Journal of Cleaner Production 479 (2024) 143917

Contents lists available at ScienceDirect

Journal of Cleaner Production

journal homepage: www.elsevier.com/locate/jclepro

Towards better understanding the economic and environmental sustainability of alternative agricultural cropping production systems through integrated modelling

Yujie Liang [b], Paul J. Brown [a,b,*], Christopher Bajada [a], Hannah Pham [a]

[a] UTS Business School, University of Technology Sydney, PO Box 123, Broadway, NSW, 2007, Australia
[b] UTS TD School, University of Technology Sydney, PO Box 123, Broadway, NSW, 2007, Australia

ARTICLE INFO

Keywords:
Sustainable production systems
Integrated modelling
Environmental accounting
Agriculture
Nitrogen management
Sustainable development

ABSTRACT

We explore and develop an integrated model to enhance understanding and evaluation of the economic and environmental sustainability performance of alternative agricultural cropping production systems. More specifically, our developed Environmentally Sustainable Residual Income Decomposition Modelling (ESRIDM) integrates accounting decomposition analysis, Water and Economic Sustainability Performance Measurement (WESM), system thinking, modular thinking, Environmentally Sustainable Residual Income (ESRI) theory, into a holistic modelling system influenced by Life Cycle Assessment (LCA) and Life Cycle Costing (LCC) approaches.

The ