these models have not been augmented by the key factors that are salient to the identification and evaluation of regulatory options. Despite there being substantial headway on how to implement co-production, until now there are no models which integrate key aspects of identifying regulation options with models such as crop simulations and economic modelling of micro and macro effect scenarios (Polk, 2015). In addition, Polk (2015)[8] suggests that the co-production of policy options is more likely to lead to better outcomes for key stakeholders.

1.3 Method and key findings

In this thesis, an integrated model is proposed and developed to aid in the identification and evaluation of GHG emission regulations. This model assimilates various approaches from distinct knowledge domains, employing a mixed-methods overarching research methodology. Chapter 2 develops a comprehensive typology to describe elements of the current regulatory systems. This typology is then evaluated against a set of extant regulatory systems. The theorised typology is based on a comprehensive literature review, and the evaluation is based on data collected from secondary archival sources. The findings indicate that the typology has some validity in describing key elements of complex regulatory assemblages, which are the sets of regulatory mechanisms implemented as part of a regulatory system.

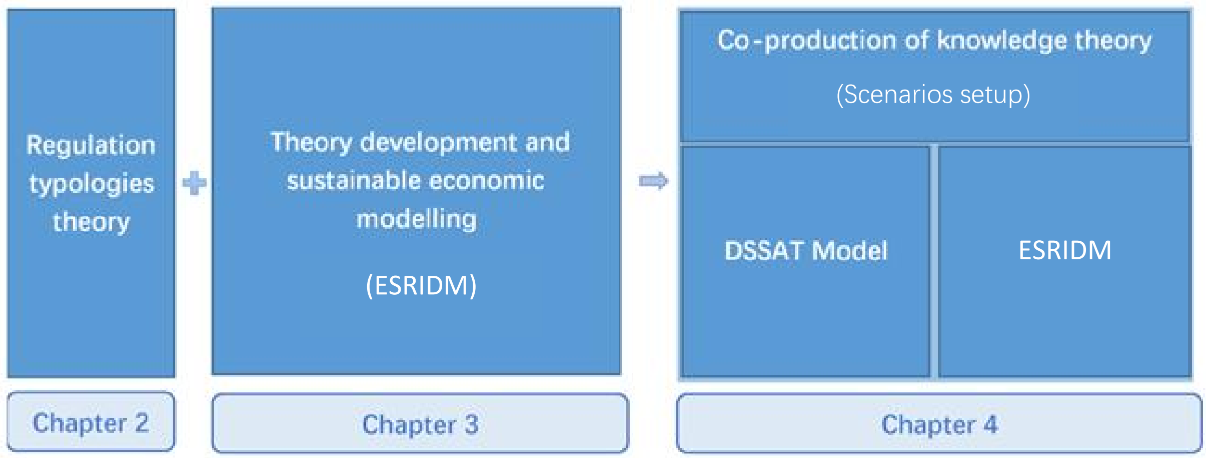
Chapters 3 and 4 develop an integrated modelling system to conceptualise the identification of regulatory options. This involves the establishment of scenarios

In addition, there are also options for evaluation of different crop management practices, like fertiliser and water options over the years (Jones et al., 2003).

[8] For brief explanation, co-production of knowledge theory (Polk 2015) is a theoretical framework in transdisciplinary research which designed for sustainability topic which involved in multiple stakeholders from different background and expertise. Detailed explanation is conducted in Chapter 4 Section 4.2.1

through the collaborative production of knowledge, as well as the testing and evaluation of scenario outcomes through simulations. This scenario and simulation set-up is based on scientific software simulation and used to test potential choice options with both economic and environmental outcomes. Figure 1.1 shows the theoretical and model structure for this thesis.

Figure 1.1: Theoretical model for this thesis



Chapter 3 predominantly focuses on objective two of this thesis, which is to develop an integrated modelling system to enable relevant and reliable evaluation of these emission regulation options. To model the potential for, and outcomes from, different regulatory options affected by different farm operation choices, it was first necessary to build an integrated model of nitrogen emissions in the context of Australian cotton farming. In the first half of Chapter 3, the Environmentally Sustainable Residual Income Decomposition Modelling (ESRIDM) builds upon Pham et al.'s (2020) WESM Model in two primary ways: first, it integrates a nitrogen management framework, and second, it refines the apex indicator (terminal output) by amalgamating the concept of Environmentally Sustainable Residual Income (ESRI). ESRI gauges the sustainable economic viability and trajectory rooted in ecosystem neutralisation and its load-bearing capacity. Initially posited by Brown (2016), ESRI, as delineated in Chapter 3, is employed to scrutinise the sustainable potential of conventional agricultural practices with chemical fertiliser, especially nitrogen fertiliser, as the main method to boost yield.

The second half of Chapter 3 presents a case study that presents (i) a quantitative analysis of emissions from nitrogen fertiliser usage; and (ii) an analysis of production to diffusion into the biosphere, drawing on the methodology used by Patzek (2004). Additionally, the potential and efficacy of emission offsets through tree planting are explored.

The main findings related to Chapter 3, based on the calculation of ESRI value using literature data, are that cotton production is not sustainable without the environmental sustaining cost (ESC) being covered by individuals or organisations. Furthermore, using a case study of tree planting, the findings suggest that offset can be costly compared to direct adjustments in farm nitrogen management and application. In addition, a more generalised environmental and economic modelling approach is assumed and analysed in the context of Pearce (1976), specifically the capacity of nature's ability to recover.

Based on the theoretical modelling presented in Chapter 3, Chapter 4 extends the design of the integrated modelling system, by implementing elements of co-production and crop simulation.

To construct emission and economic scenarios, it was essential to first acquire stakeholder insights, which for this thesis was obtained from the Australian Cotton Conference held in 2022[9] The stakeholder analysis used an approach influenced by applying the co-production of knowledge theory (see Polk, 2015). Next, scenario data was integrated into the DSSAT model to measure cropping production emission and yield calculations, which were subsequently used in the ESRIDM analysis. This approach facilitated the ability to identify several possible options for regulatory measures, or to include the analysis into regulatory assemblages for future research. The results derived

[9] A co-production desk review was conducted during the 2022 Australian Cotton Conference (held since 1982). Being held every two years by the Cotton Research and Development Corporation (CRDC), Cotton Australia is the largest cotton industry conference platform. Participants include a broad range of growers, researchers, business organisations and representatives, plus other key stakeholders in the industry, providing an ideal chance to conduct desktop research and contact key stakeholders.

from DSSAT were amalgamated with accounting and emission pricing data pertinent to agricultural activities. These data were subsequently refined and recast into financial terms, drawing upon the Environmentally Sustainable Residual Income (ESRI) theory, to compute the monetary value associated with environmental impact.

1.4. Contribution

This research aims to makes four key contributions to theory and practice.

First, this research developed a novel categorisation model that incorporated a wide set of factors for evaluating the relative merits of different emission regulations and regulation assemblages for emission reductions. This model of emission regulation assemblages for carbon reductions has been designed to assist with explanation of the relative efficiency and effectiveness of regulations before they are deployed. Accordingly, this research builds from and extends Karp and Gaulding's (1995) work in motivational underpinnings of command-and-control, market-based and voluntarist environmental policies, and the Organization for Economic Cooperation and Development (OECD) classification based on the tightness of governmental control. The findings will be of interest to the academic community, as well as being of practical interest to policymakers, farming communities and their stakeholders.

Second, this research, outlined in Chapter 3, contributes to the literature by exploring and enhancing the framework of Environmentally Sustainable Residual Income (ESRI). This research extends the work of Pearce (1976), Richard (2012), Rambaud and Richard (2015) and Brown (2016) by proposing a method to enable the evaluation of which emission regulation is more suitable under different circumstances.

The ESRIDM integrated modelling build up in Chapter 3 explores and fills the gap in the methodological application of ESRI theory and demonstrated through a case study. In addition, the ESRIDM integrated modelling level structure which builds off Penman

(2003) decomposition analysis modelling extent Penman (2003) by brings the economic analysis of regulation options and production system options into an integrated level.

It also extends the work of Pham et al. (2020) on the integrated modelling design to achieve the adaptation, combination, and improvement with current and assist with future regulation options evaluation, supported by economic, agricultural and accounting theories.

Third, the research presented in Chapter 4 theorises a novel integrated modelling system to enable the identification and evolution of more sustainable production systems, and illustrates the model in context of nitrogen management options at the farm level. The highlight of this contribution is to amalgamate the innovative integrated modelling system in Chapter 3 with a co-production of knowledge like framework. This synthesis proposes a methodology for establishing an environment conducive to the identification of sustainability improvement challenges within production systems, alongside facilitating continuous and timely feedback.

Furthermore, stakeholders were collaborated with to delineate scenarios using an approach inspired by co-production of knowledge (Polk 2015) with a view to furnishing scientific evidence to bolster various simulation scenarios via DSSAT. This research builds off and extends the work of Pham et al. (2020) and other integrated modelling research on sustainable production and evaluation (cf. Hofkes, 1996; Belcher, Boehm & Fulton, 2004; Shen, Kyllo & Guo, 2013; Chami & Daccache, 2015; Hadjimichael et al., 2016).

Four, this research contributes to transdisciplinary research by combining several knowledge domains and practices to address what is both a theoretical and practical challenge. This research combines knowledge from the fields of ecological economics, agriculture, sustainable production, among other domains. Furthermore, it integrates theories of regulation and policy; co-production of knowledge (Polk 2015); crop science; and management accounting. The work is transdisciplinary in the tradition of ecological economics, as characterised by Costanza and King (1999) who define ecological economics as “the intellectual culture which compromises different disciplines to solve complex problems in an integrated way” (Costanza & King, 1999, p.2).

In addition, Hansson and Polk (2018) explain the co-production of knowledge perspective as “the integration of different knowledge sources provided by participants in different disciplines and their links to context” (Hansson & Polk 2018, p.135). By taking these approaches, this study contributes by utilising an integrated modelling approach to synthesise and transcend the component knowledge domains in a transdisciplinary way, specifically for the construction of a knowledge construct that has a public interest aim.

1.5. Structure of the thesis

The structure of this thesis follows.

Chapter 2 addresses the first objective, namely, to develop and explore a general model to identify and describe (categorise) alternative regulatory mechanisms for managing GHG emission reductions. This chapter develops a comprehensive typology to categorise and describe elements of regulatory systems.

Chapter 3 addresses the second objective of this thesis, namely, to set up the theoretical framework and build an integrated modelling system to enable the identification and evaluation of alternative practices, technologies and other factors that may lead to more sustainable crop production through better nitrogen management.

Chapter 4 begins by constructing a theoretical model inspired by the co-production of knowledge theory, which highlights co-production’s role in decision-making and emphasises component interactions between sustainable drivers and industry-level measurements. In addition, this chapter addresses the second and third objectives of this thesis, namely, to implement and evaluate regulation options through an integrated model system based on the scientific model – Decision Support System for Agrotechnology Transfer (DSSAT). This chapter further enhances the integrated model used in Chapter 3, providing evaluations and outcomes of different regulatory options by using a combination of integrating economic modelling, accounting mechanism and agriculture system modelling.

Chapter 4 begins by constructing a co-production of knowledge theory framework that highlights co-production's role in production system options identification and emphasises component interactions between sustainable drivers and industry-level measurements. Three different production system option scenarios were decided on, based on information gleaned from a desk review and interviews with stakeholders such as growers, researchers and suppliers attending the Australian Cotton Conference in 2022. Next, this chapter addresses the second and third objectives, to implement and evaluate regulations through the integrated model system based on the science model (DSSAT). This chapter further enhances the integrated model developed in Chapter 3 with co-production to design different scenarios and simulation software, to provide scientific-based cropping data. Through these processes, Chapter 4 evaluates the outcomes of different nitrogen management options in three scenarios, including economic modelling, accounting mechanism, agriculture system modelling using DSSAT, and climate modelling weather data.

Chapter 5 of this thesis is comprised of five sections. The first to third sections encompass a comprehensive summary and recapitulation of the implications derived from this research. Subsequently, the fourth section provides a concise overview of the contributions made by this study. The last section is dedicated to delineating the limitations and delimitations encountered during the research process and potential avenues for future research.

Chapter 2 Categorising Emission Regulations

2.1 Introduction

2.1.1 Background

Australia government has set up a net 2030 emissions target of a 43% reduction below 2005 levels and net zero emissions by 2050 in legislation (Prime Minister of Australia, 2022). To attain this objective, Australia necessitates a spectrum of adjustments spanning from industrial production to economic policies. Moreover, as per the public disclosures from the 2023 Annual Progress Advice Report, Australia's greenhouse gas emissions amounted to 467 million tonnes by June of the year 2023, marking an increase of 4 million tonnes from the previous year (Climate Change Authority, 2023). This lag in adjustment response underscores the urgency for modifications related to sustainable production in emissions reduction efforts (Climate Change Authority, 2023; Byrom, Bongers, Dargusch & Garnett, 2023).

Facing this situation, some academies are of the opinion that no effective action will be taken unless there is clear guidance for relevant policies, including the management evaluation method, accounting method, degree of disposal and explicit provisions of carbon emission limits of enterprises (Sydney University, 2020). This illustrates the importance of formulating, improving and evaluating rules and regulations on emission reductions.

2.1.2 Motivations and objective

Despite the emergence of a systematic evaluation method by authors such as Liu et al. (2014), it is not clear how to best regulate greenhouse gas (GHG) emissions. Although there exists the necessity and urgency to optimise emission reduction regulations, a key challenge for "how to identify what regulatory options may or may not be more efficient in certain circumstances" is the lack of a well-developed typology of regulatory options[10].

Accordingly, the motivation for Chapter 2 that current literature lacks well-established typologies for regulatory assemblages. These ambiguities in category can result in uncertainties in foundational definitions, potentially diminishing the accuracy of quantification when comparing various system combinations.

Alongside, the objective of Chapter 2 is to address the absence of a comprehensive typology to describe and analyse different regulatory options, and to categorise and describe elements of regulatory systems.

To achieve this classification of regulations, Chapter 2 develops a comprehensive typology to categorise and describe elements of the current regulatory systems and evaluates this typology against a set of extant regulatory systems. The theorised typology is based on a comprehensive literature review, and the evaluation is based on data collected from secondary archival sources.

2.1.3 Contributions

This thesis provides one of the first studies to theorise and incorporate a wide set of factors for evaluating the relative merits of different Emission Regulation Assemblages for Carbon Reductions (ERACRs). Accordingly, this chapter builds off and extends the work of Karp and Gaulding (1995), dividing emission regulations into three categories that include Command-and-Control, Market-Based, and Voluntarist Environmental Policies. In addition, this thesis builds on the Organization for Economic Cooperation and Development (OECD) classification, based on the tightness of governmental

[10] To be specific, topics such as the following have been discussed in recent years: (i) how best to enable the optimisation to both reduce GHG emissions and enable economic prosperity (Lotjonen & Ollikainen, 2019; Goodstein et al., 2014); (ii) how to create uniform accounting and calculation mechanisms reducing overly accommodating regulation implementation (Hashmi, 2008; Freestone & Streck, 2009; Foster et al., 2017); (iii) uncertainties in the generalisability of emission reduction policy coverage and fairness across different industries and countries (d'Autume & Schubert, 2016; Xie, Yuan & Huang, 2017; Wang et al., 2019); and (iv) how best to design and implement emission reducing regulatory measures (Goulder, 1995; Bruvoll & Larsen, 2004; Freestone & Streck, 2009; Nilsson, 2009; Lotjonen & Ollikainen, 2019).

control. Regulation categories have undergone updates. These updates include refining regulation types, such as education and campaigns, into the information/education categories. A detailed division for broad economic instruments has been established, incorporating Karp and Gaulding's (1995) approach to tightness control. When analysing the command-and-control regulation unit, the foundational theory is based on the principle of environmental regulations. This principle typically mandates public policies to either limit or suspend harmful activities, known as performance-based regulations, or to introduce mitigation measures that curb negative side effects, termed technology-based regulations (Percival et al., 2013).

The establishment of regulatory categories serves as a framework for the identification, analysis and recombination of regulation assemblages. This provides the scope and classification for modular frameworks essential for further integrated modelling of these regulation assemblages. Consequently, in the ongoing study of emission regulation assemblages, Chapter 3 builds upon the categorisations presented in this chapter. It further deconstructs the regulation assemblage to its fundamental component, which is the regulation option, and conducts integrated modelling and decomposition analysis for several of its potential configurations.

2.1.4 Structure

The structure of this chapter follows.

First, this chapter examines the necessity of regulations when working towards improved social welfare and environmental protection outcomes.

Next, this chapter draws on a theoretical classification model for different emission regulations drawing from the literature.

Finally, this chapter analyses the possibility of regulatory assemblage.

2.1.5 Link to Chapter 3 and 4

Figure 2.1 is composed of four primary elements. The two large hexagons represent the "production system and integrated modelling" and the "regulation assemblages model" respectively. The purple dashed quadrilateral symbolises the co-production of knowledge process centred around the production system. A more detailed exploration of this component is provided in Chapter 4. The blue dashed quadrilateral symbolises the co-production of knowledge process anchored in the regulation assemblages. While this part is preliminarily explored in this chapter, it is not thoroughly implemented or delved into.

Figure 2.1: Co-production of regulation assemblages and production system options

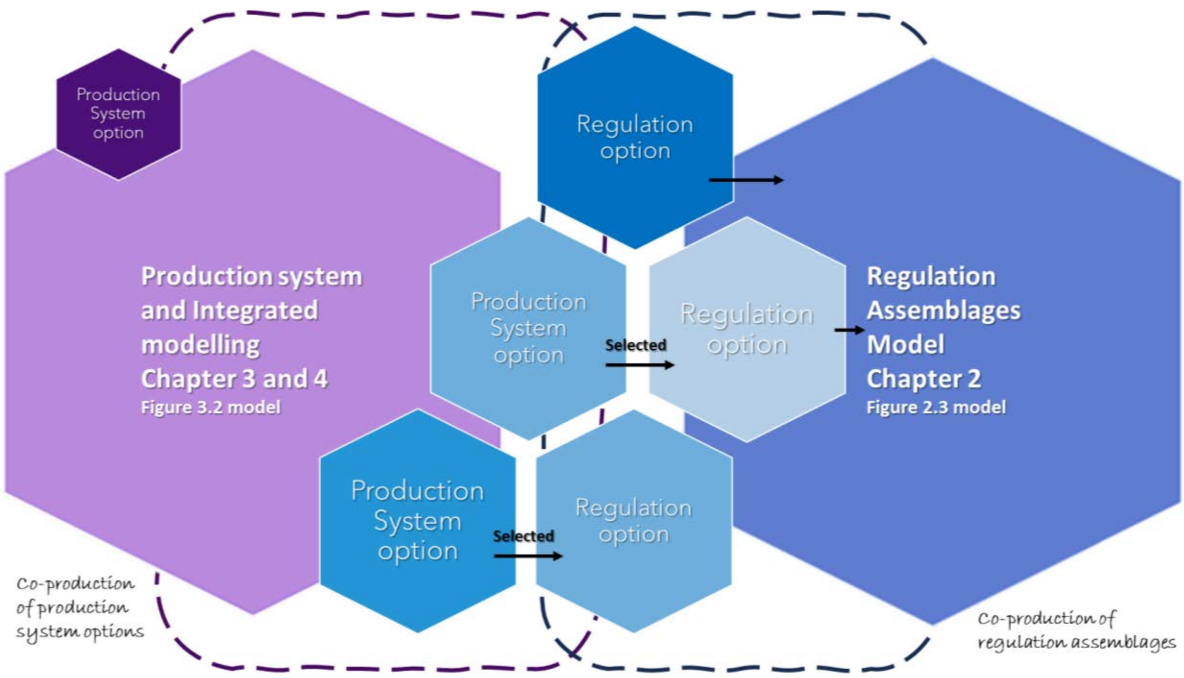


Figure 2.1 shows the integration between the production system and regulation assemblages used in this thesis. From left to right, certain production system options, either vital to production or the environment, are chosen to be modified and then regulated into regulation options, exemplified by the blue hexagons in the figure. Conversely, some production system options might not be selected, due to reasons such as being non-hazardous or related production technologies not being widely

adopted, as depicted by the dark purple hexagons in the figure. Correspondingly, there are instances where certain regulation options pre-exist or are unrelated to production options, such as policies related to sulphur compound emissions in contrast with algal biofuel production systems, represented by the dark blue hexagons.

Figure 2.1 illustrates the connection between the regulation assemblage in Chapter 2, the regulation options in Chapter 3, and the production system options under co-production feedback loop in Chapter 4. Chapter 1 showed that the selection, evaluation and production system options analyses are important for determining regulation options and assemblages. This conceptual framework is modelled and expounded upon in Chapters 3 and 4. Nevertheless, the co-production aspect of the regulation assemblage remains a subject for future inquiry. It should be noted that the analyses within Chapters 3 and 4 lay the groundwork for more advanced research into the regulation assemblage in future endeavours.

2.2. Literature review and theory development

This section provides an overview of the relevant literature on key emission regulation categories[11] to theorise a more comprehensive typology.

2.2.1 Social contract and regulation

The origin of social rules can be traced back to Plato's Utopia, Montesquieu's *On the Spirit of Law*, and Rousseau's social contract theory.

The significance of regulatory intervention has been posited for an extended period. Plato's theory on Utopia (380 BC) discusses the philosophical origin of the political and

[11] This is shown in two ways, first in following Section 2.2.2 to 2.2.4, reviews emission regulation categories methods and second in Section 2.3 further listed the selection rational of chosen emission regulation categories..

social system which he argued the community needed in order to be managed, so that it can be orderly and powerful. Plato's arguments raise the thoughts of the three waves, demonstrating the necessity of regulations and also raising ideas for community improvement from a philosophical perspective.

Furthermore, emission regulations should evolve in tandem with advancements in climate science and shifts in the current climate conditions. The formulation of effective policies necessitates consideration of the specific geographical and climatic factors of the region in question. Montesquieu argued that the geographical environment influenced the human social system and emphasised the construction and adjustment of regulations according to local conditions. Montesquieu's concept of geographical environmental determinism is not to be misconstrued as fatalism. At the beginning of Chapter 14 in *The Spirit of Law* (1748), the relationship between law and climate are articulated specifically: "If the spiritual temperament and inner feelings are greatly different due to different climate, the law should be different from these feelings and the differences of these temperaments." Montesquieu argued that climate and other geographical factors indirectly affect and determine the formulation and implementation of legal and social-political systems by influencing people's thoughts and temperament; that is, legislators should try to reduce the negative impact of climate and geographical factors or make targeted adjustments in regulations, rather than directly copying the successful legislative cases of other non-comparable regions. This indicates the importance of regional flexibility of regulations.

The rational enforcement of regulations is essential. Rousseau explained why regulations are needed to maintain freedom and put forward the “shackles of freedom” (1762). Although the concept of “forced freedom” gives birth to despotism, the pursuit and purpose in his idea of the law stated in the “general will” is emphasised in his thought: “If everyone wants to dedicate himself to all, he does not dedicate himself to anyone.” It is equivalent to “if everyone legislates from the perspective of the general will of the state, then the law applies to that person and to everyone”, where the general will is not only the will of the state but also each individual. Some might argue that Rousseau's assumption of the uniqueness of interests for general will is limited but instead the public interest itself is pluralistic in practice (Bertram, 2020). However, that still does not affect his most important argument that the existence of correct regulation is the guarantee of free will.

However, systems related to emissions are inherently complex. Singular regulatory approaches, such as carbon taxes, or more market-oriented policies, like carbon trading, may not be suitable for all emission problems. Consequently, assessing their effectiveness is also intricate. American sociologist Cooley (1918) argues that the social system is complex and comprises a large number of norms which are formalised by society (in legal form) to meet its needs. This view has been inherited and accepted by many sociologists.

2.2.2 Alternative ways to classify regulatory mechanisms

An important focus of this chapter is on the design characteristics classification approach, due to the objective of the research being to focus on developing and evaluating a comprehensive typology to categorise and describe elements of the regulatory system, to identify a more optimal regulatory design and achieve a regulatory outcome.

Since the establishment of environmental economics as a new branch of economics in the 1950s and 1960s (Pearce, 2002), there has been ongoing discussion in the literature clarifying theoretical frameworks for classifying environmental regulations. A review of the literature reveals there are two key means by which to classify environmental regulations. First, as exemplified by research papers such as Karp and Gaulding (1995), Taylor et al. (2012) and Xie et al. (2017), some authors focus on classifying regulation based on achieving regulatory goals. These authors tend to classify regulations based on key design characteristics. The other is based on the political purpose (or intent) of regulation generation (see Table 2.1 for a summary). However, this classification approach will not be used in this chapter.

Table 2.1: Summary of examples for other key types of political purpose classification

Type	Explanation
Greenwashing	That is to say, it meets the requirements of environmental protection, but in fact only covers a layer of environmental protection camouflage on the policy. However, some policies may initially be pseudo-environmental, but with the development of policies and the participation of policy roles, may become policies with real environmental protection content.
Piggybacking	That is to say, the main purpose of the policy is not environmental protection, but some environmental protection contents are attached intentionally or unintentionally.
Mainstreaming	This refers to integrating environmental objectives into public policies more seriously, comprehensively, beneficially and transparently. The government often works with environmental organisations to promote the mainstreaming of environmental policies, such as the UK's wind power policy. On the one hand, it aims to develop green energy; on the other hand, it aims to achieve the long-term goal of energy decentralisation and reducing fossil energy dependence. This is the mainstream direction of efforts in the field of environmental policy.
Green streaming	That is to say, through the efforts of the government and society, we should radically and thoroughly turn the current policy into one that fully meets the environmental requirements. This type is rare.

Source: Modified from Fitzpatrick (2011)

2.2.3 Classifying regulatory mechanisms

A review of the literature identified a broad range of the ways regulation has been classified using the design classification approach (see Tables 2.2-2.4). The comprehensive typology in this chapter extends the work of Karp and Gaulding (1995) which provides a ternary classification based on the effectiveness for motivation of organisations which are: command and control, market-based approaches, and voluntarist policies; and the OECD classification based on the tightness of governmental control: command and control, market incentive and voluntarist projects. Taylor et al.

(2012) developed a broad typology aiming to answer the question of ''what, how and why'' certain environmental regulations work/do not work, based on the emission regulations used by the UK government till 2012. In this typology, Taylor et al. (2012) listed and analysed interviews of policy stakeholders and used qualitative analysis to classify the characteristics of the contents, to give a comprehensive typology. Taylor et al. (2019) extended the classification typology to include more detailed indicators, such as risk management, while providing a broader view of the influence of each environmental regulation type.

The three types of emission reduction regulations in the table all have limitations when implemented alone (Liu et al., 2014), and highly depend on the local situation (Ren et al., 2018). According to the research on emission reduction in China's iron and steel sector, “no single economic incentive or command-and-control instrument is overwhelmingly superior” (Liu et al., 2014, p.140). Accordingly, it is not reasonable to assume “one regulation fits all”.

Table 2.2: Environment regulation classification

Regulation type	Command and control	Economic incentive instruments	Public participation/persuasion strategies
Description	Command and control are the traditional strategies to mitigate environmental problems. To achieve environmental mitigation through compulsory means and unified standards, such as making relevant laws, certification or direct bans are required to force organisations to change their behaviour (Tietenberg, 2006). There are two common ways of execution: performance-based regulations which aim to reduce the damaging actions and technology-based regulations which require actions to be done to reduce the damage and are hard to avoid.	When the application of a means is enough to affect the economic parties to evaluate the costs and benefits of alternative actions, the means can be named as 'economy' (OECD, 1994). The economic incentive instruments play the role of reducing pollution caused by human activities and guide emission enterprises to change their behaviour spontaneously through the pricing rules. There are generally several forms of carbon pricing, such as the carbon tax, the cap-and-trade or the baseline-and-trade system. In this, the top two keyways are the carbon tax and emission trade. In addition, there are two common ways in execution, which are upstream or downstream, that is, to charge for raw materials or final products of polluting items. Other less common options are deposit-refund schemes, environmental liability insurance, and so on.	The public participation/persuasion strategies refer to the regulations that the government uses to guide the public to change the cost-benefit value or environmental ethics through leading public opinion, consultation and exhortation, moral preaching, citizen participation and other non-mandatory means, so that people may prefer to take voluntary actions to improve environmental quality and finally achieve the goal of environmental governance (OECD, 1994). The two most common complementary strategies are information/education and voluntary participation.
Characteristic	1）The policy of command and control has the nature of mandatory execution, which forces polluters to internalise the cost of pollution through laws and compulsory regulations. 2）The policy of command and control has the accuracy to deal with one or several pollutants, that is, to restrict or prohibit the pollutants and generally needs to be backed	1）The economic incentive instruments can complement command-and-control policies (OECD, 2017) or be used as policy assemblages and may be mandatory, depending on the situation (Cao et al., 2017). For example, the European Union Emissions Trading System (EU ETS). However, the specific content and proportion of this combination have been controversial.	1）The essence of public participation/persuasion strategies is “the change of the concept and priority of the parties in the decision-making framework or the internalisation of the concept of environmental protection into the preference structure of the parties” (OECD, 1994) 2）Public participation/persuasion strategies take the voluntary behaviour of the parties as

Regulation type	Command and control	Economic incentive instruments	Public participation/persuasion strategies
	up by sanctions and monitoring, which highly depend on the group size (Yamagishi 1988). 3）The policy of command and control has theoretical fairness, that is, it treats different individuals equally in front of the law. 4）The command-and-control regulation has immediate environmental improvements, which can be seen in the short term (Karp and Gaulding, 1995). 5) Fines related to violations can become government revenue.	2）Compared with the command-and-control regulation, the economic incentive instruments may have higher efficiency, that is, they use the market law to encourage cost saving and improve the effectiveness of environmental resource allocation (Finkelshtain and Kislev, 1997). 3）The economic incentive instruments may have higher flexibility and may play out as an incentive instrument to industrial innovation (Porter, 1991), that is, the result-oriented system may encourage organisations to explore more self-directed and spontaneous cost-optimised emission reduction programs. 4）Although the economic incentive instruments may use the pricing strategy to try to transfer the external environmental damage to internalise, the pricing added to production caused by pollution cannot be accounted for as production cost.	the basis of environmental governance, instead of the strength of control over the participators (Karp & Gaulding, 1995). 3) This kind of regulation generally aims for long-term impact and mostly does not exist alone and is coordinated with the first two kinds of environmental policies, such as tax deduction. 4）The theoretical framework of the public participation/persuasion strategies is that the selective provision or disclosure of information is seen as not only a tool that assists other regulations to operate but also an independent policy tool in terms of its rights (Sterner & Coria, 2012). 5）The public participation/persuasion strategies have a wide range of participants and the highest degree of flexibility. However, the effort of implementation may depend on the publicity.
Limitation	1）Some scholars think that, from the economic perspective, command-and-control regulations may increase the production burden of firms and reduce the possibility for firms to reduce more than the regulated limit line (Boyd & McClelland, 1999). 2）The command-and-control regulation depends on the thoroughness of policymakers' cognition of pollutants, that is, incomplete cognition of pollutants may	1）There are concerns that the economic incentive instruments may reduce the competitiveness of involved businesses, especially in international markets, and then reduce the growth in affected economic sectors. 2) Another concern is the income distribution. Using either upstream or downstream pricing mechanisms, the burden of increasing cost will transfer to the	1）There is a low degree of assurance for the effectiveness of environmental protection. The output of the public participation/persuasion strategies is highly dependent on the consciousness of the executive subject. 2）It is difficult to measure the overall impact of the public participation/persuasion strategies (Skopek, 2010), and the executor may tend to give up halfway when the cost is higher than the budget.

Regulation type	Command and control	Economic incentive instruments	Public participation/persuasion strategies
	reduce the effectiveness of policies (Liu et al., 2014). 3）The command-and-control regulations have limited flexibility, that is, for all industries, the uniform standards of enterprises may lead to a higher risk for high-burden enterprises to apply data manipulation, so as to increase the possibility of increasing management costs and corruption (Boyd and McClelland, 1999) and impact of the time lag effects (Li & Ramanathan, 2018). 4) There is strong control between the government and formulation and operation supervision of the command-and-control regulations, which will inevitably bring political colour to the command-and-control regulations. However, the fluctuation of the political environment, such as the election of a new government, is likely to bring adjustment or cancellation of non-environmental protection purposes to such regulations.	consumers, and the living standard of the low-income groups may be impacted most. 3）The negative impact of the economic incentive instruments on the economy of less developed countries is greater than that of developed countries (Avetisyan, 2018), and the same for the poor and the rich (Wang et al., 2019). 4）The economic incentive instruments may rely more on the consciousness of firms than the command-and-control regulations, that is, the relatively higher degree of freedom of rules and regulations may cause difficulties in balancing the efficiency and cost of supervision (Fan et al., 2017). 5）The economic incentive instruments make it relatively difficult to set the original price accurately. Journal articles show there is a possibility of reaching a similar level of emission reduction at a lower carbon price than current (Ervine, 2018). While from a financial point of view, the Coase Theorem model shows that it may have little effect on the formation of final price in the long run when having an efficient market, it still might have a certain degree of impact on efficiency in the short term.	3）There are different opinions about whether public participation/persuasion strategies promote environmentally friendly innovation for different businesses. According to Demirel and Kesidou (2011), it is considered that environmental regulations have an important driving force for enterprises to carry out environmental research and development, but have a limited impact on integrated technology. 4) Moreover, based on the voluntary mechanism there are some hidden dangers, such as lack of guarantee, difficulties in measurement of the achievement, and space for free riders. At present, for most participants, the attraction of such regulations and the main propaganda direction of the government is still in the aspects of goodwill and corporate image, which are difficult to measure by accounting (Frondel, Horbach & Rennings, 2008).
Compulsion	Passive acceptance	Passive acceptance and active compliance	Active compliance

Within this classification, the economic instruments mechanism can be further classified into three parts: (i) carbon/environmental taxes, (ii) tradeable permit regulations (such as the emission reduction market), and (iii) others (for example, subsidies, incentives and low interest/no-interest loans).

I) The carbon/environmental taxes and the tradable permit regulations

Table 2.3: The carbon/environmental taxes and the tradeable permit regulations

Regulation type	Carbon or environmental tax	Tradeable permit
Description	The carbon or environmental tax is a type of Pigouvian Tax. It uses a price control method which sets a fixed price for carbon emissions. In addition, the "per-unit tax" that the government levied by the government on the users of fossil fuels addresses its purpose of using taxation law as a tool to reduce pollution. (Tietenberg, 2006). Similar to other taxation, the carbon or environmental tax is a kind of public policy tool that has the function of influencing public behaviour and preference. In addition, the taxation instruments are designed with the theoretical framework of internalising the cost of public environmental harm by setting up a mandatory pricing mechanism following the "Polluter Pays Principle", which has the intention to bring social and private costs closer together (OECD, 1972).	The tradeable permit is a market incentive emission reduction mechanism supported by the Kyoto Protocol. The theoretic base for this mechanism is the theory explained by Baumol and Oates in the property rights theory. They believe that if seeing the permit to pollute is a right, then the government when controlling this right will be able to control the overall amount of pollution (Baumol & Oates, 1988). Thus, in contrast to the carbon or environmental tax, the tradeable permit, guided by the Coase Theorem, is paying to pollute less. By establishing the total emission limit, which is the cap or baseline, and allowing emission permission trading, this may not only achieve the overall emission reduction but also meet the individual differences of emission reduction among different enterprises. Generally, the tradeable permit systems can be built under cap-and-trade and baseline-and-trade systems or hybridise with other economic incentive strategies, such as the Certified Emission Reduction (CER) mechanism.

Regulation type	Carbon or environmental tax	Tradeable permit
Characteristic	1) Carbon tax may have the features of pollution tax and import and export carbon tariff. In addition, as a kind of tax, a carbon tax can also become a part of the national financial revenue, which may give the government more financial support to reduce income tax for the low-income population, reducing the economic burden of pollution tax impact on them. This is also commented on by the European Environment Agency (EEA), saying carbon tax is “potentially the most attractive for governments” (EEA, 2005). 2) Carbon tax making may be affected by multiple factors, such as votes and political factors (Gary & Lucas, 2017). Special care should also be taken in the design and application of carbon taxes (Baranzini, Goldemberg & Speck, 2000). 3) Compared to the complexity of carbon market regulation, the regulation of carbon tax is relatively simple. For example, the carbon tax can make use of the existing national tax system. In addition, researchers propose that the carbon tax has an effective range, and reasonable and strict environmental regulations may enhance rather than reduce industrial competitiveness (Xie, Yuan & Huang, 2017). 4) The carbon taxes are expected to play two roles in environmental protection: the direct influence on products' price and consumption structures through encouraging or discouraging certain economic activities, which may result in fluctuation of production cost; and the indirect impact on the “recycling of the collected fiscal revenues” to	1）The pricing of tradeable permit systems is affected not only by the regulation makers but also by other stakeholders (Zhu et al., 2019). 2）Compared with the carbon tax, the carbon trading market is more likely to be bound by economic development. It is the prediction of researchers that tradeable permit price is expected to be dependent on the common advance and retreat of economic growth in the future (Gerlagh & Liski, 2018). 3）The tradeable permit has higher flexibility, not only for industry differences, but also for better coupling with other policies, and the effect of combined use may be higher than that of single-use (Tvinnereim & Mehling, 2018). 4）The tradeable permit policy, such as the carbon credit trade system, has the possibility of expanding the scope, that is, it has the possibility of a transaction market, not only domestically but also internationally (Yu et al., 2017). 5）The price of the permit is not unchangeable. However, the transaction in line with the original intention of the establishment of the carbon market determines the market price trend (Fan, Liu & Guo, 2016). Other reasons explain the fluctuation, such as the participation of intermediaries (Wang et al., 2019).

Regulation type	Carbon or environmental tax	Tradeable permit
	change the investment tendency (Baranzini, Goldemberg & Speck, 2000). 5）The carbon tax might be expected to cooperate with other emission reduction regulations, such as the emission trade mechanism, and function to correct free market imperfections (Chen et al., 2020).	6) In addition, some academics believe that when the emitting businesses continually have to buy permits, the market will reach the point where the cost of purchasing a permit exceeds the cost of undertaking their abatement activities (Baumol & Oates, 1988). To get rid of this high cost of pollution, businesses may have a continuing incentive to develop environmental protection technology.
Limitation	1) Carbon tariffs may bring high management costs and need strong welfare justifications (Kortum & Weisbach, 2016). 2) The negative impact of carbon emission regulations such as the import environmental tax in developed countries may cause negative impact on developing countries' exports (Avetisyan, 2018). 3) Because carbon tax can be used as a part of fiscal revenue, the motivation for setting up carbon tax and the tax pricing range may not be the only purpose of emission reduction (Gao & Chen, 2002). 4) To some extent, the carbon tax is similar to the command-and-control regulations, which are highly dependent on the overall planning ability of policymakers. For example, if the carbon tax is formulated without considering pre-existing taxes, the cost of new environmental tax plans may be greatly underestimated (Goulder, 1995); and the effectiveness of Norway's carbon tax has been	1）The tradeable permit regulations highly depend on the perfection and maturity of the market. That is, an immature market or lack of legal protection will increase the risk of leading to market failure (Bond, 2012). 2）Because it is based on the free market, the tradeable permit regulations may have similar limitations to the free market, such as the difficulty of supervision and vulnerability to the financial crisis (Lotay, 2010). 3）There are journal articles from researchers about the efficiency of the tradeable permit market, saying it lacks the ability to adjust to the risk of low prices. In addition, the participants of the emission trade market may be impacted by the uncertainty of future prediction, which is reflected in the complexity of the market and the uncertainty of regulation persistence (Knopf et al., 2018). 4）Various emission reduction projects with complex designs and new energy projects are mixed in parallel, which leads to regulatory

Regulation type	Carbon or environmental tax	Tradeable permit
	dragging on its inconsistencies within policy design (Bruvoll & Larsen, 2004).	difficulties. This may cause market confusion and regulatory difficulties, which might reduce the authenticity of emission reduction figures (Weiss, 2004).
Relationship	Passive acceptance	Acceptance and active compliance

II) Others

There are also other forms under the economic incentive instruments category. For example, under the OECD mechanism, there are other economic incentive environmental instruments classified as deposit-refund schemes, environmental liability insurance, and environmentally motivated subsidies. Some of them later became part of a series of policies on permits or carbon taxes. Table 2.4 lists some typical instances of regulations that are relatively self-contained and implemented separately. The scope of application of these regulations is relatively narrow; some systems have only been implemented in one or more states rather than federally, and the implementation effect is uncertain or limited.

However, these policy systems can be seen as bold attempts to design emission reduction systems and they provide a reference for the formulation of relevant systems in the future.

Table 2.4: Examples of other environment regulation

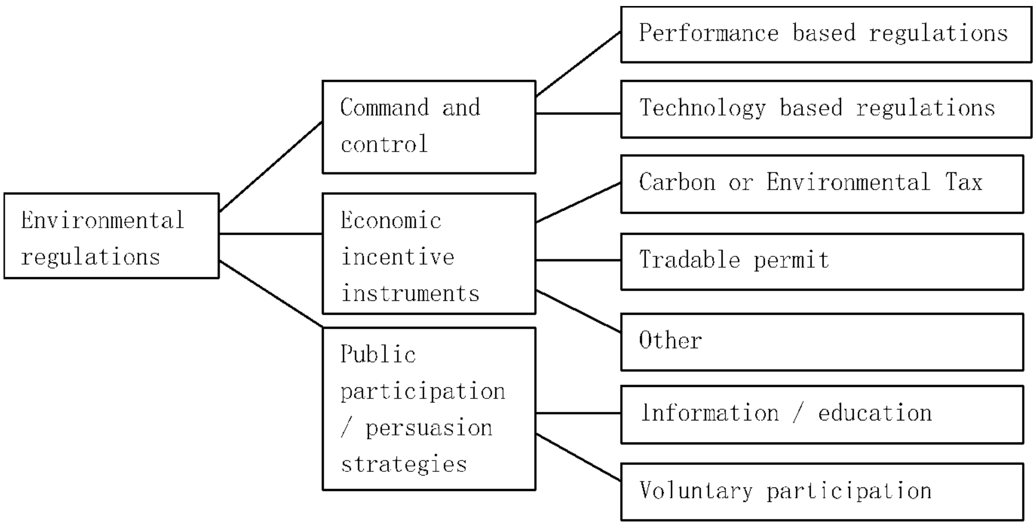
	Description	Effect	Example
Deposit-refund schemes (OECD, 1994)	There are two kinds of deposit systems: based on the willingness of business and based on the government's compulsion. The amount of government return may be greater than the amount of deposit. This type of regulation is mainly used for products or substances that can be recycled or must be recycled.	Return rate of 80%–90%	Greece, Norway, Sweden (car wreck deposit return system)
Environmental liability insurance (Qi & Zhang, 2018)	Firms purchase environmental pollution insurance to deal with the compensation for possible victims.	No data	China
Implementation incentive fund (OECD, 1994)	The implementation incentive fund for execution includes bonds and fines. The rate of the fine is not fixed, so depends on the damage caused or the benefit from violation of regulations.	No data	US, Sweden, Canada, Australia (fees and enforcement bonds)

2.2.4 Classification typology for key regulatory mechanisms

Figure 2.2 presents the comprehensive typology derived from a review of the literature review. There are two key characteristics which underpin the typology. First, the level of compulsion contained in the design of regulations, with regulations ranging from voluntary through to mandatory with and without sanctions attached; and second, the level of enthusiasm for emission reduction that each type of regulation can promote for its audience.

Environmental policy instruments can be divided into three categories, based on enforcement and incentive characteristics: (i) command and control, (ii) economic incentive instruments, and (iii) others. Generally, as shown in Figure 2.2, there are two major ways to define these "others". One way is the information-leading method (Mickwitz, 2003; OECD, 1994) and the other is the voluntary participation method (Karp & Gaulding, 1995; Howarth, Haddad & Paton, 2000).

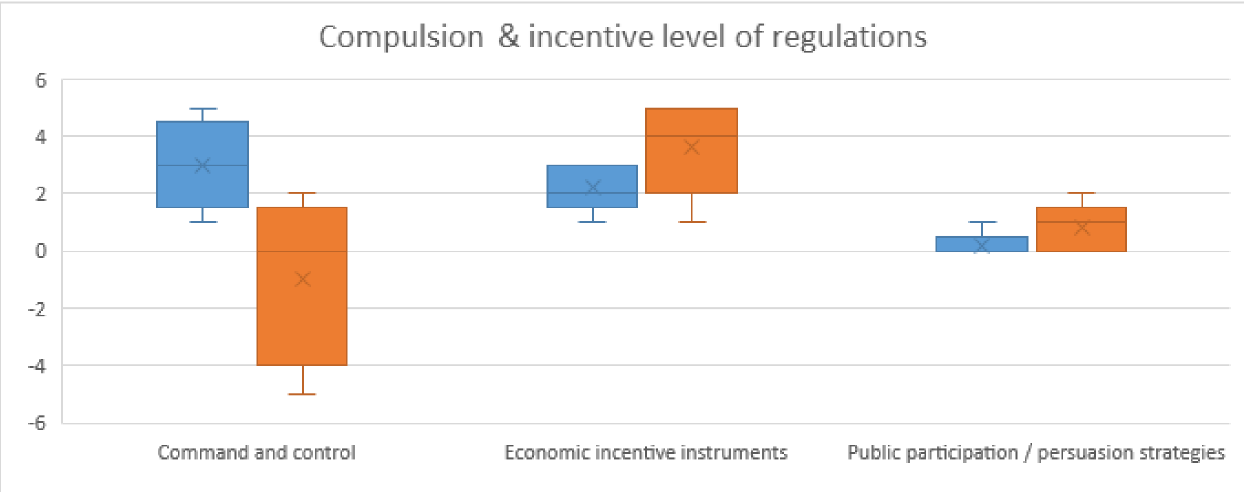
Figure 2.2: Environment regulation typology



The information-leading and voluntary participation categories have been combined into the public participation/persuasion strategies for two reasons: first, they both have low levels of coercion; and second, both are often implemented in an auxiliary fashion, along with Command and Controls and Economic Incentive mechanisms.

The impact of compulsion and incentive characteristics on different policy types is illustrated in Figure 2.3. Blue represents the level of compulsion and orange represents the incentive level. The vertical axis represents the intensity of the assumed type characteristics of the regulatory mechanism, where the scale of 0 to 6 on the compulsion scale represents weak to strong y and 0 to –6 on the incentive scale represents the strongly attraction to repulsion.

Figure 2.3: Illustration of the intensity of compulsion and incentive characteristics on different policy types



Note: This figure does not derive from a direct transformation of precise numerical data but rather from an interpretive conversion of the analyses and descriptions provided by Swaney (1992) and Sinclair (1997).

It is crucial to highlight that a high degree of compulsion and potential resistance does not necessarily imply that policies based on 'Command and Control' are inherently less effective than those based on 'Economic incentive instruments'. In practice, when appropriately planned and implemented[12], both approaches exhibit comparable efficacy and demonstrate complementary characteristics (Tuladhar, Mankowski and Bernstein 2014; Blackman, Li and Liu 2018; Wiesmeth 2022). Accordingly, it is not reasonable "one

[12] When regulations pertaining to the production system are "appropriately planned and implemented," context-aware evaluations becomes essential, necessitating modifications that are precisely adapted to the unique local/industry circumstances (such as variations in local precipitation and regulations concerning irrigation).

size fits all” regulation.

2.3 Research design

This section provides an empirical evaluation of the proposed regulation classification typology, by comparing it with a set of regulatory systems. The set of regulatory systems were chosen based on:

(i) Whether they are globally significant. The set of key Australian regulatory systems, as well as key international regulatory systems, were chosen based on these principles. Regulatory systems which are small or did not appear to have a lot of influence were excluded (One example of failed environmental regulation is the pink batts program in Australia as expressed by Crawford, 2014).

(ii) Whether they serve as an appropriate representation. There was also consideration given to selecting different sets of regulation types, including regulations from EU, US, South America and Australia emission reduction regulations.

(iii) Whether there are sufficient and reliable data sources. Data about the regulatory systems were sourced through governmental documents, publications and websites on open courses, which were used to classify them using the typology given in Section 2.4 according to structure, purpose, function, terms of reference, force, reward and punishment measures, whether there are laws and regulations to support, and whether agriculture and land use change are involved.

Please note, (i) emphasizes the applicability scope of emission reduction policies. For major countries with international influence, such as the USA, China, and Russia, policies that are nationally applicable meet the criteria for the applicability scope as required by (i). Conversely, national policies from other countries, like Indonesia (as a less affluent non-permanent member states), do not fulfill these criteria. Additionally, environmental policies of individual EU member states are not considered in isolation; instead, the European Union is regarded as a unified entity in terms of its environmental policies.

(ii) highlights the effectiveness of the policy, that is, whether the policy has been successfully implemented and operates effectively within the applicability scope defined by (i). Effective operation refers to whether the policy is widely accepted by its target audience and has been continuously enforced for a year or more (including updates or being replaced by more specific or comprehensive policies of the same type) and has achieved internationally recognized positive environmental benefits. .

Table 2.5 presents each of the selected regulatory systems, along with a description.

Table 2.5: Sample of regulatory systems analysed

Regulation	Brief introduction	Type
EU ETS	The Emissions Trading Directive (ETD) stipulates that each member country has to form a National Allocation Plan (NAP) for GHG emissions allowances, according to the parameters set by the Directive. Each member country determines the total quantity of permits needed and a specific allocation plan. The Commission of the European Communities then has to approve the NAPs submitted by member countries.	Economic incentive
International Emission Trading	The European Linking Directive (ELD) offers European companies the opportunity to invest in emissions reduction projects in developing countries and bring carbon credits back to use in the EU ETS. Therefore, companies can use credits from the Kyoto Protocol mechanisms to fulfil their obligations under the EU ETS. It suggests the recognition of credits created through CDM or JI mechanisms as equivalent to allowances.	Economic incentive
Joint implementation (JI)	Under Joint Implementation, countries with commitments under the Kyoto Protocol are eligible to transfer and/or acquire emission reduction units (ERUs) and use them to meet part of their emission reduction target.	Economic incentive + public participation
Clean Development Mechanism (CDM)	The Clean Development Mechanism (CDM), defined in Article 12 of the Protocol, allows a country with an emission-reduction or emission-limitation commitment under the Kyoto Protocol (Annex B Party) to implement an emission-reduction project in developing countries. Such projects can earn saleable Certified Emission Reduction (CER) credits, each equivalent to one tonne of CO_2, which can be counted towards meeting Kyoto targets.	Economic incentive + public participation
Regional Greenhouse Gas Initiative (RGGI)	The Regional Greenhouse Gas Initiative (RGGI) is the first mandatory program to reduce GHG emissions in North America. Ten Northeastern and Mid-Atlantic states have capped and will reduce CO_2 emissions from the power sector.	Command and control

Regulation	Brief introduction	Type
NSW Greenhouse Gas Abatement Scheme (GGAS)	National Greenhouse and Energy Reporting Scheme (NGERS) supports the Department's National Greenhouse Gas Inventory Program and underpins Australian emission reduction policies, including the Emission Reduction Fund, Safeguard Mechanism and Renewable Energy Target. It provides a national framework for corporations to report on greenhouse gas emissions, energy consumption and energy production data.	Command and control + public participation
Chicago Climate Exchange (CCX)	Chicago Climate Exchange (CCX ™) is the world's first and North America's only voluntary, legally binding greenhouse gas reduction and trading program for emission sources and offset projects in North America and Brazil.	Economic incentive
Over-the-counter (OTC)	"Over-the-counter", or "OTC" within the REMIT compliance system, means any transaction carried out outside an organised market. (This definition has relevance, for example, for REMIT reporting.) In turn, under MiFIR characteristics, OTC trades include transactions that are non-systematic, ad hoc, irregular and infrequent, are carried out between eligible or professional counterparties, are part of a business relationship that is itself characterised by dealings above standard market size, and where the deals are carried out outside the systems usually used by the firm concerned for its business as a systematic internaliser.	Economic incentive
Environment and Resource Efficiency Plans (EREP)	Commenced on 1 January 2008. Participants in the first year were required to register by March 31, 2008, and submit their EREP to EPA Victoria by 31 December 2008. It required industrial and commercial sites to identify and implement energy, water and waste efficiency actions. Closed in 2013.	Command and control
Victorian Energy Upgrades (VEU) program	The program provides households and businesses with access to discounted energy-saving products and upgrades through accredited providers. Every upgrade allows accredited providers to generate Victorian energy efficiency certificates. Each certificate represents one tonne of greenhouse gas abated. Energy retailers are then required to buy and surrender these certificates to meet legislated targets.	Economic incentive

Regulation	Brief introduction	Type
NSW Energy and Water Savings Action Plans (ESAP and WSAP)	Includes businesses and NSW Government agencies that use more than 10 GWh per year at a site, or local councils that service cities with populations larger than 50,000 people. Also includes sites of businesses and NSW Government agencies that use 50 ML or more of water in a year that is provided by Sydney Water, or all local councils located in Sydney Water's area of operation.	Public participation
The Smart Energy Savings Program (SESP)	The Smart Energy Savings Program is intended to assist medium to large-sized businesses in Queensland to unlock energy cost savings. The program requires participating businesses to undertake an energy audit, develop an energy savings plan, and publish their actions for each relevant site, on a five-year cycle.	Economic incentive + public participation
Energy Efficiency Opportunities (EEO) program	The EEO program was developed and administered by the former Department of Industry, Tourism and Resources. The new Department of Resources, Energy and Tourism will continue to administer the program. EEO Act was repealed in 2014.	Command and control
The National Greenhouse and Energy Reporting (NGER) Act	The National Greenhouse and Energy Reporting (NGER) scheme, established by the National *Greenhouse and Energy Reporting Act 2007 (NGER Act)*, is a single national framework for reporting and disseminating company information about greenhouse gas emissions, energy production, energy consumption and other information specified under NGER legislation.	Command and control + public participation
Safeguard legislation	Example: The high-level framework of the safeguard mechanism was legislated as part of the Carbon Farming Initiative Amendment Act 2014. Once commenced (July 2016), the safeguard mechanism will be part of the National Greenhouse and Energy Reporting Act 2007.	Command and control
Small-scale Renewable Energy Scheme (SRES)	The SRES creates a financial incentive for households, small businesses and community groups to install eligible small-scale renewable energy systems, such as solar water heaters, heat pumps, solar photovoltaic (PV) systems, small-scale wind systems, or small-scale hydro systems.	Economic incentive + public participation

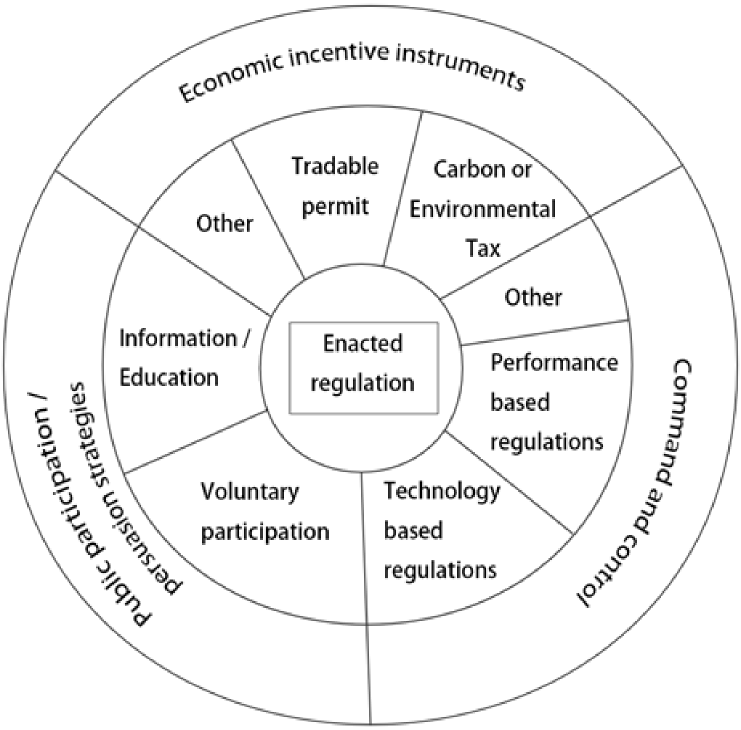
Regulation	Brief introduction	Type
Conservation covenant	Conservation covenants are agreements entered into voluntarily but require the landholder to carry out certain activities by law. To meet the regulatory additional requirement of the Australian Carbon Credit Unit (ACCU) Scheme, scheme participants must demonstrate that a reduction in emissions would not occur without their project. Under Section 27(4A) (b) of the Carbon *Farming Initiative Act 2011*, the regulatory additional requirement is met when: a project is not required to be carried out by or under a law of the Commonwealth, state or territory government; or the method that is used by a project specifies one or more requirements that are 'in lieu' of the regulatory additionally requirement. This means there are two ways participants may be able to enter a conservation covenant and run a project on the same land under the ACCU.	Economic incentive + public participation

2.4 Findings and discussion

Tables 2.6 and 2.7 present the set of regulatory systems classified utilising the consolidated typology. The process of classifying each regulatory mechanism confirmed that the environmental regulation typology (Figure 2.2) can be used to classify regulatory mechanisms.

Furthermore, an analysis of Table 2.6 and 2.7 resulted in the derivation of the model presented in Figure 2.4.

Figure 2.4: Categories of environmental regulations



The outermost layer of the circle consists of the three major categories of current emission reduction policies. According to the severity of coercion, reward and punishment, there are three regulation types: command and control, economic incentive instruments, and public participation/implementation strategies. All emission reduction policies can be classified as the domain of one type or several combinations. These three categories can be further subdivided into eight subcategories. In command and control, the most common types are performance-based regulations with strong prohibitions to stop pollution and technology-based to reduce side effects, that is, to restrict or prohibit the use of certain polluting materials, or to encourage the development of alternative energy sources, or a combination of the two.

Economic incentive instruments can be divided into three categories, of which the most common two are carbon or environmental tax and tradeable permits. Both aim to eliminate high-polluting enterprises or raw materials, by adjusting the factors in the supply and demand relationship of the market. Although carbon or environmental tax has the nature of tax enforcement, it is not suitable to be classified as command and control, because the purpose of tax collection is to change the existing supply and demand through price changes, rather than simply a fine charged for polluting.

The third category of 'other' includes all other non-mainstream emission reduction policies which may also use the market to adjust emission reduction cost. For public participation/emission reduction strategies, this type of emission reduction policy has been given more attention in recent years. It can be divided into two categories: information/education and voluntary participation. The prominent feature of these two is that both the effective time and duration are long term. They are mostly introduced as auxiliary policies of emission reduction policies.

Table 2.6: Typical regulator systems-1

	Economic incentive Mechanisms	command and control	Educational or informational	Voluntary agreements	Compliance/ mitigation	Voluntary	public disclosure/audit	plan making/execution report	Direct Ban
EU ETS	1				1				
International Emission Trading	1				1				
Joint implementation (JI)	1				1				
Clean Development Mechanism (CDM)	1				1				
Regional Greenhouse Gas Initiative RGGI	1				1				
NSW Greenhouse Gas Abatement Scheme (GGAS)	1				1				
Chicago Climate Exchange (CCX)	1					1			
Over-theCounter (OTC)	1					1			
Environment and Resource Efficiency Plans (EREP)		1	1		1			1	
Victorian Energy Efficiency Target (VEET)	1		1		1				
NSW Energy and Water Savings Action Plans (ESAP and WSAP)		1	1		1			1	
The Smart Energy Savings Program SESP		1	1		1		1		
Energy Efficiency Opportunities (EEO) program		1	1		1		1		
The National Greenhouse and Energy Reporting (NGER) act		1	1		1	1	1	1	
Safeguard legislation	1		1		1				
Renewable Energy Scheme (SRES)	1			1		1			
Conservation covenant				1		1			

Table 2.7: Typical regulator systems-2

	Tax deduction	Offset	Carbon taxes	ETS	Cap-and-trade systems	Baseline-and-credit	Baseline and adjust	Per unit pricing	Offsets originate from emissions reduction projects	Fine	Not legally binding	Legally binding	Agriculture & land use
EU ETS				1	1							1	
International Emission Trading				1	1							1	
Joint implementation (JI)				1	1							1	
Clean Development Mechanism (CDM)				1				1				1	
Regional Greenhouse Gas Initiative RGGI				1	1							1	
NSW Greenhouse Gas Abatement Scheme (GGAS)				1		1						1	
Chicago Climate Exchange (CCX)				1	1							1	
Over-theCounter (OTC)		1							1		1		
Environment and Resource Efficiency Plans (EREP)										1		1	
Victorian Energy Efficiency Target (VEET)				1	1							1	
NSW Energy and Water Savings Action Plans (ESAP and WSAP)												1	
The Smart Energy Savings Program SESP												1	
Energy Efficiency Opportunities (EEO) program										1		1	
The National Greenhouse and Energy Reporting (NGER) act											1	1	
Safeguard legislation		1					1					1	
Renewable Energy Scheme (SRES)		1				1					1		
Conservation covenant	1					1						1	1

In summary, environmental regulation has three characteristics.

First, it predominantly features a policy-centric nature within a delineated time span in a particular national or governmental system, with several secondary supporting attributes. For example, the US uses command and control to limit the use of sulphur pollutants. In addition, there is a high focus on emission reduction for agriculture and land use in literature, but a low focus in regulating practice.

Second, the transition of leading policies in different periods is not smooth, and the driving force of the transition is not a single environmental protection orientation. In addition, there is a high chance this transition will be affected by any change in the term of government office.

Third, in literature, many focus on and demonstrate the advantages of carbon pricing, especially market-oriented carbon trading, and confuse the carbon tax with command and control. The reasons given are as follows:

(i) Carbon trading is predominantly governed by market rules, which can inadvertently lead to a reduction in investments within emission-intensive industries. Specifically, the indiscriminate application of carbon trading policies might dampen the enthusiasm for investments in heavy industries that have a deeper engagement in emission investigations. On the other hand, industries that lack thorough emission investigations and regulations might remain unnoticed. For instance, the absence of emission reduction policies for agriculture and land use implies that these sectors are not obligated to participate in carbon trading and bear the associated responsibilities.

(ii) Carbon trading is more economical and efficient than other methods under equivalent levels of industrialisation. For example, direct environmental pollution fines in response to the not entirely severe, yet plausible scenario where multiple organisations surpass the prescribed emission limits.

(iii) Carbon trading exemplifies the merits of self-regulation and adaptability, while aligning with various environmental strategies. However, delving into and appraising these regulatory configurations can be daunting. Despite the presence of robust theoretical paradigms for regulations, significant challenges endure. As underscored by Gao (2019), there are limited methods for evaluating combinations of regulations or conducting pre-launch comparisons.

2.5 Conclusion

The objective of this chapter was to develop and evaluate a comprehensive typology to categorise and describe elements of regulatory systems. The origin of social rules was presented to frame the discussion. The literature review and research method extended Taylor et al. (2012), to draw out a comprehensive typology for regulation categories. With respect to achieving the regulatory goal, regulatory classification can be divided based on the compulsive strength and incentive degree (as Figure 2.3); regulation types can be summarised into the following three categories: command and control, economic incentive instruments, and public participation/implementation strategies.

In relation to this research, the evaluation of regulatory options stands as a pivotal exercise in ensuring efficacious policy measures. Chapter 2 delves into the intricate nuances of regulation, underlining the symbiotic relationship between the 'how' and 'what' of regulatory processes, emphasizing the myriad choices that converge into comprehensive regulatory assemblages. Building upon this foundational understanding, Chapters 3 and 4 delve deeper, focusing on delineating 'what might be subject to regulation'. By addressing the subject of regulation, these chapters naturally extend the discourse initiated in Chapter 2, further clarifying the intricate dynamics between the 'how' and 'what' of regulatory practices.

A limitation of the research is the lack of reflection on the possible relevance between the political intent and the design of regulatory mechanisms in practice. Furthermore, within this chapter, policy combinations introduced as a cohesive whole are partitioned into their individual components, overlooking potential disparities in efficacy between the entire assemblage and its individual elements. However, it is beyond the scope of this thesis to address these challenges. Future research is needed to address this gap in the literature.

While having a comprehensive typology will help describe regulation, the analysis illustrates that regulation is enacted in assemblages of different types of regulatory mechanisms, which makes it challenging to design, compare and evaluate different regulatory options. The following chapters go some way to exploring how this challenge may be addressed.

Chapter 3 ESRI modelling and case study

3.1 Introduction

3.1.1 Background

Scholars have long recognised that conventional agriculture production systems are not sustainable due to the environmental burden they cause (Hall & Hall 1984; Bowler 2002; Gomiero, Pimentel & Paoletti 2011). Agriculture is the main source of global nitrogen greenhouse gas (GHG) emissions[13] (Smith 2010; Skiba & Rees 2014; Reay et al. 2012; Laing et al. 2023). According to IPCC 2020 climate modelling predictions, nitrogen GHG emissions are anticipated to persist without significant reductions, due primarily to the increasing needs of agricultural outputs (IPCC 2019) and the nature of current production systems. Together with nitrogen emissions, on-farm diesel and lime have been identified as the primary contributors to GHG emissions from cropping (Sevenster et al. 2022; Brock et al. 2012)[14]. While it is known that GHG emissions, specifically nitrogen emissions, need to be reduced substantially in agricultural production systems, it remains unclear how to achieve such reductions at a global level to avert the worst impacts of climate change.

3.1.2 Motivations

The primary motivation is that current production system performance evaluation systems and models fall short of comprehensively depicting the extent of environmental degradation and possible consequences caused by different nitrogen management practices, to enable identification of what practices are most sustainable.

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[13] GHG increases through the release of N_2O and the fertiliser application process, which may result in the release of ammonia escaping into the atmosphere. This may contribute to acid rain (through HNO_3). While addressing these nitrogen emissions, I acknowledge that there exist numerous additional sources of emissions, including those attributed to CO_2 emissions resulting from changes in land use and soil carbon.

[14] Data from Brock et al. (2012) 30% fertiliser production, 7% lime production, 2.9% diesel production; on-farm 39.3% fertiliser, 1.9% lime, 13.5% diesel.

This challenge is reflected in three sub-motivations.

(I) Motivation one

First, the typical evaluations of agricultural production systems face limitations in comprehensively evaluating the extent of environmental degradation and its impact on non-renewable natural resources, due in part to an inability to provide a reliable assessment of the trade-offs between economic and environmental performance (Pearce, 1976; Davis et al., 2011; Rambaud & Richard 2015; Rambaud & Richard, 2021, Brown 2016; Pham et al. 2010)[15].

To that end, efforts have been made to ascertain the value of natural resources and evaluate the impact of production systems (cf. Bebbington & Gray, 2001; Galbally et al. 2005; Herbohn, 2005; Johnson et al. 2008; Rochester 2011; Rochester 2014; Rambaud & Richard, 2015; Taplin et al. 2006; Hadjimichael et al. 2016; Pham et al. 2010; Powell, Welsh & Freebairn 2019)[16]. However, in the context of supporting stakeholders to make decisions in favour of the achievement of substantive reductions in global GHG emissions such as net-zero targets, the depiction of agricultural production system sustainability performance is inadequate. Historically, research has largely failed to build the link between practical activities (i.e., decision-making) and holistic environmental and economic performance (Bebbington & Gray, 2001; Rambaud & Richard, 2015). Many research studies prioritise enhancing production efficiency and energy utilisation

[15] For instance, within the domain of nitrogen management practices sustainability, precision agriculture (Fairchild 1988, Pierce & Nowak 1999) and site-specific management (Cassman 1999; Plant 2001) have been demonstrated to possess the capacity to enhance nitrogen utilisation efficiency and contribute to establishing the technological groundwork for sustainable production. However, their evaluation remains constrained by variations in temporal frameworks and the assessment of indirect pollution within the supply chain.

[16] This presents a predicament for establishing emissions reduction and sustainable production targets, as under the strong sustainability theory, natural resources hold varying values, with non-renewable resources that cannot be restored considered particularly valuable (Ayres, Van Den Bergh & Gowdy 1998; Jaffe et al. 1999; Pearce, 1988; Málovics, Csigéné & Kraus 2008). However, traditional value instruments are inadequate in addressing this aspect (Rambaud & Richard, 2015, Brown 2016; Pham et al. 2020; Pearce, 1976; Davis et al. 2011; Resurreccion et al. 2012; Havemann, Negra & Werneck 2022).

(cf. Rochester 2011; Rochester 2014), overlooking the crucial aspects of economic analysis (cf. Liu et al., 2022) and broader sustainability considerations at the industry or system level (cf. Meena et al., 2022). This oversight has been highlighted by various researchers (Hueting et al., 1992; Hueting & de Boer, 2001; Pham, 2016; Pham et al., 2020) as essential for a theoretically sound approach to assessing the sustainability of a production system.

As yet, there are few examples of comprehensive evaluations of agricultural production systems that provide a multidimensional economic and environmental assessment of the sustainability of the system as a whole.

(II) Motivation two

Second, another obstacle to the transformation of agricultural production systems into sustainable practices is the complex task of developing a comprehensive information support system. Several authors argue that the information and measurement model for agriculture sustainability production should cover various activities along the entire production chain, providing decision-makers with high-quality information to guide their choices. They argue that it is essential to design the information flow within the system based on well-defined causal models linked to each other, allowing for a clear understanding of the interconnectedness and interdependencies among different elements (Baumgärtner, 2015; Brown & Bajada, 2018; Pearl, 2009; Pham, 2016; Pham et al. 2020; Cucurachi & Suh, 2017; Li, 2020).

This is because agriculture production (and similar production systems) involves multiple stakeholders and disciplines, making agricultural emission reduction a challenge that cannot be addressed solely by practitioners or researchers (Brown & Bajada, 2018; Taplin et al., 2006; Brown, 2016; Pham, 2016; Sander & Murthy, 2010). Developing such a comprehensive evaluation information system is crucial, due to the significant role of information quality in decision-making and the growing demand for integrated modelling approaches.

Research suggests that establishing a robust integrated modelling system[17] is crucial to addressing sustainability concerns, which can aid in regulatory assessments and decision-making (cf. Sharma, Carmichael & Klinkenberg, 2006; Gibbons & Ramsden, 2008; Hadjimichael et al., 2016; Pham et al., 2020). For example, in the context of the agriculture sector, nitrogen emissions are primarily attributed to cropping and other land use, and are further exacerbated by the excessive application of nitrogen fertilisers (IPCC, 2019). However, nitrogen emissions are a complex interdependent indicator, as their impact on crop yield and the level of emission is closely linked to irrigation choices. Thus, nitrogen emission modelling[18] should also consider other activities, such as irrigation choice (Halvorson, Del Grosso & Reule, 2008).

Nevertheless, the modelling of agricultural emissions within the framework of emission reduction initiatives mostly fails to capture the growing interconnection between agricultural emissions and their associated economic and upstream-downstream activities (cf. Yli-Viikari et al., 2007).

Despite the availability of current practices[19] (e.g., myBMP for Australian cotton growers and Cotton Carbon Calculator by the Cotton Research & Development Corporation CRDC), a gap persists between these decision support systems and on-farm practices to the end of achieving sustainability in GHG emission reduction. Indeed, there is a notable absence of comparative assessment on the conscientiousness of choices at the

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[17] The integration framework should address three key areas for effective integrated modelling: (i) seamless integration of existing models, (ii) plug-and-play interaction, and (iii) intuitive, real-time interaction (Muth Jr & Bryden, 2013).

[18] It is noteworthy that agricultural modelling can be developed for a multitude of purposes, such as self-learning (developers), education, scientific exploration, and decision support (engineering) (Passioura 1996). In this thesis, ESRIDM is designed for decision support.

19 Please note that: i) in the scope of AU cotton which this thesis focuses on, 75% or more of the Australian cotton crop is irrigated (CRDC, 2023; Roth et.al., 2013). ii) nitrogen management is not only occurred by irrigated or not, but also nitrogen fertiliser application, which non-irrigated farms also need to apply nitrogen fertiliser, therefore create emissions caused by nitrogen fertiliser production (although some of these fertilisers may not be produced in AU), therefore it still fits in the discussion scope of ESRIDM. While addressing these nitrogen emissions, I acknowledge that not all farms have irrigation. In future research, non-irrigated cropping system can be used by adjusting water module in ESRIDM.

operating level (i.e., on-farm) when considering various sustainable options within the context of sustainable production. This gap may be due in part to insufficient linkage and representation between environmental damage/improvements and economic value, as well as a lack of sufficient quantum of decision choice assessments, in terms of both information and modelling support. Although assistant tools such as carbon calculators are provided, platforms or tools for regulation or reduction scenarios are not. In other words, users would not be able to identify and test a comprehensive set of sustainable actions to identify which combinations are most desirable and what impact they might have. While facing multiple choices, users cannot compare different options to see which one might bring better sustainability and economic outcomes. This poses a significant challenge for the industry in achieving more sustainable outcomes.

(III) Motivation three

Third, there is a lack of an established information and evaluation framework to facilitate sustainable production decision-making[20] and the formulation and implementation of related regulations.

The lack of regulatory support is one important barrier to achieving sustainable production (IPCC, 2019; Smit & Smithers, 1993). This issue can be attributed to several factors, as identified in Chapter 2, such as lack of clarity on what to plan, how to plan effectively, the limited budget affecting the outcomes, inadequate consideration of influencing factors, failure to integrate with the local context, and misalignment with future planning objectives. As demonstrated analytically by Pearce (1976), in the absence of regulation surrounding accountability for externalities, markets driven by cost-benefit decision-making alone are prone to collapse.

Thus, the promotion of sustainable production requires regulatory measures that closely adhere to objective scientific principles. These measures should be based on the

[20] While the term 'sustainable production decision-making' may have broad meanings in a variety of contexts, in this thesis the focus is on GHG emissions reduction with respect to sustainable production decision-making.

current state and potential future development of the industry, encompassing, but not limited to, the objective progress of climate change and technological advancements.

This may be attributed to the inherent complexity of agricultural production, the fluctuating nature of the phenomenon, and the delays in information systems indicated above. These factors make it difficult for stakeholders to accurately predict, assess, and make informed decisions (Malhotra, Melville & Watson, 2013; Melville, 2010; Chen, Boudreau & Watson, 2008). Regulatory or institutional support is needed to address this issue, as current support systems have been criticised for their lack of sustainability (Gholami et al., 2016: Korsching & Malia, 1991; Keswani, Sarma & Singh, 2016).

3.1.3 Objective and method

Accordingly, the objective of this chapter is to develop and explore an integrated model to understand and evaluate the economic and environmental performance of proposed GHG emission regulation options, in the context of agricultural production. To that end, this chapter theorises the design characteristics for integrated modelling systems that enable the identification and evaluation of alternative practices, technologies and other factors that may lead to more sustainable cropping production through more informed decision-making.

This chapter employs an integrated modelling approach combined with a case study taken from the Australian cotton industry, to illustrate a method for evaluating the sustainability of an agricultural production system. The focus of the case study is on-farm nitrogen management. By integrating various modelling techniques and utilising a specific case study, this chapter aims to showcase a systematic methodological framework that can be adapted in a variety of contexts to facilitate sustainable research and innovation practices.

The proposed system extends the water management system model developed by Pham (2016), which integrates a science-derived model with accounting evaluations utilising a Penman-style (2003) decomposition model. While Pham (2016) introduced a water use and profitability efficiency evaluation model in the context of a furrow-irrigated cotton farm, the integrated model presented in this chapter expands the application scope to encompass a larger boundary around the cotton production process. It specifically focuses on the pollution effects related to nitrogen management decisions.

Furthermore, the proposed system incorporates the utilisation of the Environmentally Sustainable Residual Income (ESRI) theory (Brown, 2016) for evaluating production sustainability. This theory originates from Hueting's (1993) concept of sustainable national income and has been further expanded upon by Rambaud and Richard (2015), in the context of organisational performance evaluation. The illustrative case study is of a typical furrow-irrigated cotton farm, which offsets its nitrogen-based GHG emissions through tree planting.

Through the application of the Environmentally Sustainable Residual Income Decomposition Modelling (ESRIDM) in a case study, several notable findings emerge. First, it becomes apparent that the existing agricultural cropping practices are not sustainable, as evidenced by negative ESRI and the economic challenges associated with achieving full offset for cotton production. It demonstrates that the cost of achieving full offset for cotton production poses economic challenges for further sustainable operations. Second, the analysis of offsetting emissions solely through tree planting projects demonstrates the complexity of addressing GHG emissions without regulatory support. This underscores the importance of implementing appropriate policies and interventions. Lastly, these findings show the capacity of ESRIDM to offer quantitative evidence of the ongoing accumulation of environmental assimilative capacity, supporting Pearce's (1976) framework and underscoring the importance of considering broader environmental implications.

3.1.4 Contributions and chapter structure

This research makes several theoretical and practical contributions.

First, this research introduces an approach to advance the study of sustainable production by developing a novel integrated modelling framework to aid in the investigation and support of decisions on environmental and economic sustainability, within the context of nitrogen sustainability. The proposed approach extends Pham et al.'s (2020) work, by broadening the scope of the model from on-farm water management to nitrogen management and aspects of the supply chain. Additionally, it extends Penman's (2003) decomposition modelling to the realm of operational decision-making, fusing sustainability and its integration with agronomic and sustainability theories. This aspect is of pivotal importance for both the academic literature and practical applications. Not only does it expand the scope within which decomposition and integrated modelling can be employed to explore sustainable production in cropping from a systematic perspective, but it also illustrates the feasibility of adapting such a framework of sustainable production to other environmental studies and topics, both in academic research and practical applications.

In addition, the method development may contribute a novel method for constructing and calibrating expectations about what nitrogen management decisions may be amenable to regulation.

The approach integrates valuable insights with scientific modelling, specifically the water and nitrogen module, with adapted techniques of sustainable economic modelling previously implemented by researchers such as Hofkes (1996), Shen, Kyllo and Guo (2013), Chami and Daccache (2015), Nguyen et al. (2021) and Belcher, Boehm and Fulton (2004). However, none of these studies have incorporated the integration with scientific software or provided evaluations for different simulation scenarios.

With the advancement of computer science, data science techniques and software technologies (e.g., machine learning capability) have increasingly been employed to bolster research in sustainable production. This enables researchers to work with larger datasets and higher levels of modelling complexity to support their research topics. For instance, studies by Barthel et al. (2008), Welsh et al. (2013) and Guo et al. (2021) have utilised a combination of scientific models and economic impact/benefit analyses in their integrated modelling approaches. However, these works do not explore the ability to integrate the modelling of a set of possible sustainable production scenarios. More importantly, they lack a design intent and capability for the decomposition, attribution, tracing and analysis of the phenomena generated. For example, Guo et al. (2021) explicitly state that the variability of climate conditions complicates the adaptation of models in practice, thus restricting them to offering reasonable forecasts over short-term future time spans only.

In this chapter, the emphasis of the integrated modelling discussion centres on its design and structural framework. Further elaboration on this methodology, including scenario configurations and simulations, are presented in Chapter 4.

Second, this study contributes to the existing literature by exploring probable outcomes and the calculation process within the framework of environmentally sustainable residual income (ESRI). This approach builds upon prior research conducted by Hueting (1993), Rambaud and Richard (2015), Brown (2016) and Pham et al. (2020), which highlighted the significance of considering the ecological value of resources in sustainable income assessments. In this chapter, the model extends this concept by enhancing the calculation methodology, integrating inputs from life cycle costing (LCC), life cycle assessment (LCA), management accounting, and economic modelling within the agricultural domain.

Third, this chapter gives a demonstration of the usage of integrated modelling, using cotton farms and tree offsets as a case study.

The findings of this study will be of interest to scholars, policymakers and practitioners who are interested in understanding which regulations are likely to effectively address climate change.

This chapter is structured as follows:

I. Section 3.2 provides a literature review and theory development;
II. Section 3.3 describes the model design, mapping and formula, followed by a case study to demonstrate how the module can be used; and
III. Section 3.4 presents the findings and concludes with a discussion of the findings.

3.2 Literature review and theory development

3.2.1 Identification of more sustainable[21] policy options

3.1.1.1 Strong and weak sustainability

Sustainable development comprises both weak and strong sustainability[22] (Ayres, Van Den Bergh & Gowdy, 1998). Weak sustainability[23] implies that as long as the total capital can be maintained, even if the quantity of natural capital decreases due to the creation of human-made capital, the sustainability criteria are met (Opschoor, 1998; Hediger, 1999). On the other hand, strong sustainability[24] holds that some natural

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[21] The term "more sustainable" in this thesis specifically related to improvement in GHG emissions reduction. In addition, while acknowledging the significant role of chemical fertilisers in enhancing crop yields, the discourse within this chapter is constrained to analyses related to GHG emissions, around the context of "strong sustainability." Given that cotton is not a staple crop, the focus remains specifically on the environmental impacts associated with GHG emissions, without delving into broader agricultural sustainability aspects.

22 The term "weak sustainability" and "strong sustainability" are specialized terms in environmental economics to describe the transformation and substitution relationship between economic value and natural resources (c.f. Dietz & Neumayer, 2007).

[23] Hediger (1999) explained weak sustainability as: "*'weak' sustainability is an economic principle which is founded within the body of neoclassical capital theory. It is a value principle with the necessary condition that some suitably defined value of aggregate capital - including human-made capital and the initial endowment of natural resources - must be maintained intact over time*". This means that if the total warfare is considered unchanged, the acceptability of environmental damage can be observed, such as in the case of mining coal and natural gas for energy. Although these activities contribute to pollution, the provision of electricity enhances people's living standards, thereby maintaining the balance between human capital and natural resources in the overall welfare.

[24] The concept of "strong" sustainability, rooted in the paradigm of ecological economics, recognises the economy as an open subsystem within the finite and non-growing global ecosystem, necessitating the preservation of certain attributes of the physical environment based on thermodynamic laws, specifically

capital cannot be substituted (or only to a limited extent) by human-made capital and may suffer irreversible harm (Hediger, 1999; Leach, Newell, Scoones & Mehta, 1999; Pearce, 1988; Málovics, Csigéné & Kraus, 2008). Consequently, this implies an "ecological value principle" that quantifies the overall "value" of the diverse array of natural capital from an ecosystem perspective (Hediger, 1998).

In the analysis and evaluation of sustainable projects and policies, strong sustainability should be adopted as the theoretical framework[25]. This is because, unlike the basic theoretical assumption of weak sustainability, that "all types of capital and the services and welfare generated by them to be expressed in the same monetary unit" which has been criticised by many researchers (e.g., Faucheux, O'Connor & Van Der Straaten, 1998; Ekins et al., 2003), strong sustainability distinguishes natural resources and defines them in different value categories (Turner & Pearce, 1990).

This distinction highlights one of the advantages of establishing an integrated modelling framework that integrates economic indicators and scientific models, as it aligns with the natural sciences' recognition of the varying renewability and recyclability of different natural resources.

Strong sustainability highlights the need to distinguish between natural and other capital resources, as economic development should occur within the limits of natural resource availability and regenerative capacity. It has the possibility of reflecting the

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requiring the constancy of the total stock of natural capital over time (Costanza, 1991; Daly, 1991; Pearce et al., 1994; Hediger, 1999). From the perspective of strong sustainability, damage such as N_2O causing harm to the ozone layer cannot be equated to the benefits gained from the increased application of chemical nitrogen fertiliser, in terms of human capital. Therefore, it is necessary for growers to take measures to restrict the use of chemical fertilisers or explore alternative nitrogen sources.

[25] According to Turner and Pearce's (1990) explanation, the first function of natural capital is to provide resources for production, while the second is to absorb waste generated during production processes and from the disposal of consumer goods. These functions act as a medium for environmental degradation and can be considered negative investments, depreciation or capital consumption. The third function of the environment is to provide the basic environmental conditions and requirements necessary for production, contributing to human welfare through what may be called "amenity services". These four functions of natural capital are closely related to sustainability (Ekins et al., 2003).

fundamental reality of the natural world through economic/accounting modelling integrated with scientific modelling, where different resources have varying degrees of renewability and recyclability. Failing to account for this distinction in such a framework could lead to inaccurate or misleading results, as it would fail to capture the differential impacts that various resources have on both the economy and the environment.

3.2.1.2 Understanding strong sustainability.

Assessing the economic and environmental sustainability of production methods and regulations is a challenging task. Sustainability encompasses both environmental and economic aspects. This has been recognised and discussed by researchers and international organisations; for example, it is interconnected through the Social Cost of Carbon theory. This theory quantifies the cost of environmental damage in monetary terms and has been studied extensively (Pearce, 1996; Pearce, 2003). In addition, the Brundtland Commission defined sustainable development as "development that meets the needs of the present without compromising the ability of future generations to meet their own needs" (United Nations Brundtland Commission, 1987).

Arguably, to be economically sustainable over the long term, a project must cover its costs and internalise any negative externality; and to be environmentally sustainable, an organisation must either have no uncompensated negative environmental impact from its emissions or reverse any environmental degradation caused from the production/consumption cycle (Patzek, 2004). This concept has been discussed in various studies (Gray, 1992; Richard, 2012; Brown, 2016). As theorised by Gray (1992): "Additional costs must be borne by the organisation if the organisational activity were not to leave the planet worse off" (p. 419).

To operationalise sustainability, it is essential to incorporate environmental considerations as responsibility within an organisation's decision-making processes (Málovics, Csigéné & Kraus, 2008; Robertson & Swinton, 2005). This requires a comprehensive evaluation framework that considers the environmental impact of a product throughout its lifecycle (de Boon, Sandström & Rose, 2022). Key factors to consider include the ecological cost of primary materials, emissions from producing intermediate products, future emissions during the product's use, and waste generated during disposal (Pham et al., 2020). Such an approach enables stakeholders to gain a comprehensive understanding of the true cost of a product and make informed decisions that support sustainable practices. By integrating environmental considerations into decision-making processes, operationalising sustainability can promote the adoption of environmentally friendly practices and enable the emergence of sustainable systems (cf. Davies & Simonovic, 2011; Van Delden et al., 2010).

In summary, I argue that to assess the sustainability of a given product and its system of production, it is necessary to consider its environmental impact together with its economic performance. This includes taking into account the ecological cost of primary materials, emissions from producing intermediate products, future emissions during the product's use, and waste generated during disposal. By considering these factors, decision-makers and other stakeholders can gain a comprehensive understanding of the true cost of a product and make informed decisions that support sustainable practices. Operationalising sustainability requires integrating these considerations into decision-making processes throughout a product's lifecycle. Such an approach promotes the adoption of environmentally friendly practices and enables the emergence of sustainable systems. Ultimately, operationalising sustainability requires an integrated approach that incorporates environmental considerations into decision-making processes from production to disposal at the organisational level (Richard, 2012; Brown, 2016; Gray, 1992) and at the economy level (Hueting, 1993; Hueting, 2011).

3.2.2 Evaluation mechanisms

As summarised above, sustainability encompasses a multiplicity of perspectives. To maintain tractability in this analysis, the focus is on economic and environmental sustainability. The question of how to assess the economic and environmental sustainability of production systems has been extensively investigated separately (cf. Pearce, 1976; Penman & Zhang, 2002; Stockle et al., 1994; Faeth, 1993). The results of these studies have implications for the development of information models for production systems. The climate challenge confronting humanity today has presented novel obstacles that demand adaptive adjustments and innovative sustainability of production systems, building upon prior research, to meet emerging challenges and achieve sustainability requirements.

Nevertheless, although efforts have been made to advance existing approaches, the majority of these have been criticised for their incompleteness in addressing the needs of policymakers, investors and other stakeholders, and need non-financial information/models to assist with production system options decision-making (cf. Kimbro,2013; Sinha & Datta, 2020; Luthra et al., 2018).

The subsequent section presents and evaluates several key models and methods that have been utilised to assess the sustainability of production systems. The aim is to identify the most notable aspects of each model that lend themselves to integrated modelling, which are highlighted.

3.2.2.1 Critical analysis of key economic performance indicators

Key performance indicator (KPI) models have widely adopted approaches to guide decision-making involving the allocation of resources. A fundamental principle that underpins these models is that for a project (or decision) to be deemed economically sustainable, it ought to be profitable and provide a return on capital investment. Key models used include Accounting Rate of Return (ARR), internal rate of return (IRR), Net Present Value (NPV) and Cost-Benefit Analysis (CBA). Appendix A provides a summary of these terms.

However, as mentioned in the introduction, financial analysis methodologies are subject to criticism for their inadequacy in addressing sustainability-related concerns. To that end, it is important that these approaches are augmented with other models that cater to environmental characteristics (Ryszawska, 2016; Van Delden et al., 2010).

ARR has a significant advantage over IRR and NPV in terms of its ease of calculation, allowing users to quickly gain an initial perspective on the investment's profitability in its early stages. However, the concept of “time value of money”, which is addressed by both IRR and NPV, is not considered in ARR. As a result, ARR is criticised for its inability to evaluate the performance of long-term investments. This can pose a challenge for sustainable projects, especially in the context of agriculture where operations do not typically end in the short term. Given the high capital requirements involved, overlooking the opportunity cost of capital can result in an incomplete assessment of the true change in project value.

NPV and IRR are both analysis methods based on predicted cash flows, with the assumption that returns gained from investments can be reinvested at the cost of capital. In contrast to IRR, NPV takes into consideration the opportunity cost of capital investment. However, both models are heavily reliant on the estimates of the required rate of return prediction. Thus, the quality of the analysis is highly dependent on the investor's comprehension of expectations and their willingness to select sustainable projects for investment.

Furthermore, these capital investment instruments have long been subjected to criticism for their lack of specificity and adaptability to different industries and projects. Although organisations attempt to adjust their corresponding discount rates for different project types, the risk assessment process is considered to be inaccurate, which may not ensure the use of unique risk-adjusted discount rates for individual projects (cf. Brigham, 1975).

In addition, all these measurements place a high focus on economic sustainability and do not incorporate measures of environmental sustainability. None of these models alone can fully account for sustainability concerns. Rather, they are useful for evaluating the economic value and predicting the outcomes of existing environmental projects and regulated standards.

However, under the strong sustainability framework, production systems and organisations should share more costs than just production costs, as they cause damage to natural resources (cf. Málovics, Csigéné & Kraus, 2008). These costs are currently borne by society, and many of the losses are not linked to economic value. Therefore, relying solely on these accounting indicators might result in an underestimation of production costs and a failure to accurately reflect losses incurred by the entire system. This could potentially lead to biases in policymaking and decision-making.

3.2.2.2 Cost-Benefit Analysis (CBA)

According to Pearce, the CBA method was first proposed by Jules Dupuit (1844) and designed to measure the social worth of a specific project (Pearce, 1998). The concept of CBA is widespread and utilised by researchers and policymakers in various countries for projects related to natural resources or environmental damage. For instance, the United States Flood Control Act (1936) and the "Green Book" (1950) are examples of such regulations aimed at regulating water projects.

The concept of CBA was expanded to connect with social wellbeing, in the belief that compensation could lead to an overall net gain in wellbeing. However, this approach faces significant challenges, as the accuracy of estimations relies heavily on two factors: first, the proper identification of stakeholders and second, the scope of “loss” that is specified by the regulation.

In addition to the aforementioned limitations, the project-based measurement method fails to account for the wider impact and interconnectivity of the entire industry system. This oversight can lead to a phenomenon known as pollution transfer, wherein the pollution is shifted rather than reduced. Therefore, project-based measurement methods may not be sufficient to comprehensively assess the sustainability implications of a project.

Failure to address these issues can result in an imbalance of justice. Consequently, CBA may not be the most suitable measurement method for assessing sustainability.

In the context of agriculture regulations, the CBA method has a worldwide impact on researchers and is often used to examine the impact of energy efficiency improvement plans or to compare different plans (cf. de Gorter & Just, 2010; Balmford et al., 2018). However, as researchers have pointed out, CBA does not fully account for opportunity costs or costs that occur over time, and therefore additional analysis may be required to complement the analysis. These issues indicate that CBA may not be the optimal measurement method for evaluating sustainability performance.

3.2.2.3 Life Cycle Costing (LCC)

Goh and Sun (2016) state that the origins of LCC can be traced back to the late 1950s in the UK, but its specific origin document is not clear. What is certain is that the LCC method was invented earlier than LCA, and was more widely adopted. The LCC concept provides only a framework for cost management methods, and its specific implementation is highly dependent on the particular domain and field of discipline (Blanchard, 1978; Swarr et al., 2011). However, the primary focus of LCC research, including sustainability topics, remains cost savings (cf. Woodward, 1997; Zou et al., 2019; Gorjian et al., 2022). The research in Australia using LCC for sustainability contents mostly focuses on buildings; few are in agriculture, which shows the implication is quite new.

LCC is designed to calculate the product cost "cradle to grave". Some try to combine it with the sustainability topic, as they believe the extent of product usage life and more accurate costs will lead to less waste and therefore improve the energy efficiency, resulting in improved sustainability (cf. AbouHamad, Mona, & Metwally Abu-Hamd, 2019; Weldu & Assefa, 2017). However, even with this theory's extent, LCC is still not an indicator of, nor involved in, sustainability measurements but is a cost measurement indicator. Some see this limitation and start to research the combination of LCC and other models, such as LCA (cf. Heidari, Heravi & Esmaeeli, 2020; Heijungs, Settanni & Guinée, 2013) and energy use efficiency (cf. Zhou, Jiang & Qin, 2007).

In the agriculture background, LCC can be used to analyse the competitive advantage of sustainable products (cf. Zhou, Jiang & Qin, 2007; Pergola et al., 2018), cost analysis of sustainable innovation (Peña, Rovira-Val & Mendoza, 2022), and practical improvement with LCA (Mohamad et al., 2014). However, the focus of LCC research is still on cost assessment and cost saving.

3.2.2.4 Life Cycle Assessment (LCA)

The notion of conducting an LCA first emerged in the 1960s, as a response to growing concerns about environmental degradation and the increasingly limited accessibility of resources (Guinée et al., 2011). During its early development, the primary focus of LCA remained centred on energy usage and the generation of solid production waste. The analysis of LCA is based on the framework provided by the International Organization for Standardization (ISO) standards[26], but it also allows for flexibility in its application, as it enables stakeholders to assess the environmental impacts of new technologies, products or services throughout their entire life cycle. By analysing the life cycle stages, from raw material extraction to end-of-life disposal, LCA can identify potential areas of improvement and help innovators make informed decisions that promote sustainability.

In the agriculture sector, researchers have used LCA to analyse the new challenges brought by sustainability requirements (cf. Matos & Hall, 2007), compare sustainable plans (cf. Tricase et al., 2018), internationalise food production standards (cf. Roy et al., 2009), analyse possibilities for newly innovated cropping technology (cf. Hanafiah et al.,

[26] The build-up of LCA clarifies the responsibility system and the upstream and downstream of industrial production, which identifies that the emission calculation of the final product should include historical emissions that occurred due to the production of its primary materials (ISO 1440).

2022), and provide decision-making information on sustainability (cf. Hasler et al., 2015; De Backer et al., 2009).

However, the applicability of LCA is limited by the availability of data, as there may not be readily accessible data available, and the quality of data directly impacts on the quality of the LCA analysis (Curran, 2014).

This can result in the issue of inadequate adaptation to changing environments. Specifically, while LCA may help identify the most ecologically efficient solution from a range of alternatives, the practical ecological efficiency that can be achieved through redesign and technological innovation may often fall short (Bjørn et al., 2015). Moreover, LCA itself does not encompass all relevant aspects of sustainability. Many sustainability researchers, such as Jevons (1865), argue that focusing solely on ecological efficiency is insufficient, as improvements in ecological efficiency at the product or technological level may be offset by increasing demand (York & McGee, 2007).

Furthermore, LCA by itself does not offer any economic analysis or costing, which makes it challenging to support decision-making processes that require balancing sustainability and economic considerations. It is a tool that can identify the most ecologically efficient way of delivering a particular service from a range of predefined alternatives, but it does not recognise the importance of various services (Moltesen & Bjørn, 2018).

As noted earlier, several researchers have attempted to integrate LCA with LCC, due to their shared focus on analysing the production process and its external impacts. However, as the standards for such calculations have not yet been developed to align with each other, it is challenging to conduct both processes within the same system (Heidari, Heravi & Esmaeeli, 2020).

Furthermore, from an agricultural perspective, due to the complexity of stakeholders and the ecosystem itself, relying solely on the application of LCA and LCC without integration with other models (such as crop models) can significantly bias the reliability of sustainability analysis outputs (De Luca et al., 2017).

3.2.3 Penman decomposition model

Penman (2003) is one of the most well-theorised decomposition models. This model extends earlier attempts to decompose causal factors relevant to explaining financial performance, with links to common valuation models. The underlying concept of the Penman (2003) decomposition model is that a company's financial performance can be decomposed into operating performance (accounting measurements) and financial performance. This separation is desirable, as it enables the demonstration, allocation and tracing of specific performance outputs. The operating performance, driven by net operating profit after tax (NOPAT), reflects how well the company is generating profits from its operations; and the financial performance built on equity reflects how well the company manages its financial activities.

The overall value of the model lies in its provision of a comprehensive approach for exploring and explaining the factors affecting a company's profitability and risk level. It integrates various financial indicators and analysis tools, including cash flow, balance sheet and income statement, among others, to offer more comprehensive analytical results[27].

Another value of the model is its ability to assist analysts and investors in better understanding and evaluating a company's future profitability and cash flow. Through the utilisation of this model, analysts and investors can get better information support when predicting a company's future performance and making corresponding investment decisions.

[27] For Penman (2003) decomposition modelling, the calculation of return on equity needs equity data which is based on the financial statement which include none-cash loss such as depreciation. Thus, as depreciation generally does not necessarily appear as a standalone element in Penman's decomposition, it is inherently part of the financial statements which feed into the decomposition analysis.

In this research, a specific value adopted from Penman's work is the decomposition level, which offers both a comparable indicator of the desired evaluation outcome and lower indicators that explain how the outcome is achieved. By incorporating variables and their relationships, this decomposition structure can provide an evidence-based analysis that improves information accuracy.

Breaking down complex systems into several lower-level indicators can help different stakeholders improve their understanding of the interaction between different components, thus building a common framework of understanding. Furthermore, the design of this model allows for flexibility, which is particularly important when applied to the sustainability field, where continuous integration of rapidly evolving technologies is necessary. The overall system framework and levels of relative independence in the decomposition model provide flexibility in applying the decomposition model in different ways as needed. This allows for adjustments and involvements under the system framework, with adjustments only occurring at certain levels.

Combined with the above, the decomposition-structured model can assist decision-makers in quickly understanding the relationships between different components of a complex system and identifying key variables that need adjustment. Such adjustments can occur at various levels, from low to high, providing corresponding results and insights into the underlying causes of problems. Moreover, the flexibility of this model allows multiple industries to fill the model's framework with their industry-specific content, to provide decision-makers with greater depth and breadth of information, enabling them to develop strategies for optimisation and make better-informed decisions.

3.2.4 Integrated modelling and WESM model

3.2.4.1 Integrated modelling

Some authors have proposed methods to construct integrated economic and environmental modelling aimed at addressing sustainability challenges through the incorporation of economic and environmental indicators. For instance, Hofkes (1996) proposed an integrated economy-ecology model that incorporates pollution as an efficiency indicator in the measurement of production outcomes; Shen, Kyllo and Guo (2013) developed an urban sustainability measurement model which integrated three primary domains (urban economy, society and environment); Sarker et al. (2021) developed a theoretical model for sustainability assessment by combining the Balanced Scorecard and the Fuzzy multiple-criteria decision-making approach in a weighted manner.

To meet the characteristics of effectively integrated modelling, the integration framework should address three areas: (i) "plug-and-play" integration of existing models; (ii) plug-and-play interaction; and (iii) intuitive, real-time interaction (Muth Jr & Bryden, 2013). For further discussion, see Pham et al (2020) for a discussion of the relative merits of simulation and integrated modelling.

In the agriculture sector particularly, El Chami and Daccache (2015) have attempted to integrate climate modelling, LCA and the cropping model to assess sustainability for winter wheat, using CBA, NPV and IRR as economic indicators.

Belcher, Boehm and Fulton (2004) analysed the necessity of comprehensive modelling for agricultural ecosystems and established an integrated model for agricultural ecology, based on the Sustainable Agroecosystem Model framework. They also suggested that the economic and environmental sustainability of the system is contingent upon biophysical constraints, which determine feasible management strategies, in terms of technology, agronomy and economics.

3.2.4.2 Water and Economic Sustainability Performance Measurement (WESM) model

Pham (2016) and Pham et al. (2020) has attempted to fill the gap by building the Water and Economic Sustainability Performance Measurement (WESM) model[28] to assist with decision-making, combining Penman (2003), science-based modelling of a cropping system, sustainability theories and accounting research.

The salient feature of this model is the integration of the Penman (2003) decomposition modelling with key sustainability measures and accounting assessments. This framework provides a mechanism for utilising high-level indicators that combine economic performance (net profit after tax) with considerations for primary production systems. It elucidates a theoretical model for enhancing the quality of sustainability-related information and decision-making through the use of integrated modelling.

However, this model still has room for improvement as certain aspects need to be addressed. Its major limitation lies in using the water efficiency rate as the highest indicator, which carries a risk of being ineffective.

As pointed out in the LCA analysis section, improvements in resource use efficiency can increase overall resource consumption, thus exacerbating pollution levels instead of reducing them (Jevons, 1865). Additionally, using the "profit-to-water cost ratio" as the primary criterion for determining an organisation or industry's sustainability can lead to significant problems, including the risk of falling prey to the Jevons Paradox, where an increase in efficiency in resource use generates an increase in resource consumption, rather than a decrease (Jevons, 1865; Sears, Louis et al., 2018; Ceddia et al., 2013; Hadjimichael et al. 2016). In addition, from an agricultural perspective, WESM is simulated under Australian irrigated cotton farm assumptions. Using this ratio limits its

[28] I note that models are developed for many reasons, and accordingly integration of different models requires attention to the purpose for which both the models were designed for, and the purpose for the integration.

direct comparability to other cropping options such as rainfed cotton due to differences in the production systems.

Furthermore, the model does not reflect economic value, as it only calculates the productivity of water at a predetermined price without factoring in the capital, which may pose a challenge for long-term evaluation of economic performance.

3.2.5 Two economic theoretical approaches - Pearce and Hueting

While each of the models discussed above is limited in some way, as demonstrated by Pham et al. (2020) and others, they may be augmented with additional elements to address key limitations of the original model. To that end, Pearce (1976) and Hueting, Bosch & de Boer (1992) provide insight into several elements that are amenable to integration with the WESM and other models, to form a more complete integrated modelling framework to understand and evaluate the economic and environmental sustainability of cropping systems.

Pearce (1976) introduced an analytical theoretical model to elucidate how organisational decision-making can impact on the overall assimilative capacity of the environment. The model integrates observations on the assimilative capacity of ecosystems to accommodate industrial impacts and incorporates the essential characteristics of cost-benefit analysis (CBA) in policy evaluation. This model shows that even in cases of ecologically unsustainable resource utilisation, it is possible to achieve a Pareto-optimal allocation and achieve sustainable production. The pollution level should be no larger than the environment's assimilative capacity. This occurs when organisations apply CBA to activities that impose costs on the environment's assimilative capacity, without bearing these costs themselves. Consequently, an irreversible process of ecological degradation persists until the remaining assimilative capacity is depleted. However, Pearce (1976) presents the theoretical model without explicit formulas, calculations or case study demonstrations, which may hinder the practical application of this concept (Brown, 2016).

Hueting, Bosch & de Boer (1992) builds up a theoretical framework to enable the integration of environmental information into more conventional economic analysis, utilising the concept of competing functions. He develops and applies a novel method for calculating environmentally sustainable national income (eSNI). This calculation involves estimating the opportunity costs incurred at the national level for restoring any natural capital (environmental sustaining costs) and subtracting this from the national income. In the theoretical framework of calculating eSNI, the environment is defined as the non-manufactured physical environment, specifically ecosystems encompassing water, air, soil, plant and animal species, as well as life support functions.

Hueting argues that economic production engenders competition with environmental functions, such as the conversion of forests into agricultural land, resulting in a decline in their relative abundance and limited availability (Hueting, 2011). Consequently, the shadow prices of environmental functions increase and their values (derived from prices multiplied by quantities) ascend from zero to progressively higher positive values. This presence of opportunity costs precludes the assignment of environmental values without market prices (e.g., clean air), based on their "restorative value" subsequent to impairment caused by economic production.

One of the advantages of eSNI is its conceptual inclusion of ecosystem services, which are used by various groups to assess the value of natural capital and provide a theoretical foundation for integrating environmental capital into economic models. Moreover, eSNI allows for flexibility in valuing environmental costs that vary due to technological advancements, based on the valuation of governance and restoration costs.

3.2.6 Environmentally sustainable residual income (ESRI)

ESRI is a method that allows market participants and other stakeholders to apply the modification to balance sheets and income statements, according to private and preferential information, to calculate a better reflection of value (Brown, 2016: calculation of environmentally sustainable residual income).

The background of this theory is based on the recognition that most of the natural resources accessible are constrained by the earth's sustaining limit (Boulding, 1966). In addition, a shadow cost for environmental resources is more than what has been identified and should be identified as a cost that can be seen. Furthermore, a shadow cost for environmental resources remains underestimated and should be disclosed as a visible cost (Hueting, Bosch & de Boer, 1992).

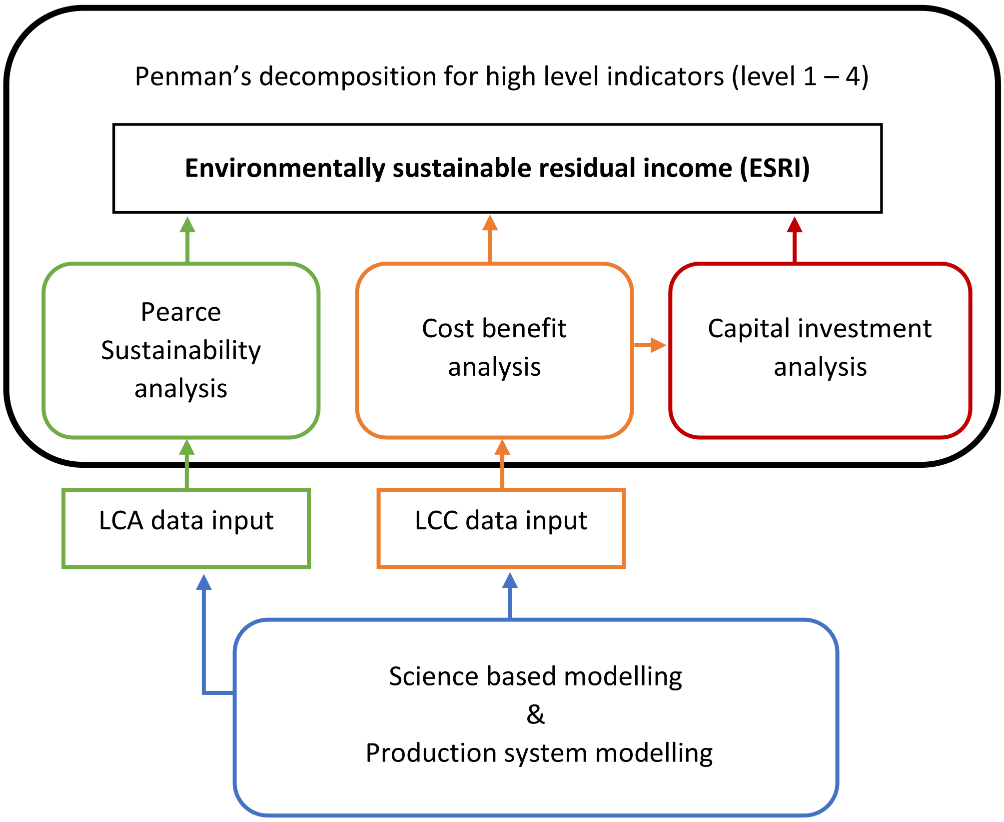
The concept of environmental residual is first described by Gray (1992) as “calculations of which additional costs must be borne by the organisation if the organisational activity were not to leave the planet worse off” (p. 419). Richard (2012) and Rambaud and Richard (2015) extend this insight, proposing that the key actors in the utilisation of natural resources, businesses and individuals bear a responsibility to “decrease their absolute degradation” and the value of natural resources as public assets in national accounts (Obst et al., 2016; ABS, 2017). Thus, pollution is treated as a cost/damage of desirable public assets and should be included in the accounting framework (Rambaud and Richard, 2015).

Brown (2016) further developed the theory by introducing ESRI value as a high-level indicator of the environmental value system, with the additional valuation and calculation for environmental damage as the environment residual account. The ESRI filled this gap by introducing the capital charge and environmental charge in the accounting model, in which the capital charge represents the opportunity cost on capital and environment sustaining cost (ESC) as the shadow cost identified and remediation cost if it is reversible.

However, while Brown (2016) explains the theory of ESRI, he does not offer any supported formulas, calculations or applications for sustainability practices, nor does he provide any specific industry examples.

Figure 3.1 depicts the extended structure of the theories within ESRIM, as conceptualised in the model.

Figure 3.1: Model 1 Conceptual model



This chapter proposes that the approach is to integrate LCA, LCC, ESRI and Penman-style decomposition, to build an integrated modelling system which enables a novel form of economic and environmental performance evaluation. This integration enables the estimation of high-level indicators and causally linked modelling that allows for the identification, explanation and evaluation of alternative options.

3.3 Decomposition modelling

3.3.1 Overview

This study builds upon the WESM decomposition model of Pham (2016) and Pham et al. (2020). I extend that work by utilising a similar financial ratio analysis decomposition method to link the overall financial status to the operational level of a crop business and provide the articulation to include[29] nitrogen fertiliser production and on-farm nitrogen management, alongside water management. In this chapter, the decomposition model is further expanded to reflect the ESRI value as the highest-level indicator, instead of the efficiency-focused ratios in Pham (2016), further linking Penman (2003) to integrated agriculture modelling.

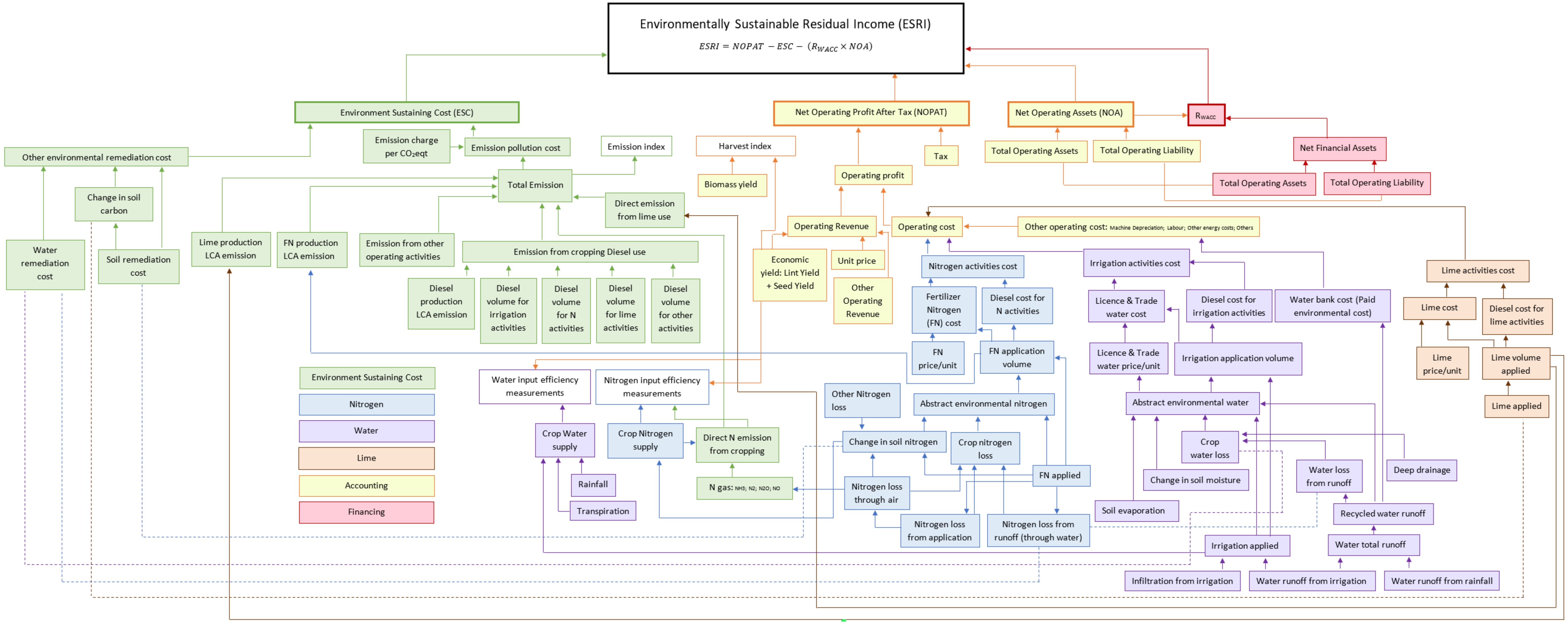
Figure 3.2 presents the structure of the ESRIDM model developed in this chapter. The decomposition model presented herein depicts a comprehensive overview of the potential interconnections between farm operating level activities and changes in high-level indicators. The model serves as a tool for identifying and tracking the specific steps or processes that have the greatest potential to make a significant impact.

As noted in the introduction and literature review and in contrast to Pham et al. (2020), the highest-level indicator in this model is ESRI, which is a more comprehensive measure of environmental and economic performance (as theorised above). The model separates the high-level indicators of ESRI into six sub-modules: farm operation accounting module, farm performance financing module, nitrogen management module, water management module, environment sustaining cost (ESC) module, and indicators module. Each module includes any environmental remediation cost that has already been regulated as mandatory (e.g., water bank), and others remain in the ESC module (e.g., nitrogen emission). The ESC module includes the LCA emission of

[29] The decomposition method is one of the general practices in accounting and financing to analyse and compare a firm's financial statement and operations. It was adopted by Pham (2016) to apply in the cotton farm water management sector.

nitrogen fertiliser production cost, farm operating pollution cost and any other environmental remediation cost. The model presented in Figure 3.2 focuses on estimating and analysing the operational activities of a cotton farm and is designed to provide insights into economic and environmental performance in the context of cotton production. Doing so may assist stakeholders to optimise their decision-making processes and improve overall farm performance from both economic and sustainability perspectives, by providing the final output ERSI and step-by-step outputs in different modules.

Figure 3.2: Structure of ESRIDM integrated modelling



3.3.2 Model functions

Necessary input data for calculation are shown in Figure 3.3.

Farm management and practices data is needed as the framework of calculation, which includes decisions related to cropping and other agricultural activities. Examples of such decisions include the size of the farm, the amount of nitrogen fertiliser used, the quantity of diesel consumed, and the method of irrigation employed. These data are inputted into cropping models to calculate the output of these decisions and information during crop growth. The output needed should include information on yield, emissions, nitrogen usage and water usage.

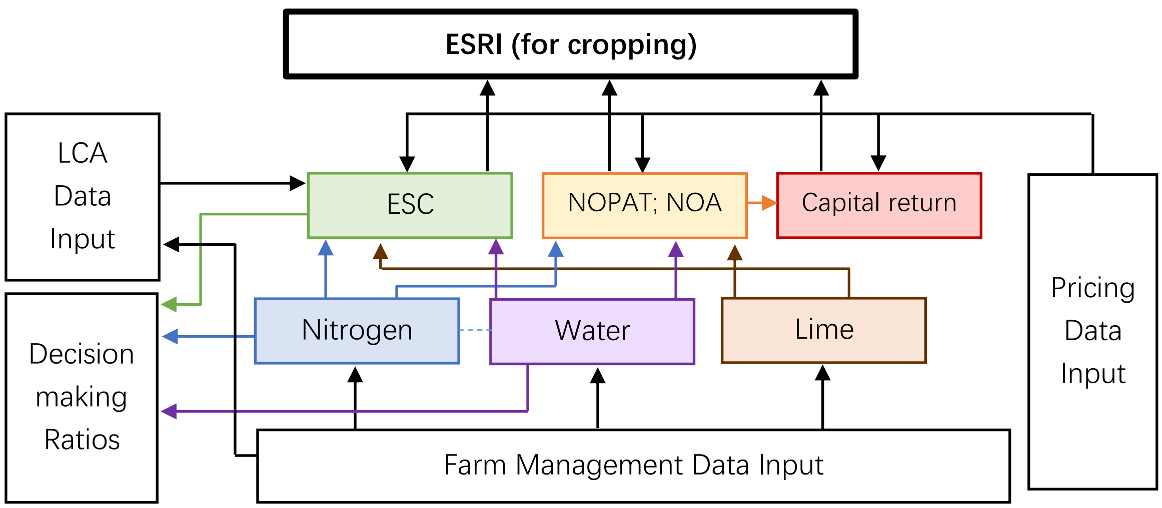
The model incorporates a variety of inputs, including historical and current data related to pricing, farm management and LCA. These inputs are integrated into the model's algorithms, which simulate the operations of the farm and the impact on the environment over a specific period. The outputs generated by the model provide information that enables stakeholders to make informed decisions through useful ratios (e.g., nitrogen emission per yield ratio) and values (e.g., profitability, accumulated nitrogen emission, ESC value) which reflect both farm operation conditions and the environmental impact that might transit to society.

These decisions may involve choices regarding fertiliser selection, irrigation planning, allocation of inputs and pricing strategies. Through the analysis of these outputs, stakeholders can evaluate different scenarios and strategies to optimise their objectives. For example, stakeholders can test various scenarios to enhance resource use efficiency, such as exploring the use of dripping irrigation for improved water efficiency. Similarly, they can examine the use of slow-release nitrogen fertiliser to enhance environmental sustainability.

These output data affect the higher levels directly, and irrigation methods affect the nitrogen module. For example, the amount spent on water licences impacts on operating costs, thereby affecting the net operating profit after tax (NOPAT).

By utilising the model and analysing its outputs, stakeholders gain insights into the potential outcomes of different decisions and actions. This enables them to assess the economic and environmental implications of various strategies and make informed choices that align with their goals for optimisation, efficiency and sustainability.

Figure 3.3: Data input for cropping system with focus on nitrogen and water



Because the capital return part of the calculation is simplified, market risk and price data could also be an input to that finance module.

(I) Sustainability outputs

The highest-level output of integrated modelling is the potential combined value output of the selected industry production model and the selected sustainability strategy. This means that one function of this integrated modelling is to measure the feasibility and effectiveness of the sustainability solutions or regulations selected by decision-makers and to assess the sustainability of a certain industry/organisation.

(II) Relationship and links focus

Using the Penman (2003) level system and management modules linked to accounting performance, this integrated modelling approach provides output indicators and illustrates the underlying relationships between different management decisions (e.g., production and fertiliser decisions) and their effects on related modules and final output.

This modelling approach enables the tracing of any indicators/output at a high level, providing explanations for the results and highlighting the strength of the correlation with relevant objects associated with the computed result.

(III) Systems thinking and modular thinking in decision-making

This integrated modelling approach provides the industry with a supply chain map, LCA analysis of ingredients, and scientific analysis of activities related to cropping operations. It enables decision-makers to assess the profit or loss, cost, and risk of management decisions, from both economic and sustainability perspectives.

Furthermore, the modular design of the model allows users to selectively focus on specific modules, as each module within the model has complete functionality within the scope of the analysis.

3.3.3 Modelling explanation

Table 3.1 provides an overview of levels in the ESRIDM model. In this table, the ESC module and accounting module (NOPAT and NOA) start from level 2; the nitrogen module, water module (irrigation) and lime module start from level 4.

Values such as Rwacc and traded water licence price are not in the level category, because they are values used for calculation, not outputs of management decisions.

Table 3.1: Level list demonstration for nitrogen management related emissions

Top-level	ESRI			
Level 2	Environment Sustaining Cost (ESC)	Net Operating Profit After Tax (NOPAT)	Net Operating Assets (NOA)	Rwacc[30]
Level 3	Emission pollution cost	Operating Revenue Operating Cost Other Operating Revenue Other Operating Cost	Total operating assets Total operating liabilities	
Level 4	Total emission	Economic/ Lint/ Seed yield Cost of Nitrogen (N) activities Cost of irrigation activities Cost of lime activities		
Level 5	FN production LCA emission Lime production LCA emission Direct N emission from cropping Emissions from cropping diesel use Emission from other operating activities Emission from other sources	Cost of EnergyN Cost of EnergyW Cost of EnergyL Cost of FertiliserN Seed cost Irrigation cost Lime cost		

[30] The formula is as follows:

$$WACC = (Equity\ Value\ /\ Total\ Firm\ Value) * Required\ Rate\ of\ Return + (Debt\ Value\ /\ Total\ Firm\ Value) * Cost\ of\ Debt$$

In this equation, "Required Rate of Return" represents the expected return on equity investment, "Equity Value" refers to the market value of equity, “Debt Value” is the market value of debt, and “Cost of Debt” is the expected rate of return required by debt holders.

The net operating assets (NOA) represent the operating assets (e.g., inventory, accounts receivable and fixed assets) net of operating liabilities (e.g., accounts payable) and can be used as a measure of the total capital employed in the farm (equity value).

Thus, in this model, multiplying NOA by the required rate of return can estimate the equity component of a farm's Weighted Average Cost of Capital (WACC) or determine the expected return on equity investment.

Note: I) FN refers to nitrogen fertiliser

II) For NOA in this chapter and case study, to simplify the calculation, an assumption of cotton farm market value is used instead of the calculation amount.

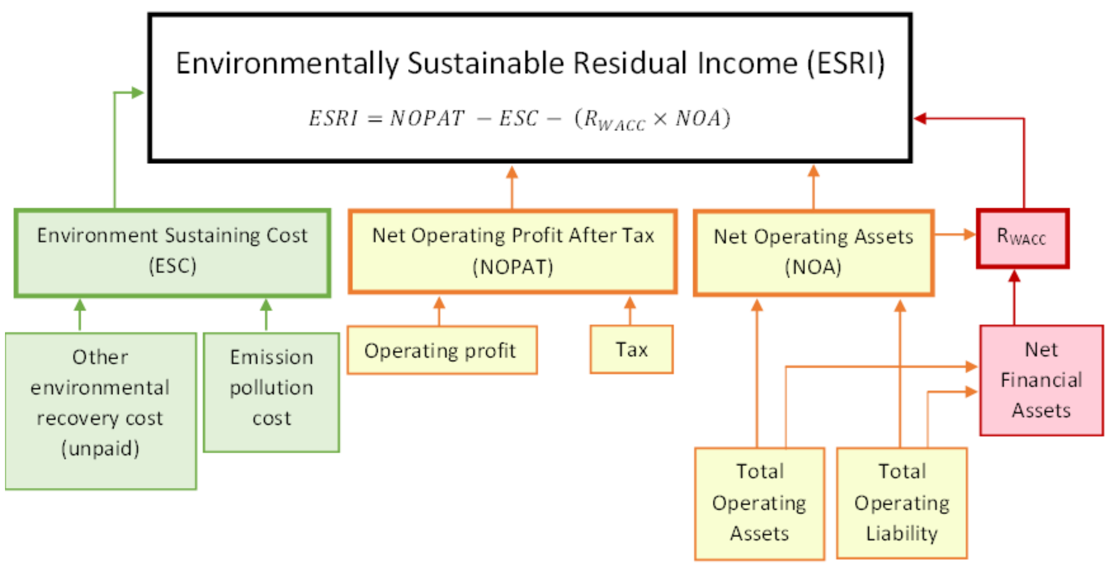
In addition, additional ratios in the ESRIDM model are designed to evaluate performance and conduct comparisons between different organisations or different years within one organisation. Thus, the ratios are independent of levels and should be adjusted to fit the industry ESRIDM is used to test.

III) It should be noticed that the various items mentioned in Table 3.1 are limited to nitrogen emission focus of this thesis and following the formulae developed and explained in this chapter. Other environmental effects such as soil carbon and deforestation/afforestation have been summarised in Figure 3.2 as 'other environmental remediation cost'. A more detailed treatment of these factors is beyond the scope of this thesis and could be part of future research.

3.3.3.1 High-level indicators

This section provides an explanation for level 1 and level 2 indicators.

Figure 3.4: High level indicators – level 1



Level 1:

$$ESRI = NOPAT - ESC - (R_{wacc} \times NOA)$$

Where:

NOPAT = net operating profit after tax

ESC = estimate of the impact/cost of irreversible damage to environmental resources and remediation cost of reversible damage to environmental resources. (e.g., cost of water purification, mark-up of water price and fertiliser price due to regulation)

R_{WACC} = expected return rate on investment

$$NOA = Net\ Operating\ Assets$$

$$= (Total\ Operating\ Assets) - (Total\ Operating\ Liabilities)$$

This indicator is useful because it shows the profitability of the farm. Here, the capital cost in the original formula – "the book value of owners' equity" in Brown (2016) – has been replaced by "net operating asset" to simplify the calculation. One reason for this simplification is that the main focus of this research is on nitrogen management. While acknowledging the significance of considering capital return, fully demonstrating the financial model would veer away from the primary objective of this study. Further detailed calculations should utilise the market price of the cotton farm to perform calculations of NOA.

Level 2:

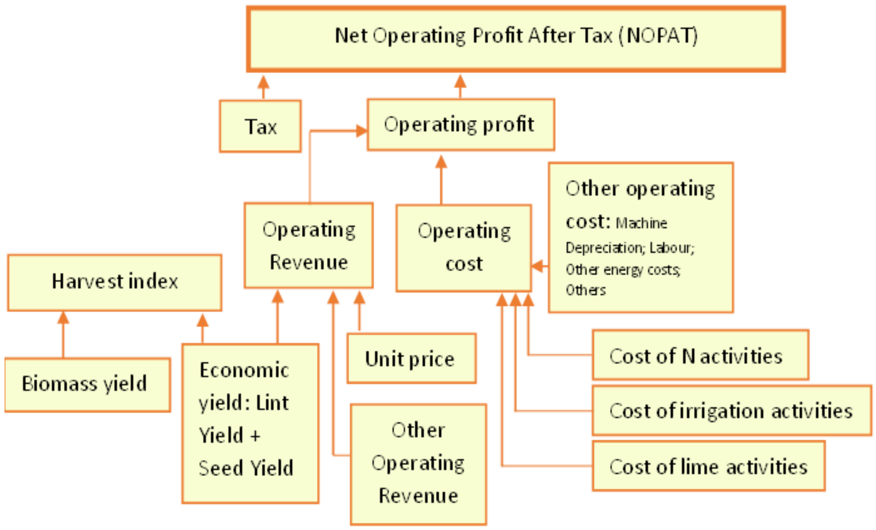
There are three components in this level: Environment Sustaining Cost (ESC), Net Operating Profit After Tax (NOPAT) and Net Operating Assets (NOA).

The ESRI, NOPAT, NOA and ESC are all present as value, and R_{WACC} is present as rate. Due to the complexity of the model, the following levels are explained under the NOPAT and ESC modules separately.

3.3.3.2 Level 2 NOPAT module

Net Operating Profit After Tax (NOPAT) is determined by the operating profit and taxes paid. A higher NOPAT value corresponds to a greater annual profit per hectare.

Figure 3.5: Level 1 indicator – Net operating profit after tax (NOPAT)



$$NOPAT = NOPBT * (1 - Tax)$$

$$\begin{aligned} &Operating\ Profit \\ &\quad = (Operating\ Revenue - Operating\ Cost) \\ &\quad + (Other\ Operating\ Revenue - Other\ Operating\ Cost) \end{aligned}$$

Level 3:

Operating Revenue

$$\begin{aligned} &Operating\ Revenue \\ &\quad = Lint\ yield\left(\frac{kg}{ha}\right) * Lint\ yield\ price\left(\frac{\$}{kg}\right) + Seed\ yield\left(\frac{kg}{ha}\right) \\ &\quad * Seed\ price\left(\frac{\$}{kg}\right) \end{aligned}$$

Operating Cost

$$\begin{aligned} &Operating\ Cost \\ &\qquad = Cost\ of\ N\ activities + Cost\ of\ irrigation\ activities \\ &\qquad + Cost\ of\ lime\ activities \end{aligned}$$

In this formula, "Operating Cost" should include the costs for main operating activities that can be identified as on-farm emission drivers. This includes costs incurred for the top three GHG emission drivers of cropping (Sevenster et al., 2022).

"Cost of N activities" is the total of costs related to on-farm nitrogen input management activities, including nitrogen fertiliser, seed and fertiliser application costs, such as diesel cost.

"Cost of irrigation activities" is the total costs related to on-farm water input management activities including purchase of water licence, traded water licence cost and irrigation costs, such as diesel cost for pumping. In this research, to simplify the calculation, "traded water licence cost" is assumed to be zero. It is only used for demonstration purposes in the theory development chapter.

The term "lime activities cost" refers to the total costs associated with on-farm soil improvement management activities, specifically the process of liming. This includes the cost of purchasing lime as well as the energy costs associated with the application of lime, such as diesel expenses.

Other Operating Revenue

"Other Operating Revenue"[31] includes other revenue from farm operating activities.

[31] In Chapters 4, the calculation of other operational revenue is omitted to streamline the calculations, ensuring an emphasis on emissions rather than a focus on water management.

For example: in the Australian cotton farming content, one of the "other operating costs" can be the gain from a traded irrigation licence, which can be calculated by:

$$Gain\ from\ traded\ irrigation\ licence = (Traded\ price - Licence\ price) * Traded\ amount$$

Other Operating Cost

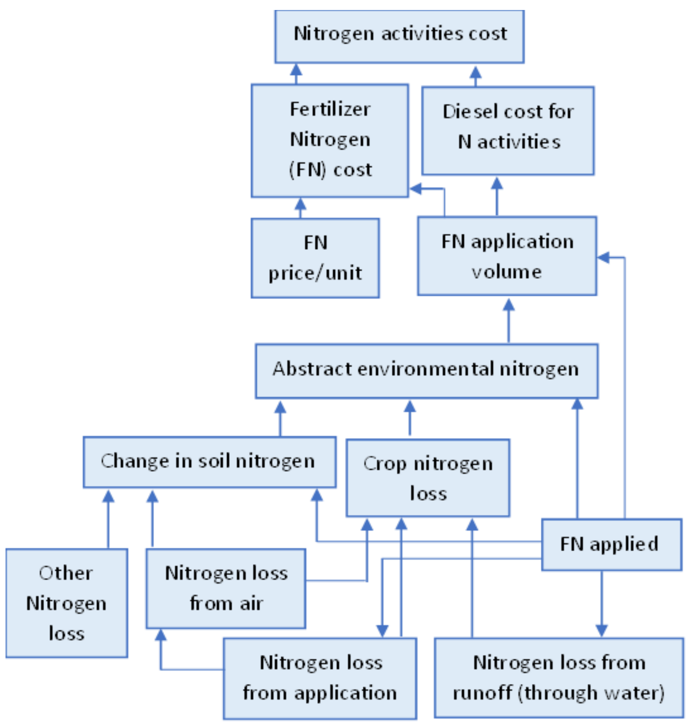
"Other Operating Cost" includes costs incurred for other farm operating costs.

For example: Machine Depreciation, Labour and Other Energy costs.

Level 4:

Cost of Nitrogen (N) activities

Figure 3.6: Level 4 indicator: Nitrogen module



$$Cost\ of\ N\ activities\ =\ Cost\ of\ FN\ +\ Cost\ of\ energy\,N\ +\ Seed\ cost$$

$$=\ FN\ price\ \left(\frac{\$}{kgN}\right) * Applied\ amount\ \left(\frac{kgN}{ha}\right) +\ Diesel\ cost\ \left(\frac{\$}{ha}\right)$$

$$*\ Treatment\ numbers\ +\ Seed\ cost\ \left(\frac{\$}{ha}\right)$$

Cost of FN: total cost for nitrogen fertiliser applied.

In an ideal situation, the required amount of nitrogen fertiliser is decided by soil nitrogen and the amount of abstract environmental nitrogen required by the crop and other indicators.

However, in practice, the purchase activities before planting are highly likely decided by growers' experience, which creates a gap between the purchased cost and the cost of nitrogen fertiliser applied.

The situation with energy costs and the total energy purchased is similar.

One of the significant differences from Pham's (2016) WESM model is the inclusion of the concept of "abstract environment resources", specifically water and nitrogen. In contrast to Pham's model, where rainfall is considered "free water" and licensed water is drawn from rivers, creating both economic cost and environmental loss, this model defines them distinctively.

All nitrogen resources on farms are abstracted from the ecosystem in various ways, making them all "abstract environment nitrogen". To elaborate further, the production of nitrogen fertiliser, which converts nitrogen gas to ammonia molecules, results in a decrease in total nitrogen in the ecosystem and the conversion of a non-polluting gas to a pollutant. Another primary source of nitrogen for crops is soil nitrogen, which decreases over time as growers continue to cultivate crops.

Cost of energyℕ: total cost for diesel used to run equipment in the fertiliser application process. For example, dry fertiliser blended into the soil; or the mixture of liquid fertiliser to irrigation water. The application of fertiliser after being mixed into irrigation water is not included here but should be accounted for in the irrigation energy costs, unless liquid fertiliser application is separate from irrigation activities.

Seed cost: total cost for seed purchased and planted.

Level 5:

$$Cost\ of\ FN = \ FN\ price\ \left(\frac{\$}{kgN}\right) * Applied\ amount\ \left(\frac{kgN}{ha}\right)$$

$$Cost\ of\ energy\mathbb{N} = \ Diesel\ cost\ \left(\frac{\$}{ha}\right) * \ Treatment\ numbers$$

$$Seed\ cost\ = \ Seed\ amount\ \left(\frac{kg}{ha}\right) * \ Seed\ price\ \left(\frac{\$}{kg}\right)$$

With further decomposition:

$$Cost\ of\ energy\mathbb{N}\ = \ Diesel\ price\ \left(\frac{\$}{L}\right) * \ Diesel\ use\mathbb{N}\ \left(\frac{L}{ha}\right) * \ Treatment$$

$$Diesel\ use\mathbb{N}\ \left(\frac{L}{ha}\right) = \ Machine\ hours\ \left(\frac{h}{ha}\right) * \ Engine\ power\ \left(\frac{kw}{h}\right) * \ \alpha\ \left(\frac{g}{kwh}\right) * \frac{1}{1000} * \ 1.192\ \left(\frac{L}{kg}\right)$$

The engine power is listed on the machine instructions.

Treatment/Treatment numbers are the terms used in DSSAT to describe a specific treatment for crops, such as a fertilisation or irrigation scheme, is referred to as a management operation or agronomic practice. These terms encompass various agricultural interventions designed to optimize crop growth and productivity, such as fertiliser application management, irrigation methods/amount, planting strategies, and pest control.

α: Fertiliser adding machine energy use (g/kwh)

1.192 (L/kg) is the conversion indicator to transfer from weight units to volume units.

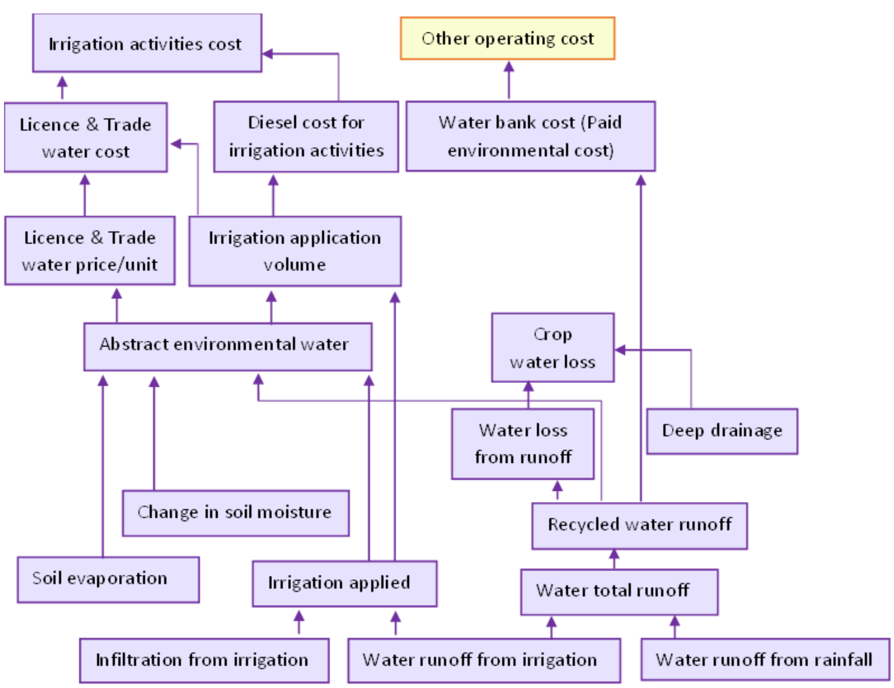
Diesel price: the market/purchase price of diesel

If the farm uses labour to add fertiliser to water, then this aspect does not entail energy (diesel) costs.

Irrigation-related activities, level 4:

Cost of irrigation activities

Figure 3.7: Level 4 indicator – Water module



$$Cost\ of\ irrigation\ activities\ = Irrigation\ cost\ +\ Cost\ of\ energy$$

This aspect builds on Pham's model and extends it by introducing the concept of water bank cost as a “paid environmental cost”. The state government mandates that farms build this facility to collect contaminated water, and the cost of establishing and maintaining the water bank is considered part of the farm's “other operating cost”.

Irrigation-related activities, level 5:

$$Irrigation\ cost = \ Licensed\ irrigation\ cost\ +\ Traded\ irrigation\ cost$$

There are two kinds of irrigation costs in the Australian cotton context:

1. **Licensed irrigation cost**

$$Licensed\ irrigation\ cost$$
$$= \ licensed\ irrigation\ volume\ \left(\frac{L}{ha}\right) *\ Treatment\ number\ *\ Price\ \left(\frac{\$}{L}\right)$$

2. **Traded irrigation cost**

$$Traded\ irrigation\ cost$$
$$= \ Traded\ irrigation\ volume\ \left(\frac{L}{ha}\right) *\ Treatment\ number\ *\ Price\ \left(\frac{\$}{L}\right)$$

Note*: During cropping practices, it is reasonable to assume licensed water should be used first, then the activity be required to buy/sell traded licence water with a different price which will create a gain/loss. The gain will go to other operating revenue and the loss will go to other operating costs.*

At Level 5:

$$Cost\ of\ energy_{W} = Diesel\ price \left(\frac{\$}{L}\right) * Diesel\ use_{W} \left(\frac{L}{ha}\right) * Treatment$$

In which:

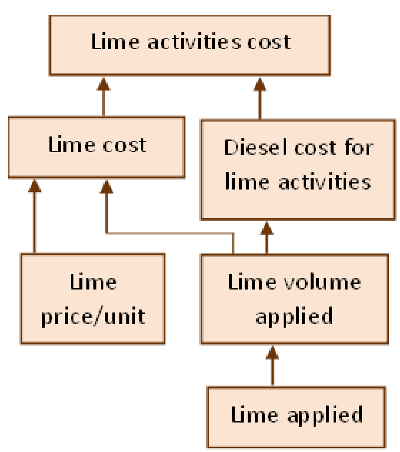
$$Diesel\ use_{W} \left(\frac{L}{ha}\right) = Machine\ hours \left(\frac{h}{ha}\right) * Engine\ power \left(\frac{kw}{h}\right) * \beta \left(\frac{g}{kwh}\right) * \frac{1}{1000} * 1.192 \left(\frac{L}{kg}\right)$$

β: Irrigation pumping energy use (g/kwh). If the farm uses the same machine for both fertilising and irrigation, then the fertiliser activities' energy cost belongs N and the irrigation component belongs W.

For lime-related activities, level 4:

Cost of lime activities

Figure 3.8: Level 4 indicator: Lime module



$$Lime\ activity\ cost = Lime\ cost + Cost\ of\ engergy_{D}$$

For lime-related activities, level 5

$$Lime\ cost\ = \left[Lime\ price\ \left(\frac{\$}{kg}\right) *\ Applied\ amount\ per\ treatment\ \left(\frac{kg}{ha}\right)\right] * Treatment$$

$$Cost\ of\ energy_{\square}\ =\ Diesel\ price\ \left(\frac{\$}{L}\right) *\ Diesel\ use_{\square}\ \left(\frac{L}{ha}\right) *\ Treatment$$

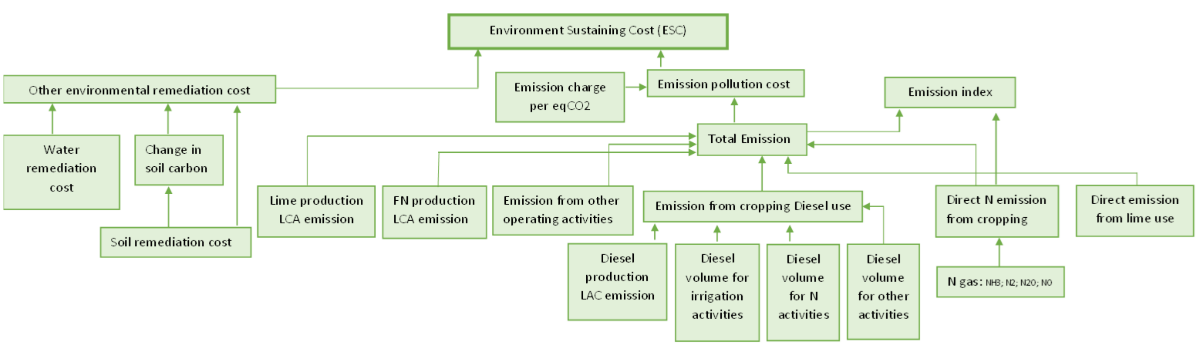
In which:

$$Diesel\ use_{\square}\ \left(\frac{L}{ha}\right) = \ Machine\ hours\ \left(\frac{h}{ha}\right) *\ Engine\ power\ \left(\frac{kw}{h}\right) * \ \gamma\ \left(\frac{g}{kwh}\right) * \frac{1}{1000} *\ 1.192\ \left(\frac{L}{kg}\right)$$

γ: Lime application activity energy use (g/kwh). Diesel is used for lime application machines.

3.3.3.3 Level 2 ESC module

Figure 3.9: Level 2 indicator – Environment sustaining cost (ESC)



$$ESC\ =\ Emission\ pollution\ cost\ + Other\ enbironmental\ remediation\ cost\ (unpaid)$$

Which in next level, **level 3**:

$$Emission\ pollution\ cost\ =\ Total\ emission\ *\ Emission\ charge\ per\ CO_2eq$$

$$Emission\ index\ =\ Total\ emissions\ /\ Desirable\ outcome$$

Note: *The emission index functions as a measurement of performance, thus the "desirable outcome" should be customised to fit the most suitable measurement indicator, according to needs. For example, for nitrogen emissions and farm performance measurement, the "desirable outcome" can be yield, lint yield or NOPAT.*

Level 4:

$$\begin{aligned} Total\ emission \\ &= FN\ production\ LCA\ emission\ +\ Lime\ production\ LCA\ emission \\ &+ Direct\ N\ emission\ from\ cropping \\ &+ Emission\ from\ cropping\ diesel\ use \\ &+ Emission\ from\ other\ operating\ activities \end{aligned}$$

Level 5:

$$Direct\ N\ emission\ from\ cropping\ =\ N_2O\ emission\ +\ NO\ emission$$

$$= \left(22\ *\frac{N_2O}{7}\right) *\ 298\ +\ \left(15\ *\frac{NO}{7}\right) *\ 296$$

Note: The coefficient values are calculated by the relative atomic mass of N and O atoms.

$$Emission\ from\ diesel\ use\ (CO_2eq) =\ Cropping\ diesel\ amount\ (L) *\ 2.63\ \left(\frac{kg}{L}\right) + Diesel\ production\ LCA\ emission\ (CO_2eq)$$

In which:

2.63 (kg/L): Diesel use emission CO_2eq (kg/L Diesel)

298 and 296: CO_2eq multiplicative coefficient of N_2O and NO

$$Cropping\ diesel\ amount\ (L)$$
$$= Diesel\ use\ N\ \left(\frac{L}{ha}\right) + Diesel\ use\ W\ \left(\frac{L}{ha}\right) + Diesel\ use\ L\ \left(\frac{L}{ha}\right)$$

$$Diesel\ Production\ LCA\ emission$$
$$= Cropping\ diesel\ amount\ (L)$$
$$* Diesel\ production\ LCA\ emission\ per\ unit\ \left(\frac{CO_2eq\, kg}{L}\right)$$

If **Urea** is applied (Kumar et al., 2021):

$$FN\ production\ LCA\ Urea = \frac{FN\ applied\ amount}{46\%} * 0.714\ \left(CO_2eq \frac{kg}{kg}\right)$$

Because the FN applied the amount kgN as the unit, while the other uses kg as the unit for urea. So, a unit conversion should be made based on the nitrogen content of urea. The same conversion is applied to ammonia.

If liquid **Ammonia** is applied, then:

$$FN\ production\ LCA\ Ammonia\ = \frac{FN\ applied\ amount}{82.4\%} * 60\% * 3.03\ \left(CO_2eq \frac{kg}{kg}\right)$$

60%: Liquid ammonia selected, 60% Ammonia 40% Water.

3.03: Ammonia production 3.03 (CO_2eq kg/kg) using Methane and 3.85 (CO_2eq kg/kg) using Coal (Singh et al., 2018).

$$Emission\ from\ other\ operating\ activities$$
$$= Electricity\ amount\ * Emission\ CO_2eq + Transportation$$
$$* Emission\ CO_2eq$$

Electricity production Emission CO_2eq: 656.4 (gCO_2/kwh) (AU Statistics, 2021)

$$\begin{aligned} &Lime\ Emission \\ &\quad = Lime\ production\ LCA\ emission\ +\ Emission\ direct\ from\ lime\ activities \\ &\quad = 0.9\left(CO_2eq\frac{kg}{kg}Lime\ production\right) * Lime\ use\ amount\left(\frac{kg}{ha}\right) * Treatments \\ &\quad + 0.12\left(CO_2eq\frac{kg}{kg}\right) * Lime\ use\ amount\left(\frac{kg}{ha}\right) * Treatments \end{aligned}$$

0.9 CO_2eq kg/kg Lime production (Laveglia et al., 2022)

0.12 CO_2eq kg/kg Lime application emission: If growers use limestone, then the multiplicative coefficient is 0.12; if growers use dolomite to lime, then the multiplicative coefficient is 0.13.

3.3.4 Demostration of model

The objective of this section is to illustrate the practical application of the model in a simplified manner. A case study is employed to demonstrate the use of the Nitrogen module for conducting the relevant calculations[32]. The case study is of a typical irrigated cotton farm of 467 ha, where the owner is remediating the environmental degradation of nitrogen emission by planting Blue Mallee Eucalyptus trees[33].

3.3.4.1 Data and assumptions

Chapter 3 presents a simple analysis that targets nitrogen emissions from cotton farms in Australia. To address the issue of sustainable nitrogen fertiliser output, this case study

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[32] In this demonstration, simulation is not involved, as it is beyond the scope of this chapter. This is intended to showcase the potential capabilities of this modelling approach and the type of analysis that can be performed with it. Simulations will be performed in the next chapter.
Although this is not ideal, as in the ideal situation data should be provided by experiments, field tests or simulations, this approach can still provide a reasonable and reliable analysis.

[33] Blue Mallee Eucalyptus as the carbon offset tree choice is recommended by the Australian Government Clean Energy Regulator, CO2 Australia and other tree offset projects. For detailed explanation and analysis see Appendix B.

investigates the possibility of offsetting nitrogen emissions through tree planting projects. Data source and calculation is presented in Appendix B.

As the literature only provides experimental data focusing on the link between nitrogen management and yield, the water module is simplified, focusing solely on the calculation of the economic impact pertaining to the total cost and diesel consumption associated with irrigation activities. Data to calculate water usage is based on: used irrigation volume (Pham, 2016), diesel usage volume (UTS, 2015), and water licence market price (ABARES, 2022).

The assumption for this case study is as follows:

(I) The prices of seeds, nitrogen fertilisers, fibre and all other costs are assumed to be constant among different farms and areas.

(II) The cotton farms in the study all utilised furrow irrigation and chemical fertiliser as their primary cropping practices. The only difference is the fertiliser application amount.

(III) There are no differences between tree planting projects regarding costs, labour, efficiency and others.

(IV) Other environmental issues that may be caused by agriculture or tree planting are not in the scope of consideration, such as water pollution and the destruction of natural vegetation. Thus costs, such as other environmental remediation costs, are assumed to be zero in this case study.

3.3.4.2 Sample results

The values for the key variables are presented in following Tables 3.2 to 3.8. For this case study, three scenarios of furrow irrigation were considered, with varying application amounts of nitrogen fertiliser (urea) at 200 kgN/ha, 250 kgN/ha and 300 kgN/ha. The emission of cropping production is expected to be fully offset through the blue eucalyptus[34] planting project.

Note: *(i) Analysis, explanation and calculation related to nitrogen balance and tree planting cost are demonstrated in Appendix B.*

(ii) In Tables 3.2 to 3.9, the data types are indicated as follows: "C" represents calculated results, "A" represents assumptions, and "M" indicates measurements obtained from the literature or based on calculated data from the literature.

The basic unit for this model is per hectare, which means all value is the per hectare amount, and total means total cost/revenue per hectare.

Table 3.2: ESRI for cotton farm emission and tree planting offsets

Nitrogen Fertilizer application amount			S1	S2	S3	Units
			200.00	250.00	300.00	kgN/ha
Level 1						
ESRI						
	Net Operating Profit After Tax (NOPAT)	C	1,139.40	1,340.97	1,583.79	$/ha
	Environment Sustaining Cost (ESC)	C	2,018.82	2,203.95	2,389.09	$/ha
	Net Operating Assets (NOA)	C	7,087.00	7,503.88	7,990.25	$/ha
	R_{WACC}	A	6%	6%	6%	Rate
			-1,304.64	-1,313.22	-1,284.72	$/ha

Table 3.3: Calculation of NOPAT

Nitrogen Fertilizer application amount			S1	S2	S3	Units
			200.00	250.00	300.00	kgN/ha
Level 2						
NOPAT						
	Tax	C	379.80	446.99	527.93	$/ha
	Operating profit	C	1,519.21	1,787.96	2,111.72	$/ha
			1,139.40	1,340.97	1,583.79	$/ha

[34] "The Blue Mallee Eucalyptus is the most commonly planted tree in Australia as part of carbon offset programs" (CO2 Australia, 2013).

Table 3.4: Calculation of total operating revenue (operating revenue + other operating revenue)

			S1	S2	S3	Units
Nitrogen Fertilizer application amount			200.00	250.00	300.00	kgN/ha
Level 3						
Total operating revenue						
	Other operating revenue	A	-	-	-	$/ha
	Operating revenue	C	6,367.24	6,697.24	7,082.24	$/ha
	Harvest index	C	0.49	0.50	0.52	Rate
	Economic yield	C	6,367.24	6,697.24	7,082.24	$/ha
	Lint yield	M	2,550.00	2,700.00	2,875.00	kg/ha
	Seed yield	M	2,704.42	2,704.42	2,704.42	kg/ha
			6,367.24	6,697.24	7,082.24	$/ha

Table 3.5: Calculation of total operating cost (operating cost + other operating cost)

Nitrogen Fertilizer application amount			S1	S2	S3	Units
			200.00	250.00	300.00	kgN/ha
Total operating cost						
Operating cost						
	Total fertilizer nitrogen (FN) cost	M	244.98	306.22	367.47	$/ha
	Total water cost	M	751.41	751.41	751.41	$/ha
	Lime cost	M	280.00	280.00	280.00	$/ha
	Energy (diesel) cost	M	526.33	526.33	526.33	$/ha
	Seed cost	M	129.00	129.00	129.00	$/ha
Other operating cost						
		C	2,916.31	2,916.31	2,916.31	$/ha
			4,848.03	4,909.28	4,970.52	$/ha

Table 3.6: Calculation of cotton farm equity

Nitrogen Fertilizer application amount			S1	S2	S3	Units
			200.00	250.00	300.00	kgN/ha
Level 2						
Net Operating Assets (NOA)						
	Total Operating Assets	A	7,087.00	7,503.88	7,990.25	$/ha
	Total Operating Liability	A	-	-	-	$/ha
	Expect growth rate in Operating Assets	C		5.88%	6.48%	

Table 3.7: Calculation of Nitrogen activities, Irrigation activities and Lime activities

Nitrogen Fertilizer application amount			S1	S2	S3	Units
			200.00	250.00	300.00	kgN/ha
Level 4						
Nitrogen						
	Nitrogen activities cost	C	252.18	313.42	374.67	$/ha
	Total fertilizer nitrogen (FN) cost	C	244.98	306.22	367.47	$/ha
	FN application volume	M	434.78	543.48	652.17	kg/ha
	Abstract environmental nitrogen	C	200.00	250.00	300.00	kgN/ha
	Crop nitrogen loss	M	131.10	164.60	198.10	kgN/ha
	Nitrogen loss from application	M	34.00	42.50	51.00	kgN/ha
	Nitrogen loss from runoff (through water)	M	50.00	75.00	100.00	kgN/ha
	Nitrogen loss through air	M	47.10	47.10	47.10	kgN/ha
	Change in soil nitrogen	M	62.00	62.00	62.00	kgN/ha
	Nitrogen input efficiency		51.45%	51.16%	50.97%	Rate
Water						
	Irrigation activities cost	C	917.01	917.01	917.01	$/ha
	Total water cost	M	751.41	751.41	751.41	$/ha
	Licensed water	M	751.41	751.41	751.41	$/ha
Lime						
	Total lime activities cost	C	280.78	280.78	280.78	$/ha
	Lime cost	M	280.00	280.00	280.00	$/ha
	Lime volume applied	M	4.00	4.00	4.00	t/ha
			1,276.39	1,337.63	1,398.88	$/ha

Table 3.8: Environment sustaining cost (ESC)

			S1	S2	S3	Units
Nitrogen Fertilizer application amount			200.00	250.00	300.00	kgN/ha
Level 2						
Environment Sustaining Cost (ESC)						
	Other environmental remediation cost	A	-	-	-	$/ha
	Emission charge per ha using tree offsets	C	2,018.82	2,203.95	2,389.09	$/ha
	Emission pollution cost	C	2,018.82	2,203.95	2,389.09	$/ha
	Emission charge per CO2e	C	0.18	0.19	0.21	$/CO2ekg
	Break-even price for emissions	C	0.07	0.09	0.11	$/CO2ekg
	Total emission	C	11,441.29	11,518.90	11,596.51	CO2ekg/ha
			2,018.82	2,203.95	2,389.09	$/ha

Table 3.9: Calculation of emissions from operating activities and production emission of materials

			S1	S2	S3	Units
Nitrogen Fertilizer application amount			200.00	250.00	300.00	kgN/ha
Level 3						
Total emission						
	Emission index	C	1.80	1.72	1.64	CO2ekg/kg
	Direct emission	M	543.90	543.90	543.90	CO2ekg/ha
	Emission from farm energy use	M	10,526.66	10,526.66	10,526.66	CO2ekg/ha
	LCA emission from production	C	370.73	448.34	525.95	CO2ekg/ha
	Emission from other operating activities	A	-	-	-	CO2ekg/ha
			11,441.29	11,518.90	11,596.51	CO2ekg/ha
Level 4						
Direct emission						
	Direct N emission from cropping	M	535.86	535.86	535.86	CO2ekg/ha
	Direct emission from lime use	M	8.04	8.04	8.04	CO2ekg/ha
			543.90	543.90	543.90	CO2ekg/ha
Emission from farm energy use						
	Total energy cost	C	526.33	526.33	526.33	$/ha
	Pre-planting	M	81.00	81.00	81.00	CO2ekg/ha
	Planting	M	94.50	94.50	94.50	CO2ekg/ha
	Energy - nitrogen	M	8.10	8.10	8.10	CO2ekg/ha
	Energy - water	M	186.30	186.30	186.30	CO2ekg/ha
	Energy - lime	M	0.32	0.32	0.32	CO2ekg/ha
	Other diesel emission	M	264.20	264.20	264.20	CO2ekg/ha
	Diesel production LCA emission	C	9,892.24	9,892.24	9,892.24	CO2ekg/ha
			10,526.66	10,526.66	10,526.66	CO2ekg/ha
LCA emission from production materials						
	FN production LCA emission	C	310.43	388.04	465.65	CO2ekg/ha
	Lime production LCA emission	C	60.30	60.30	60.30	CO2ekg/ha
			370.73	448.34	525.95	CO2ekg/ha
Emission from other operating activities						
			-	-	-	CO2ekg/ha
Other environmental remediation cost						
	Soil remediation cost	A	-	-	-	$/ha
	Water remediation cost	A	-	-	-	$/ha
			-	-	-	$/ha
Total emission			11,441.29	11,518.90	11,596.51	CO2ekg/ha
Total remediation cost			2,018.82	2,203.95	2,389.09	$/ha

3.4 Findings and discussion

3.4.1 Findings and reflections from the case study

3.4.1.1 Findings

This case study reveals two specific findings.

First, the top-level indicator, ESRI, is negative. This indicates the net economic value generated from production is insufficient to cover the cost of environmental degradation caused. Accordingly, it is apparent that the existing agricultural cropping practices are not sustainable under the modelled conditions, particularly when considering the emissions stemming from the supply of raw materials and the generation of production waste, such as nitrogen leakage.

Second, upon analysing the feasibility of offsetting emissions entirely through tree planting projects, it becomes evident that the GHG emission issue is unlikely to be easily resolved in isolation without the support of regulatory measures to support the production systems to evolve, such as to lower the impact of production practices and support the emergence of lower cost environmental remediation. There is insufficient surplus value from the typical cotton farm to cover these costs.

In addition to those main findings, one interesting rate is observed, as it needs an estimated 2.5 m^2 of land for one tree to grow (Gumeracha Farm Forestry Management Area, 2008). Hence, under the extreme assumption that tree offsetting is the sole option for remediating nitrogen atmospheric pollution and that nitrogen emissions are completely offset, the estimated area required to sequester the emissions resulting from the application of 200 kgN, 250 kgN and 300 kgN fertiliser per farm for one year of cotton production is 92.63 hectares, 101.12 hectares and 109.62 hectares respectively over 25 years. This may result in less land being available for future activities, which increases the cost of the accounting book and increases the risk of keeping the offset.

The NOPAT component provides a cost analysis from an accounting perspective, while the ESC is elevated due to various underlying factors.

(I) The ESC's outcome is influenced by two key indicators: pollution quantity and remediation cost. In this case, the focus is exclusively on evaluating emission offset solutions, assuming no charges are associated with water and soil remediation. However, the total emissions remain considerably high, due to the substantial life cycle emissions associated with farm production materials. This factor should be duly incorporated into the market mechanism (possibly akin to LCC), to establish sustainability as a significant market indicator. This implies that producers bear the responsibility of considering both their output and the choice of their suppliers.

(II) To ensure the success of GHG emission mitigation solutions, it is imperative to consider not only the project's design but also multiple indicators, stakeholders and the external climate change context. Furthermore, within this case study, several concerning findings have emerged that warrant further investigation. A disparity exists between the 'true cost' of emissions and the market price of emission offsets, indicating that a substantial portion of the pollution cost is borne by society, rather than the polluter. This is of course a much larger discussion than a doctoral thesis and has been subject to much debate. When growers are obligated to bear the direct cost of remediation of emissions, the phenomenon of being able to afford higher emission charges as they engage in greater pollution diminishes, which might be due to the exorbitant cost involved.

This prompts the question of whether the government should strive to make the emission charge more accessible. However, if the government insists on ensuring that the emission charge is affordable for all growers, it may inadvertently disadvantage those who have lower levels of pollution. In such cases, a considerable portion of the emission cost is shouldered by society, inadvertently fostering a tacit endorsement of pollution.

In this particular case, the adoption of extant approaches to carbon credit systems may not present a viable solution for addressing the issue at hand. This is primarily attributed to the significant disparity between the emission charge that ought to reflect responsibility and the affordable price. Consequently, the carbon credit system alone may prove inadequate in achieving substantial reductions, unless there is heavy reliance on government subsidies or other cross-subsidisation regimes.

(III) The utilisation of ratios such as the nitrogen input efficiency rate and the water input efficiency rate present certain limitations in this context. These ratios overlook the aspect of scale and primarily indicate the level of efficiency within an organisation, without providing more generalisable insight into underlying mechanisms that drive such efficiency levels or establishing connections to other components of the system, particularly within the agricultural sector. Furthermore, due to the inherent nonlinear dynamics of climate change in agriculture, the variability in these rates is likely to contain less informative content, compared to financial indicators. This challenge is explored in Chapter 4.

This experimental case study highlights the potential of integrated modelling as a valuable tool for comprehensively understanding complex systems. The constructed integrated model facilitates a holistic analysis of the system, which involves diverse disciplines, stakeholders and nonlinear climate change dynamics. The model offers high-level indicators for comparison purposes and lower-level indicators for explaining observed differences.

By utilising the integrated modelling system to investigate specific indicators, there is a significant likelihood of unveiling unexpected phenomena within the system. This can be attributed to the functional representation of the system, which provides insights into its operational mechanisms, thus revealing previously unidentified interconnections or elucidating reasons behind the effectiveness or inefficiency of certain processes.

3.4.1.2 Reflections

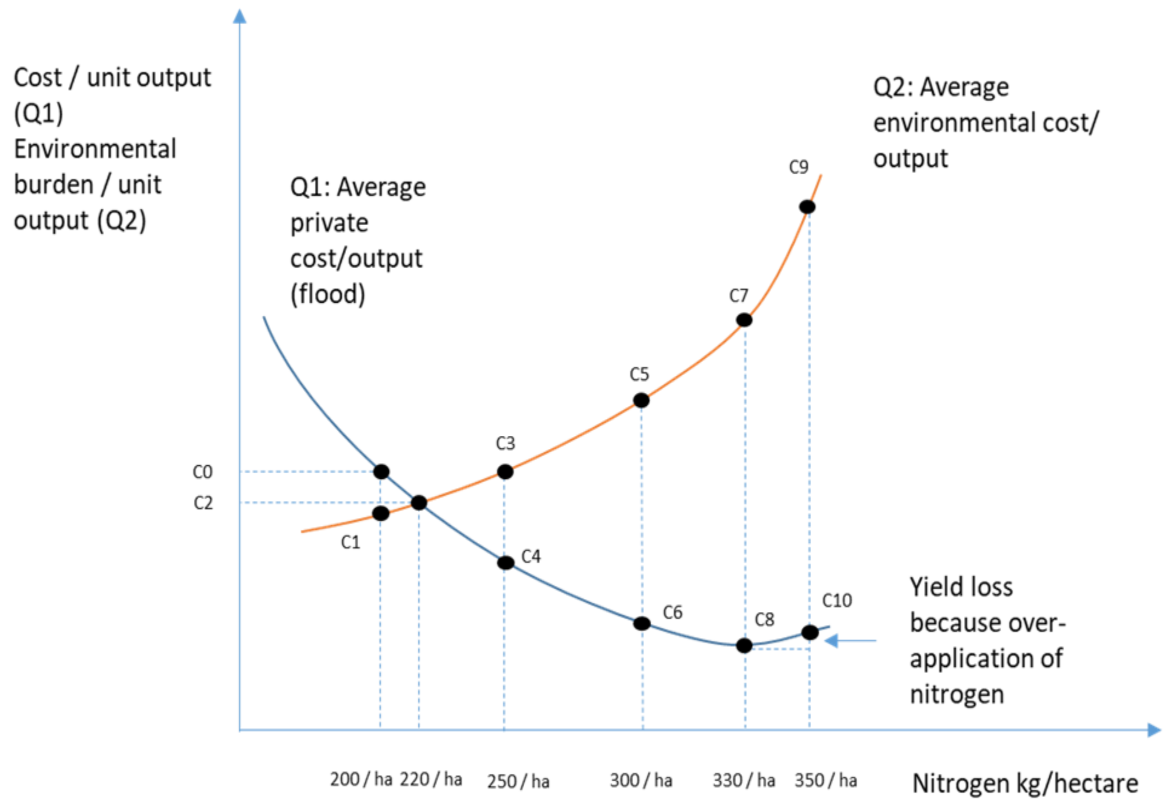
The simplified model shown in Figure 3.10 demonstrates the relation between farm management outcomes and corresponding environmental cost changes. Before the planting season, farms evaluate the water level prediction, including rainfall and the price of the water licence. When the water supply is sufficient and affordable, farmers might prefer to plant cotton. As for nitrogen fertiliser, because the price of chemical fertiliser is relatively low, the price per unit is not counted as a significant factor in decision-making[35]. However, many growers believe that the overapplication of nitrogen fertiliser may act as a safeguard for yield. The fact is, as farmers only choose prosperous water years to plant cotton and insist on using furrow irrigation, with a large flow of surface water, a large percentage of nitrogen nutrition will be washed away (Rochester, 2003; See Appendix B for data and support).

Thus, any decision regarding farm nitrogen fertiliser application is mainly determined by water supply/price and yield prediction.

As shown in Figure 3.10, when having different nitrogen application rates, Q1 represents the changes in average farm cost per output when using furrow irrigation with increasing fertiliser application, and Q2 represents the changes in the corresponding environmental cost per output of cotton yields in the same fertiliser application level with Q1.

[35] Although the chemical fertiliser price has increased from 2022 to 2023, it is not high enough to raise alarm in growers' minds causing them to decrease the application amount significantly. However, the increasing price has already made some growers start to reconsider the application cost-benefit analysis for nitrogen fertiliser (2022 Australian Cotton Conference; AgEcon, 2023).

Figure 3.10: Motivation of N fertilizer application and over application



In addition, because the amount of fertiliser applied would not change with one additional unit of cotton yield, the fertiliser cost per output is considered the total fixed cost. Thus, when the application amount increased from 200 to 330 kgN/ha of nitrogen (c0 to c8), yield increased significantly, so the cost per output for farmers decreased.

From 330 to 350 kgN/ha, the nitrogen absorption capacity of crops reaches saturation, so the effect of increasing yields by increasing nitrogen application is minimal. In addition, overapplication may cause problems, such as late ripening of cotton balls and damage to fibre quality. As shown in the above figure from C8 to C10, fixed cost increased while total output decreased, thus cost per output for farmers increased.

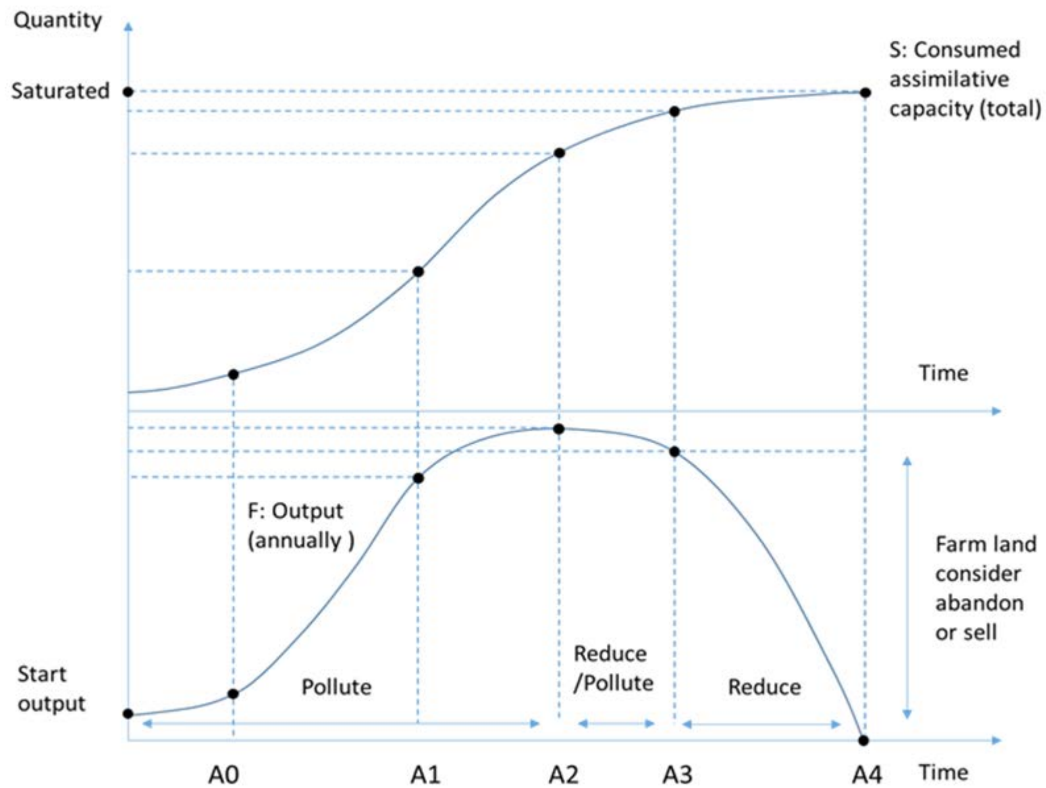
However, cotton farms choose to apply over 350 kgN/ha, even up to 500 kgN/ha, because the costs of nitrogen fertiliser and water licences are relatively low, the damage to yield and fibre is minimal, and farms are not required to compensate for environmental damage (difference from C9 to C10). Thus, growers would be motivated by the decrease in cost per unit output and push the application amount from 200 kgN/ha to 350 kgN/ha and even more from the economic perspective.

As for Q2, the average environmental cost per output (cotton yield), the nitrogen-related pollution sources include nitrogen oxide emissions, fertiliser production emission, water eutrophication caused by nitrogen fertiliser loss, soil quality decline and/or salinisation caused by excessive fertilisation.

This observation is in line with Pearce's suggestion (1976) that these effects exhibit long-term damage characteristics and do not easily dissipate naturally. As these damages accumulate from C1 to C9, the net growth continues to accumulate with the increasing application amount.

Assume that growers are required to compensate for the environmental damages caused by nitrogen fertilisers, in a tree offset situation. Theoretically, without considering the cost of land and labour, through previous calculations, only the 200 kgN/ha nitrogen applied method may have a chance to achieve sustainability, where C0 is more significant than C1. However, there is not much profit margin left for owners. Compared to C8, C0’s cost per output is higher, which means farms are unlikely to accept this application amount without regulations or adjustment of matched irrigation.

Figure 3.11: Projection of relation between traditional agriculture production and consumed assimilative capacity (total) over the long term



However, if the total environmental capacity and long-term sustainability were considered, Figure 3.11 shows how farmland output (annually) (curve F) and consumed assimilative capacity (total) change in the long term (curve S). In the modelling of emission trance from IPCC AR6 (2021, p.13), the emission of nitrous oxide is expected to increase, or in the most optimistic forecasts, show a slight decrease.

From A0 to A2, farmers are more motivated to pollute/waste than reduce/offset.

A0 and the area to its left represent the agriculture output before chemical fertilisation, with only slow growth in yield but also a smaller amount of assimilative capacity burden. A1 to A4 represent modern agriculture practice, in which yields are boosted by chemical fertiliser application but at the same time create a larger environmental burden.

From A0 to A1, since the absorption capacity of crops has not reached the upper limit, the enrichment of nutrients provided by chemical fertiliser and the popularisation of mechanical farm tools (instead of manual operation) has led to a rapid increase in crop yield. At the same time, pollution caused by the increasing amount of chemical fertiliser application and fossil fuel usage (fertiliser production and agricultural tool fuel) has also increased. Thus, there is rapid growth for both curves.

From A1 to A2, the absorption capacity of crops is almost saturated. Continuing to increase the application rate does not lead to a large increase in yield, but leads to waste and pollution (volatilisation, soil salinisation, water pollution, etc.). During this period, the growth rate in farm output (F curve) has slackened, yet the consumption of assimilative capacity is still rising. At the same time, during this period people began to be aware of the harm caused by the abuse of chemicals to the natural environment and their health. Relevant laws and regulations are constantly introduced to avoid large-scale hazards, such as DDT and sulphide acid rain.

However, because of the large economic benefit payback of performing non-illegal waste or pollution, such as furrow irrigation and overapplication of fertilisers, farmers have the motivation to pollute rather than offset or reduce from A0 to A2.

From A2 to A3, internal and external reasons cause choice differences to appear. In farm management practice, previously accumulated pollution and destruction begin to cause minor damage to output, with either increases in cost, decreases in yield or both. In addition, people recognise the urgency of emission reduction and environmental protection externally; through regulations and public opinion, they encourage farms to consider emission reduction and environmental protection measures.

Under this double-pressure situation, some farmers may choose to try emission reduction or offset projects, because simply adding chemical input can no longer bring the previous growth rate in output. However, despite the diminished economic benefits, individuals remain motivated by both polluting and wasteful behaviours; they tend to engage in polluting and wasteful actions unless appropriate regulations are implemented to restrict such behaviours.

From A3 to A4, the accumulated pollution has led to a serious decline in land quality, scarcity of water resources, frequent extreme weather and other problems. Farms are no longer suitable for cropping, and the production level has fallen irreparably. In this period, growers' motivations would be dominated by offset/reduction, if they decide to keep the land for agricultural use. However, unless scientific and technological means have made some breakthrough (such as improved varieties that can adapt to the harsh environment, soilless cultivation that can be popularised on a large scale, etc.), few rescue measures can turn the tide.

Theoretically, the best opportunity to reduce waste/pollution is from A0 to A1 before any damages have been made. However, because the economic margin growth is high, people would be more likely to seize the opportunity to gain economic benefit.

Therefore, the reduce/offset option is more likely to take off from A1 to A2 and is primarily accepted by audiences from A2 to A3.

3.4.2 To what extent is this model generalisable?

In the context of agriculture, sustainability challenges arise from various factors, such as eutrophication of water bodies, land use impacts, habitat destruction for native species, and greenhouse gas emissions (IPCC, 2019). In the specific case of cotton production in Australia, the nitrogen emission challenge significantly affects the industry's sustainability, with about 60% of GHG emissions caused by nitrogen fertiliser (CRDC, 2022, p.8)[36].

[36] Report name: Australian Cotton Sustainability Update

However, due to the impacts of crop rotation and fluctuating water supply, Australian cotton farms require assessments and choices to be made before the planting season. This implies that growers and other stakeholders need a comprehensive system that can provide timely evaluations based on climate change, market conditions and crop growth, for informed management decision-making and long-term planning.

Similarly, policymakers also need to be able to understand what can and could be regulated, to support the emergence of sustainable production and to assess policy effectiveness based on changing climate conditions, market trends and growers' choices. Policymakers need to ensure long-term effectiveness, which means that models limited to short-term assessments have limitations in identifying critical points for regulation and evaluating whether the outputs meet expectations.

Regarding the ESRIDM comprehensive model, first it embodies the characteristics mentioned in the introduction, which identify a sufficient integrated modelling system by three dimensions: seamless, interactive and intuitive (Muth & Bryden, 2013). Modules in this modelling may be applied to different cropping systems with different nitrogen level input and irrigation choice[37].

Second, the model is based on scientific modelling outputs, which provide a robust data framework for analysis and can illustrate the connections and links between various modules. This feature reflects and elucidates the causal relationship between management decisions made and output delivered[38]. As an illustration, in the case study, varying amounts of nitrogen application led to distinct emission liabilities, consequently yielding

[37] The demonstration in this chapter shows the mapping of chemical fertiliser and furrow irrigation; in the next chapter, more scenarios with different nitrogen sources and irrigation practices will be modelled and compared.

[38] See the different nitrogen application amount scenarios that cause differences from fertiliser cost to yield amount, revenue and emission output.

divergent offset costs. In this chapter, the model function is demonstrated through a case study that sources its data from literature and is calculated using formulas. However, to fully meet application standards, data input to the integrated modelling system should be provided through field tests or cropping simulation software, to ensure data consistency and comparability. This is demonstrated in the next chapter, using three scenarios applying Decision Support System for Agrotechnology Transfer (DSSAT) software.

3.4.3 Extending the early work on sustainability modelling

The call by academics for integrated modelling to improve agricultural sustainability has been ongoing for many years. As early as the 1990s, the necessity of economic and ecological integrated modelling for achieving sustainability objectives was recognised by academics (cf. Hofkes, 1996). However, a key limitation of this approach was that it did not distinguish between various environmental costs.

Subsequently, the concept of strong sustainability emerged, leading to the establishment of a theoretical framework that enabled the differentiation of different types of environmental costs (Baumgärtner & Quaas, 2009). This framework has facilitated the advancement of sustainability modelling, providing impetus for the recognition of varying degrees of natural costs and the attempt to link them with economic value (cf. Hysa et al., 2020).

With the establishment of sustainability tools like Life Cycle Assessment (LCA) being used on sustainability topics, there has been a renewed focus on identifying areas for improvement and aiding informed decision-making to promote sustainability (Moltesen & Bjørn, 2018; Ding, 2014). Nevertheless, this approach only superficially combines economic and ecological factors and falls short in describing and evaluating sustainability value (cf. Corsten et al., 2013; Finnveden, 2000).

To address this, Richard (2008) and Rambaud & Richard (2015) have combined insights such as these with Pearce's (1976) critique of CBA to propose a theoretical accounting model for natural resources. In this model, natural capital and production are represented as depreciating with natural losses in the form of capital depreciation.

This improvement provides a theoretical foundation for further valuation of sustainability value. Building on this work, Brown (2016) proposed the ERSI theoretical model, which values natural capital using the opportunity cost of restoring it, calculates the capital value generated by consuming natural costs using everyday business accounting, and considers changes in financial capital. However, Brown did not develop the model further with additional formulas or case studies.

Prior to this, academics in the field of sustainability had already proposed that, given that sustainable development is a complex issue that involves multiple fields, its assessment and valuation models should integrate multiple disciplines, particularly integrating natural science models with accounting models (cf. Bebbington & Larrinaga, 2014).

Building on these theoretical foundations, Pham et al. (2016) developed an Environmental Performance Measurement Systems (EPMS) transdisciplinary approach that integrates science and sustainability theories into accounting research. This approach was developed to create a decision-making assistant framework that could assist in making decisions based on environmental performance.

The model utilised Penman's (2003) multi-level financing analysis framework, which separates the profits and losses generated by daily operations from those resulting from water resource usage. However, the highest-level indicator of the model is water usage efficiency, which carries the risk of triggering the Jevons Paradox (Jevons, 2007).

This research proposes an integrated modelling approach that aims to aid policy decision-making in sustainability measurement, scenario assessment and investment evaluation. It builds on the practical aspects of Brown's (2016) work, which extends Richard (2012) by developing a nitrogen management module and providing detailed calculation formulae and case study examples.

To further advance these theories and models, this study proposes an integrated modelling approach that aims to assist with policy decision-making in sustainability measurement, scenario assessment and investment evaluation. The study extends Brown's (2016) work, which extends Richard's (2012) work, by developing a specific nitrogen management module and providing calculation formulas and case study examples.

Furthermore, this research extends Pham et al.'s (2020) water efficiency management model into the nitrogen management field and improves the highest-level indicator from an efficiency ratio to the ESRI value. This value includes the development of environmental sustaining costs, operational accounting maturity, and capital investment, providing a more comprehensive coverage of the operating status and environmental impact of farm-level production. This is beneficial for simulating the long-term effects of decision-making more accurately. The analysis of resource utilisation ratios is also included, which can support short-term and rapid decision-making.

Furthermore, this research has significant value in enhancing the detail and accuracy of data, while maintaining its forward-looking nature, information accessibility and integrity. This is particularly relevant given that the highest-level indicator in Pham et al.'s (2020) framework only indicates the level of efficiency in utilising natural resources, but fails to provide insights into the reasons and mechanisms underlying such efficiency.

According to Ramos (2019), collaborative scientific development and innovation should serve as the foundation for change in sustainability assessment tools. This research extends the methodology of Pham et al. (2020) by proposing novel approaches for designing integrated modelling that incorporates different farm management modules. Specifically, the research suggests incorporating additional agriculture operation management modules, such as soil health, which is highly connected to fertiliser and irrigation management.

3.4.4 The Coupled Human and Natural Systems (CHANS) Nitrogen Cycling Model vs. Grains Greenhouse V10.8 carbon calculator

Numerous studies have been devoted to the development of computational models aimed at capturing, calculating and analysing greenhouse gas (GHG) emissions. Notable examples include The Coupled Human and Natural Systems (CHANS) Nitrogen Cycling Model (Gu et al., 2009; Gu et al., 2013; Gu et al., 2015) and Grains Greenhouse V10.8 carbon calculator.

(I) CHANS Nitrogen Cycling Model

The CHANS Nitrogen Cycling Model, developed by Baojing Gu, serves as a computational tool for calculating nitrogen emissions. The model incorporates modules that calculate key aspects of farm operations, with a particular focus on nitrogen fertiliser, as well as environmental and atmospheric considerations related to agricultural activities, such as forest and water nitrogen pollution. In its trade module, the model quantifies the nitrogen emission ratios of imported and exported agricultural products, based on their nitrogen content. However, the model does not encompass any economic calculations at any level. Furthermore, the data presented in the trade module are sourced from publicly available datasets, without additional calculations being conducted by the nitrogen calculator itself. Moreover, compared to ESRIDM modelling which includes other emission activities, all computations and analyses in the nitrogen calculator are narrowly centred on nitrogen emissions. For instance, while it considers the impact of agricultural activities on the surrounding environment, its focus is solely on the infiltration and pollution consequences of waste nitrogen, excluding other factors such as carbon emissions.

(II) Grains Greenhouse V10.8 carbon calculator

The Cotton Research and Development Corporation (CRDC) has proposed the utilisation of a carbon calculator as a tool for cotton growers to assess their decision-making processes by calculating emissions and gross margin. The carbon calculator incorporates modules that account for various activities associated with cotton farm production, as well as the carbon emissions related to the production and transportation of raw materials.

Comparatively, ESRIDM integrated modelling and the Grains Greenhouse V10.8 carbon calculator consider additional emission factors related to production activities. These factors, which typically remain consistent across different years, are based on private data for growers and publicly available data for external users in all models.

(III) Comparison between two calculators and ESRIDM integrated modelling

In comparing the ESRIDM integrated modelling approach to other calculators, all models consider additional emission factors stemming from production activities, such as lime activities. These factors, which generally remain consistent across different years, are addressed by both models using publicly available data.

All models take into account production emissions resulting from nitrogen fertiliser usage and adopt a LAC-like process for emission calculation. The Grains Greenhouse calculator utilises publicly available data for calculations, whereas the CHANS Nitrogen Cycling Model and ESRIDM integrated modelling are designed to expand capabilities by incorporating the LCA system for production pre-cropping, depending on its accessibility to test broader, newly appeared technology and production resource choices. The LCA system is known for its ability to customise calculations based on the specific fertiliser brand and type chosen, the LCA-like process has a broader application scope and may be used to evaluate the environmental impact of future technology products. If using assumed LCA calculation results for emission calculation, the analysis cannot be involved with the latest technology in assessing the possibility or potential benefits brought by technology that has not yet been released to market.

In addition, both models are constrained by the quality of input data, in which high-quality agricultural data on a large scale may be difficult to access. Laing et al. (2023) conducted a comprehensive literature review focusing on two decades of research into nitrogen emissions in Australian agriculture. In that paper it finds, although a national database has been constructed based on this review, that its accessibility remains limited. Also, the functionalities of both the carbon calculator and the nitrogen calculator are primarily confined to computational tasks and certain forms of quantitative analysis based on the results, such as emission ratios. Thus, these models may only have limited influence in an economic and macro-analysis context compared to the ESRIDM model, as the emission calculation is only part of the ESRIDM analysis. The ESRIDM model has the capability for both pattern recognition and qualitative analysis of events. Additionally, the ESRIDM model is amenable to approximate analysis based on publicly available macro-level data, as demonstrated in the case study included in this chapter. Conversely, the two calculators are geared towards purely quantitative analysis.

Although the current demonstration of the ESRIDM model does not encompass calculations for other fertiliser types, its design is capable of adopting such applications within the fertiliser section in future research.

Furthermore, the ESRIDM model may extend the capabilities of emission calculators by incorporating long-term effects and multiple climate situations. While the Grains Greenhouse calculator and the CHANS Nitrogen Cycling Model predominantly focus on annual outcomes, the ESRIDM model, benefiting from scenario implementation (simple case study in this chapter) through various time lengths (more detailed simulation in Chapter 4) has the potential to simulate a broader range of impacts resulting from decision-making over several decades. By incorporating the effects of climate change, it has the potential to enhance the accuracy of simulation and prediction.

Moreover, the incorporation of co-production of knowledge practices in the ESRIDM model can enhance the Grains Greenhouse calculator and CHANS Nitrogen Cycling Model in two ways. First, it would enable refinement and improved stakeholder engagement, allowing model users to identify obstacles to the adoption of sustainable practices more easily. Additionally, it facilitates the set-up of scenario testing, enabling users to simulate different scenarios and evaluate their potential outcomes. Second, by integrating scenarios into the model's design via a multi-tiered analytical structure, the ESRIDM model establishes a clearer connection between decision-making processes and their resulting consequences, particularly in the context of diverse regulatory settings.

3.5 Conclusion and limitations

Chapter 3 provides a detailed account of the ESRIDM integrated modelling framework, elucidating its design and implementation. Through the presentation of a pertinent case study focusing on three distinct nitrogen application scenarios, the efficacy of the ESRIDM integrated model is showcased. The model demonstrates its ability to generate credible, dependable and practical information. This information, in turn, aids policymakers and growers in making informed decisions and exerting greater control over sustainability-related aspects.

The limitation of the design for the ESRIDM model is that pest control is not integrated, which is also essential to cotton yield. Pest control may be affected by activities such as the reduction of usage of chemicals and lime application, where the action of emission reduction may need a corresponding method on pest control. The ESRIDM model is not yet able to reflect this link. Perhaps future research might be interested in this.

The limitations of this chapter are aligned with the use of publicly available data and the constrained scope, which only encompasses activities until the harvest stage, excluding costs and pollution incurred after harvest. While the design of the ESRIDM integrated modelling framework incorporates scientific modelling and proposes to enhance data quality through simulation methods, the analysis presented in this chapter relies solely on scientific model calculations and publicly available data. These limitations are further addressed and expanded upon in the next chapter.

Chapter 4 Integrating co-production and DSSAT with ESRIDM

4.1 Introduction

Apart from the severe predicament of agricultural emissions reduction discussed in Chapter 3, the endeavour to advance agricultural emission reduction policies is marked by clear difficulties and significance. Nonetheless, a formidable obstacle confronted by policymakers and the wider set of stakeholders revolves around the effectiveness, efficiency and proficiency with which choices about 'what is being regulated' are selected, evaluated and compared.

4.1.1 Motivation

Drawing on the groundwork established in Chapter 3, the primary motivation of this chapter is to address the limitations of current evaluation systems and models in capturing the extent of environmental degradation and potential consequences associated with different nitrogen management practices in agricultural production systems. The overarching aim is to explore a method to identify and evaluate which production resources, technology and practices (herein decision options) are likely to be most sustainable, in that they can effectively mitigate environmental degradation in the production system.

Chapter 3 partly addressed this motivation through the theory development of the Environmentally Sustainable Residual Income (ESRI) integrated modelling and case study demonstration.

However, a notable limitation of Chapter 3 is the absence of guidance on the process of identifying and designing pertinent information and decision system options, essential for their integration into the valuation-oriented integrated modelling framework. The content mentioned herein also correlates with two fundamental challenges elucidated in Chapter 2.

These challenges are imperative for the enhanced practice of regulatory assemblages, especially concerning the initiation and assessment of regulatory options that constitute the assemblages, denoted as: (i) what could or should be regulated, and (ii) which combination of regulatory options are more likely to be effective (cf. Łuczka & Kalinowski, 2020; Campuzano, 2023; Laurett, Paço & Mainardes, 2021).

Furthermore, to evaluate regulatory options, an analysis of decision options should be undertaken which may serve as the subjects of regulation options. To elucidate, a decision option is a defined set of resources, technology and practices utilised in a given production scenario. As discussed in Chapter 2, once decision options that are most likely to lead to more sustainable production have been identified, they may be subject to regulation to support their adoption and diffusion through the production system.

These challenges give rise to the following two sub-motivations: (i) the identification of possible decision options amenable to regulation; and (ii) the identification of how they can be effectively compared in the constantly evolving and intricate circumstances.

Sub-motivation (i): There is a compelling necessity to adopt new approaches that can enhance the effectiveness and efficiency of identifying problems and designing worthwhile scenarios to be tested in improving sustainable production.

One of the big challenges to the emergence of sustainable production for Australian cotton is the complexity of the context, which can be partly attributed to the presence of multiple stakeholders with diverse expertise and roles. These stakeholders, including researchers, policymakers, industry experts and growers (Edwards, Foley & Jasovska, 2022), each contribute valuable experiments and research output to the industry. However, the intricate nature of their interactions and involvement poses non-trivial challenges in fostering effective stakeholder engagement (Barbosa, Mushtaq & Alam, 2012; Mackrell, Kerr & Von Hellens, 2009).

Some research methods have been widely used by researchers for problem identification and worthwhile scenario design in the domain of sustainable production (cf. Taylor et al., 2012; Ma & Zhang, 2022; Giunipero, Hooker & Denslow, 2012; Stewart, Bey & Boks, 2016). These methods have limitations in providing a comprehensive understanding in a timely manner, when facing complex tasks like agricultural sustainable production (Layzer, 2008; Lo, Li & Chen, 2020; Saravade et al., 2022)[39].

Consequently, the establishment of collaborative platforms and processes becomes imperative, facilitating stakeholder engagement in joint knowledge generation, analysis and decision-making (Head, 2014; Tseng et al., 2020; Likens, 2010; Brock & Carpenter, 2007).

In response to this situation, diverse attempts have been made to address this need from various directions (cf. Innes & Booher, 1999; Hage, Leroy & Petersen, 2010; Lang et al., 2012; Ahlborg et al., 2019; Maasen & Lieven, 2006; Francis et al., 2008). While these approaches have shown some effectiveness in improving communication and cooperation within sustainability projects, it remains unclear how they can be integrated with analysis and evaluation modelling systems. Furthermore, the current utilisation of these methods is often isolated; their potential is unclear in conjunction with complementary analysis and evaluation models.

[39] For example, data-driven research relies on empirical evidence and quantitative analysis provides a more objective foundation and the possibilities of comprehensive understanding, when data and modelling of different disciplines are provided (Wang, Li & Wang, 2021; Esty & Porter, 2005). However, it poses its own set of risks, potentially neglecting subtle factors that defy easy quantification (Niemeijer, 2002; Yang, Li & Huang, 2020). It also lacks the capacity to offer profound insights into the sociocultural and political dimensions that shape regulatory landscapes, thus compromising the ability to make well-informed decisions and identify regulations suitable for specific contexts (Waas et al., 2014; Likens, 2010).

Sub-motivation (ii): To explore the effective leverage of simulation advantages for option evaluation and comparison in Environmentally Sustainable Residual Income Decomposition Modelling (ESRIDM).

Chapter 3 utilises publicly available data for analysis. While publicly available data is widely used by researchers, concerns have been raised regarding its reliability and comprehensiveness, particularly in the context of decision-making (Dekker, Groot & Schoute, 2012; Ittner & Larcker, 2001; Susha et al., 2015; Wilkes et al., 2020; Pham et al., 2020). Researchers underscore the criticality of accessing high-quality data[40] to enhance the likelihood of successful decision implementation and sustainability improvement (Blichfeldt & Eskerod, 2008; Elonen & Artto, 2003; Citroen, 2011; El Bilali & Allahyari, 2018; Andreoli & Tellarini, 2000; Blackmore et al., 1995).

Pham et al. (2020) analysed this issue and proposed a method to incorporate cropping modelling and scenario simulation into the Water and Economic Sustainability Performance Measurement (WESM) model they developed. While simulation methods have been widely adopted in sustainability practices by various researchers (cf. Belcher, Boehm & Fulton, 2004; Kotir et al., 2016; Bongiovanni & Lowenberg-DeBoer, 2004; Whitbread et al., 2010), Pham et al. (2020) made notable contributions by designing and theorising a scientifically and hierarchically valid framework that can be applied to support decision-making in environmental and economic sustainability.

However, Pham et al. (2020) only address a single module (irrigation module), so it remains unclear how simulations can effectively leverage their advantages in the context of Environmentally Sustainable Residual Income Decomposition Modelling (ESRIDM) which

[40] Bovee et al. (2003) discussed the theoretical potential to conceptualise attributes of high-quality data, including interpretability, relevance, accessibility and integrity.

incorporates multiple correlated modules, such as broad economic, environmental influence, irrigation and nitrogen.

4.1.2 Objective

Chapter 3 of this study presents a theoretical framework for designing integrated modelling systems that facilitate the identification and evaluation of alternative practices, technologies and other factors that can contribute to a more sustainable approach to crop production.

Chapter 4 aims to accomplish the same overarching objective as Chapter 3. Chapter 4's objective is to develop and explore a method to identify, evaluate and evolve plausible production system options amenable to GHG emission regulation in the context of agriculture production.

Specifically, it delves into the exploration of the design of an integrated modelling system that enables the identification and evaluation of alternative practices, technologies and other factors that can promote more sustainable crop production, with a specific focus on nitrogen management.

This chapter delves deeper into the exploration of sustainability with respect to various methodologies for problem identification and the corresponding scenarios for evaluation. Such approaches potentially enhance the efficacy of integrated analysis models. While simulations in the domain of sustainability have been extensively researched and implemented, as evidenced by their proven capability to enhance data quality (Pham et al., 2020), there is still ambiguity surrounding how these simulations can efficiently harness their strengths within the context of ESRI-integrated modelling. This chapter is predominantly centred around these two pivotal themes.

In addition, it should be emphasised that the complete adoption and integration of co-production falls outside the purview of this thesis[41]. For the segment pertaining to co-production, this chapter aims to investigate its role in theoretical development. Certain facets of co-production have been incorporated during the experiment design phase, and there is an anticipation of further research to assimilate this modelling within a more expansive co-production process.

4.1.3 Method and data source

4.1.3.1 Method

Chapter 3 identified the absence of a comprehensive information system in the field of sustainability as a crucial missing component hindering sustainable development in the medium to long term.

This chapter introduces two methods aimed at further enhancing the integrated modelling presented in Chapter 3. First, the co-production of knowledge (Polk 2015) of framework is partly introduced to ERSI scenarios/assumptions, when developing scenarios and regulatory assemblages. The co-production of knowledge theory on data collection is particularly advantageous in modelling nonlinear and multi-core sociological research with interdisciplinary cooperation (Weimer and Ruijter, 2017). It effectively addresses complex

[41] In the paradigm of co-producing knowledge, particularly within the realm of transdisciplinary research, certain frameworks and corresponding methods of information collection and interaction have been established. These are evident in practices such as workshops, as outlined by Polk (2015). However, the present study deviates from these established frameworks and activities. For instance, the interaction with stakeholders detailed in this chapter was initiated through preliminary contacts and brief dialogues established at the Australian Cotton Conference in 2022. Subsequent interactions were facilitated through electronic mail and video conferences, coupled with continuous sharing of the research progress and drafts to obtain feedback. Furthermore, the design of ESRIDM incorporates broader stakeholder engagement feedback and exhibits greater reflexivity, as presented by Polk (2015). Yet, this exceeds the scope of this paper, which primarily focuses on the design based on ESRIDM. More detailed design and implementation regarding this might be explored in future studies.

problems that require the integration of multidisciplinary professional knowledge and participation (Lino et al., 2019). These characteristics align with the scenario setting and data collection requirements for ESRI-integrated modelling.

Second, the ESRIDM model proposed in Chapter 3 is integrated with cropping software, namely Decision Support System for Agrotechnology Transfer (DSSAT). DSSAT facilitates comprehensive analysis and extensive simulation of scenarios while being closely linked to climate modelling, thus providing reliable analyses within the required timeframe and climate prediction model. Furthermore, DSSAT offers cost and time savings compared to other experimental methods, such as field tests.

Here is a short explanation of the variables that exhibit variations among the different identified scenarios. A more detailed explanation follows in the research design section.

S1: Furrow irrigation + Slow-release fertiliser

Sulphur-coated urea is one kind of controlled-release fertiliser that reduces fertiliser leakages, by controlling the dissolution rate to make the nitrogen fertiliser gradually provide nitrogen for crops in the process of slow dissolution (Jarrell & Boersma, 1980).

In this scenario, slow-release fertiliser may be used as fertilising material, and furrow irrigation is applied. The outcome is expected to show the difference between S1 and S2, to see whether the adoption of slow-release fertiliser can cover the negative impact on GHG, and which scenario is more likely to achieve sustainability.

S2 and S3: Furrow irrigation + Urea

The furrow irrigation method is selected, because it is the most adopted irrigation choice on Australian cotton farms (CottonInfo, n.d.).

Urea is selected because it is one of the widely adopted fertiliser choices in general farm operation practice, and this is for the need to control variables to form an effective comparison with the more sustainable fertiliser group (Schwenke et al., 2022).

4.1.3.2 Data source

The evidence to be presented in Chapter 4 is derived from four key sources of data collection: (i) engagement with key stakeholders during the 2022 Australian Cotton Conference; (ii) co-production process and project cooperation with key industry stakeholders to derive a set of plausible alternatives; (iii) the ESRI modelling calculations developed in Chapter 3, with a crop simulation (using DSSAT) to provide the outputs of those scenarios and production system options decision-making; and (iv) public available data for economic analysis, such as pricing data.

A key output of this chapter is a simulation experiment to demonstrate the integrated model, as well as explain and predict what options for reducing GHG emissions are most likely to be environmentally and economically sustainable in the context of primary production. The model provides estimates of the likely magnitude of impact for each of the modelled scenarios, as well as quantification of the benefit and sacrifice for each proposed option.

Thus, Chapter 4 is expected to be able to (i) identify a set of plausible emission regulation scenarios; (ii) develop expectations about what the likely impact of those scenarios would be on stakeholder decisions and actions; and (iii) evaluate the likely long-term sustainability dynamics of these scenarios. In Chapter 4, the scenarios identified are used as examples to show how the decomposition model functions.

4.1.4 Findings and contribution

There are three findings for this chapter.

(1) The model and simulation uncover trade-offs or synergies between crop production and sustainable production goals. It identified some circumstances where reducing costs may impact crop yield but result in better overall profit and environmental outcome. These findings could help in making informed decisions or regulatory targets by considering the trade-offs involved. This situation may occur when the drop in economic yield is smaller than the saving from less fertiliser purchased. In other words, this situation is more likely to be observed when: (i) no significant decline in economic yield, due to either insignificant decrease in lint yield or low lint sales revenue; (ii) significant increase in nitrogen fertiliser market price; and (iii) the excessive application of nitrogen fertiliser has reached a threshold where incremental increases result in minimal yield enhancement, or conversely, lead to yield degradation due to over-nutrition. In this case, when fertiliser application is reduced, it may not trigger a significant reduction in crop yield or even improved yield.

(2) The model assists with revealing correlations or relationships between crop production decisions (e.g., input usage), financial indicators (e.g., revenue, expenses, profitability) and sustainability indicators (e.g., nitrogen emissions). These findings could help explain the impact of various factors regarding different leverage to raise certain regulation controls for sustainable crop production.

(3) The model provides insights into the effects of different management decisions (e.g., changes in input methods or types) on crop production and financial performance. These findings could indicate the effectiveness of specific practices or strategies in achieving desired sustainable outcomes.

This chapter makes a number of theoretical and practical contributions to literature.

These contributions entail two main aspects: (i) a segment of the co-production theory is integrated into the domain of sustainable cropping and nitrogen management on problem identification and scenario settings, and (ii) this theory is utilised to enhance the efficiency and effectiveness in assisting with identification of possible regulatory targeting and facilitating decision-making processes.

First, a contribution is to integrate the co-production of knowledge theory into problem identification and scenario setting of sustainable agriculture production. Co-production of knowledge theory, as a transdisciplinary research theory, has immense potential for bridging stakeholder engagement and facilitating multi-domain transdisciplinary research. The co-production of knowledge has demonstrated notable efficiency and effectiveness in enhancing communication within sustainable projects involving multiple disciplines and framework development (cf. Zarei, Karami & Keshavarz, 2020; Schneider et al., 2019; Djenontin & Meadow, 2018; Vincent, Daly, Scannell & Leathes, 2018; Kliskey et al., 2023; Davis et al., 2011; Yorgey et al., 2017). However, its implementation in the context of cropping systems and nitrogen management remains limited.

To be specific, studies utilising this theory have not sufficiently explored its applications in the areas of environmental policy formulation, evaluation and implementation – especially in the context of agriculture and, more specifically, nitrogen management decisions and applications. In academic discourse, Polk's (2015) co-production framework allows for flexibility in research methodologies and is adaptable to various sustainability issues. While Chapter 4 diverges to some extent from the exact parameters of Polk's (2015) framework, the inherent flexibility of this model allows for varied methodological adaptations. Within this context, this chapter can be construed as contributing to filling this particular gap to link co-production theory to the context of agriculture and integrated modelling.

Second, this chapter expands the scope of the simulation experiments conducted by Pham et al. (2020). Pham et al. (2020) recognise the benefits that cropping software-based simulation brought to information providing, sustainability evaluation and decision-making. However, Pham et al. (2020) fail to reflect on the inherent connection between irrigation methods and the effectiveness of other production resources, such as nitrogen fertilisers, which also exert a significant influence on both crop profitability and production sustainability. Also, Pham's work primarily focuses on water use efficiency, which carries the risk of triggering the Jevons Paradox (Alcott, 2005).

In comparison to Pham et al. (2020), this research makes two contributions in extending their work: (i) Chapters 3 and 4 collectively present modelling to extend the simulation scope of Pham et al. (2020) from irrigation management to nitrogen management and other significant cropping emission drivers (for example, lime activities and diesel usage), while demonstrating the link between sustainability outcomes and production system options choices; and (ii) Chapters 3 and 4 develop modelling approaches and formulas to calculate the ESRI indicator, which represents the overall sustainability performance of production systems. Furthermore, Chapter 4 builds on the advancements made in Chapter 3, by providing additional demonstrations that incorporate multiple variables into the calculations and offer further identification, comparison and analysis of different regulatory scenarios.

In addition, as elaborated in Chapter 3, despite the ongoing iterative advancements in integrated modelling and software simulations, extant research predominantly focuses on simulation accuracy and short-term projections （Guo et al., 2021). It does not engage extensively with simulations under hypothetical scenarios of policy or decision-option implementation. Furthermore, there is a notable absence of design intent and capability for the decomposition, attribution, tracing and analysis of emergent phenomena.

Third, this chapter extends the utilisation of the DSSAT, by introducing a novel method that demonstrates how DSSAT can be employed to analyse cropping operations within the realms of accounting and sustainable production. Up to this point, DSSAT has played a substantial role in advancing research on sustainable production, with a primary emphasis on the field of botany. Limited research has been conducted on the application of the DSSAT in the context of cropping management decisions (cf. Jones et al., 1998; Thorp et al., 2008; Phetheet et al., 2021). Some have attempted to apply DSSAT to accounting or economic evaluations within the domain of sustainable production (cf. Sarkar, 2009; Sarkar & Kar, 2008; Thorp et al., 2008). Nevertheless, the investigation is limited as to how DSSAT can be used in exploring the potential alternatives for sustainability regulatory assemblages and evaluating possible outcomes in different scenario settings.

Additionally, for DSSAT, one of its constituent modules is an economic module, specifically engineered for executing rudimentary profit-and-loss calculations during a single simulation run (DSSAT, v4.8). Nevertheless, due to its limited incorporation of basic pricing elements, this module is not equipped to evaluate large-scale or intricate planting plans or conduct long-term scenario simulations. However, it does hold the potential for further refinement and development to fully realise its intended functionalities.

The contributions of this chapter hold significant importance for two reasons. First, they offer the capability to deconstruct sustainable production regulations into discrete decision-making options through a co-production process. This enables a feedback loop that facilitates enhanced communication, as well as iterative adjustments and improvements to the said regulations. Second, the chapter establishes a nexus between the identified options and the quantification of associated risks and opportunities, employing quantitative scientific modelling and simulations. Consequently, this chapter might be of interest to those who wish to advance the efficiency and standardisation of regulatory planning in the realm of sustainable agricultural production.

4.1.5 Structure

The chapter follows the following structure: Section 4.2 supplements the literature review presented in Chapter 3, by including an additional literature review on the co-production of knowledge and an analysis of the integration of simulation within the design of integrated models. Specifically, the chapter aims to (i) theorise the role of co-production of knowledge in integrated modelling; (ii) provide a theoretical exploration of the reasons for and mechanisms through which simulation contributes to integrated modelling; and (iii) present a visual representation in the form of a figure that demonstrates the integration of ESRI modelling with co-production of knowledge and simulation.

Section 4.3 describes the research design, including experiment setting, enacted co-design and scenario set-up, integration of a specific simulation package DSSAT including configuration, and overall simulation experiment design.

Section 4.4 delves into the outputs, findings and subsequent discussion of this chapter.

4.2 Literature review

4.2.1 Co-producing theory, regulation assemblages, regulatory options and production system options

Numerous studies have made efforts to identify regulatory assemblages that effectively achieve regulatory objectives. Notable examples include Coase's (1960) influential work on social cost, which employed transaction cost economics and economic modelling to emphasise the significance of pricing externalities. Tietenberg (2006) analysed the pros and cons of carbon taxes and tradable permits, considering their suitability in different economic contexts. Zhou et al. (2018) aimed to evaluate emission levels and reductions, by analysing environmental data from 71 cities between 2005 and 2012.

Perman (2011) quantified and formulated various emission reduction systems, presenting diverse economic models to assess the relationship between social welfare and externalities arising from environmental pollution. Litman (2013) conducted a comparative study on energy management improvement and the promotion of clean energy sources. The research revealed that the effectiveness and outcomes of different emission reduction schemes varied, depending on whether external influences were excluded or considered. Surprisingly, the scheme that appeared advantageous may, in practice, have been less effective than expected which can mislead expectations and may result in an overall emission reduction outcome that is less effective than initially assumed. This finding was corroborated in Chapter 3, where the tree offset project was not capable of fulfilling the offset need of cotton production.

While the above publications have refined and calibrated our understanding of regulation, it has been observed that enacted regulation is complex but has similar characteristics when regulation design is first, within a specific timeframe, one policy type predominated, with others serving complementary roles. Second, the change of policy is not smooth and generally driven by multiple leading forces. Third, some literature demonstrates the advantages of carbon pricing while the confusion of regulation classification definitions exists and analysing the output data of regulation assemblages but not singular regulation as they claimed. Thus, it is difficult to generalise from these papers to specific settings.

In more recent years, researchers such as Polk (2015) have introduced alternative approaches to tackle this challenge, by incorporating co-production methods. Polk (2015) integrated the co-production of knowledge theory with economic modelling, to assess the micro and macro impacts of scenarios involving multiple stakeholders. While the co-production of knowledge offers advantages in theory development, it falls short in evaluating the long-term efficacy of policies and decisions, necessitating the inclusion of scientific backup for comprehensive analyses.

The methodology of "knowledge co-production", introduced by Polk (2015) and initially applied in the field of urban sustainable development, employs inclusive processes to capture the diverse perspectives of stakeholders from various professional fields. This approach is particularly valuable for modelling multi-core sociological research that necessitates cross-domain collaboration (Weimer and Ruijter, 2017). Typically, co-production processes are conducted collaboratively within research groups, effectively gathering data from a multi-professional standpoint while focusing on interdisciplinary research topics.

Polk (2015) introduced a method of co-production and delineated its procedural steps. This approach is an extension of Godemann's (2008) three activities concerning knowledge integration, which encompass: (i) the exchange of information and knowledge; (ii) the establishment of a shared foundational understanding; and (iii) meta-reflexivity within groups, as described by Godemann (2008).

While the theoretical framework of co-production and the methodology delineated by Polk (2015) have significant applicability to sustainable production research, the present study only partially incorporates Polk's approach, drawing inspiration rather than adhering fully to its precepts. The primary advantage of knowledge co-production theory, in conjunction with this research, lies in its ability to address complex problems that require input from multiple disciplines and stakeholder participation, such as public policy (Lino et al., 2019). These complex issues cannot be easily categorised solely as social or natural science problems, as each field's professional knowledge holds equal importance. For instance, environmental regulation challenges call for the expertise of diverse professionals. Knowledge co-production facilitates the effective exchange and application of professional knowledge through collaborative knowledge output, thereby bridging the gap between academic research and policy implementation (Polk, 2015). This methodology not only provides a theoretical basis, but also offers a successful case study that can serve as a valuable reference for this research.

With the above advantages, the co-production of knowledge methodology has the potential to contribute to sustainable production at the broader regulatory assemblage level, by aiding in the identification of policies that can effectively achieve sustainability goals or assess risks associated with those regulatory arrangements. Nevertheless, the initial step in establishing potential regulatory assemblages is to identify feasible regulatory options that are desirable and amenable to regulatory support. This chapter aims to explore how the ESRIDM model may be further adapted to enable the identification and evaluation of possible regulatory options in various of production system options through co-production of knowledge. In the following sections, a set of identified production system options are tested and evaluated.

The co-production of knowledge approach plays a crucial role in identifying specific components of environmental regulation options, such as various cropping practices including fertiliser selection or irrigation methods. Attempts have been made to explore the use of co-production on regulations (cf. Maiello et al., 2013; Lemos & Morehouse, 2005; Galende-Sánchez & Sorman, 2021).

However, one area that remains less clear is how the co-production method enables the modelling of alternative scenarios. To address this gap, the research incorporates elements of co-production theory with ESRI modelling to identify decision options, which may be amenable to specific regulatory options through a separate co-production process. This supports simulation set-up for DSSAT crop science modelling, addressing the void between co-production theory and simulation-based investigation. Collaborative efforts between scientists and growers, employing the co-production of knowledge approach, are expected to produce meaningful simulations for further analysis (Carolan, 2006a; Carolan, 2006b; Bartels, Furman, Diehl et al., 2013; Pham, 2016; Reed & Paivi, 2018; Pham et al., 2020).

4.2.2 Using simulations for emissions and regulations

In the realm of research concerning nitrogen emissions and policy recommendations for emission reduction, the utilisation of modelling and multi-year simulations to investigate the possibilities for policy intervention has precedence. For instance, Luo et al. (2019) employed modelling and simulation techniques to approximate a 37-year nitrogen emissions profile across multiple provinces in China, as well as to analyse potential scenarios for N_2O reduction. This study illuminated the prospective impact of long-term modelling and simulation on emissions mitigation and strategic alleviation, as well as its utility for identifying policy entry points. However, a significant limitation, akin to many other similar studies, is the lack of consideration given to the economic aspects, which constitute another vital dimension of policy analysis (cf. Gu et al., 2015; Liu et al., 2020).

Other studies have proposed policy recommendations based on emissions modelling and also incorporated economic evaluations. However, these economic assessments frequently employ public macroeconomic data, such as Gross Domestic Product (GDP). This approach implies that the correlation between emissions mitigation options and macroeconomic variables is not necessarily robust[42] (Gu et al., 2012; Zhang et al., 2020; Wu et al., 2018).

[42] Two primary issues arise in relying on macroeconomic data for modelling emissions and policy analysis. First, macroeconomic variables are influenced by a myriad of factors and cannot definitively attribute changes to shifts in agricultural practices. This is also applicable to farmers' choices, as variations in land and fertilizer usage may sometimes be triggered by extraneous factors, such as the output of the chemical fertilizer industry or wage levels in factories. For instance, within the socio-economic context of China, if

Consequently, it is challenging to accurately estimate the changes in emissions resulting from shifts in choices by regional cultivators. Similarly, this approach limits the capacity for nuanced, sector-specific policy guidance.

In this section of the chapter, an analysis is conducted on the application of simulation, building upon previous research in the field. The discussion elucidates the appropriateness and advantages of employing simulation in the context of ESRIDM modelling. The utilisation of simulation offers inherent advantages in assessing and evaluating the effectiveness of policies in diverse contexts. To be specific, to analyse cropping system-related decision-making, a simulation system is necessary for the following three reasons:

(I) The success of cropping production relies on plant growth.

Researchers have identified key elements in cropping production, encompassing soil preparation (tillage), planting, nutrient and pest management, irrigation, harvest and post-harvest processes. This delineation implies that the production process yields discernible outcomes.

Moreover, the adoption of different management practices and decision-making when selecting among these practices significantly influences the economic and environmental outcomes of the system. The relationships between these indicators' interactions and crop

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factory work gains higher income than farming, agricultural lands may be left fallow or inadequately attended to during that year.

Second, there exists a temporal lag and ambiguous boundaries between macroeconomic statistics and the production decisions of cultivators. For example, emissions stemming from the production phase, such as CO2 emissions from nitrogen fertiliser manufacturing, are often not considered. This lack of comprehensive data integration complicates the formulation of precise and targeted policy directives. Therefore, policy recommendations derived from models that employ only macroeconomic variables may not be fully equipped to address the complexities inherent in emissions mitigation and economic considerations.

output exhibit non-linear characteristics (Avissar et al., 1985; Forbes & Watson, 1992; Guo et al., 2021)[43].

Thus, the crop production process outcomes exhibit variability across regions and years, owing to the inherent characteristics of agricultural systems. This variability stems from diverse environmental and climatic conditions, as well as the dynamic nature of agricultural practices, contributing to fluctuations in the data (Pham, 2016).

This reflects one of the reasons why simulation is essential. Simulation enables the reflection of the consequences of such variability, allowing for a comprehensive understanding of the impacts and outcomes of crop production processes under different conditions.

(II) The data quality of crop simulation has been validated and endorsed by researchers in related fields.

The utilisation of cropping production simulation models has been prevalent for several decades, with their origins traced back to the 1950s (Reynolds et al., 2018). These models have continuously evolved and been refined through rigorous comparisons with field tests and experiments, benefiting from advancements in related disciplines (e.g., Heeren, Werner & Trooien, 2006; Yakoub et al., 2017; Abedinpour, 2021; Saseendran et al., 2007).

(III) Simulation provides greater flexibility and cost savings compared to field experiments.

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[43] This phenomenon arises due to the non-linear nature of many physiological processes. These processes often display a threshold response, wherein plant performance remains unaffected within a certain range of stimulus, but declines as the stimulus exceeds this range. Additionally, plants possess the capacity to maintain stability in the face of changing external environments, resulting in non-linear responses to inputs rather than a linear relationship (Avissar et al., 1985; Forbes & Watson, 1992).

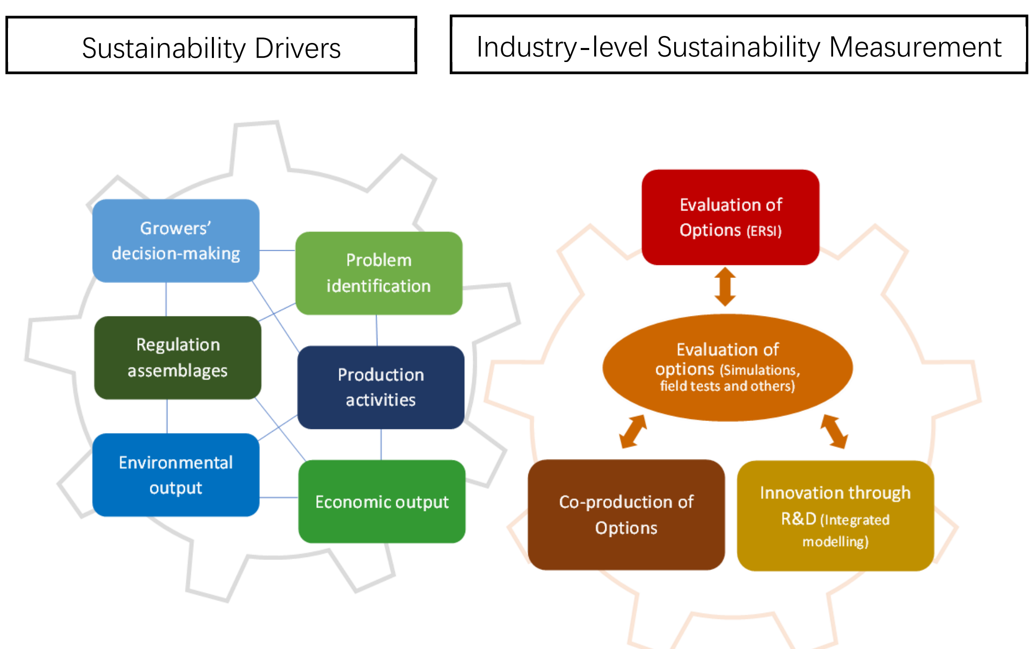
Substantial work has shown that crop simulators (e.g., DSSAT) are able to model different crop production scenarios, including yield and links between crop production and climate change, with a level of validity, at lower cost and timelier than field trials (cf. Jones et al., 2003; Thorp et al., 2008; Whisler et al., 1986; Brisson, Seguin & Bertuzzi, 1992).

Furthermore, cropping simulation models are developed as process-based simulations, focusing on the planting production process. This approach enables stakeholders to assess and analyse environmental and economic risks, profitability and sustainability of practices. By employing process-based cropping system simulation, a comprehensive understanding and effective management of these aspects can be achieved (Pham, 2016).

4.2.3 Figure illustration for theorised co-production model

Figure 4.1 presents a theoretical model illustrating the role of co-production in both the sustainable drivers and industry-level sustainability measures in the decision-making process, highlighting the interactions among various components.

Figure 4.1: Nomothetic model of theorised interactions of drivers and information in dynamic co-production of sustainable production systems



The left segment represents sustainability drivers at the societal level, which is the system of factors relating to regulation and crop production. There are six elements selected as drivers: growers' decision-making, problem identification, regulation assemblages, production activities, environmental output, and economic output.

It is essential to incorporate both growers' decision-making and regulatory assemblages when aiming to achieve sustainable production. Previous studies have indicated that although improvements in input usage efficiency can facilitate the transition to sustainable cropping practices, the successful attainment of sustainable production in agriculture is contingent upon the introduction of relevant policies (Manhire et al., 2012; Boland et al., 2006).

However, in this model, growers and regulation are treated as separate components due to the direct involvement of growers in production activities and the specific challenges they encounter, which may result in barriers or failure in regulations. Empirical evidence suggests that the resistance exhibited by growers contributes to the delay in adopting sustainable production methods and technologies (Pannell, 2003; Manhire et al., 2012).

Within this part, sustainability drivers interact with economic and environmental considerations, influencing decisions and actions, and subsequently provide feedback or receive recommendations and measurements from industry-level sustainability measures through information channels.

The right segment represents sustainability measures at the industry level.

Within the segment representing sustainable drivers on the left side, the decision-making process of growers exerts a direct influence on production activities, leading to diverse environmental and economic consequences. Policymakers, utilising information systems such as professional reports and surveys, observe these outcomes and subsequently advance or modify corresponding regulatory frameworks, thereby further influencing the decision-making of growers. During these processes, problems continue to be identified, and all sustainable drivers communicate information with sustainable measurements through various information channels/systems.

Within the industry-level sustainability measures section on the right side, researchers and professionals receive requests or encounter issues via information channels. They subsequently formulate testable options through co-production, fostering the advancement of evaluation systems and models. These advancements are realised through diverse methodologies such as simulation and field tests, ultimately yielding quantifiable outputs and industry-level sustainability measures.

4.3 Research design

4.3.1 Overview

This research follows a similar method to Pham et al. (2020), whereby this research develops and integrates (i) a theoretical model of a typical Australian cotton farm; (ii) LCA of nitrogen fertiliser production; and (iii) a management accounting model which combines financial and non-financial data. In addition, this research modelled the sustainability effects in the long term from macro and theoretically analysed how regulation and nitrogen practices may affect agriculture sustainability performance. Pham et al. (2020) also evaluate their model using simulated data, as is done in Chapter 4 of this thesis. Building upon the work of Pham et al. (2020), this chapter models three scenarios, delving deeper into a comprehensive assessment of both environmental and economic impacts, while also providing an estimation of financial returns.

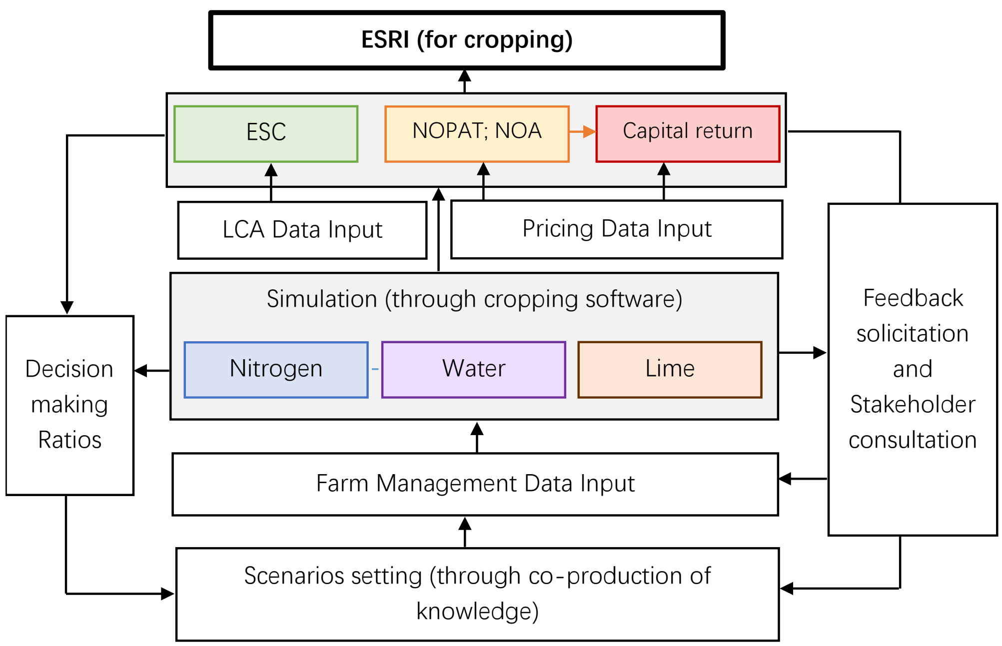
Cotton is an important cash crop in many economies (Baffes, 2005) and the primary source of natural fibre (Gordon & Hsieh, 2006). However, cotton production has brought an environmental burden that cannot be ignored, especially agricultural chemical overuse and water resources pollution (Chapagain et al., 2006).

Australian irrigated cotton nitrogen application ranges from 93 to 500 kg/ha (Macdonald et al., 2016), and the mean of Australian cotton farming nitrogen application is 335 kg/ha (Baird et al., 2019, p.23) and assessed to be "low efficiency" (Antille and Moody, 2021)[44].

This chapter focuses on the direct and indirect nitrogen emissions brought by the production and application of nitrogen fertiliser and crop growth.

Figure 4.2 integrates the Figure 4.1 model with the model developed in Chapter 3.

Figure 4.2: Data flow for co-production of sustainable crop production decision options



[44] It is a harmful effect because most of the nitrogen is used in agricultural production and all the nitrogen formed in the combustion of fossil fuels is lost to the environment, causing greenhouse warming and shifting ecological balances of natural ecosystems. (Follett and Hatfield, 2001).

In this context, the scenarios are devised through the collaborative process of co-production of knowledge with stakeholders. This entails identifying and analysing simulation options, taking into consideration the stakeholders' perspectives on feasibility and acceptance, ultimately resulting in the formulation of scenarios.

Once the scenarios have been defined, simulations are employed to assess the nitrogen, water and lime modules, utilising software and additional tools like LCA calculations. The outcomes obtained during this phase are subsequently fed back into the co-production process to evaluate and, if necessary, adjust the settings. This iterative approach enhances the accuracy of the scenario simulations and improves data quality, enabling a comprehensive analysis of the potential effectiveness and risks associated with implementing the regulatory options identified at the outset.

This chapter devises and evaluates three scenarios, by using the following procedure for scenario selection and experiment. First, identify scenarios through a co-production of knowledge process with key stakeholders. Second, simulation design and data collection in collaboration with key stakeholders. Third, estimate simulation of the scenarios utilising crop simulation software, such as DSSAT. Fourth, estimation of Lifecycle Assessment (LCA) for pre-farm emission, economic impact and environment residual modelling to evaluate the scenarios. The focus of this chapter concentrates on data gathering, texting and experiment outcome analysis.

To provide a more elucidating demonstration of the integration of the co-production of knowledge theory and DSSAT into a unified framework of integrated modelling, the following Figure 4.3 depicts the process and scenarios selected.

Figure 4.3: Experiment working flow

① Scenarios setup

Australian Cotton Conference 2022 (bullet points)

- It is important to improve sustainability in cotton industry
- Attempts still focus on two strategies: irrigation management and nitrogen management
- Dry land (or rain grow) cotton is promoted
- Economic and policy change has an obvious impact on cotton planting profit-cost
- High proportion of innovations are around sustainability improvements
- It is still yet not clear which measures are the most effectives and the overall impact on broader economic/environment

② Data collection

Scientists

Basic DSSAT settings:
- Weather station
- Soil detail
- Fertiliser detail
- Cultivar
- Suggestions (promoted nitrogen and irrigation plan/any other N emission reduction measures)

Growers

Nitrogen management and Irrigation plan:
- Soil analysis report (optional)
- Fertiliser management plan
- Irrigation infrastructure and irrigation plan

③ DSSAT

S1:
Furrow + slow-release urea fertiliser

S2:
Furrow + urea fertiliser

S3:
Furrow + urea fertiliser

④ Proceed evidence based integrated modelling

Question to be answered:

- Can effective nitrogen emission can be achieved through the restriction of fertilizer usage? How effective can be for regulatory effects?

Step 1: LCA analysis for fertilizer production
Step 2: DSSAT output management
Step 3: Profit and loss analysis (annual) and residual (accumulate)
* If either of the residual turn negative (This means the farm cannot keep business running by itself or 2 ℃ target cannot be reached with "this century's" technology level).
* Add value/unit will be compared between different scenarios.

Notably, the approach whereby I use a statistical test to evaluate the output in this chapter follows a long research tradition (c.f. Pham et. al., 2020; Malik, Jiménez-Aguirre, & Dechmi, 2020; Dzotsi, Basso, & Jones, 2013; Tyagi, Singh, Sonkar, & Mall, 2019; Kipkulei et al., 2022; Li et al., 2015; Pathak, 2017.

4.3.2 Identification of plausible scenarios

4.3.2.1 Conference and scenario designs

The processing of the co-production of knowledge method is designed as follows. (i) Desktop review during the Australian Cotton Conference 2022, including discussion with growers and other stakeholders about general cropping processes such as actions, options and management practices. (ii) Refinement of options and scenarios drawn from desktop review. (iii) Collaboration with cotton scientists to first, assess rationality and possibility of several sustainable practices and second, flesh out scenarios to run simulations on. (iv) After the conference, establish theoretical modelling and run simulations to produce results and seek further feedback from stakeholders. Please note that the seeking of feedback has been simplified due to the scope of PhD research. By design, this requires further research, such as behaviour research and economic modelling.

The Australian Cotton Conference 2022 was held in August on the Gold Coast. During the 2022 Australian Cotton Conference, consultation conversations were applied to help with setting up the co-produce scenarios of different options and a choice experiment in collaboration with key stakeholders, including growers and other stakeholders.

This presented a favourable opportunity to employ the co-production method for scenario development, as it leveraged the extensive engagement of diverse stakeholders within a prominent and expansive platform. By employing the desktop review method under these circumstances, a comprehensive and efficient comprehension of current technologies and management strategies in the Australian cotton industry could be achieved with a high degree of effectiveness.

For stakeholders' selection and feedback, five steps have been taken. First, prior to the Australian Cotton Conference, information regarding the forthcoming speakers and their respective topics was made available on the official website. This data was collected, categorized, and then filtered to select speakers whose presentations were related to "nitrogen management, irrigation, GHG emissions, and soil health." The initial screening identified 24 policymakers, 17 farmers, and 25 researchers/experts. Subsequently, a comprehensive collection of publicly accessible information about these stakeholders was undertaken, encompassing biographies, publications, research experiences, and contact details. This process involved categorizing them based on a subjective assessment of their relevance to "nitrogen management and GHG emissions" into three priority levels: "a" (must not miss), "b" (hope to converse with), and "c" (others). It is noteworthy that while the contact details for most policymakers and researchers/experts were obtainable, those for the majority of farmers were not.

Notably, for targets at level "a," some had already engaged in online discussions with us prior to the conference, and we had received referrals from their contacts. The primary focus during the Australian Cotton Conference was to exchange updates on the current progress of my work and solicit feedback. For targets at levels "b" and "c," our attention was directed towards gathering information from them, such as current management/cultivation practices and inquiring about their interest in responding to similar queries in the future and to gauge their interest in our research.

The third step involved engaging with stakeholders not presenting at the conference, primarily conducted at the commercial exhibition related to the cotton industry held on the first floor, where inventors, commercial partners, and their clients (farmers and staff) were the main participants. Inquiries and conversations with these stakeholders were predominantly held at the exhibition on an availability basis whereby we approached people and asked for referrals, consistent with a snowball sampling approach. Information was documented through notetaking, recording/photography, saving online links, and collecting business cards.

The fourth step entailed organizing, reviewing, and discussing the content of these conversations post-conference, leading to refinements of the original three scenarios. The fifth step involved drafting the refined scenarios and research design into documents, which were then sent to stakeholders willing to review and provide feedback, primarily targeting researchers and experts specialised in cropping nitrogen management, nitrogen emissions, and carbon trading.

Presentation topics were gathered, and short open questions were asked. For example: What do you think is the best practice to improve nitrogen sustainability? What would you believe to be the most sustainable practice in the future? Would you like to adopt that for your farm? Why? What is your current nitrogen and irrigation plan?

The outcomes of this conference have been collated and integrated into scenario-based data utilisation, subsequent to comparison and optimisation through an extensive review of the relevant literature. Detailed explanations can be found in Sections 4.3.3 and 4.3.4. Table 4.1 below shows a tentative list of key stakeholders.

Table 4.1 2022 Australian Cotton Conference consultation conversations

Conference participants	No. of people
Cotton growers	10
Independent experts	12

Conference participants	No. of people
Commercial partners	14
Total	36

After communication with the participants of the Cotton Conference, scenarios are designed to assess the impact on cropping output, and whether it is necessary to engage in more sustained practice under the current technology level, while considering the influence of climate conditions. Additionally, potential changes in management practices, such as the direct reduction of fertiliser use and alteration of fertiliser choice, should be incorporated and examined. This approach provides insights into the potential effects on cropping output and enables a comprehensive evaluation of the interplay between these factors.

Proposals to enhance sustainability have been put forward. Yet, there's a noticeable lack of assessment for suggested strategies, like drip and spray irrigation, considering the prevailing technological and economic standards. This deficiency extends to understanding the potential economic and environmental consequences of implementing measures such as the universal application of drip irrigation or the compulsory use of slow-release fertilisers.

The general practice of nitrogen fertiliser production and on-farm usage is to simulate typical farm production and management activities without adjustments beyond their operation.

From these situations, Scenario 1 is designed as: furrow irrigation and more sustainable (slow-release) fertiliser, using the most economically efficiency application amount (234 kgN/ha from Antille & Moody, 2021); Scenario 2: furrow irrigation and chemical fertiliser with the same application as S1; and Scenario 3: designed as a comparison group, using the average fertiliser application amount (275 kgN/ha from Scheer et.al., 2023). This control group is designed to compare the difference between general practices on the field by growers (S3) and possible differences brought by changing to more sustainable practices (S1). A reduction in the amount of fertiliser application, without altering the fertiliser type, is implemented as part of the experimental design (S2), to investigate the impact on the outcomes of DSSAT and ESRIDM simulations. This approach ensures that any variations observed in the results between the two models are not solely attributed to the decrease in fertiliser quantity. Moreover, this approach showcases the capability of ESRIDM to simultaneously explore multiple scenarios and test diverse assumptions concurrently. The selection of data has been presented and communicated to scientists in the field of cropping. Additionally, it has been compared with the total amount of fertiliser used, as well as the specific amounts applied in the first and second instances, as communicated by growers at the Australia Cotton Conference 2022.

Furrow irrigation is the most generally used practice and though growers hold a positive attitude to emission reduction and environmental protection, they also show strong views on keeping yield level and limiting costs associated with farm management practices.

4.3.2.2 Abstracts collected from conference on enhancing sustainability for nitrogen fertiliser

There are three propositions to improve nitrogen fertiliser towards sustainability: change source, change practices and offset pollution (analysed in Chapter 3).

(i) Change source

As explained above, hydrogen (H_2) is made from methane gas, which creates a large part of material maintenance and usage-related emissions.

If hydrogen can be produced by nuclear power or solar power, the emissions caused by methane will be reduced. Nuclear power also produces toxic waste among other challenges. Further, large-scale hydrogen clean production such as electrolytic water is not mature enough to support the widely used industrial output at the current scientific and technological level (Mazloomi and Gomes, 2012).

Thus, although this method has the potential to change the whole table of nitrogen fertiliser production from emission source to carbon taker if a clean hydrogen supply can be achieved, achieving sustainability through this strategy needs heavy investments and a technical breakthrough, to reduce the cost to a reasonable level.

(ii) Change practices

Fertiliser industries are trying to design more advanced nitrogen fertilisers, such as slow-release fertilisers, to solve on-farm fertiliser loss (Christian et al., 2020).

Chemical fertiliser enterprises have made progress in designing and optimising slow-release fertilisers to improve the loss of chemical fertiliser caused by the environment (IFA, 2010). Compared with traditional fertiliser, slow-release (or controlled-release) fertiliser can effectively alleviate nutrition loss and crop dry matter accumulation (Wang et al., 2021). This could potentially serve as one approach to mitigate nitrogen emissions arising from the cropping industry, albeit without fundamentally resolving the issue. The emissions resulting from the production and utilisation of chemical fertilisers remain present. Nevertheless, when compared to the other two methods, the slow-release fertiliser method demonstrates clear advantages, due to its lower investment requirements. Additionally, since there are currently no established regulations associated with the set-up of dripping systems, the slow-release fertiliser method may be considered a gentler solution for growers.

Another main focus for practice change is the on-farm fertiliser usage, especially the irrigation method. Widely used flooding irrigation and overapplication may cause up to 71% nutrition loss (Wang et al., 2010).

According to experimentation for furrow irrigation, up to 45% of nitrogen fertiliser might be lost due to irrigation and heavy rain (Terman, 1980). Two practices can improve this situation:

1) Change irrigation to a more effective water-saving method, such as dripping irrigation or drip fertigation. For example, drip fertigation can reduce the fertilisation rate by 1/3 without yield decline (Yao et al., 2019).

2) More precise fertiliser type choice, application timing and depth of fertilisation (Smith, McTaggart and Tsuruta, 1997). Applying fertiliser using deep placement can efficiently reduce fertiliser loss by around 44% to 48%; and precise timing to avoid days of heavy rain would effectively minimise nitrogen loss (White et al., 2002).

An important challenge associated with implementing this method pertains to the constraint of DSSAT, as highlighted in Pham (2016). The irrigation module within DSSAT considers the entire irrigated area as a single entity, lacking the ability to capture specific field characteristics such as field length and failing to adequately reflect irrigation performance indicators. Consequently, the implementation of different irrigation methods yields insignificant differences. Furthermore, the installation of a dripping system requires a significant investment, estimated at approximately $7000–9000 per hectare on an annual basis (CRDC, 2023).

To synthesise the findings garnered from the abstracts presented by the conference stakeholders, through conversations, presentations and written materials, two primary directions have been discerned that hold practical implications for sustainable production and nitrogen management. Due to the high technological requirements for changing sources (to green hydrogen sources ammonia), they are assessed as economically unfeasible for large-scale production in the current context. Additionally, these changes have an indirect influence on on-farm nitrogen management decision options. Consequently, the policy options related to this direction lean more towards long-term development and government project support.

Conversely, changing practices has a direct and closely related impact on on-farm nitrogen management decision options. Thus, scenario designs are inclined to select projects within the broad category of changing practices. Notably, the available options for selection include changing irrigation practices and altering the nitrogen type and quantity. Due to previously mentioned constraints (limitations in the DSSAT irrigation model) and stakeholder opinions, experiments designed around altering nitrogen type and quantity can yield more accurate and rational DSSAT outcomes to underpin subsequent computations and evaluative analyses.

4.3.3 DSSAT set-up

This section offers a concise overview of the DSSAT setup pertaining to the selection of weather station location, soil type, cultivar, crop management practices and others.

Table 4.2: DSSAT experiment set-up

Category	**Set-up**
Cultivar	
Cotton	Deltapine 555 B[45]

[45] It is important to note that while the selection may not align with conventional choices, it is the closest approximation to Bollgard II cultivars available within the DSSAT framework. Bollgard II cultivars are prevalently used in the Australian cotton industry, as evidenced by studies such as Braunack, Bange and Johnston (2012) and Pham (2016).

Category	Set-up
Weather station	
Station name	CSIRO[46]
Location	Latitude 30.2 South; Longitude 149.6 East
Elevation zone	262 (cm)
Time length[47]	2010 to 2098
Primary soil conditions[48]	
Soil name	Clay Vertosol
Soil type	Vertosol
Surface texture[49]	Clay[50]
Slope %	12
Drainage	Poorly (0.05)
Run-off potential	Moderately high
PH in water	8~8.9
Total Nitrogen %[51]	0.98
Cation exchange capacity	38.4
Soil adsorption coefficient (0 to 1 scale)	1
Depth, cm	200

[46] Myall Vale Weather Station, New South Wales, Australia (300.200'S, 149.600'E)

[47] The future weather data is generated using DSSAT Weatherman, based on past weather data from 1924–2014. This function is explained in Pickering et al. (1994) and has been used in research papers such as Sarkar & Kar (2006), Sarkar & Kar (2008), Nicoloso, Amado & Rice (2020).

[48] The soil data in this segment has been optimised in accordance with the literature, drawing specifically from the soil data presented in Pham (2016).

[49] Calculation reference: Saxton et al. (1986), Baumer and Rice (1988)

[50] Source: Vervoort (n.d.)

[51] Calculation method in DSSAT code: NH_4=Total nitrogen * Bulk density * Soil layer base depth * 1000 * 0.004. (Ritchie & Godwin, n.d.)

Note: this BD * DEPTH is aimed to transfer % to reserves, and the output should be in (kg/ha).

Category	Set-up
Initial conditions	
Initial soil water (0-80 cm)	0.43
Initial soil NH_4 (0-80 cm)	1.1
Initial soil NO_3 (0-80 cm)	7
Planting Details[52]	
Planting date	October 15th
Planting method	Dry seed
Planting distribution	Rows
Planting population at seeding	12 (plants/m^2)
Row spacing (cm)	100
Plant depth (cm)	5
Fertiliser management	
Fertiliser material[53]	S1: Sulphur-coated urea-thick S2, 3: Urea
Fertiliser application method[54]	Bend beneath surface

[52] The Planting data, Irrigation management and Harvest in this segment have been compared with information gathered from the Australian Cotton Conference 2022 and examined by cropping sciences; these data are originally drawn from Pham (2016).

[53] The use of more sustainable nitrogen fertiliser is one of the options to improve sustainable cotton production proposed at the Australian Cotton Conference 2022. Details of the selection of this option and comparisons with other options have been explained in Section 4.3.2.2.

[54] As explained in Section 4.3.2.1, for the fertiliser application method detailed, the total application amount of S1 and S2 is 234 kgN/ha; and S3 is 275 kgN/ha. The application amount of S1 and S2 is chosen because it is the most economically efficient amount proposed by Antille & Moody (2021); and the application amount of S3 is the average amount of Australian cotton farm nitrogen application, referenced from Scheer et al. (2023). The date for fertiliser application is determined based on an integrated consideration of information gathered from dialogues with growers, expert opinions, and a review of existing literature, all of which are generally aligned.

Category	Set-up
First fertiliser application Amount (kg N/ha)	S1,2: 180 S3: 200
First application date	On planting date
Second fertiliser application Amount (kg N/ha)	S1,2: 54 S3: 75
Second application date	2 months after the planting date
Irrigation management	
Irrigation option	Furrow
Application	Automatic when needed
Management depth (cm)	40
Threshold, %	60
Harvest	
Harvest option	Harvest at maturity

Note: The suboptimal application of nitrogen fertiliser in terms of its amount and distribution is acknowledged. As the purpose of this chapter is to demonstrate the measurement of large-scale simulation outcomes, the exploration of optimal nitrogen fertiliser plans can be conducted by further research.

Table 4.3: Treatment details for nitrogen fertilisers S1, S2, S3

Scenario	Time length	Fertiliser type	Fertiliser application[55]	Fertiliser (total)
S1	2010 to 2098	Sulphur-coated urea-thick (41% N)	1) 180 kgN/ha on planting date 2) 54 kgN/ha 60 days after	234 kgN/ha

[55] The choice of application amount for S1 and S2 (234 kgN/ha) is the most economically application amount from Antille and Moody (2021). The choice of application amount of S3 (275 kgN/ha) is the average application amount from Scheer et.al. 2023. Application amount for each application is decided through interviews with growers during Australian Cotton Conference 2022, Pham (2016) and distribution in *CottonInfo on-farm Nitrogen trials and N use practices 2017*.

Scenario	Time length	Fertiliser type	Fertiliser application[55]	Fertiliser (total)
S2	2010 to 2098	Urea (46% N)	1) 180 kgN/ha on planting date 2) 54 kgN/ha 60 days after	234 kgN/ha
S3	2010 to 2098	Urea (46% N)	1) 200 kgN/ha on planting date 2) 75 kgN/ha 60 days after	275 kgN/ha

4.3.4 Data source of indicators for ESRIDM calculation

Table 4.3 above shows more details of treatments for each scenario, and Table 4.4 lists critical indicators for ESRIDM calculation.

By design, the LCA assessment for fertiliser should be done by LCA calculation, to reflect how technology involvement might influence emission reduction options. However, (i) the scope of this determination is focusing on demonstrating how decision-making can assist assets through ESRIDM, which using LCA calculation is not part of on-farm decision-making; and (ii) for the scenarios selected, the production techniques and industry chain for urea and sulphur-coated urea are relatively mature, leading to a corresponding level of maturity in LCA studies based on them. Therefore, unlike hydrogen-produced urea which may be highly dependent on the technology the production method is based on (e.g., Xie et al., 2022; Lubitz & Tumas, 2007), the utilisation of natural gas-produced nitrogen fertiliser LCA results from the literature is unlikely to deviate significantly from the industry's current state.

In addition, the water remediation cost (marine eutrophication - Great Barrier Reef) is assumed to only count the remediation cost on the water flow end (Great Barrier Reef) for marine eutrophication effect. Other effects such as human health and ecological hazards in freshwater systems are not considered because the damage is highly dependent on diversity settings, such as farm location.

Table 4.4: Additional data source for ESRIDM calculations

Name	Amount	Unit	Reference
Emission cost	28.5	AUD	Market price from Carbon Credits (2023)
Lint rate (by weight)	42	%	Cotton Australia (2023)
Lint price	2.98	$/kg	Department of Primary Industries (2023)
Seed price (sell)	0.28	$/kg	AgEcon (2022)
Seed price (buy)	130	$/ha	Australian Cotton Comparative Analysis (2021)
Diesel price	1.73	$/L	GlobalPetrolPrices (2023)
Fertiliser price (Urea)	0.88	$/kg	Market price from Index Mundi (2023)
Fertiliser price (Sulphur-coated Urea)	0.88	$/kg	Market price from Alibaba (2023)
Licensed water price	99	$/ML	Australian Government Department of Agriculture, Fisheries and Forestry (2022)
Lime price	9.9	$/kg	Market price from Alibaba (2022)
Lime application cost	300 to 400	$/ha	Historical price from Groundcover (2023)
Diesel use emission	2.7	CO_2eqkg/L	Chen & Baillie (2009)
Diesel production emission	42.1	CO_2eqkg/L	Brock et.al. (2012)
Urea production emission	2.18	CO_2eqkg/kg	Shi et al. (2020)
Sulphur-coated urea production emission	1.03	CO_2eqkg/kg	Calculated by Da Costa et al. 2019 and Shi et al. (2020)
Water remediation cost (marine eutrophication - Great Barrier Reef)	150	$/kgDIN	Kavehei et.al. (2021)

Name	Amount	Unit	Reference
Tax rate	25	%	Australian Taxation Office (2023)
Rwacc	6	%	ABARES (2023)

Note: I) Dissolved Inorganic Nitrogen (DIN)

II) Initially, I assumed a significant price disparity between the two fertilisers; however, upon reviewing the trading market presented in Table 4.4 on the source website, it became apparent that the market prices for both fertilisers are remarkably similar. It is a plausible assumption that in markets such as those approximating perfect competition, customers are likely to welcome lower prices.

4.4 Results and findings

4.4.1 DSSAT

4.4.1.1 DSSAT S1, S2, S3 results

Table 4.5 presents a summary of key indicators generated by DSSAT. Each scenario includes 88 years (2010–2098) of observations derived from DSSAT-generated output.

Three scenarios have been entered into DSSAT, with differences in nitrogen fertiliser choice listed in Table 4.3. In this study, assessment and comparison focus on the most sustainable choice between the three scenarios: (S1) to sub-sustainable choice (S2), and the most sustainable choice within three scenarios (S1) to general practices (S3).

(I) Output overview

As presented in Table 4.5, under varying nitrogen fertiliser application amounts (S1, S2 and S3), the average lint yield exhibits little discernible difference. Conversely, substantial distinctions are evident in N_2, N_2O and NO losses. Specifically, in terms of nitrogen emissions, there is a significant difference in nitrogen emission between S1 vs. S2 and S1 vs. S3, respectively. This shows that the change of nitrogen fertiliser application choice to a more sustainable option may have more emission reduction effects in the long term (88 years), which may not have a significantly negative impact on farm productivity, while significantly reducing nitrogen emissions.

Table 4.5 also shows nitrogen emissions reduction can be observed through both reduction in application amount, which is an 18% reduction from 275 (S3) to 243 kgN/ha (S1 and S2) and to a more sustainable fertiliser choice, which is urea (S2 and S3) to sulphur coated urea (S1).

(II) Note on output (carbon) and nitrogen leakage

In the following table, "output (carbon)" is not compared, due to two reasons: (i) the main focus of this research is nitrogen fertiliser-related emission, thus the simulation output of soil carbon and carbon dioxide emission is aimed to serve the ESRIDM calculation; the differences and possible reason are not in the scope of this chapter; and (ii) in the default settings of DSSAT, the initial simulation values for soil carbon do not inherit from the previous year's consumption; instead, they are based on fixed values derived from the soil-related data entered before the experiment begins. Additionally, all scenarios have the same default treatment for crop residue.

Furthermore, it is acknowledged that there may be discrepancies between the nitrogen-leached data and the simulated leached data, due to limitations within the DSSAT model (Ritchie and Godwin, n.d.). As a result, these conditions render them devoid of comparative value.

4.4.1.2 Statistical test on S1 vs S2 and S1 vs S3

Table 4.5 presents the p-value of the t-test for DSSAT results.

The data reveals negligible disparities in yield across scenarios S1, S2 and S3. However, notable differences are evident in nitrogen loss in all three gaseous forms and nitrogen emissions. As previously explained in Section 4.4.1.1 (II), carbon emissions remain relatively consistent across the scenarios. Therefore, the pronounced variations in total emissions can be attributed primarily to nitrogen emissions.

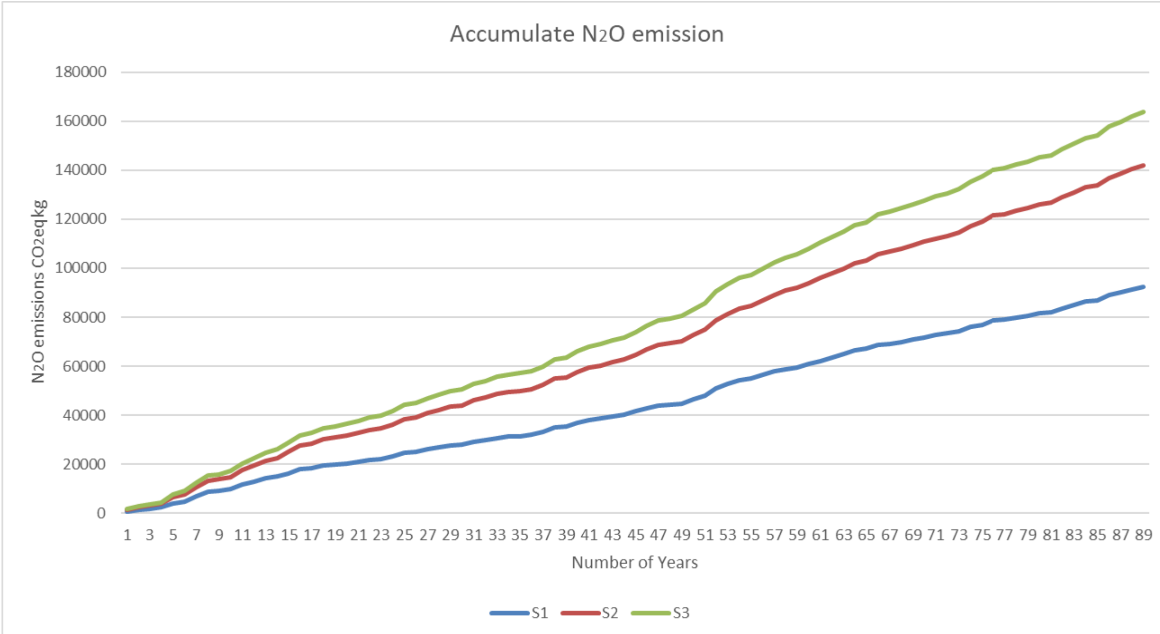
Table 4.5: 2010-2098 output from DSSAT and statistical test (S1, S2, S3)

	S1 Average	S2 Average	S3 Average	Units	P value S1 & S2	P value S1 & S3	
Setting							
Irrigation during growing season	565.84	565.84	567.19	mm			
Rain	310.67	310.67	310.67	mm			
Growing season length	161.96	161.96	161.96	days			
Irrigation method	Furrow	Furrow	Furrow	-	-	-	-
Fertilizer type	Coated urea	Urea	Urea	-	-	-	-
Nitrogen fertilizer	180+54	180+54	175+100	kgN/ha	-	-	-
Total N fertilizer	234	234	275	kgN/ha	-	-	-
Output (desired)							
Yield	5,118.37	5,133.43	5,143.99	kg/ha	0.829	0.738	-
Lint %	0.42	0.42	0.42	kg/kg	-	-	-
Lint yield	2,149.72	2,156.04	2,160.48	kg/ha	0.829	0.738	-
Biomass yield	15,287.29	15,302.34	15,628.24	kg/ha	0.931	0.074	-
Output (carbon)							
Change in soil C	207.77	207.83	207.57	kgC/ha	-	-	-
CO_2 CO_2eq	8426.31	8,425.90	8,406.52	CO_2eqkg/ha	0.987	0.422	-
N loss							
N leached	0.06	0.06	0.06	kgN/ha	-	-	-
N_2 loss	2.22	3.41	3.94	kgN/ha	0.008	0.001	*
N_2O loss	5.41	7.00	7.63	kgN/ha	0.000	0.000	*
NO loss	0.88	1.25	1.45	kgN/ha	0.000	0.000	*
GHG							
N_2O emission CO_2eq	1,038.72	1,596.76	1,840.44	CO_2eqkg/ha	0.000	0.000	*
Total GHG	9,465.03	10,022.66	10,246.96	CO_2eqkg/ha	0.000	0.000	*

4.4.1.3 Figure demonstration

Figure 4.4 depicts the cumulative nitrogen emissions for scenarios S1, S2 and S3, spanning the period from 2010 to 2098. Evidently, S3 consistently exhibits the highest nitrogen emissions throughout, while S2's emissions surpass those of S1. As illustrated in the figure, the differential between S1 and both S3 and S2 intensifies with the accumulation of years. By 2098, this divergence is notably significant.

Figure 4.4: Accumulation of N_2O emissions (accumulated), S2-S1 and S3-S1, year 2010 to 2098



Note: The future weather data was generated employing the DSSAT Weatherman, predicated on historical weather data spanning from 1924 to 2014. For further citation, refer to Table 4.2.

Figure 4.5 illustrates annual differences between S1 vs. S2 and between S1 vs. S3. On a broad scale, the difference from S3 to S1 is greater than that from S2 to S1. Yet, when considering the overall trend, the annual disparities between S1 and S2, as well as between S1 and S3, remain relatively stable.

Figure 4.5: Difference in N_2O emissions (annual change), S2-S1 and S3-S1, year 2010 to 2098

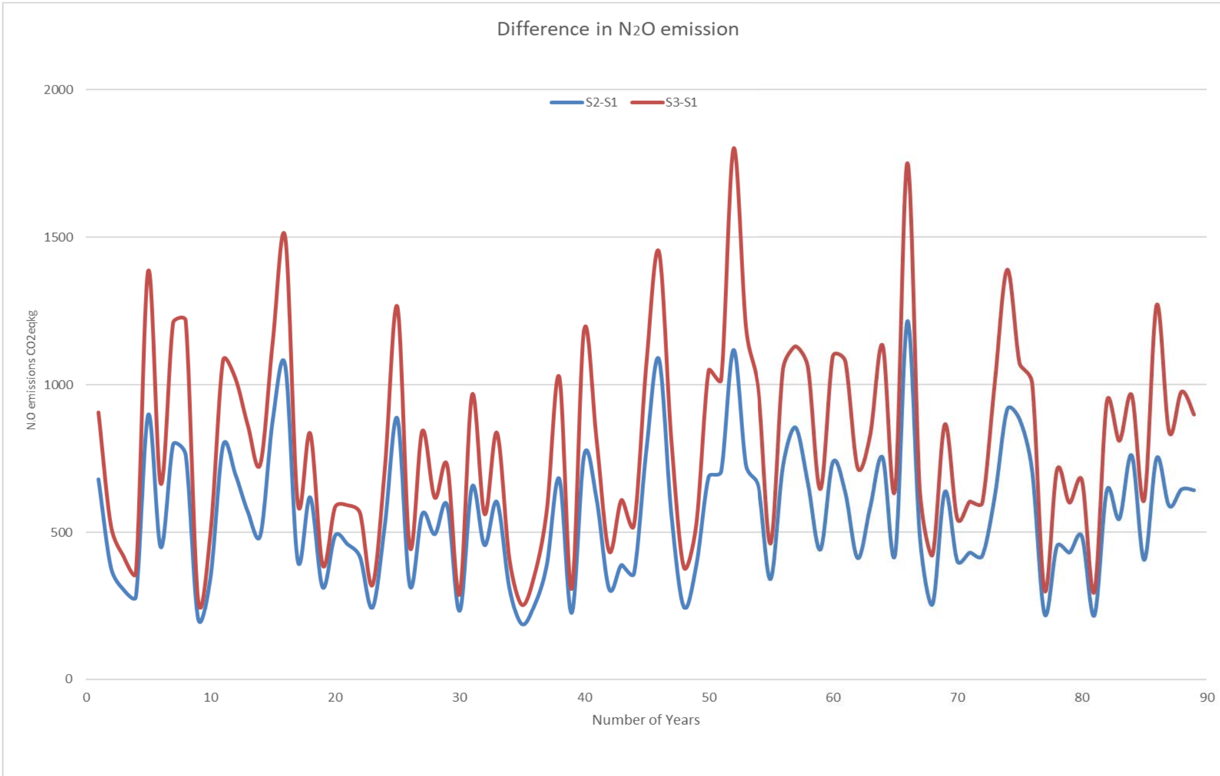
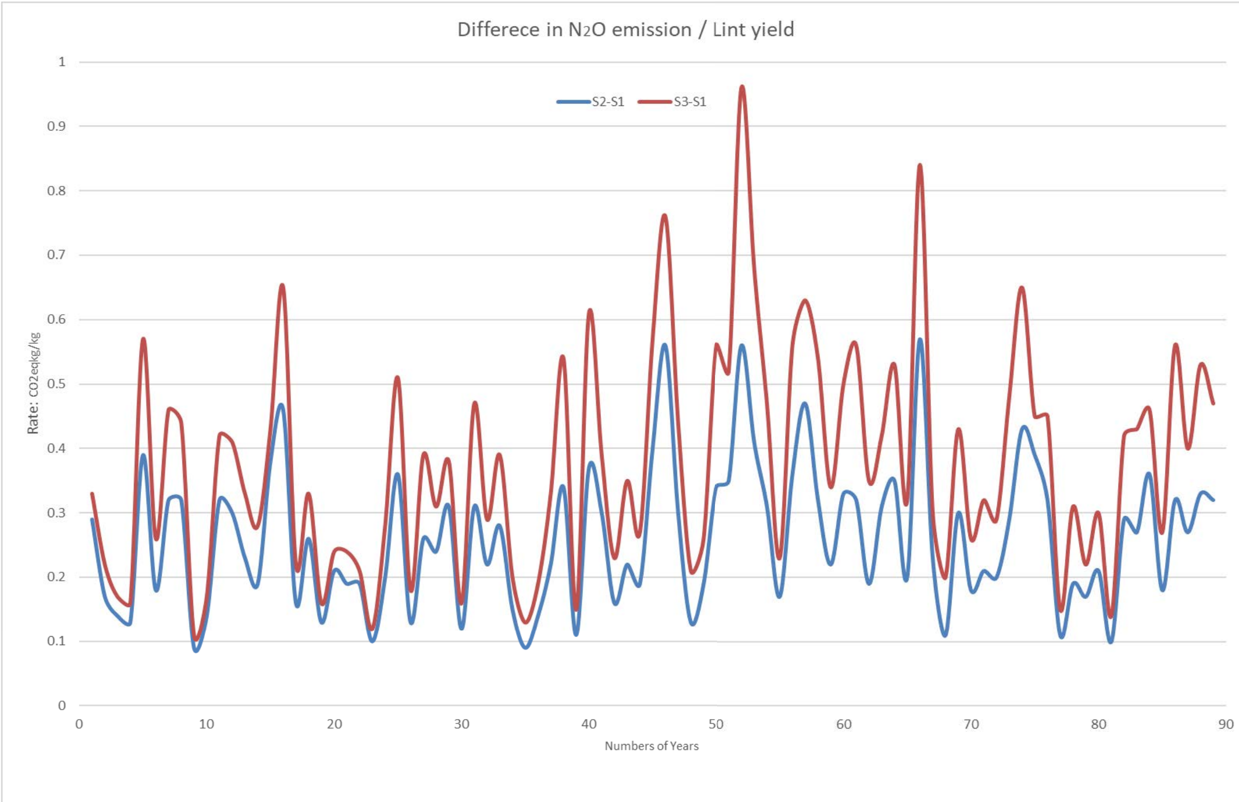


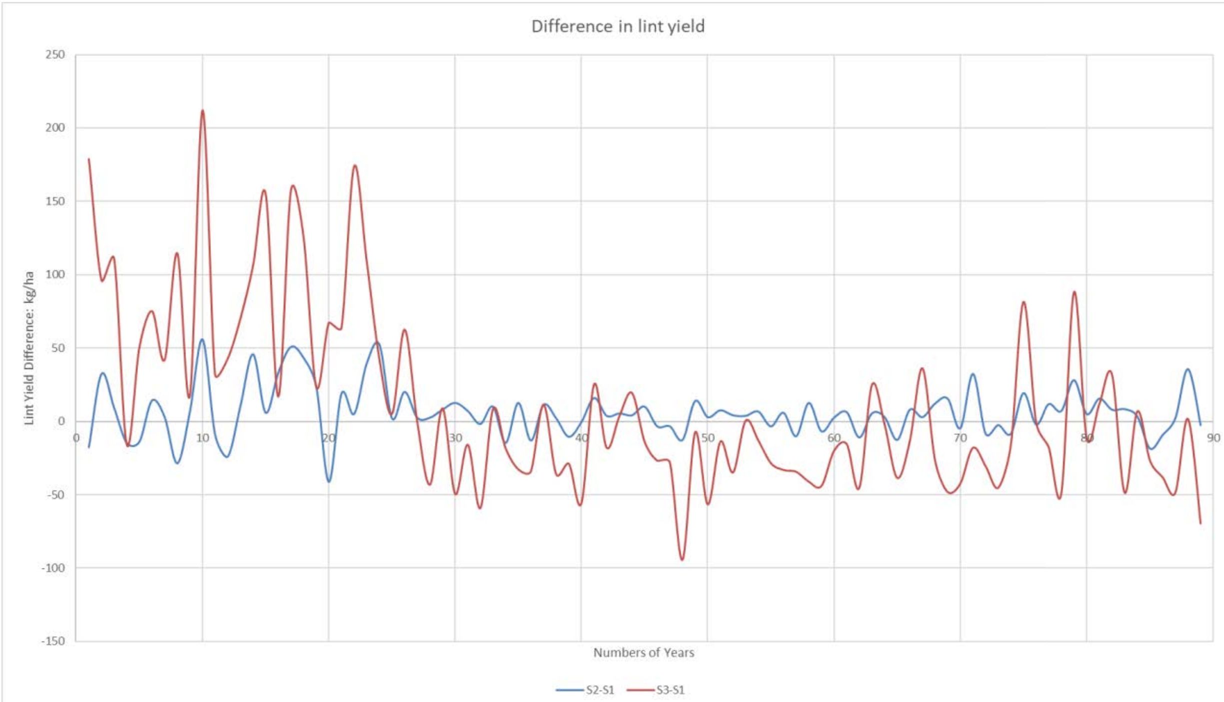
Figure 4.6 presents the annual difference between the N_2O emission/lint yield ratios of S1 to S3 and S1 to S2. The general trajectories of the red and blue lines closely mirror those observed in Figure 4.5. In conjunction with the DSSAT results from Table 4.5, which indicate no significant yield variation between S1, S2 and S3, one can infer that the patterns in Figure 4.6 are predominantly influenced by nitrogen emissions.

Figure 4.6: Difference in N_2O emission/lint yield (annual change), S2-S1 and S3-S1, year 2010 to 2098



In Figure 4.7, the annual variation in yield difference between S1 and S3, as well as between S1 and S2, is depicted. The rationale for presenting yield as a difference is based on Table 4.5, which indicates no significant yield variations across all three scenarios. Representing it in this manner facilitates a clearer discernment of the yield disparities among scenarios S1, S2 and S3, as illustrated in Figure 4.7. Notably, the yield for S3 surpasses that of S1 and S2 in the initial years but subsequently declines, aligning with or even falling below the levels of S2. Drawing on the weather data provided in Appendix C, one could hypothesise that increased precipitation (rainfall) might have influenced the plants' nitrogen nutrition. This might be attributed to factors such as nitrogen leaching, an improved soil nitrogen balance, or a shift in the equilibrium between nitrate and ammonium forms of nitrogen.

Figure 4.7: Difference in yield (annual change), S2-S1 and S3-S1, year 2010 to 2098



This change in yield difference is assumed to be caused by the increase in rainfall. For more information, see Appendix C.

In the next section, simulated results generated by DSSAT for S1, S2 and S3 are used as part of input data for ESRIDM analysis and comparison between S1, S2 and S1, S3 respectively.

4.4.2 ESRIDM

4.4.2.1 High-level summary indicators

This section contains the high-level summary indicators of environmental and economic implications of moving between S1, S2 and S3.

The key findings are based on 88 years for the period from 2010–2098. They include:

Environmentally Sustainable Residual Income (level 1), Net Operating Profit After Tax, Environmental Sustaining Cost, Net Operating Assets, Required rate capital return (level 2) and useful ratios (independent measurements), in which:

$$Harvest\ index\ =\ Yield\ at\ harvest\ maturity\ /\ Tops\ weight\ at\ maturity$$

$$Irrigation\ water\ use\ efficiency = \ Yield/Irrigation\ during\ growing\ season$$

$$Nitrogen\ fertiliser\ use\ efficiency\ =\ Yield\ /\ N\ fertiliser\ applied$$

$$Emission\ index = Total\ emission\ /Economic\ yield$$

$$Emissions\ intensity\ =\ Yield\ /\ Total\ emission$$

Please note: The harvest index can be seen in the DSSAT documents named OVERVIEW, provided by DSSAT software.

Table 4.6 shows that there is a statistically significant difference in ESC, nitrogen input efficiency measurements (nitrogen fertiliser use efficiency) and emission index, and no significant difference in NOPAT and ESRI. Other indicators, which are NOA and irrigation water use efficiency, were not compared. NOA is an estimated figure derived from the difference in total operating revenue, rendering statistical testing of limited value. Regarding irrigation water, the DSSAT set-up employs automatic irrigation, which means it does not incorporate water efficiency controls. Consequently, conducting statistical tests on this indicator would not gain significant insights.

Table 4.6: High-level summary indicators

		S1	S2	S3	S2-S1	S3-S1	Unit	S1 & S2		S1 & S3	
		Average	Average	Average	Difference	Difference		P value		P value	
Nitrogen Fertilizer application plan		234.00	234.00	275.00	-	41.00	kgN/ha				
Level 1											
ESRI											
	Net Operating Profit After Tax (NOPAT)	1,741.02	1,799.82	1,751.51	58.80	10.49	$/ha	0.455	-	0.861	-
	Environment Sustaining Cost (ESC)	597.09	627.90	639.83	30.81	42.74	$/ha	0.000	*	0.000	*
	Net Operating Assets (NOA)	7,087.00	7,107.85	7,122.47	20.85	35.47	$/ha	-	-	-	-
	RWACC	6%	6%	6%	-	-	Rate	-	-	-	-
		718.71	745.44	684.33	26.74	-34.38	$/ha	0.739	-	0.725	-

While these indicators in Table 4.6 and 4.7 are helpful in identifying which scenario is most likely to lead to sustainability, they are incomplete in that the causal drivers are not explained.

Table 4.7: Useful indicators

		S1	S2	S3	S2-S1	S3-S1	Unit	S1 & S2		S1 & S3	
		Average	Average	Average	Difference	Difference		P value		P value	
Nitrogen Fertilizer application plan		234.00	234.00	275.00	-	41.00	kgN/ha				
Ratio											
Useful ratios											
	Harvest index	33%	33%	31%	0%	-2%	kg/kg	-	-	-	-
	Irrigation water use efficiency	9.05	9.07	9.07	3%	2%	kg/ML	-	-	-	-
	Nitrogen fertilizer use efficiency	21.87	21.94	18.71	0.06	-3.17	kg/kgN	0.00	*	0.00	*
	Emission index	2.85	2.99	3.04	0.14	0.19	CO2eqkg/$	0.00	*	0.00	*
	Emissions intensity	25%	24%	23%	-0.01	-0.02	kg/CO2eqkg	-	-	-	-

4.4.2.2 Level 1 ESRI

Table 4.8 shows the ESRI results for scenarios S1, S2 and S3, using the 2010–2098 average output from DSSAT and data from Table 4.5.

This calculation result shows that S2 has the highest ESRI output. Under the current regulation option shown in Table 4.4, this suggest that S2 generates the highest sustainable value, S3 has the lowest sustainable value, and S1's ESRI is allocated in between. In addition, all three scenarios get a positive ESRI value, and the p-value suggests insignificant differences between S1 vs. S2 and S1 vs. S3. This might suggest that when the regulation tool (carbon pricing mechanism, $28.5) for emissions is sitting at a relatively low pricing level, all scenarios remain sustainable under the condition that the ESC can be adequately covered by a supporting entity, such as the government or growers. In light of the negative ESRI observed across all scenarios in the tree offsets projects, there is merit in conducting a more extensive investigation into diverse emission remediation costs. Further analysis is presented in the discussions section.

As demonstrated in Chapter 3, the ESRI value has four drivers: Net operating profit after tax (NOPAT), ESC and financial measurement indicators Net operating assets (NOA) and the capital return rate (Rwacc). The Rwacc is a fixed rate retrieved from ABARES (2023). The difference in result should be caused by NOPAT, ESC and NOA.

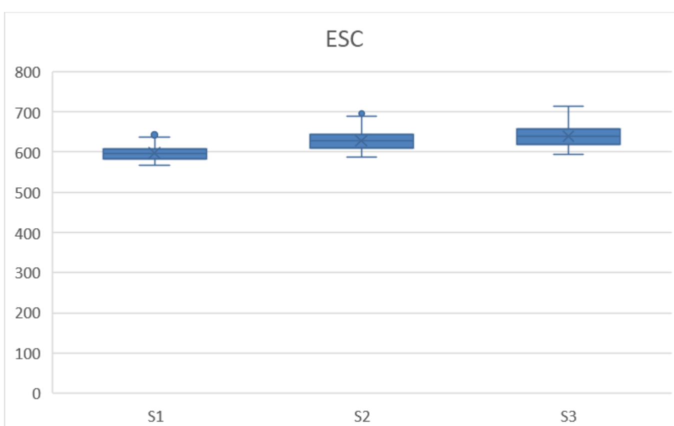
From Table 4.8, it seems the reason for S2 to be outstanding is that it has the highest NOPAT when compared to the other two scenarios. While there is a significant difference in ESC, ESRI value has no significant difference due to the difference in ESC (as reflected in Figure 4.8) being smaller than NOPAT when S1 is compared to S2 and S3. To understand more details about the specific drivers of change and their underlying causes, it is necessary to look at lower levels of ESRIDM modelling.

Table 4.8: Level 1 ESRI S1, S2, S3 calculated by 2010–2098 average

		S1 Average	S2 Average	S3 Average	S2-S1 Difference	S3-S1 Difference	Unit	S1 & S2 P value		S1 & S3 P value	
Nitrogen Fertilizer application plan		234.00	234.00	275.00	-	41.00	kgN/ha				
Level 1											
ESRI											
	Net Operating Profit After Tax (NOPAT)	1,741.02	1,799.82	1,751.51	58.80	10.49	$/ha	0.455	-	0.861	-
	Environment Sustaining Cost (ESC)	597.09	627.90	639.83	30.81	42.74	$/ha	0.000	*	0.000	*
	Net Operating Assets (NOA)	7,087.00	7,107.85	7,122.47	20.85	35.47	$/ha	-	-	-	-
	RWACC	6%	6%	6%	-	-	Rate	-	-	-	-
		718.71	745.44	684.33	26.74	-34.38	$/ha	0.739	-	0.725	-

Figure 4.8 and 4.9 shows the boxplot of ESC and NOPAT for S1, S2 and S3[56].

Figure 4.8: Boxplot of ESC for S1, S2 and S3



[56] I would like to express my gratitude to the anonymous examiner for the recommendation to employ boxplots for certain variables.

4.4.2.3 Level 2 NOPAT

Table 4.9 shows that S2 has the highest NOPAT and S1 is the lowest, while the p-value shows no significant difference in S1 vs S2 and S1 vs S3. NOPAT is driven by net operating profit (NOP) and tax. As the tax rate is a fixed ratio for small businesses (see Table 4.4), the main driver for NOPAT is NOP. Thus, the main driver for the difference in NOPAT is NOP.

Table 4.9: Level 2 NOPAT S1, S2, S3 calculated by 2010–2098 average

		S1 Average	S2 Average	S3 Average	S2-S1 Difference	S3-S1 Difference	Unit	S1 & S2 P value	S1 & S3 P value
Nitrogen Fertilizer application plan		234.00	234.00	275.00	-	41.00	kgN/ha		
Level 2									
NOPAT									
	Tax	580.34	599.94	583.84	19.60	3.50	$/ha	-	-
	Net operating profit	2,321.36	2,399.76	2,335.35	78.40	13.99	$/ha	0.455	0.861
		1,741.02	1,799.82	1,751.51	58.80	10.49	$/ha	0.455	0.861

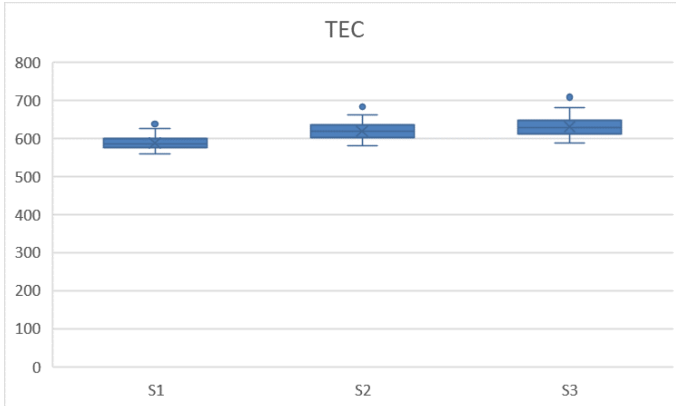
4.4.2.4 Level 2 ESC

As demonstrated in Table 4.10, S1 has the lowest ESC, and S3 has the highest. ESC shows a significant difference between S1 vs. S2 and S1 vs. S3 by p-value. ESC is driven by total emission cost (TEC) (as reflected in Figure 4.9) and other environmental remediation costs. As the other environmental remediation cost is much lower than TEC and assumed to have the same amount, the main driver for ESC in this calculation should be TEC.

Table 4.10: Level 2 ESC S1, S2, S3 calculated by 2010–2098 average

		S1 Average	S2 Average	S3 Average	S2-S1 Difference	S3-S1 Difference	Unit	S1 & S2 P value		S1 & S3 P value	
Nitrogen Fertilizer application plan		234.00	234.00	275.00	-	41.00	kgN/ha				
Level 2											
Environment Sustaining Cost (ESC)											
Total emission cost											
	Total emissions	20,610.54	21,691.64	22,110.23	1,081.09	1,499.69	CO2eqkg/ha	0.000	*	0.000	*
		587.40	618.21	630.14	30.81	42.74	$/ha	0.000	*	0.000	*
Other environmental remediation cost											
	Soil remediation cost	-	-	-	-	-	$/ha	-	-	-	-
	Water remediation cost (marine eutrophication)	9.69	9.69	9.69	-	-	$/ha	-	-	-	-
		9.69	9.69	9.69	-	-	$/ha	-	-	-	-
		597.09	627.90	639.83	30.81	42.74	$/ha	0.000	*	0.000	*

Figure 4.9: Boxplot of TEC for S1, S2 and S3



4.4.2.5 Level 2 NOA and Rwacc

NOA is driven by total operating assets (TOA) and total operating liabilities (TOL). In this chapter, the value of financial indicators is based on assumption. Where TOA is assumed to be market value, add the mark-up (expected growth rate in operating assets) based on the changing rate brought by the difference in Operating Revenue, TOL is assumed to be 0, Rwacc is assumed to be 6% (ABARES, 2023).

Table 4.11: Level 2 NOA and Rwacc S1, S2, S3 calculated by 2010–2098 average

		S1	S2	S3	S2-S1	S3-S1	Unit	S1 & S2	S1 & S3
		Average	Average	Average	Difference	Difference		P value	P value
Nitrogen Fertilizer application plan		234.00	234.00	275.00	-	41.00	kgN/ha		
Level 2									
Net Operating Assets (NOA)									
	Total Operating Assets	7,087.00	7,107.85	7,122.47	20.85	35.47	$/ha	-	-
	Total Operating Liabilities	-	-	-	-	-	$/ha	-	-
	Expect growth rate in Operating Assets		0.29%	0.21%	-	-		-	-
		7,087.00	7,107.85	7,122.47	20.85	35.47	$/ha	-	-
Level 2									
Rwacc									
		6%	6%	6%	-	-		-	-

To this level, the main driver for differences in S1 vs. S2, and S1 vs. S3 ESRI output is TEC and NOP.

4.4.2.6 Level 3 Net operating profit (NOP)

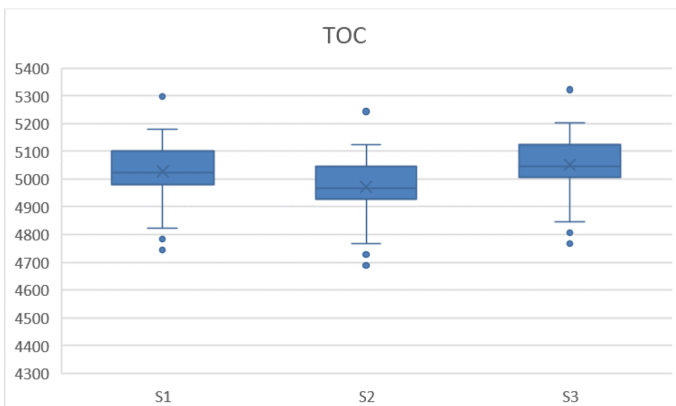
NOP is driven by total operating revenue (TOR) and total operating cost (TOC) ((as reflected in Figure 4.10). S2 has the highest and S1 has the lowest NOP. P-value suggests a significant difference in TOC and an insignificant difference in both TOR and NOP.

Table 4.12 shows the difference in TOR covers the difference brought by TOC. However, as the differences are relatively close to each other, and the p-value suggests significant differences in TOC, both TOR and TOC need further analysis.

Table 4.12: Level 3 Net operating profit S1, S2, S3 calculated by 2010–2098 average

		S1	S2	S3	S2-S1	S3-S1	Unit	S1 & S2		S1 & S3	
		Average	Average	Average	Difference	Difference		P value		P value	
Nitrogen Fertilizer application plan		234.00	234.00	275.00	-	41.00	kgN/ha				
Level 3											
Net operating profit											
	Total operating revenue	7,228.28	7,249.55	7,264.46	21.26	36.18	$/ha	0.83	-	0.69	-
	Total operating cost	4,906.93	4,849.79	4,929.12	-57.14	22.19	$/ha	0.00	*	0.11	-
		2,321.36	2,399.76	2,335.35	78.40	13.99	$/ha	0.455	-	0.861	-

Figure 4.10: Boxplot of TOC for S1, S2 and S3



4.4.2.7 Level 3 Total emission cost (TEC)

TEC ((as reflected in Figure 4.11) is driven by carbon emissions price and total emissions amount. As the carbon price is a fixed amount for all three scenarios (see Table 4.4), the main driver for TEC should be total emissions. Thus, the main driver for the difference in ESC is the total emissions amount.

If the emissions charge is different based on its sources, TEC decomposition can also be based on direct emissions (from cotton growth) ((as reflected in Figure 4.12), emission cost from farm energy use, emission cost from other operating activities, and LCA emission cost from producing direct materials (fertiliser and lime).

Table 4.13: Level 3 Total emission cost S1, S2, S3 calculated by 2010–2098 average

		S1	S2	S3	S2-S1	S3-S1	Unit	S1 & S2		S1 & S3	
		Average	Average	Average	Difference	Difference		P value		P value	
Nitrogen Fertilizer application plan		234.00	234.00	275.00	0	41	kgN/ha				
Level 3											
Total emission cost											
	Direct emissions cost	269.89	285.78	292.18	15.89	22.28	$/ha	0.000	*	0.000	*
	Emission cost from farm energy use	300.01	300.01	300.01	-	-	$/ha	-	-	-	-
	Emission cost from other operating activities	-	-	-	-	-	$/ha	-	-	-	-
	LCA emission cost from producing direct materials	17.50	32.42	37.96	14.92	20.46	$/ha	0.000	*	0.000	*
OR											
	Total emissions	20,610.54	21,691.64	22,110.23	1081.09	1499.69	CO2eqkg/ha	0.000	*	0.000	*
	Carbon price (AU)	28.50	28.50	28.50				-	-	-	-
		587.40	618.21	630.14	30.81	42.74	$/ha	0.000	*	0.000	*

To this level, the main driver for differences in S1, S2 and S3 ESRI output is total emissions, TOR and TOC.

Figure 4.11: Boxplot of Total emissions for S1, S2 and S3

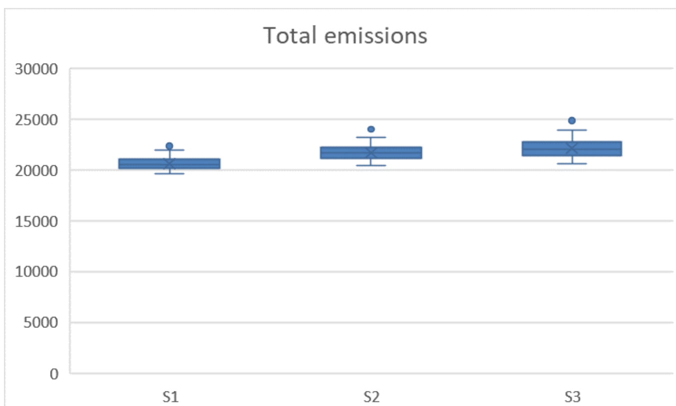
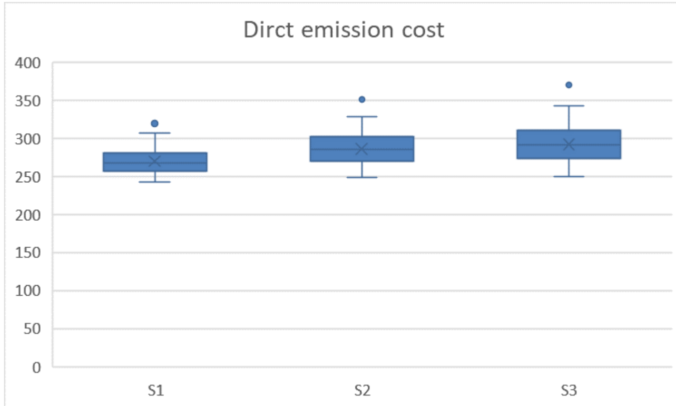


Figure 4.12: Boxplot of Direct emission cost for S1, S2 and S3



4.4.2.8 Level 4 Total operating revenue and total operating cost

In Table 4.14, TOR and TOC are combined for further analysis.

TOR is driven by operating revenue and other operating revenue, in which other operating revenue is assumed to be 0. As for operating revenue, it is equal to economic yield, which is calculated by lint yield and seed yield. Because lint yield is higher per unit price and total sales (see Table 4.4), lint yield is the main driver of economic yield and therefore the main driver of operating revenue and TOR.

TOC is driven by operating costs and other operating costs, in which other operating costs and seed costs are assumed to be a fixed amount (Boyce, 2022). Lime cost is also assumed to be a fixed amount (Ground Cover, 2023). The main driver of operating cost is fertiliser cost and energy cost. As nitrogen fertiliser application choice is the main driver of nitrogen emissions, nitrogen management activities are further analysed in the following Table 4.14.

Table 4.14: Level 4 Total operating revenue and total operating cost S1, S2, S3 calculated by 2010–2098 average

		S1	S2	S3	S2-S1	S3-S1	Unit	S1 & S2		S1 & S3	
		Average	Average	Average	Difference	Difference		P value		P value	
Nitrogen Fertilizer application plan		234.00	234.00	275.00	-	41.00	kgN/ha				
Level 4											
Total operating revenue											
	Other operating revenue	-	-	-	-	-	$/ha	-	-	-	-
	Operating revenue	7,228.28	7,249.55	7,264.46	21.26	36.18	$/ha	0.829	-	0.690	-
	Economic yield	7,228.28	7,249.55	7,264.46	21.26	36.18	$/ha	0.829	-	0.690	-
	Yield	5,118.37	5,133.43	5,143.99	15.06	25.62	kg/ha	0.829	-	0.738	-
	Lint yield	2,149.72	2,156.04	2,160.48	6.32	10.76	kg/ha	0.829	-	0.738	-
	Seed yield	2,968.66	2,977.39	2,983.51	8.73	14.86	kg/ha	0.829	-	0.738	-
		7,228.28	7,249.55	7,264.46	21.26	36.18	$/ha	0.829	-	0.690	-
Level 4											
Total operating cost											
Operating cost											
	Total fertilizer nitrogen (FN) cost	502.24	445.11	523.10	-57.14	20.85	$/ha	-	-	-	-
	Total water cost	560.18	560.18	561.52	-	1.33	$/ha	-	-	-	-
	Lime cost	396.00	396.00	396.00	-	-	$/ha	-	-	-	-
	Seed cost	130.00	130.00	130.00	-	-	$/ha	-	-	-	-
	Energy (diesel) cost	406.50	406.50	406.50	-	-	$/ha	-	-	-	-
		1,994.93	1,937.79	2,017.12	-57.14	22.19	$/ha	0.001	*	0.105	-
Other operating cost											
	Other operating cost	2,912.00	2,912.00	2,912.00	-	-	$/ha	-	-	-	-
		4,906.93	4,849.79	4,929.12	-57.14	22.19	$/ha	0.001	*	0.105	-

Note: For this part, because yield losses caused by harvesting machinery operations and other circumstances, as well as the losses from seed dehulling, have not been taken into account in the calculation, the yield is simply divided into lint yield and seed yield. This could result in deviations in the estimation of fibre and seed harvests, leading to an overestimation of the income generated from the seeds. For example, in Boyce's (2022) report, the revenue from seed yield is 1,015 $/ha for the 2016 to 2021 average.

4.4.2.9 Level 4 Total operating cost explanation – Production activity details

In Table 4.15, on-farm production activities are further analysed. From all activities listed, water cost is driven by licensed water cost, which is driven by water irrigated provided by DSSAT results.

As indicated in Table 4.5 of the DSSAT results, nitrogen emissions (two critical components of nitrogen emissions, namely Nitrogen loss through air and Nitrogen loss as enumerated in Table 4.15, are visually represented and analysed in Figures 4.13 and 4.14, respectively) serve as a significant factor influencing the system. Hence, the analysis should prioritise the examination of nitrogen costs as one of the primary targets. Nitrogen cost is driven by both application choice in nitrogen amount (kgN/ha) and fertiliser type (urea or sulphur-coated urea).

When comparing S1 and S2, the N% in urea is higher than sulphur-coated urea. With the same nitrogen level, the fertiliser amount of sulphur-coated urea is larger than urea, which leads to a higher FN application volume and therefore higher total fertiliser cost.

When comparing S1 to S3, although the difference caused by N% still exists, in S3 the application rate significantly exceeded S1, which led to a higher total cost in fertiliser than S1. Therefore, while S3 has the highest nitrogen emissions (see Table 4.5), S3 also has the highest fertiliser cost.

Table 4.15: Level 4 TOC explanation - Production activity details S1, S2, S3 calculated by 2010–2098 average

		S1 Average	S2 Average	S3 Average	S2-S1 Difference	S3-S1 Difference	Unit	S1 & S2 P value		S1 & S3 P value	
Nitrogen Fertilizer application plan		234.00	234.00	275.00	-	41.00	kgN/ha				
Level 4											
Operating cost - Production activities details											
Nitrogen											
	Nitrogen activities cost	507.43	450.30	528.29	-57.14	20.85	$/ha	0.000	*	0.000	*
	Fertilizer application energy cost	5.19	5.19	5.19	-	-	$/ha	-	-	-	-
	Total fertilizer (FN) cost	502.24	445.11	523.10	-57.14	20.85	$/ha	0.000	*	0.000	*
	FN application volume	570.73	508.70	597.83	-62.04	27.09	kg/ha	-	-	-	-
	N leached	0.06	0.06	0.06	-	-	kgN/ha	-	-	-	-
	Nitrogen loss through air	8.50	11.66	13.01	3.16	4.51	kgN/ha	0.000	*	0.000	*
	Nitrogen loss	8.57	11.72	13.08	3.16	4.51	kgN/ha	0.000	*	0.000	*
Water											
	Irrigation activities cost	679.55	679.55	680.89	-	1.33	$/ha	-	-	-	-
	Irrigation energy cost	119.37	119.37	119.37				-	-	-	-
	Total water cost	560.18	560.18	561.52	-	1.33	$/ha	-	-	-	-
	Licensed water cost	560.18	560.18	561.52	-	1.33	$/ha	-	-	-	-
	Water irrigated	5.66	5.66	5.67	-	0.01	ML/ha	-	-	-	-
Lime											
	Lime activities cost	396.08	396.08	396.08	-	-	$/ha	-	-	-	-
	Lime application energy cost	0.08	0.08	0.08				-	-	-	-
	Lime cost	396.00	396.00	396.00	-	-	$/ha	-	-	-	-
	Lime volume applied	40.00	40.00	40.00	-	-	kg/ha	-	-	-	-
		1,583.07	1,525.93	1,605.25	-57.14	22.19	$/ha	0.001	*	0.105	-

Figure 4.13: Boxplot of Nitrogen loss through air for S1, S2 and S3

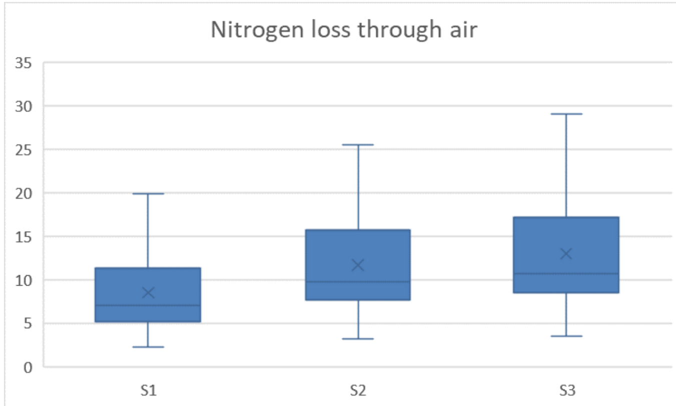
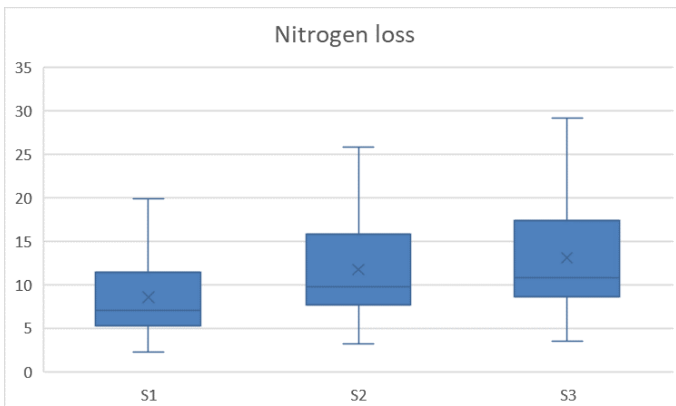


Figure 4.14: Boxplot of Nitrogen loss for S1, S2 and S3



4.4.2.10 Level 4 Total emissions

Total emissions are driven by direct emissions (from cotton growth, the result generated by DSSAT), emissions from farm energy use, emissions from other operating activities and LCA emissions from producing direct materials (fertiliser and lime).

As the emissions from energy use are assumed to be a fixed amount for each hectare, the difference between S1 vs. S2 and S1 vs. S3 is caused by direct emissions and LCA emissions from producing direct materials (fertiliser and lime).

Table 4.16: Level 4 Total emissions S1, S2, S3 calculated by 2010–2098 average

		S1	S2	S3	S2-S1	S3-S1	Unit	S1 & S2		S1 & S3	
		Average	Average	Average	Difference	Difference		P value		P value	
Nitrogen Fertilizer application plan		234.00	234.00	275.00	-	41.00	kgN/ha				
Level 4											
Total emission											
	Direct emissions	9,469.83	10,027.46	10,251.76	557.63	781.92	CO2eqkg/ha	0.000	*	0.000	*
	Emission from farm energy use	10,526.66	10,526.66	10,526.66	-	-	CO2eqkg/ha	-	-	-	-
	Emission from other operating activities	-	-	-	-	-	CO2eqkg/ha	-	-	-	-
	LCA emission from producing direct materials	614.05	1,137.52	1,331.82	523.47	717.77	CO2eqkg/ha	0.000	*	0.000	*
		20,610.54	21,691.64	22,110.23	1081.09	1,499.69	CO2eqkg/ha	0.000	*	0.000	*

4.4.2.11 Level 5 Emissions

Level 5 serves as a supplementary and detailed explanation for level 4, explaining direct emissions (two critical components of direct emissions, namely Direct GHG emission from cropping and Nitrogen emission from cropping as enumerated in Table 4.15, are visually represented and analysed in Figures 4.15 and 4.16, respectively), emissions from farm energy use, and LCA emissions from producing direct materials.

For direct emissions and LCA emissions from producing direct materials, the main drivers are direct GHG emissions from cropping and direct emissions from lime use. Direct emissions are calculated using nitrogen emissions and carbon emissions from cropping provided by DSSAT results (see Table 4.5). The main driver of direct emissions is nitrogen emissions, which show significant differences between S1 vs. S2 and S1 vs. S3 respectively. Thus, differences in direct emissions are driven by fertiliser type and application rate.

LCA emissions from producing direct materials are calculated by LCA emissions per unit of lime/fertiliser and application rate. Thus, they are also driven by fertiliser type and application rate.

For emissions from farm energy use, despite assuming a fixed amount per hectare for all three scenarios, it is worthwhile to investigate their composition structure. As shown in Table 4.17 below, the primary proportion of energy utilisation stems from the consumption of diesel during production and cultivation activities, with the two most significant contributors being irrigation practices and other diesel consumption.

Table 4.17: Level 5 Direct emission, emission from farm energy use, LCA emission from production and emission from other operating activities S1, S2, S3 calculated by 2010–2098 average

		S1	S2	S3	S2-S1	S3-S1	Unit	S1 & S2		S1 & S3	
		Average	Average	Average	Difference	Difference		P value		P value	
Nitrogen Fertilizer application plan		234.00	234.00	275.00	-	41.00	kgN/ha				
Level 5											
Direct emission											
	Direct GHG emissions from cropping	9,465.03	10,022.66	10,246.96	557.63	781.92	CO2eqkg/ha	0.000	*	0.000	*
	Nitrogen emissions from cropping	1,038.72	1,596.76	1,840.44	558.04	801.72	CO2eqkg/ha	0.000	*	0.000	*
	Carbon emissions from cropping	8,426.31	8,425.90	8,406.52	-0.42	-19.80	CO2eqkg/ha	0.987	-	0.422	-
	Direct emissions from lime use	4.80	4.80	4.80	-	-	CO2eqkg/ha	-	-	-	-
		9,469.83	10,027.46	10,251.76	557.63	781.92	CO2eqkg/ha	0.000	*	0.000	*
Emission from farm energy use											
	Diesel production LCA emissions	9,892.24	9,892.24	9,892.24	-	-	CO2eqkg/ha	-	-	-	-
	Diesel on-farm emissions	634.42	634.42	634.42	-	-	CO2eqkg/ha	-	-	-	-
	Pre-planting	81.00	81.00	81.00	-	-	CO2eqkg/ha	-	-	-	-
	Planting	94.50	94.50	94.50	-	-	CO2eqkg/ha	-	-	-	-
	Energy - nitrogen	8.10	8.10	8.10	-	-	CO2eqkg/ha	-	-	-	-
	Energy - water	186.30	186.30	186.30	-	-	CO2eqkg/ha	-	-	-	-
	Energy - lime	0.32	0.32	0.32	-	-	CO2eqkg/ha	-	-	-	-
	Other diesel emissions	264.20	264.20	264.20	-	-	CO2eqkg/ha	-	-	-	-
		10,526.66	10,526.66	10,526.66	-	-	CO2eqkg/ha	-	-	-	-
LCA emission from producing direct materials											
	FN production LCA emissions	585.49	1,108.96	1,303.26	523.47	717.77	CO2eqkg/ha	0.000	*	0.000	*
	Lime production LCA emissions	28.56	28.56	28.56	-	-	CO2eqkg/ha	-	-	-	-
		614.05	1,137.52	1,331.82	523.47	717.77	CO2eqkg/ha	0.000	*	0.000	*
Emission from other operating activities											
		-	-	-	-	-	CO2eqkg/ha	-	-	-	-

Figure 4.15: Boxplot of Direct GHG emission from cropping for S1, S2 and S3

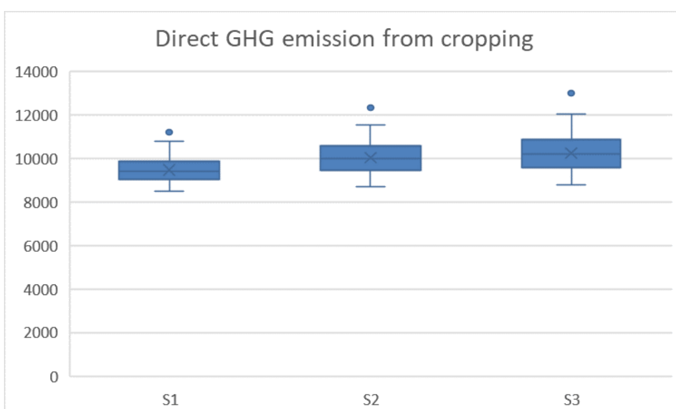
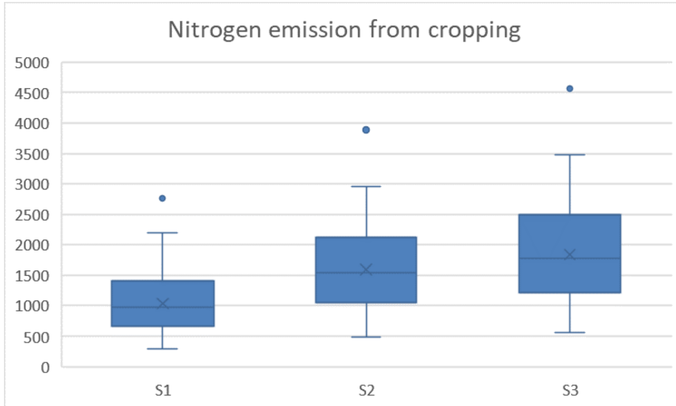


Figure 4.16: Boxplot of Nitrogen emission from cropping for S1, S2 and S3



To this level, the main drivers for differences in S1, S2 and S3 ESRI output is lint yield, fertiliser cost, direct emissions and LCA emission from producing direct materials. Taking a more comprehensive approach to examine these drivers, as identified in the nitrogen cost section, reveals that the disparity in total fertiliser cost is influenced by both the type of fertiliser and its application rate. These factors, in turn, contribute to variations in LCA emissions and nitrogen emissions. This means the difference in ESRI to this level is driven by fertiliser type, application rate and lint yield. Nevertheless, upon linking these findings back to the tree offsets discussed in Chapter 3, the variations in emission remediation costs result in significant differences in ESRI values. Consequently, it becomes intriguing to explore the ESRI value under varying carbon prices as another driver for ESC and ESRI as part of further investigation.

4.4.2.12 Other useful ratios

Table 4.18 presents several informative ratios that aid in comprehending the overall ESRI values.

As illustrated in Table 4.18, S2 exhibits the highest performance in nitrogen usage-related ratios (N productivities and N undertake efficiency); its N_2O CO_2eq/Lint yield ratio is significantly higher than that of S1. This observation may indicate that the most productive option is occasionally distinct from the most sustainable choice.

In addition, for the Breakeven price for emissions, S1 shows the highest and S3 shows the lowest, which contrasts with the observations made in Chapter 3. The observation in the Chapter 3 tree offsets case study shows the low carbon pricing mechanism might benefit entities with the greatest ability to generate profit, rather than those adopting the most sustainable practices in certain circumstances, such as when yield is highly sensitive as an input to nitrogen application rate, which can be further addressed when ability to generate profit is significantly reduced by adopting more sustainable practices. Nevertheless, in the simulations conducted in Chapter 4, since the yield does not exhibit significant differences among the three scenarios, and yield is the main driver of revenue, the most sustainable practices may demonstrate greater adaptability to potential increases in emission reduction policies, such as carbon prices.

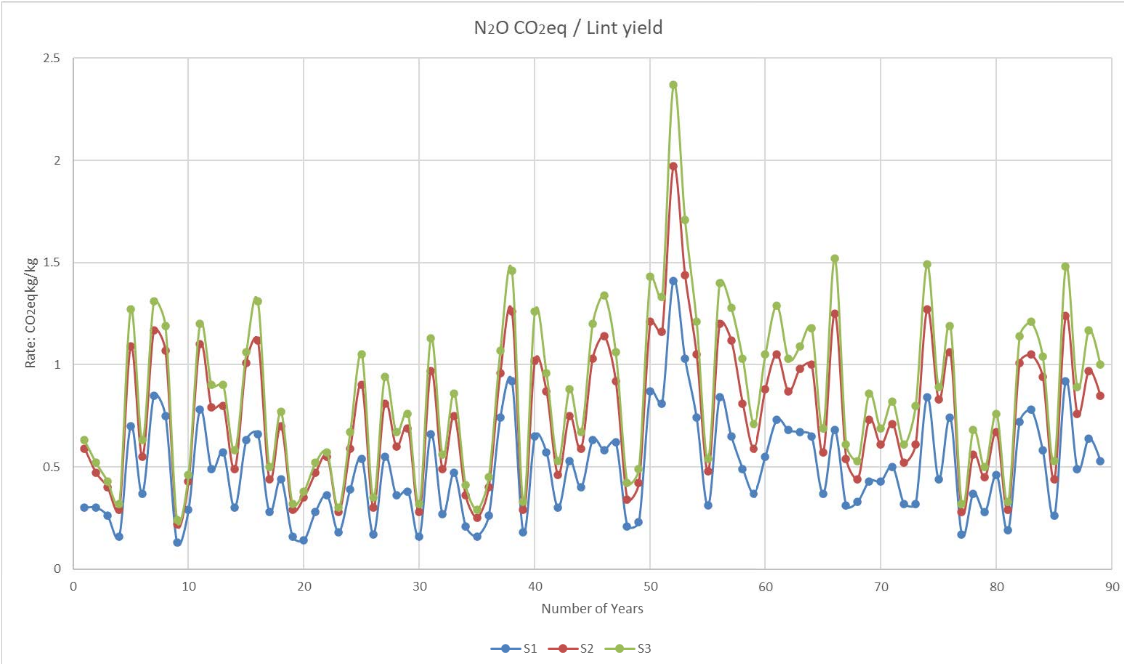
Table 4.18: Other useful ratios, S1, S2, S3 calculated by 2010–2098 average

		S1	S2	S3	S2-S1	S3-S1	Unit	
		Average	Average	Average	Difference	Difference		Formulas
Nitrogen Fertilizer application plan		234.00	234.00	275.00	-	41.00	kgN/ha	
Ratio								
Useful ratios								
	N Productivity	21.87	21.94	18.71	0.06	-3.17	kg/kgN	Crop Yield/Nitrogen Applied
	N uptake Efficiency	1.43	1.44	1.58	0.00	0.15	kgN/kgN	Nitrogen Uptake/Nitrogen Applied
	GHG CO_2eq/Lint yield	4.44	4.69	4.80	0.25	0.36	CO2eqkg/kg	Total emission/Lint yield
	N_2O CO_2eq/Lint yield	0.49	0.75	0.86	0.26	0.38	CO2eqkg/kg	Nitrogen emission/Lint yield
	Breakeven price for emissions	84.47	82.97	79.22	-1.50	-5.26	$/CO2eqt	NOPAT/Total emission
	Profit margin	32%	33%	32%	0.01	0.00	$/$	NOP/Revenue

Figure 4.17 shows the nitrogen emissions per lint yield ratio (CO_2eqkg/kg), the formula shown as:

$$Nitrogen\ emissions\ per\ lint\ yield\ ratio\ =\ Nitrogen\ emissions\ /\ Lint\ yield$$

Figure 4.17: N_2O carbon emission equivalent to lint yield ratio for S1, S2 and S3 from year 2010–2098 (88 years)



4.4.3 Validation of the ESRIDM model

For integrated multidisciplinary modellings like WESTM (Pham et al., 2020) and ESRIDM, the validation extends beyond the individual verification of a singular model or a specific process description; rather, it emphasises the validation of the entire coupled system (Barthel et al., 2008). Accordingly, the validation of the ESRIDM modelling has been explored in three ways: (i) comparison of the simulation with empirical data; (ii) experts' examination of the whole model; and (iii) sensitivity analysis using changing carbon price.

First, in regard to the general comparison of the observation-simulation approach on DSSAT outputs, the output from the simulation is consistent within a reasonable cotton yield level (Pham et al., 2020; Boyce, 2022).

Second, the ESRIDM model was subjected to thorough evaluation by both the supervisory team and external experts specialising in business analysis, crop science and life cycle assessment.

Third, the sensitivity test result is presented in this section 4.5.4.3 and the following section 4.5.4.4.

This validation will not explore the validation of the DSSAT cropping science models in depth. The DSSAT output validation will essentially adhere to the traditional "compare observation-simulation" approach, comparing with real production data to see if the outcome fits within the reasonable level[57].

The focus of this chapter's validation is directed towards assessing verification options applicable to the economic outcomes within the ESRIDM modelling. Constraints of the model mainly pertain to challenges in data collection, as well as issues concerning accuracy and dependability.

Data that can be employed for the validation of economics at various levels often does not maintain the same degree of accuracy, relevance and reliability as equivalent data available in the realm of natural sciences. There are primary concerns regarding these datasets as follows.

The spatial and temporal precision of existing observational data is frequently limited. Much of the data is premised upon assumptions and logical inferences, which might compromise its reliability. In addition, some input data remains either undisclosed to the general public or proves challenging to access. Consequently, the results of such data collection might reside across disparate locations (such as different fertiliser or carbon credit providers) or timing (e.g., the price change within times is not reflected). Lastly, a

[57] This follows the same approach done for DSSAT validation by Pham (2016); Pham et al. (2020); Barthel et al. (2008).

portion of this data remains proprietary in nature, exemplified by datasets like the valuation of farm capital assets and other operating activities (referenced from Boyes, 2022).

Initially, given that the calculations and formulas are interconnected and influence the precision within the ESRI modelling system, it is imperative to validate them to ensure their correctness. This aspect has been scrutinised by the supervisory team.

Subsequently, within the architectural framework of the ESRIDM model, there exist modules pertaining to irrigation and nitrogen, both of which fall under the domain of cropping science. These modules are addressed and processed by DSSAT in the simulation outlined in Chapter 4. Our model has been submitted to domain experts for further evaluation.

It must be emphasised that the output of the ESRI model is intrinsically linked to the outputs of science models (DSSAT). Consequently, it inherits the respective uncertainties and variances from these models. The main inputs for the entire cropping system are characterised by significant uncertainty, such as the climate scenarios, which rank among the most pivotal inputs. Thus, the scenarios utilised should be viewed as representations of various possible futures. Accordingly, simulation results should be considered as methods for exploring potential developments within the confines of these future scenarios. It is crucial to recognise that the results are not predictions; ESRI does not put forth solutions but instead provides a foundation for discussing potential challenges on the horizon and the essence of either sidestepping or addressing those issues.

4.4.4 Discussions

4.4.4.1 Recall

The objective of this chapter is to construct a theoretical framework for integrated modelling systems that facilitate and assist with the identification and assessment of diverse practices, technologies and variables conducive to a more sustainable approach to crop production, with production system option scenario setting and simulations. Additionally, this chapter integrates the co-production and simulation model into the design of the ESRIDM integrated modelling system proposed in Chapter 3. It specifically focuses on nitrogen management, which facilitates the identification and evaluation of alternative practices, technologies and other factors that foster sustainable crop production.

In the theoretical framework development section in Chapter 3, reviews have been done on sustainability, evaluation mechanisms, integrated modelling, decomposition model and environmentally sustainable residual income. While all these theories proposed measurements towards sustainability, limitations occurred over which traditional evaluation mechanisms fell short on addressing non-financial information (cf. Kimbro, 2013; Sinha & Datta, 2020; Siddikee, 2018). ESRI (Brown, 2016) proposed a theoretical framework to build sustainable management but did not provide supported formulas, calculations or applications for sustainability practices; and Pham (2016) proposed integrated modelling for water use efficiency measurements, which the irrigation module in ESRIDM builds on. Pham (2016) built a science-based modelling of a cropping system, sustainability theories and accounting research to assist with decision-making, and also made attempts to decomposition (Penman, 2003) on sustainable production. However, as the highest indicator is an efficiency ratio measurement, it has a gap on measuring sustainability levels, due to the possibility of leading to Jevons Paradox.

In this chapter, ESRIDM modelling has been further refined, by integrating the co-production of knowledge theory into modelling design for scenario identification and a nomothetic model of the theorised interactions of drivers and information in the dynamic co-production of sustainable production systems. In addition, simulation software provides scientific evidence and autonomy in customising simulation scenarios when designing, conducting and testing different scenarios. This significantly improves the flexibility, generality and accuracy of ESRIDM modelling applications.

4.4.4.2 Findings in general

To summarise the findings highlighted in Section 4.4:

(i) Under a relatively low price of the carbon pricing mechanism and ESC covered by entities, S1, S2 and S3 show no significant difference in ESRI value. Also, the differences in ESRI are mainly driven by nitrogen application rate and fertiliser type. In comparison to S3, both S1 and S2 demonstrate comparable or even superior performance, in terms of both profitability and environmental sustainability. If looking at ESC and ratios, the most sustainable practice (S1) provides the lowest environmental sustaining costs and (nitrogen) emissions per lint ratio. However, as the carbon price is relatively low and S1's ability to generate profit is lower than S2 (due to a higher fertiliser cost), growers may be encouraged to move from S3 to S2 which is the sub-sustainable choice, but may not be motivated to take a further step to the most sustainable choice of the three scenarios.

(ii) Comparison between the calculations of tree planting and ESRI valuation models reveals contrasting profitability outcomes. The significance lies in the identification of underlying factors influencing this disparity.

Integrating Chapter 3's analyses on tree offsets and the three scenarios yields valuable insights. When a sustainable option exhibits no considerable difference in yield compared to other choices, adopting the most sustainable practice may offer enhanced adaptability and risk resilience in response to potential environmental policy changes. However, when yield is more sensitive to nitrogen fertiliser management changes (in Chapter 3, a decrease in nitrogen fertiliser input brought a significant decrease in yield), placing a relatively low carbon pricing mechanism may cause less economic loss on highly polluted entities than less polluted entities (see Chapter 3, Table 3.9).

4.4.4.3 Summary of ESRIDM model

The evolution of research in the domain of sustainability has provided a plethora of diverse avenues to pursue sustainable production. Nevertheless, evaluating these alternatives based solely on cost, benefits or pollution reduction may not sufficiently address their systemic implications. In Chapter 3, a comprehensive analysis of various measurement methodologies was executed. While these methodologies offer insightful measurements in select domains, they manifest deficiencies in comprehensiveness, flexibility or systematicity, especially in light of the continually advancing and shifting paradigms of sustainable production (cf. Pearce, 1976; Penman, 2003; Stockle et al., 1994; Faeth, 1993).

In the context where the theoretical and practical demands for sustainable production are steadily escalating, numerous authors have put forth methods to devise integrated models. These models are designed to tackle sustainability challenges by integrating economic and environmental indicators (cf. Hofkes, 1996; Shen, Kyllo & Guo, 2013; Sarker et.al, 2021; Muth Jr & Bryden, 2013; Belcher, Boehm & Fulton 2004).

Additionally, Hueting, Bosch & de Boer (1992) introduced and employed an innovative technique for computing environmentally sustainable national income (eSNI). This method was subsequently expanded on by Richard (2012) and Rambaud and Richard (2015), who advocated for its key role in the utilisation of natural resources. This concept was further developed by Brown (2016) in the theory of ESRI. Pham et al. (2020) ventured into the calculation and modelling of sustainable production, specifically focusing on the integrated modelling of water usage in cotton farming.

This research extends Brown (2016) and Pham et al. (2020) and builds up ESRIDM modelling. This modelling emerges as a viable solution to conduct comprehensive systemic and co-designed scenarios, run through agricultural software simulations, to examine these options. It subsequently offers economic financing evaluations, along with environmental valuation analyses. Moreover, after the main drivers are identified through decomposition, the ESRIDM model can provide analysis on different levels for those drivers to assist with key points that are worth investigating.

For example, as demonstrated in Table 4.19, S2 has the highest output under the current emission reduction option (AU carbon price $28.5). Compared to S3 and S1 which have lower ESRI output, this may suggest that the regulation option currently gives some motivation for cropping to move to more sustainable practices, but not enough motivation to move to the most sustainable production choice.

However, if assumed that the carbon price reaches $63, which is near the cap of the AU carbon pricing mechanism ($75), S1 has the highest output, while S3 has negative ESRI. This means that even if ESC can be covered by entities, S3 is no longer sustainable. In this situation, it may suggest that cropping would have stronger motivation to move to more sustainable practices and would prefer the most sustainable practices among the three scenarios: (S1) over sub-sustainable practices (S2).

To probe this further, carbon pricing in the EU reaches $151.91, when applying this pricing to ESRI calculation. This makes all three scenarios negative in regions with higher carbon prices, such as the EU (carbon price $151.91[58]), so cropping in EU has more motivation to consider and adopt sustainable practices.

[58] Please note that the $ represents Australian dollar.

This observation suggests that there is merit in conducting further investigation into the interplay between carbon pricing and sustainability measurements. *Table 4.19: ESRI prices for S1, S2, S3 under different carbon pricing*

		S1	S2	S3	S2-S1	S3-S1	Unit	(S2-S1)/S1	(S3-S1)/S1
	Carbon price	Average	Average	Average	Difference	Difference			
Nitrogen Fertilizer application plan		234	234	275	0	41	kgN/ha	0	18%
ESRI	**$ 28.50**	718.71	745.44	684.33	26.74	-34.3773	$/ha	4%	-5%
	$ 63.00	7.64	-2.92	-78.47	-10.56	-86.12	$/ha	-138%	-1127%
	$ 75.00	-239.68	-263.22	-343.80	-23.53	-104.11	$/ha	10%	43%
	$ 151.90	-1824.63	-1931.3	-2044.07	-106.67	-219.44	$/ha	6%	12%

4.4.4.4 ESRI value, change of behaviours and sensitivity test regarding carbon price

Table 4.20 presents a sensitivity test regarding various carbon pricing scenarios.

When conducting a deeper investigation into the influence of carbon pricing on ESRI value, we discerned two intersection points at which S1 vs. S2 and S1 vs. S3 resulted in identical ESRI values. As demonstrated in Table 4.20 when the carbon price is around $5.5 S1 is about equal to S3. When carbon pricing reaches $53, S1 starts to give higher ESRI than S2. Based on Table 4.20, it seems that under the $53 carbon price, the motivation for taking sustainable action is cost (efficiency) driven, rather than sustainable production driven, while the most resources efficiency plan may not be the most sustainable.

However, this does not necessarily indicate a pivotal turning point in the policy option. What remains uncertain is the changing pattern of behaviour induced by sustainability policies and the potential market reactions of both fertiliser types. In other words, this shows that there is the possibility that carbon pricing increases might incentivise growers to more sustainable practices which benefit the overall sustainability of cropping production. However, modelling how a change in carbon pricing might trigger nitrogen management choices, behaviour change of growers, and possible turning points may need further investigation and co-production on the sustainable drivers' side of the nomothetic model demonstrated in Figure 4.1.

In addition, the ESC value in this chapter does not directly affect the profitability of cropping. One important reason is that, at the current stage, carbon emissions remediation cost is not enforced to execution at the farm level. Thus, the social burden of the negative consequences resulting from ESC is accepted as default.

This could potentially create a disparity wherein ESC remains inadequately compensated and acknowledged, consequently leading to the misclassification of unsustainable practices as sustainable.

Table 4.20: ESRI value under different carbon price S1, S2, S3

		S1	S2	S3	S2-S1	S3-S1	Unit	(S2-S1)/S1	(S3-S1)/S1
	Carbon price	Average	Average	Average	Difference	Difference			
Nitrogen Fertilizer application plan		234	234	275	0	41	kgN/ha	0	18%
ESRI	**…**	…	…	…	…	…	$/ha	…	…
	$ 5.50	1192.75	1244.35	1192.87	51.60	0.12	$/ha	4%	0%
	…	…	…	…	…	…	$/ha	…	…
	$ 53.00	213.75	214.00	142.63	0.25	-71.12	$/ha	0%	-33%
	…	…	…	…	…	…	$/ha	…	…
	$ 63.00	7.64	-2.92	-78.47	-10.56	-86.12	$/ha	-138%	-1127%
	$ 63.50	-2.66	-13.76	-89.53	-11.10	-86.87	$/ha	417%	3264%

4.5 Practical implications

The effort to enhance nitrogen emission reduction in crop production, particularly in the Australian cotton sector, is an area of considerable focus. Industry associations like the Cotton Research and Development Corporation (CRDC) have recognised the critical need for reducing GHG emissions, with a special emphasis on nitrogen emissions. In listed research programs since 2017, CRDC has been working with scientists and stakeholders to research improving nitrogen management in a research program called More Profit from Nitrogen (MPfN) (CRDC, 2021).

However, their most recent sustainability report *Australian Cotton Sustainability Update 2022* shows “slightly lower yield and higher fertiliser increased emissions per bale'' and no significant reduction within a five-year trend (CRDC, 2023, p.5). This may propose more challenges to the reduction of nitrogen emission than merely focusing on improving nitrogen use efficiency.

CRDC (2023, p. 9) proposed two opportunities to reduce nitrogen emissions while maintaining stable cotton yields: to use more sustainable nitrogen fertilisers; and to enhance the efficiency of fertilisers. Interestingly, the experimented scenarios of Chapter 4 simulated, where S1 is more sustainable fertiliser and S2 improved nitrogen use efficiency, which commenced in 2022, align remarkably well with the sustainable report's strategies, just released in 2023. This synchronisation, however, appears to be more of a fortunate coincidence.

The findings listed in Chapter 4 may give some insights and possible ways to evaluate the sustainability and economic outcomes for their proposed pathways.

In addition, CRDC promotes the adoption of carbon calculators, such as GrainsGreenhouseV.10.8 (Lopez, Ekonomou, Eckard, 2024) and Cotton Greenhouse V.1.35 (Lopez, Ekonomou & Eckard, 2023), which encompasses various on-farm activities and LCA emission calculations. As discussed in Chapter 3, they are becoming increasingly comprehensive, nuanced and well-tailored to the actual production circumstances of relevant industries. However, they remain confined within certain choice frameworks, lacking the capability to conduct a holistic and systematic assessment in conjunction with the costs and output of economic activities.

The ESRIDM model in this research serves to partially enhance the functionality, akin to that of a carbon calculator, while providing a comprehensive review and assessment of the sustainability value of the selected plan. This function is also explained in Chapter 3. For example, in assessing S1 most sustainable plan (within the three scenarios) and S2 sub-sustainable plan (with the aim of resource efficiency improvement), ESRIDM explained the drivers for differences that occurred and showed that, under the situation with a relatively low carbon pricing mechanism, cropping may be more motivated to choose sub-sustainable plan (S2) than most-sustainable plan (S3).

4.6 Limitations and future directions

4.6.1 Extend DSSAT software functionality

The weather data of the current simulation is provided by a single weather station and weatherman prediction in DSSAT, which is a good demonstration but not representative enough to simulate what its impact might be for cropping under possible future climate conditions. It would largely increase the simulation accuracy and data quality if future research integrated with broader climate models, such as the Coupled Model Intercomparison Project Phase 6 (CMIP6).

ii) Adjust for a more accurate irrigation method and model fertiliser accumulation in soil

As explained in Section 4.3.2.2, the dripping irrigation method is a good choice to be investigated. The current version (v 4.8) of DSSAT's calculation model could not provide accurate calculations on irrigation amounts. However, as DSSAT is designed to run interconnection or integration between different agricultural data sets, its functions can be expanded. Other simulation software can be used to do this and then feed data back into DSSAT.

Pham (2016) adopted a more accurate method to measure furrow irrigation water usage by introducing Surface Irrigation Simulation Evaluation and Design (SIRMOD), dividing the soil into small zones and running multiple tests on one hectare. This method might be applied to more irrigation methods, such as dripping irrigation.

Regarding nitrogen fertiliser, because the soil accumulation model of solid nitrogen fertiliser is different from furrow irrigation (Baird 2022), there should be further investigation into building the solid nitrogen fertiliser in the soil accumulation model to various situations such as combine with Pham et al. (2020).

In addition, regarding fertiliser application impact simulation, it would be valuable to simulate the impact on soil nutrition pass-on effects of nitrogen management choice. However, as DSSAT has limitations on the nitrogen component variation, further research might consider integrating other software to measure more suitable nitrogen input for simulation.

4.6.2 Improvements to ESRIDM modelling

i) Improvement of on-farm practices

a) Pest control and after-harvest activities have not been integrated into the design of the ESRIDM model. As they are essential to cotton farm operations, future research might investigate and model these activities into ESRIDM modelling.

b) The planting date is set as a fixed date in this chapter. This could result in missing the optimal sowing date. This option is chosen because DSSAT automatic fertilisation does not have options to fertilise on the planting date or days after planting with different treatments (different fertiliser types and amounts). This implies that if the optimal planting date is adopted, it necessitates conducting at least two simulations over one year. The first time is to find out the optimal planting date, and the second time is to adopt the planting date and set up fertilising dates and treatments to get results. This might provide more accurate simulations of growers' planting activities. However, no critical distinctions are expected to occur.

ii) Improvement of remediation cost

As shown in Chapters 3 and 4, the different choices of compensation and remediation of environmental damage (e.g., Chapter 3 tree offsets and Chapter 4 findings on emission reduction) show a substantial difference between their ESC output. This might be caused by many factors, such as differences in trading in the futures market or the extent of government subsidies. In addition, the environmental cost for society has not been fully subscribed to by emission offsets or carbon pricing schemes, which means the current remediation cost has a limitation on reflecting unidentified/priced environmental impact. This would constitute a worthwhile subject for subsequent academic investigation.

(iii) Further exploration of relationship between behaviour changes and regulation adjustments

The current version of the ESRIDM model provides a platform for systematic simulation of decision-making and analysis of possible regulation options. However, although it is designed to conduct co-production and feedback loops to reflect the relationship between possible choice changes (e.g., nitrogen management or regulation adjustments) and behaviour changes of stakeholders, this aspect is not present in this thesis. However, it has merit for conducting further analysis to complete the structure of ESRIDM modelling.

Chapter 5 Summery, limitations and future research

5.1 Research objective

The research objective of this study is to investigate more effective approaches for identifying and evaluating the achievement of GHG emission reductions through regulatory measures.

To do this, this research develops an integrated model and method to enable the emergence and evaluation of alternative options to achieve GHG emission reductions through regulation.

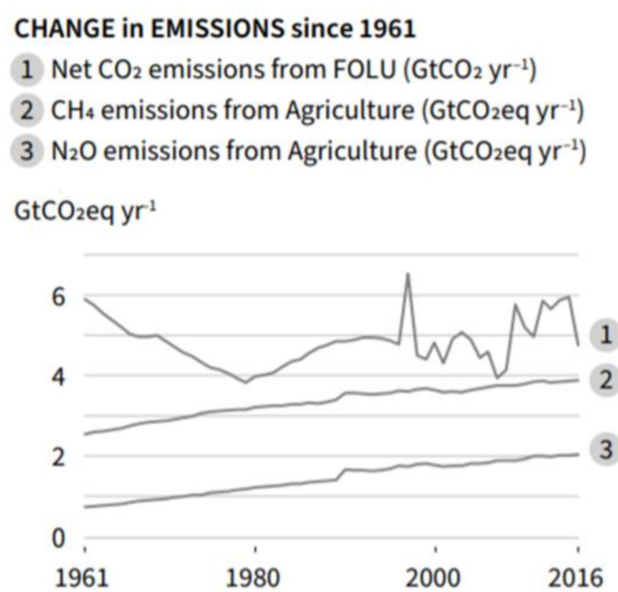
The setting of the research is based on Australian primary producers and stakeholders of a production supply chain, specifically the Australian cotton production industry.

5.2 Background

5.2.1 Agriculture and greenhouse gas emissions

Agriculture is a major source of GHG emissions, which have been increasing steadily over time, as illustrated in Figure 5.1. According to *IPCC special report: a special report on climate change and land* (2019), an estimated 23% of total anthropogenic greenhouse gas emissions (2007–2016) derive from Agriculture, Forestry and Other Land Use (AFOLU) (IPCC, 2019, p. 8). Life Cycle Analysis (LCA) and agriculture production simulators, such as DSSAT, consolidate diverse knowledge about crop production, including links between crop production and climate change. Despite substantial work showing that crop simulators are able to model different crop production scenarios with a level of validity, uncertainties remain about how crop production can be regulated to minimise climate change impacts (Hill et al., 2008; Olesen & Bindi, 2002; Bruulsema et al., 2009). For example, despite being able to evaluate the effect of key decisions on farm performance, models like DSSAT have not been augmented with the key factors that are salient to the identification and evaluation of regulatory options.

Figure 5.1: Global greenhouse gas emissions from agriculture and land use



Source: *IPCC special report: Special report on climate change and land* (2019). N_2O and CH_4 from agriculture are from the Food and Agriculture Organization Corporate Statistical Database (FAOSTAT). Net land-use change emissions of CO_2 from forestry and other land use (including emissions from Pearland fires since 1997) are from the annual Global Carbon Budget, using the mean of two bookkeeping models. All values expressed in units of CO_2-eq are based on AR5 100-year Global Warming Potential values without climate-carbon feedback.

5.2.2 GHG emission reduction regulations

The identification of greenhouse gas (GHG) emission reduction as a significant issue has prompted extensive efforts to facilitate emissions reductions. Key approaches in this endeavour include the establishment of agreements and regulations, such as the Paris Agreement (UNFCCC 2015) and the Tokyo Agreement (UNFCCC 1997).

The emission reduction regulation topic has been investigated from four different angles: (i) to explore the optimisation method to achieve reduction of GHG emissions and enable economic prosperity (Lotjonen & Ollikainen, 2019; Goodstein et al., 2014); (ii) to optimise accounting and calculation mechanisms to assist with emission reduction supervision needs (Hashmi, 2008; Freestone & Streck, 2009; Foster et al., 2017); (iii) to address ongoing discussions of equity challenge in implementing widespread emission reduction policies across regions and industries (Xie, Yuan & Huang, 2017; Wang et al., 2019; Dautume & Schubert, 2016); and (iv) to explore the most effective approach to enforce emission reduction regulations (Freestone & Streck, 2009; Bruvoll & Larsen, 2004; Goulder, 1995; Nilsson, 2009; Lotjonen & Ollikainen, 2019).

However, there remained a key challenge. The question of "how to identify what regulatory options are more efficient in certain circumstances" remained uncertain, due to the lack of a well-developed typology of regulatory options.

To address this challenge, this study prioritises the identification and assessment of GHG emission reduction regulatory options, by engaging in production system option identification, evaluation and comparison. The discernment and evaluation of these GHG emission reduction regulatory options establishes the framework for potential regulation assemblages in subsequent research.

5.2.3 Australian cotton: yield and climate change

With a focus on the Australian context, the analysis of agriculture and climate change involves three parts: (i) address the importance of agriculture in economics and its emission impact; (ii) summarise the impacts of climate change and policy changes on agriculture; and (iii) analyse the importance and necessity of agriculture in emission reduction.

(i) The importance of agriculture in Australia is shown from two key perspectives: (a) economic, and (b) social stability and national health perspectives.

First, from an economic perspective, agriculture is an important part of the Australian economy. According to the Australian Bureau of Statistics (ABS), in 2018 there were 378 million hectares of agricultural land and $59 billion gross value for Australian agriculture commodities produced during 2017–2018 (ABS, 2019). The sector supported approximately 85,000 agricultural businesses (ABS, 2019) and 278,591 agricultural employees (ABARES, 2019). In addition, Australia is one of the top ten export countries of agricultural products in the world. According to the Australian Government Department of Agriculture (ABARES), agricultural export earnings are forecast to be $44 billion (2019–2020) (ABARES, 2019). Although both the trade war between China and the United States and the drought in Australia have had an impact on agricultural output and exports, in general, agriculture is still an important pillar of the Australian economy.

Second, from the perspective of social stability and national health, emission reduction has become an inevitable development requirement in the future (Hanna, n.d.). This has become an unavoidable challenge for modern agriculture, which is the second largest emission-producing economic sector and highly dependent on chemical products. In addition, as agriculture is one of the supporters of the basic demands of human survival, it is going to hold its importance for a long time (Henzell, 2007). So seizing opportunities may bring new advantages to the agriculture industry. Furthermore, with the urgent need for carbon control, there is a growing call suggesting the necessity of carbon emission intervention flows in international trade (Duro, 2013) including concerns about carbon leaks caused by international trade (Charles & McLure, 2014).

(ii) Climate change has a range of effects on agriculture; crop yield may be more sensitive to its effects which may lead to yield decline in total rather than yield increase (IPCC, 2019).

There are three external factors that determine crop yield: climate (including radiation, temperature, rainfall, CO_2 and others), pests and diseases, and soil.

Some proposed that, as one of the main raw materials of photosynthesis, the increase of CO_2 concentration in the atmosphere will promote its fertiliser effect on crop growth, which has a positive effect on crop yield (McGrath & Lobell, 2013). However, there are more factors that may cause yield to decline even when there is an increase of CO_2 concentration. First, the CO_2 fertilisation effect on plant photosynthesis is limited by the carbon dioxide compensation point[59]. In addition, it is also limited by regions that may provide uneven daylight time for radiation supply (McGrath & Lobell, 2013). Second, increasing levels of CO_2 in the atmosphere will result in higher temperatures, which might lead to changes in sowing time and accelerate crop growth and maturity (Olesen et al., 2012). The shorter crop growing period may also result in less productivity yield.

According to the report *The Global Climate in 2011-2015* produced by the World Meteorological Organization, global warming can lead to increased extreme weather. In their paper in *Nature*, Stott, Stone and Allen (2004) show that human-induced climate change is likely to more than double the risk of heat waves, similar to 2003. According to Stone et al., there exists a significant relationship between the El Niño-Southern Oscillation (ENSO) and wheat yield, showing a negative impact (Stone, Hammer & Marcussen, 1996). In addition, the negative impact of extreme weather on crop production is expanded and supported by Lesk (Lesk, Rowhani & Ramankutty, 2016). Fourth, increasing temperature may lead to drought, which will cause decreases in yields. In addition, it might also cause a

[59] The carbon dioxide compensation point refers to the concentration of carbon dioxide in the external environment, when the amount of carbon dioxide absorbed by photosynthesis and the amount of carbon dioxide released by leaves reach a dynamic balance under light conditions (cf. Tolbert et al., 1995).

large-scale outbreak of pests and diseases, which will also have a negative impact on crop yield.

In addition to increasing or decreasing crop yields, there is another influence that will largely depend on the crops' regions. Increasing temperatures in regions will also lead to a shift in the agriculture belt (Qi & Zhang, 2018). This might lead to large-scale damage in production, due to the decrease in suitable planting regions for crops the growers have already planted, not only in a decrease in unit yield for a single crop. However, in northern Asia and North America for example, the shift of the agricultural belt will decrease winter frost damage and expand the regions which allowed double cropping to the north, which might increase the potential yield (Zhou, 2015).

Thus, the crop yield is sensitive to climate change and there is a larger likelihood that climate change will lead to a decline in future yield. According to IPCC's *Special Report Climate Change and Land and ASBP*, the crop yield is more sensitive to the yield reduction effect than the yield increasing effect brought by climate change; therefore, a decrease in future yields is predicted (Ainsworth & Donald, 2010). In addition, this shows the possibility of modelling emission reduction effects through sensitivity analysis in crop yield.

(iii) The importance of agriculture in emission reduction can be analysed from three perspectives: (a) to protect biodiversity, (b) to mitigate climate change, and (c) to develop bio-energy (FAO, 2007). Driven by the increasing population and need for food supply, agriculture expansion is predicted to continue to grow. This may cause a set of consequences, including converting natural ecosystems to farmland and the increasing the usage of chemical fertiliser, leading to a high risk of destroying the water nutrient balance while promoting the increase of crop yield (Tilman et al., 2001). According to Tilman et al. (2001), such eutrophication and habitat destruction will lead to "unprecedented simplification of ecosystems, loss of ecosystem services and extinction of species" (Tilman et al., 2001, p.284). In order to minimise the damage of agricultural expansion on the environment, significant scientific progress and changes to regulations, technology and policies are needed (Tilman et al., 2001). However, a great breakthrough, such as soilless crops in plant science, is not easy to achieve. This means that adjustment in regulations is becoming more urgent and needs to be fulfilled.

Researchers are trying to find the means to decrease emissions and environmental damage in agriculture, by analysing the crop life cycle and growth principle through building climate and agriculture simulation models (IPCC, 2019). This may affect the emission reduction and sustainable economy simulation. As Tilman et al. (2001) point out, to achieve the environmental sustainability aim, agriculture will require policy guidance at the national level, to improve the agricultural production structure (Renard & Tilman, 2019). This is supported by two premises: first, the regulation deficiency of existing emission systems for agriculture and land use-related modules; and second, the achievable emission reduction regulation output simulation, under the existing framework of science and technology.

5.3 Research summary

Chapter 2 focused on the analysis and categorisation of emissions regulations. Instead of advocating for the implementation of specific regulatory assemblages, the objective of the research was to develop an integrated model aimed at aiding policymakers and decision-makers to evaluate and analyse prevailing issues or challenges. This model facilitates the exploration of potential opportunities and risks associated with various policy or decision outcomes through the employment of scenario-based analysis.

However, the discourse surrounding regulatory assemblages is a macroscopic and expansive subject. Analogous to the assembly of timepieces from individual components, the study of regulatory assemblages requires further deconstruction into more minute units to clarify how they interact, influence, or repel one another. Moreover, the categorisation within regulatory assemblages should be contingent upon the specific real-world scenarios they address or embedded within simulations constructed to assess the stipulated or hypothesised conditions. In this context, the fundamental units of categorisation for regulatory assemblages should be the regulation options. Within the discourse of production systems, regulation options are frequently selected from a pool of extant or anticipatable production options. Therefore, the subsequent chapters concentrated on the filtration of production options and the edification of simulations, thus scaffolding an integrated modelling framework for the analysis of regulatory assemblages in prospective research.This aim was further explored in Chapters 3 and 4. Chapter 3 is devoted to the theoretical construction, framework establishment, formulas and demonstration of the Environmentally Sustainable Residual Income Decomposition Modelling (ESRIDM), through a tree offset case study. Chapter 4 extends the capabilities of the ESRIDM model by incorporating part of the co-production approach for scenario configuration and feedback adjustments, which is relevant for refining modelling inputs in this thesis, and maybe for designing regulatory options in future research. Additionally, the chapter integrates cropping software simulation as a constituent part of the ESRIDM model.

While Chapters 3 and 4 do not provide explicit recommendations for the specific formulation and combination of emissions reduction policies, they effectively demonstrate the capabilities and potential of the ESRIDM model as a tool for policy and choice analysis. Regulatory assemblages, when deconstructed, consist of specific regulations, which in turn evolve from a myriad of regulatory options. The proposal, resolution and analysis of these regulatory options are intrinsically linked to more granular choice options. Thus, when the ESRIDM model exhibits its proficiency in simulating, constructing, deconstructing and analysing choice options, it signals its utility as an instrument for shaping effective emissions reduction policies. Moreover, the ESRIDM model displays its credibility and flexible applicability, by grounding its operations and decision-making steps in scientific principles. This confers upon the ESRIDM model significant potential for both theoretical and practical advancement and development.

5.4 Contributions

This thesis makes four main contributions.

5.4.1 Contribution one

This research makes both theoretical and practical contributions by articulating and incorporating a broad range of factors in the evaluation of diverse emission regulation frameworks for emission reductions. More specifically, this research focuses on the comprehensive evaluation and assessment of production system options with different scenarios with a view to informing the selection, evaluation and adaptation of regulatory approaches in an agricultural context.

In addition, to explore the key challenge of "how to identify what regulatory options are more efficient in certain circumstances", This research identified the absence of typology modelling of regulatory assemblages as problematic due to ambiguities and uncertainty while comparing different system combinations. Chapter 2 addresses the first motivation, which is the lack of consensus in the literature on how best to categorise and regulate GHG emissions.

Chapter 2 develops a novel categorisation model to categorise and describe a wide set of factors for evaluating the relative merits of different emission regulation assemblages for carbon reductions. Chapter 2 categorisation model builds off and extends Karp and Gaulding's (1995) work in Motivational Underpinnings of Command-and-Control, Market-Based, and Voluntarist Environmental Policies. In addition, this chapter presents a new model that synthesises both conceptual and applied frameworks of emission reduction regulations. This model is intended to elucidate the comparative efficiency and effectiveness of emission reduction regulations.

The establishment of this chapter's regulatory categories is imperative from two perspectives. (i) For literature, this study synthesises, analyses and highlights how international environmental regulations may fit into the framework of regulation assemblages. It emphasises the synergistic interactions between various regulations, the complexity of the assemblage structure; the variability in implementation possibilities; and the importance of systemic analysis at the system level. Additionally, it underscores the intricacies of testing and evaluation, as well as the significance of tracing back outcomes. This, in turn, further substantiates the necessity of scenario design and simulation, as conducted in Chapter 4.

(ii) For this research, it acts as a foundational framework, guiding the identification, analysis and recombination of regulation assemblages. Moreover, it elucidates the further decomposition of these assemblages into distinct elements, such as regulations, regulatory options and production system options.

Further more, Chapter 2 contributes to the literature by providing the scope and classification for modular frameworks essential for further integrated modelling of these regulation assemblages[60]. Consequently, in the ongoing study of plausible emission regulation options, Chapter 3 builds upon the categorisations presented in this chapter. It further deconstructs the regulation assemblage and conducts integrated modelling and decomposition analysis for several of its potential configurations.

5.4.2 Contribution two

This research presented in Chapter 3 contributes to the framework of Environmentally Sustainable Residual Income (ESRI). Hueting, Bosch & de Boer (1992) initially emphasised the ecological value of natural resources and proposed the framework of sustainable national income, a concept further developed by Richard (2012) then Rambaud and Richard (2015) with an economic and accounting domain. Brown (2016) reinforced this idea's practicality and proposed the theory of Environmentally Sustainable Residual Income. In addition, Pham et al. (2020) highlight the significance of considering the ecological value in sustainable income assessments.

This approach progressively builds on previous research and extends it to integrated modelling approaches; it enhances the calculation approach by incorporating inputs from Life Cycle Costing (LCC), Life Cycle Analysis (LCA), management accounting and economic modelling, within the agricultural context. Additionally, it applies Pearce (1976) on reflection, calibrate and analyses for case study in Chapter 3.

This contribution is pivotal because the ESRI theory, although predicated on the notion of national sustainable income, fails to elucidate clear theoretical or practical steps for calculating the ESRI value within a specific industry. This represents a substantive step for

[60] To elucidate, the categorisation modelling presented in Chapter 2 establishes a framework for further deconstruction and classification of regulatory assemblages, distilling them into their fundamental elements—namely, the regulations themselves—and facilitating a classification of the various options inherent to these regulations, which are referred to as the regulatory options in Chapter 3. In addition, the link between regulation options and production system options has been demonstrate in Section 2.1.5.

the further development of the theory. While Pham et al. (2020) have made preliminary explorations into quantifying sustainable production evaluation in integrated modelling, using the opportunity cost of natural resources as a basis, their articulation of this aspect remains notably ambiguous. Against such a backdrop, this contribution addresses this challenge, advancing both the theoretical development and practical application of an ESRI theory.

5.4.3 Contribution three

The study presented in Chapter 4 further enhanced the novel integrated modelling system in Chapter 3 to enable the identification and evolution of more sustainable production systems and illustrates the model in context of nitrogen management options at the farm level. The integrated modelling design partly builds upon the work of Pham et al. (2020) and other integrated modelling research on sustainable production and evaluation (cf. Hofkes, 1996; Belcher, Boehm & Fulton, 2004; Shen, Kyllo & Guo, 2013; Chami & Daccache, 2015; Hadjimichael et al., 2016). Chapter4 enhances this modelling approach by integrating scientific software (e.g., DSSAT) and offering evaluations for various simulation scenarios set up through an approach inspired by co-production of knowledge (Polk 2015).

This contribution is significant because, while numerous studies have employed integrated modelling or scenario simulation approaches as previously discussed, they have not exhibited the same breadth and flexibility in the design and simulation of scenarios for evaluation and validation as in Chapter 4. Compared to existed studies which also utilise scientific models and economic analyses such as Barthel et al. (2008), Welsh et al. (2013), and Guo et al. (2021), they primarily focus on short-term forecasts and simulation accuracy and lack the capability for in-depth analysis and attribution of phenomena, as well as the ability to simulate diverse sustainable production scenarios.

This flexibility and reliability are brought about by the setting up of co-production scenarios, demonstrated in the nomothetic model of the theorised interactions of drivers and information in the dynamic co-production of sustainable production systems (Chapter 4, Section 4.2.3). This approach potentially contributes to the presentation of regulation options and production system options evaluations, offering invaluable insights and thorough accounting assessments. Evaluations and comparisons can be conducted before, during and after decisions are made, assisting in the identification and testing of possible alternative outcomes.

5.4.4 Contribution four

Contribution four is a contribution to transdisciplinary research.

Transdisciplinary research has been described as "problem focus, evolving methodology and collaboration" (Wickson, Carew & Russell, 2006). This research aligns with these characteristics and adeptly illustrates the attributes of transdisciplinary research.

The value of engaging in transdisciplinary research within the domain of ecological economics is explored in Costanza and King (1999). Moreover, the significance of aligning analysis with production processes or scientific methodologies has been underscored in sustainable production research. Such endeavours necessitate that researchers possess a certain level of proficiency in science or related technologies of industrial production (cf. Hueting, Bosch & de Boer, 1992; Richard, 2012; Brown & Bajada, 2016; Pham et al., 2020).

Transdisciplinary approaches offer certain advantages in studying topics related to sustainable production, predicated on the notion that achieving sustainability necessitates the concurrent sustainability of both scientific-technological and economic activities, as elaborated in Chapter 3. Consequently, research projects claiming to achieve or premise sustainability will invariably involve at least economic and technological domains. Either directly or indirectly, this necessitates that research related to sustainable production incorporates transdisciplinary research thinking and capabilities (Weimer & Ruijter, 2017; Lino et al., 2019; Maiello et al., 2013; Lemos & Morehouse, 2005; Galende-Sánchez & Sorman, 2021). Yet their limitations fell short in modelling and alternative regulatory/decision-making option scenarios.

Nevertheless, there has been limited research undertaking such an expansive transdisciplinary approach on the subject of agricultural nitrogen emissions reduction, regulation and production system options identification, evaluation, and comparison.

With respect to this issue, the present study contends that although research on sustainable production is highly amenable to a transdisciplinary research framework, the specialised integrity of the subject matter should still be ensured by experts in the respective fields. On the question of how to contribute to the entire integrated modelling system while ensuring scientific rigour and specialisation within their domains, Polk (2015) offers a compelling approach. Coincidentally, the theory presented by Polk (2015) has been applied to sustainable production research, such as Wyborn et al. (2019), Norström et al. (2020) and Chambers et al. (2021). However, despite its immense potential for bridging stakeholder engagement and facilitating multi-domain transdisciplinary research, studies utilising this theory have not sufficiently explored its applications in the areas of environmental policy formulation, evaluation and implementation – especially in the context of agriculture and, more specifically, nitrogen management decisions and applications. This study contributes to filling this particular gap.

To be specific, this research aims to contribute to transdisciplinary research by integrating knowledge from the fields of ecological economics, agriculture, sustainable production, among other domains. In Chapters 3 and 4, multiple knowledge domains and practices were amalgamated in a transdisciplinary manner to confront the sustainable production challenge in agriculture, which is both theoretical and pragmatic in nature. This research amalgamates four pivotal knowledge domains: regulation and policy theory; co-production of knowledge (Polk, 2015); crop science; and management accounting theory. This contribution is significant, because it lays the methodological groundwork for future research in this area, as established by the theoretical model presented in Chapter 4.

5.5 Limitations and future research

5.5.1 Limitations for this research

(i) This model is highly dependent on the science model's ability to source data and the reliability of that dataset. This means that, the more difficult it is to collect reliable data, the less accurate the model's output will be. For example, there might be difficulties while trying to apply an agriculture model dominated by a small-scale economy.

(ii) The complexity of this system may limit its general usage and increase its dependence on second-hand data and theory, which might affect the output accuracy.

(iii) The model system in this thesis is limited, because there are only three sets of alternatives simulated and it focused on cotton production, which is only one aspect of production. That might not be enough to capture the whole system of ecological and economic systems in reality.

(iv) This thesis relies on extant models to make the work trackable. More detailed model designs are required, according to the requirements of accuracy and measurements to local conditions.

(v) There are time limits and scale limitations for this thesis. Designs like applying the full LCA model, decision model, co-production workshops, and input-output modelling cannot be realised in this thesis.

5.5.2 Speculations on future research

Potential avenues for future research may proceed in the following four directions:

(i) Further validation of ESRIDM modelling

This study explores the understanding that the DSSAT cropping software has undergone rigorous validation by numerous scholars in the scientific community, thereby establishing its reliability. The validation associated with DSSAT pertains specifically to the inputs and outputs generated within various scenarios. The output fidelity is confirmed, using a "compare observation-simulation" methodology to ascertain whether the results align with acceptable norms. Additionally, the ESRIDM has been subject to expert evaluation for its validation.

It should be emphasised that the current study is principally designed as a proof-of-concept, centring its attention on the establishment of structural elements and theoretical underpinnings. Consequently, a substantial portion of the study relies on data derived from academic literature, cotton conference contents and conversations. For more robust validation, future studies could benefit from the incorporation of empirical data obtained from field experiments or operational data gathered from multiple real-world agricultural settings. As a subsequent avenue of research, there is the potential to extend the validation procedures for the ESRIDM model, exploring its applicability within a more comprehensive and nuanced framework.

(ii) Improvements in scientific modelling integration to other disciplines

As elucidated in Chapter 4, despite DSSAT being widely accepted by agricultural scientists as a proficient cropping simulation software, it presents limitations in simulating irrigation details and soil nutrition. Consequently, it is unable to provide substantive analysis and comparison between two distinct irrigation methods and the associated nitrogen balance (Pham, 2016; Ritchie & Godwin, n.d.).

A resolution to this, as previously proposed by Pham (2016), involves adopting alternative agricultural simulation software capable of generating data that meets detailed computation requirements. This strategy would enhance DSSAT's capacity for detailed simulation and computation by dividing the farmland into multiple measurement points (e.g., 72 test points in Pham, 2016), thereby fulfilling the need for more accurate simulations.

In the initial design phase for scenarios explored in Chapter 4, one scenario considered was dry-land cotton – a topic that was also a focal point at the Australian Cotton Conference 2022. This scenario was ultimately discarded following expert recommendation for two primary reasons: a) the applicability of dry-land cotton is confined to a limited, small-scale context; and b) the scenario is more relevant when examining the nitrogen impact of various irrigation plans. Given that the impact of nitrogen emissions influenced by different irrigation methods has already been addressed and subsequently set aside in Chapter 4, pursuing the dry-land scenario further seemed redundant. Nonetheless, future research may find it valuable to revisit and test this particular scenario, especially when incorporating economic indicators such as market share and production efficiency.

In this research, given that the primary objective is to illustrate the application of ESRIDM modelling, rather than justifying scientific modelling, such an approach is not employed.

In addition, this research uses prediction weather data; this method has also been used by other research (cf. Sarkar & Kar, 2006; Sarkar & Kar, 2008; Nicoloso, Amado & Rice, 2020). If future research is able to integrate with broader climate models, then adopting the weather data of a wider climate modelling, such as Coupled Model Intercomparison Project Phase 6 (CMIP6), would largely increase the simulation accuracy and data quality.

(iii) Improvement of remediation cost measurements

As demonstrated in both Chapters 3 and 4, the outcome of ESRIDM is closely related to environmental value identification and pricing mechanisms. In many instances, these mechanisms show substantial disparities across various nations and regions. If evaluations are based on local standards for valuing natural resources, it could lead to restricted comparability of sustainable production across different countries (e.g., the ESRI difference under EU and AU carbon pricing in Chapter 4). Future research may focus on improving the valuation and evaluation of natural resources through the establishment of more comprehensive and universally applicable value systems, grounded in cost-of-restoration or scientifically based assessments. Such an improvement could potentially increase the versatility of ESRIDM modelling and facilitate comparisons and analyses between regions at different stages of development.

(iv) Extent of co-production interaction with stakeholders and further economic modelling

This study has not extensively delved into the co-production feedback loop subsequent to ESRI calculations. Ideally, this phase should examine and investigate stakeholder behaviour, preferences and performance, in response to varying regulatory options. Additionally, scenarios have been primarily centred on nitrogen management options, which constitute a subordinate stratum of regulatory options. In turn, these regulatory options form a lower tier within the regulation assemblages identified and modelled in Chapter 2. Future investigations might extend to the co-production of regulation and regulation assemblage scenarios, as well as changes in behaviour elicited by different identified regulatory options.

Appendix

Appendix A

Name	Description	Formular	Output
ARR	ARR is used in capital budgeting to compare different investment opportunities and is easy to calculate, but it does not consider the time value of money.	ARR= Initial investment / Average annual profit	If the outcome is higher than the minimum acceptable rate of return (hurdle rate), the investment is considered profitable, and the investor may choose to proceed with the project.
IRR	IRR is used to evaluate the potential profitability of an investment by calculating the discount rate at which the present value of the investment's cash inflows equals the present value of its cash outflows	IRR = (Cash Flow- Economic depreciation) / Last period's depreciated economic value of the asset	If the output is greater than the required rate of return or cost of capital, the investment is considered profitable, and the investor may choose to pursue it.

Name	Description	Formular	Output
NPV	NPV is a finance instrument to determine the profitability of an investment by calculating the present value of all expected cash inflows and outflows associated with the investment.	$NPV = \sum_{t=0}^{n} \frac{\mathrm{Rt}}{(1+i)^t}$ Rt=Net cash inflow-outflows during a single period (t) i=Discount rate or return rate t=Number of time periods	NPV positive = present value of the expected cash inflows is greater than the present value of the cash outflows, the investment is expected to generate more cash inflows than outflow NPV 0 = the investment is expected to maintain its value in t time periods. NPV negative = the investment is expected to incur more loss of value than generate value.

Source: Hillier et al. (2014)

Appendix B

Mass Balance of Cotton Nitrogen

Cotton yield optimum is 11.82 bales (2,683.14 kg) per hectare in Australia (CRDC, 2019), and the average cotton planting size is 467 hectares per farm (Cotton Australia, 2022). The increase in yield is primarily because of chemical nitrogen fertiliser application.

There are two significant resources for cotton nitrogen supply: fertiliser and soil nitrogen. Given that furrow irrigation is most commonly used on Australian cotton farms, the nitrogen output mainly has three channels: in the soil as organic matter, absorbed by cotton, or lost to the environment (primarily rain and irrigation water).

There are many differences in the literature's specific distribution proportion of nitrogen loss. However, it can be agreed that the total N recovered when lint is mature is about 40% to 45% of aboveground biomass (e.g., Macdonald et al., 2016). For the remaining data on the ratio of preservation in the soil to loss, Angus and Grace (2017) data are that 35% is retained in the ground and 20% is lost. Experiment (Rochester, 2003) shows that 40% is lost in the environment.

According to Rochester and Bange (2016), yield increased with nitrogen absorption when applied 200 to 250 kg N/ha. However, from 220 kgN/ha to 300 kgN/ha, the economic gain from yield increase is less than the cost of input (Antille & Moody, 2021). This assumes that the 200-220 kgN/ha nitrogen application rate is the most efficient.

The nitrogen source of all nitrogen fertiliser is ammonia (NH_3).

Optimum 70% of global ammonia production uses natural gas to perform the steam reforming concepts. This is considered to be the best massive production choice with technology available for both energy efficiency and GHG emission reduction (Brightling, 2018). The natural gas method is dominant in the Australian ammonia industry (Fertiliser Australia, 2022).

Even though this method is 36% to 58% higher in emission reduction than other methods (IPCC, 2014), the process still leads to 2.6 metric tons of GHG emissions per metric ton of ammonia produced (Liu, Elgowainy & Wang, 2020).

To roughly calculate the demand for nitrogen in a high-efficiency situation, use liquid ammonia (anhydrous liquid ammonia), with a nitrogen content of 14 / 17 = 82.35%, to estimate. Thus, this becomes 3.15 times production emission per unit of nitrogen.

The carbon footprint from the complete life cycle analysis for urea production will be close to 0.714 tons of carbon dioxide equivalent (CO_2eq) (Kumar et al., 2021); urea has 46% nitrogen, which is the highest nitrogen content in solid nitrogen fertiliser.

This is not the optimal emission because when using ammonia to further process into ammonium nitrogen fertiliser, nitrate-nitrogen fertiliser and amide nitrogen fertiliser, there will be additional emissions during the production process. However, urea or slow-release nitrogen fertiliser will decrease the nitrogen loss to both soil and water, rather than liquid ammonia.

Table B.1: Ammonia and Urea Production Emission

N applied kg N/ha	200 kgN/ha	250 kgN/ha	300 kgN/ha	
Ammonia required (82.35%)	242.87	303.58	364.23	kg N/ha
Production emission	735.90	919.85	1103.62	CO_2eqkg/ha
Urea required (46%)	434.78	543.48	652.17	kg/ha
Urea production emission	310.43	388.04	465.65	CO_2eqkg/ha

Note: It is assumed that there is no unexpected nitrogen loss in nitrogen fertiliser production.

Nitrogen inputs to cotton production

Various research shows crops take a large percentage of nitrogen from the soil, around 101-162 kgN/ha (Rochester, 2011). For all applied nitrogen fertilisers, about 46 kgN/ha is lost as non-GHG N_2 (Rochester, 2003), and 62 kgN/ha is retained in the soil (Macdonald et al., 2016).

Because the N in nitrogen fertiliser comes from the air (N_2), the nitrogen loss turns the non-emission gas into pollution.

Table B.2: Urea production and usage ratio

N applied kg N/ha	200 kgN/ha	250 kgN/ha	300 kgN/ha	
Urea required (46%)	434.78	543.48	652.17	kg/ha
Production emission (urea)	310.43	388.04	465.65	CO_2eqkg/ha
Soil fixation	62.00	62.00	62.00	kgN/ha
Urea loss to water/air (wasted)	211.09	265.43	319.78	kgN/ha
Production emission (urea)	149.87	188.46	227.05	CO_2eqkg/ha
Urea soil fixation (used)	223.70	278.04	332.39	kgN/ha
Ratio of (wasted vs. used)	0.94	0.95	0.96	Rate

This means one unit of urea absorbed by crops or stays in the soil to benefit crop bears at least 0.94 to 0.96 is wasted for pure pollution.

According to Chen et al. (2008), fertiliser N recovered in the crop absorption rate is optimum 40% in Australia.

Since most Australian cotton-growing soils are medium to heavy clay, ponding is easy to occurs after furrow irrigation or heavy rain; an estimation of 50-100 kgN/ha may be lost (Rochester, 2003).

Calculation of on-farm nitrogen-related emission

Simplified from Antille and Moody (2021):

$$\begin{aligned} &On - farm\ application\ waste \\ &\quad = Fertiliser\ loss\ to\ air + Fertiliser\ loss\ to\ water \\ &\quad + Fertiliser\ loss\ to\ soil + Nitrogen\ emission \end{aligned}$$

$$\begin{aligned} &Fertiliser\ loss\ to\ soil \\ &\quad = Nitrogen\ kept\ in\ soil - Nitrogen\ absorbed\ by\ next\ year's\ crop \end{aligned}$$

$$Pre - application\ emission = On - farm\ application\ waste$$

The application loss happens within one week from the application day, and the volatile rate is around 14.52%~17.64% (Yang et al., 2017).

Because liquid nitrogen is highly volatile and challenging to store, few farms apply it or its diluted product, ammonia. Most farms choose urea or other ammonium nitrogen fertilisers.

For nitrogen fertiliser, pH volatilisation loss is one of the main reasons for the failure during application. When the soil pH value is less than 7, there is almost no volatilisation loss of ammonia; with an increase of pH, ammonia loss increases (Martens & Bremner, 1989). Temperature also affects the solubility of ammonia in water and the diffusion rate of ammonia in soil. With high temperatures, the solubility of ammonia in water is low, the diffusion rate in the soil is significant, and the volatilisation loss of ammonia increases. In contrast, the volatilisation loss of ammonia is slight (Liu et al., 2019). Many studies on fertilisation depth showed that the volatilisation loss decreased when ammonia nitrogen fertiliser was applied to a depth of 10 cm of topsoil (Blackshaw, Molnar & Janzen, 2017).

If applied on the surface, there will be about 17.7% of urea loss. Even by application beneath surface method, the amount was still about 14% when the fertiliser was incorporated, primarily as ammonia gas (NH_3) (Palma et al, 2008).

Table B.3: Application loss – Urea

N applied kgN/ha	200 kgN/ha	250 kgN/ha	300 kgN/ha	
Urea required (46%)	434.78	543.48	652.17	kg/ha
Production emission(urea)	310.43	388.04	465.65	CO_2eqkg/ha
Application loss (Nitrogen)	28.00	35.00	42.00	kgN/ha
Application loss (urea 14% loss rate)	60.87	76.09	91.30	kgN/ha
Application loss (mostly as NH_3)	34.00	42.50	51.00	kgN/ha

Table B.3 presents the estimated application nitrogen loss, and is linked to the formula：

$Pre - application\ emission\ = On - farm\ application\ waste$.

In this step, nitrogen fertiliser is lost to water, soil and air. However, in contrast to the nitrogen emission in the next step of cotton growth (through nitrification and denitrification), the amount of nitrogen loss in the air is NH_3. This would not cause environmental problems and is not counted as GHG.

Thus, for the application step, although there is 60.87 to 91.30 kgN/ha urea loss, no additional GHG emission occurred, other than production emission.

Cotton growth

Cotton growth is the main step of nitrogen absorption, soil fixation, nitrogen emission (nitrification and denitrification), and other pollution (such as water pollution). It is also the most complicated and important step to calculate.

The nutrition absorbed by the plant has two sources in the cotton growth stage: soil and fertiliser. During this process, nutrition is also lost to the environment within three significant channels: loss to irrigation, loss to air, and soil fixation through nitrification and denitrification. In addition, due to nitrification and denitrification effects, the nitrogen loss to air takes the form of nitrogen dioxide, which is GHG.

This can be represented in this formula (modified from Macdonald et al., 2016):

$$\begin{aligned} Nitrogen\ demand \\ &= crop\ absorption - soil\ supply + irrigation\ loss + emission\ loss \\ &+ soil\ fixation \end{aligned}$$

$$iNUE = [N\ (have\ fertilizer) - N\ (no\ fertilizer)]\ /\ Fertilizer\ N$$

Fertiliser recovery by the crop is 37% (Macdonald et al., 2016).

N_2O emission is estimated to be 1.1 kgN/ha (Grace et al., 2016).

And about 46 kgN/ha is lost as non-GHG N_2 (Rochester, 2003).

Due to the furrow irrigation and the heavy clays that most Australian cotton grows on, nitrogen losses to water can occur around 50–100 kgN/ha (Rochester, 2003).

In the Macdonald team's experiments, about 22 kgN/ha applied nitrogen fertiliser loss is found in the run-off water of the soil profile. At the end of the cotton season, 62 kgN/ha from the fertiliser was retained in the soil (Macdonald et al., 2016).

Table B.4: Cotton Growth N usage

N applied kgN/ha	200 kgN/ha	250 kgN/ha	300 kgN/ha	
Crop absorption	172.4	197.4	222.4	kgN/ha
Soil nitrogen supply	131.5	131.5	131.5	kgN/ha
Loss to irrigation	50	75	100	kgN/ha

N applied kgN/ha	200 kgN/ha	250 kgN/ha	300 kgN/ha	
Loss as N_2	46	46	46	kgN/ha
N loss as N_2O emission	1.1	1.1	1.1	kgN/ha
Loss in air	47.1	47.1	47.1	kgN/ha
Cotton N_2O emission	1.73	1.73	1.73	kgN/ha
CO_2eq of N_2O	535.86	535.86	535.86	CO_2eqkg/ha
Soil fixation	62	62	62	kgN/ha
Total loss in growth	97.1	122.1	147.1	kgN/ha
Cotton growth direct emission	535.86	535.86	535.86	CO_2eqkg/ha
N change in soil	62.00	62.00	62.00	kgN/ha

In Table B.4, activities are labelled using different colours: green means positively contributing to yield; white means no effect on GHG or yield but may still contribute to fertiliser waste; and yellow and orange means emission/pollution. The cotton growth direct emission in Table B.4 is the same as the CO_2eq of N_2O, which means the emission effect in cotton growth is mainly caused by bacteria activity. However, it also shows that the main provider of cotton growth nitrogen is soil nitrogen, but not fertiliser nitrogen. As soil fixation (which may benefit next year) only covers about half of the soil nitrogen, a large amount of applied nitrogen fertiliser is wasted and causes pollution problems.

Add the nitrogen production emission back to calculate the total emission related to waste or pollution. Then the total nitrogen-related ratio of "Wasted vs. Used" for Australian cotton farms is shown on the following Table B.5.

Table B.5: N related emission and pollution

N applied kgN/ha	200 kgN/ha	250 kgN/ha	300 kgN/ha	
Urea production emission	310.43	388.04	465.65	CO_2eqkg/ha
CO_2eq of N_2O	535.86	535.86	535.86	CO_2eqkg/ha
CO_2eq of Total emission	846.29	923.90	1001.51	CO_2eqkg/ha

N applied kgN/ha	200 kgN/ha	250 kgN/ha	300 kgN/ha	
Non-emission Nitrogen loss	124.00	156.00	188.00	kgN/ha
CO_2eq of Total emission (t/ha)	0.85	0.92	1.00	CO_2eqt/ha
Total cotton per farm annual emission (t)	395.22	431.46	467.70	CO_2eqt/ha

Table B.6 shows the compression ratio for the urea applied wasted (lost) to used (absorbed by crop or soil fixation).

Table B.6: "Waste vs Useful" for Australian cotton farms

Total Urea application	434.78 kg/ha	543.48 kg/ha	652.17 kg/ha	
Wasted	271.96	341.52	411.09	kg/ha
used	162.83	201.96	241.09	kg/ha
Ratio (wasted vs used)	1.67	1.69	1.71	Rate

Each nitrogen molecule absorbed by crop and soil fixation bears 1.67 to 1.71 units of nitrogen pollution or wasted debt.

Offset/remediation through tree planting

In this section, the previously discussed modelling will be enhanced by incorporating a singular regulatory option. This is to exemplify the merit of the integrated modelling approach adopted throughout this thesis.

Background of tree planting

One method that has been proposed to address the negative externality of GHG pollution is tree planting, to provide carbon offset (Yosef et al., 2018). This has also been adopted in the Australian regulatory context via the Carbon Credits (Carbon Farming Initiative) Act 2011. However, in recent research, more and more evidence has shown that if people expect to achieve the IPCC 2°C purely on the tree planting offset, the existing tree planting scale and method is less than enough to achieve the goal (Griscom et al., 2017).

Though the photosynthesis of plants is considered to have excellent carbon capture potential, research shows that the ability of carbon capture and oxygen production of algae in the ocean makes a greater contribution than trees (Tkemaladze & Makhashvili, 2016).

In addition, even assuming the tree project is limited to wasteland without any vegetation cover, it is difficult for trees to provide positive emission reduction in the first few years (Yosef et al., 2018).

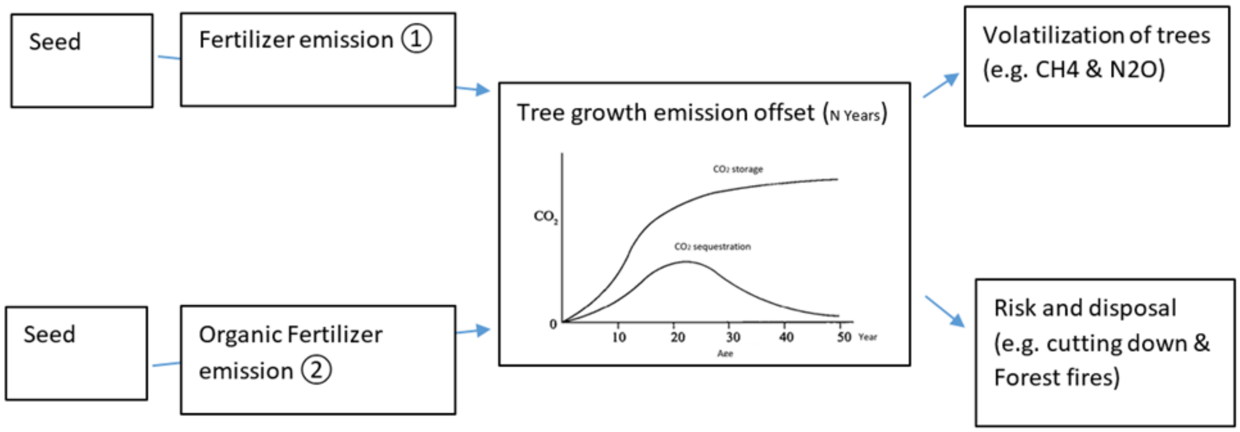
Also, the effectiveness of the tree planting offset is highly dependent on the geographical condition and existing vegetation cover (Yosef et al., 2018). This can be explained in the following three ways.

First, if the aim is to gain carbon credit, there are only narrow paths to achieve this. For example, existing habitat would not be recognised as carbon credits, and only limited commercial tree types can be counted as carbon credits. This creates the concern about destroying existing plants and whether the tree species required by the policy can survive in different geographical environments (Van Kooten et al., 1999).

Second, offsets through tree planting follow Figure B.1. Carbon absorption is commonly recognised as starting from 0 and follows a rapid growth as the seed turns into a tree and then maintains stability. However, if LCA calculates fertiliser (either organic or chemical) and GHG emissions from trees together, the opening balance for this project will be negative offset.

The system of tree planting and emission offset is shown in Figure B.1.

Figure B.1: Emission and tree carbon sequestered quantity



(The relation between tree age and sequestered amount of CO_2 is referenced from Wu, 2015.)

Note: Fertiliser emission includes production emission, transport emission and application waste emission.

Third, another primary benefit provided by afforestation is to protect biodiversity. However, for offset projects, especially commercial offset projects, project managers prefer to plant a single, fast-growing commercial tree species, such as the Blue Mallee Eucalyptus which is the most commonly grown tree in Australia (CO_2 Australia, 2013), rather than restore the natural growth forest with diversity and hierarchy. This will likely lead to higher risks of forest diseases and pests, forest fires, soil degradation and water shortage (Cao et al., 2015). In addition, it has no benefit for biodiversity protection and cannot improve local water and soil conditions.

Much research has been done to study the offset ability of tree planting programs, showing that carbon offsets need at least 30 years to reach the optimum carbon storage level (Iizuka and Tateishi, 2015).

If the farm chooses to offset these emissions through tree planting.

$$Cotton\ farm\ emission\ per\ operating\ year\ =\ X\ ha * CO_2eqt/ha$$
$$=\ 467\ *\ (2.77\ or\ 3.33\ or\ 3.89)$$

15 trees and 25 years to offset one tonne of CO_2eq

$$15\ trees\ in\ 25\ years\ *\ Y\ unit\ =\ X\ ha * CO_2eqt/ha$$

Table B.7 shows the calculation of numbers of trees needed to cover carbon offsets needs of cotton farms, without taking fertiliser production emissions into account.

Table B.7: Trees – no. needed without N fertiliser production emissions

Application option	200 kgN/ha option	250 kgN/ha option	300 kgN/ha option
Year 1 (t)	1,295.92	1,557.34	1,818.76
…	…	…	…
Year 25 (t)	32,397.98	38,933.44	45,468.90
No. of Trees needed	485,970	584,002	682,034
76% of trees survive	639,434	768,423	897,413

Calculation of tree offset project and Possibility for sustainability

For the section without fertiliser production

If the farm chooses to offset through carbon credit, the cost is about $150 per tree to maturity (Taylor, 2012; Koala Clancy Foundation, 2021).

Table B.8: Tree offset cost per farm (old)

Application option	200 kgN/ha option	250 kgN/ha option	300 kgN/ha option
Year 1 (t)	1,295.92	1,557.34	1,818.76
…	…	…	…
Year 25 (t)	32,397.98	38,933.44	45,468.90

Application option	200 kgN/ha option	250 kgN/ha option	300 kgN/ha option
No. of Trees needed	485,969.70	584,001.63	682,033.56
76% of trees survive	639,433.82	768,423.20	897,412.58
Cost/tree	$ 150.00	$ 150.00	$ 150.00
Cost/premature seedling	$ 2.95	$ 2.95	$ 2.95
Cost/farm	$ 73,348,174.07	$ 88,144,287.95	$ 102,940,401.84

Organic or not, all nitrogen fertilisers are considered to produce N_2O emissions (Bouwman et al., 1997) and provide a similar emission amount (Akiyama et al., 2004). However, chemical fertilisers bear more emission debt from the production process.

For the section with fertiliser production (commercial eucalyptus)

Most forestry companies apply about 100 kgN/ha (Laclau et al., 2009).

About 150–200 trees per hectare (Australian Government Clean Energy Regulator, 2014).

The N_2O emission factor of urea is 2.07% (Ibarr et al., 2021).

Fertiliser production emission of tree planting

Table B.9 calculates the emissions for tree planting when using nitrogen fertiliser to assist with tree growth.

Table B.9: Tree offset ability after added N fertiliser production emission

	100 kgN/ha application	
Urea	217.39	kg/ha
Production emission	1,119.57	CO_2eqkg/ha
Tree/ha	175	Tree No./ha
Urea production Emission/tree	6.4	CO_2eqkg/tree

	100 kgN/ha application	
The emission factor of Urea	2.07%	Rate
N_2O Emission/ha	4.5	CO_2eqkg/ha
N_2O Emission/tree	0.03	CO_2eqkg/tree
Total emission/tree	6.43	CO_2eqkg/tree
15 trees 25 years (t)	0.7	CO_2eqt/tree

Because nitrogen fertiliser is only applied in the first 3 years (Milthorpe, Hillan & Nicol, 1994), the emission sequestration of 15 trees in 25 years should be:

$$CO_2\, Sequestration = 1 - (15 * 6.4 * 3 + 15 * 25 * 0.03) / 1000$$

$$= 0.70\, t$$

Thus, after adjustment:

Table B.10: Tree offset cost per farm (new)

Application option	200 kgN/ha	250 kgN/ha	300 kgN/ha
Year 1	1,295.92	1,557.34	1,818.76
…	…	…	…
Year 25	32,397.98	38,933.44	45,468.90
Trees needed	694,242	834,288	974,334
76% of trees survive	913,477	1,097,747	1,282,018
Cost of premature seedling	$ 2.95	$ 2.95	$ 2.95
Cost/tree	$ 150.00	$ 150.00	$ 150.00
Cost/farm	$ 104,783,105.82	$ 125,920,411.36	$ 147,057,716.91

Compared to the profit they earned from cotton

Rochester and Bange (2016) referenced the relationship between nitrogen absorption and yield data.

The cotton market price on 2022/1/14 is 3.55 AUD (2.58 USD)/kg on Market Insider (2021).

Table B.11: AU Cotton farm income

N applied	200 kgN/ha	250 kgN/ha	300 kgN/ha
Crop absorption (kgN/ha)	177	196	221
Yield (kg/ha)	2,550	2,700	2,875
Yield/farm/year	$ 1,190,850.00	$ 1,260,900.00	$ 1,342,625.00
Income/farm/year	$ 4,227,517.50	$ 4,476,195.00	$ 4,766,318.75
Income/farm (accumulate 25 years)	$ 105,687,937.50	$ 111,904,875.00	$ 119,157,968.75

Note: the calculation of 25 years of accumulated income does not consider inflation and time values money.

In Table B.11, the yield level per farm was first calculated based on nitrogen absorption level, then the output was transferred to a dollar amount based on the market price per kg, then accumulated 25 years to compare with the tree offset cost.

Table B.12 presents a comparison of the costs associated with complete emission offsets, employing three distinct offset methods, across a total of 1,500 farms.

Table B.12: Profit and loss for tree offsets choice

Application option	200 kgN/ha	250 kgN/ha	300 kgN/ha
Year 1	1,295.92	1,557.34	1,818.76
…	…	…	…
Year 25	32,397.98	38,933.44	45,468.90
Trees needed	694,242	834,288	974,334
76% of trees survive	913,477	1,097,747	1,282,018

Application option	200 kgN/ha	250 kgN/ha	300 kgN/ha
Premature seedling cost	$ 2.95	$ 2.95	$ 2.95
Cost/tree	$ 150.00	$ 150.00	$ 150.00
Cost/farm	$ 104,783,105.82	$ 125,920,411.36	$ 147,057,716.91
Income per farm	$ 105,687,937.50	$ 111,904,875.00	$ 119,157,968.75
Difference	$ 904,831.68	-$ 14,015,536.36	-$ 27,899,748.16
1,500 farms in AU (Trees No.)	1,370,215,318	1,646,621,134	1,923,026,950
Total Cost	$ 157,174,658,724.15	$ 188,880,617,046.42	$ 220,586,575,368.68
Average cost per year	$ 6,286,986,348.97	$ 7,555,224,681.86	$ 8,823,463,014.75
Average cost/farm/year	$ 4,191,324.23	$ 5,036,816.45	$ 5,882,308.68

From Table B.12, only the “200 kgN/ha application plan” can offset its nitrogen-related emission with a remaining positive balance.

Even the estimation shows cotton farms can offset their nitrogen-related emissions through tree planting, which is one farm's data. There are up to 1,500 cotton farms in Australia (Cotton Australia, 2022). This means that even with the 200 kgN/ha option, to offset the whole industry's nitrogen-related emission, 968,239,286 trees should be planted, kept alive for at least 25 years, and end up in deep earth.

However, with each 50 kgN/ha nitrogen fertiliser reduced, a large amount of environmental cost is saved, yet yield has no significant decline.

Thus, encouraging direct emission reduction might be more beneficial than enabling offset projects.

Sensitivity test

Table B.13 represents additional sensitivity analysis utilizing tree planting cost on a per-tree basis as the variable. The trend of the results aligns with the trends observed in Chapter 4 through the employment of the DSSAT software across three different scenarios, indicating a negative correlation between tree planting cost and ESRI values.

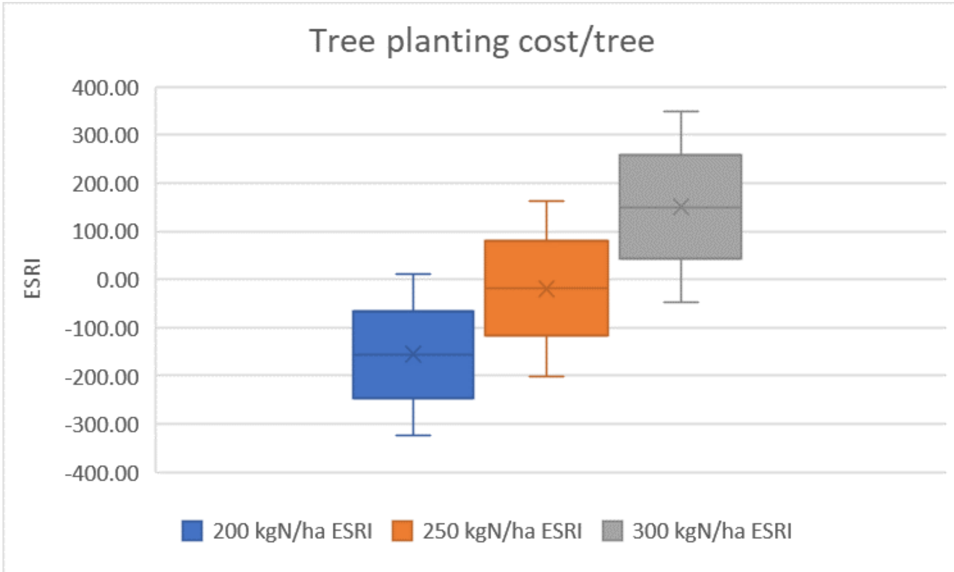
The data presented in Table B.13 illustrates that under a scenario where tree planting is posited as the sole mitigation solution and costs are imposed directly upon cotton growers, those with lower yields due to reduced fertiliser application bear the least capacity to endure the policy, particularly when output remains constant. When production is low (200 kgN/ha) and in the absence of any foundational emission allowances, cotton growers possess negligible capacity to bear the costs of emissions management, with the ESRI value only turning positive when they are either exempt from or subjected to minimal emissions-related expenses. Conversely, high-yielding cotton growers, who are associated with higher emissions, exhibit increased capacity to absorb the policy costs. However, growers across all three levels of fertiliser application find themselves unable to fully shoulder the true relative costs of emissions management, calculated in the previous section as $150/tree.

Shows in Figure B.2, examining the shifts in nitrogen application from 300 to 250 kgN/ha yields the most substantial benefit in terms of ESRI. The benefit diminishes when the application decreases further from 250 to 200 kgN/ha, indicating a scenario where the marginal utility of emissions reduction burden begins to decline. This suggests a diminishing marginal effect, a phenomenon where incremental reductions in emissions result in progressively lesser ESRI benefits.

Table B.13: Tests based on different tree planting cost

Application option	200 kgN/ha	250 kgN/ha	300 kgN/ha	250-200	300-200	300-250
Cost/tree $	ESRI	ESRI	ESRI	Difference	Difference	Difference
0	11.88	163.63	348.50	151.74	336.62	184.88
1	-1.49	149.02	332.68	150.52	334.17	183.65
2	-14.87	134.42	316.85	149.29	331.71	182.43
3	-28.24	119.82	301.02	148.06	329.26	181.20
4	-41.62	105.22	285.19	146.84	326.81	179.97
5	-55.00	90.61	269.36	145.61	324.35	178.75
6	-68.37	76.01	253.53	144.38	321.90	177.52
7	-81.75	61.41	237.70	143.16	319.45	176.29
8	-95.12	46.81	221.87	141.93	317.00	175.07
9	-108.50	32.20	206.04	140.70	314.54	173.84
10	-121.87	17.60	190.21	139.48	312.09	172.61
11	-135.25	3.00	174.39	138.25	309.64	171.39
12	-148.63	-11.60	158.56	137.02	307.18	170.16
13	-162.00	-26.20	142.73	135.80	304.73	168.93
14	-175.38	-40.81	126.90	134.57	302.28	167.71
15	-188.75	-55.41	111.07	133.34	299.82	166.48
16	-202.13	-70.01	95.24	132.12	297.37	165.25
17	-215.50	-84.61	79.41	130.89	294.92	164.03
18	-228.88	-99.22	63.58	129.66	292.46	162.80
19	-242.26	-113.82	47.75	128.44	290.01	161.57
20	-255.63	-128.42	31.93	127.21	287.56	160.35
21	-269.01	-143.02	16.10	125.98	285.10	159.12
22	-282.38	-157.63	0.27	124.76	282.65	157.89
23	-295.76	-172.23	-15.56	123.53	280.20	156.67
24	-309.13	-186.83	-31.39	122.30	277.74	155.44
25	-322.51	-201.43	-47.22	121.08	275.29	154.21
…	…	…	…	…	…	…

Figure B.2: Boxplots for tree planting cost/tree ($) and ESRI value



Appendix C

Figure C.1: Growing season length (S1, S2, S3), year 2010 – 2098

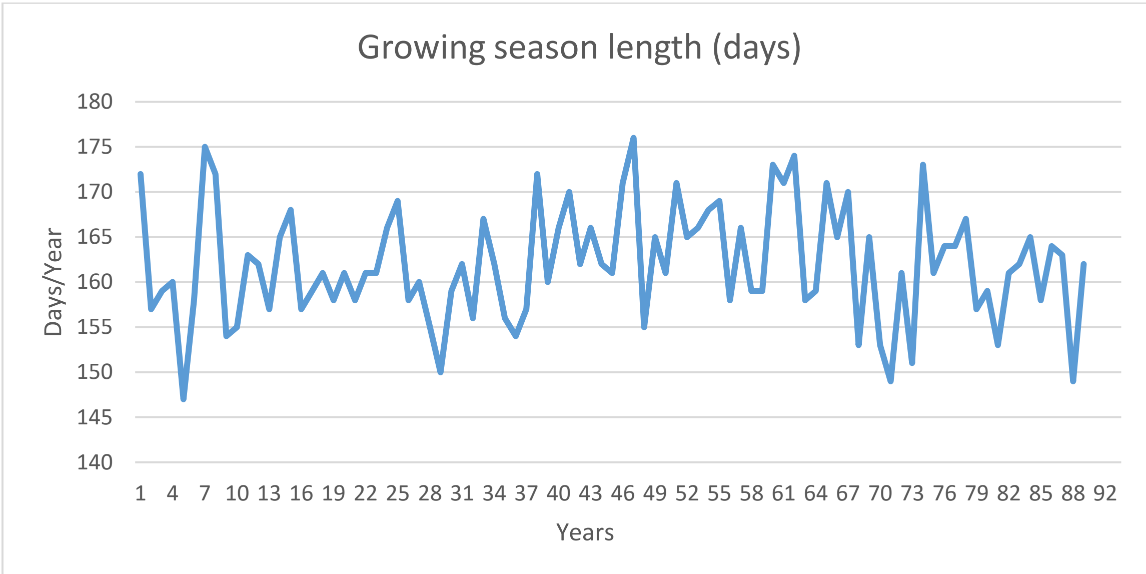


Figure C.2: Irrigation amount (S1, S2, S3), year 2010 – 2098

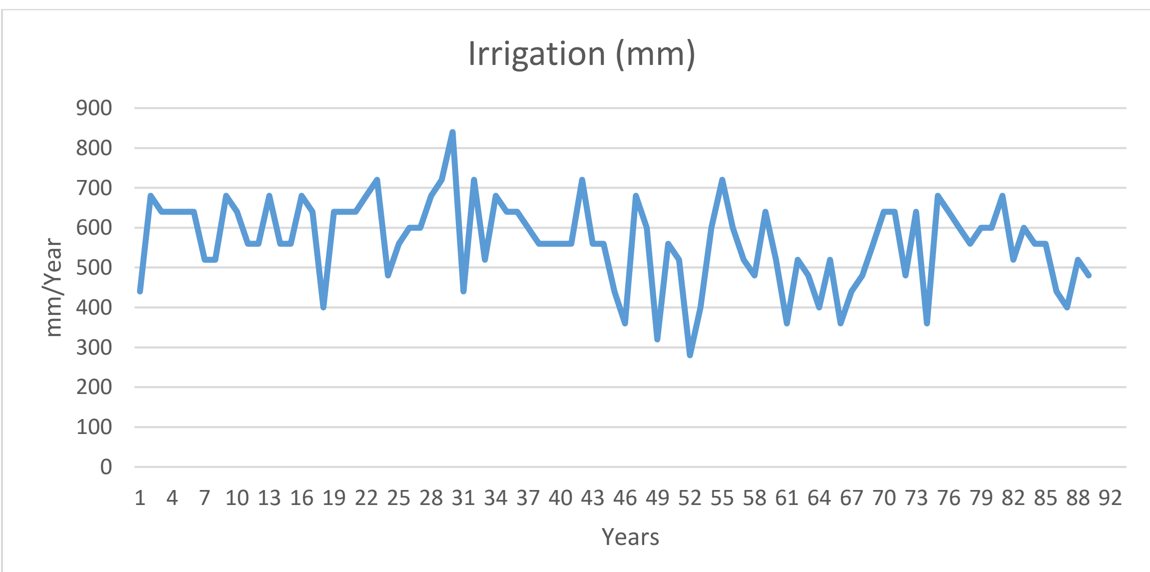
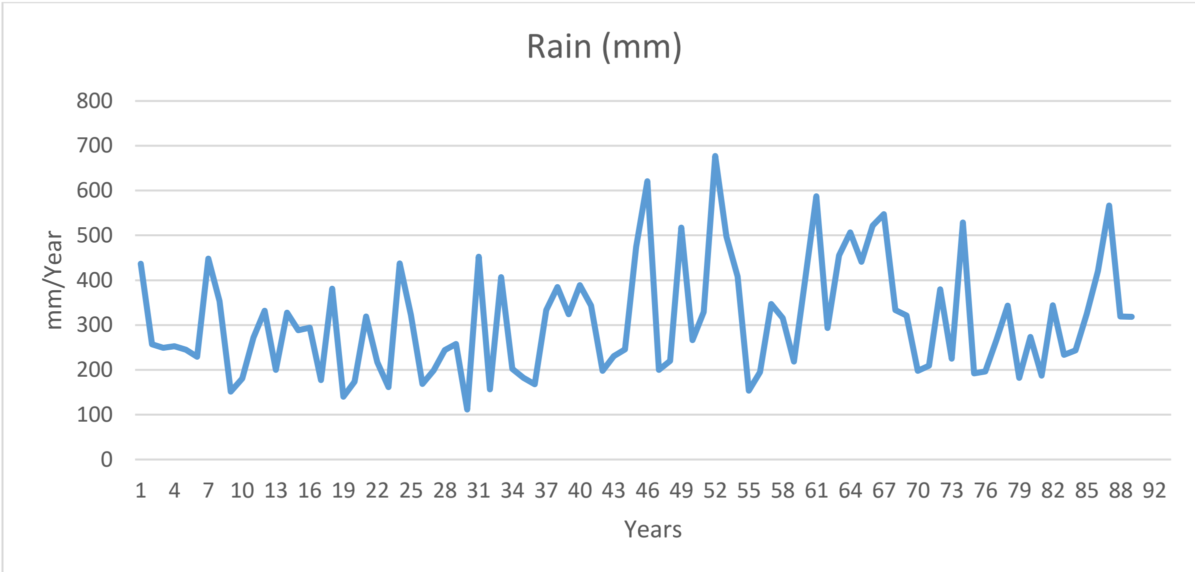


Figure C.3 Raining amount (S1, S2, S3), year 2010 – 2098



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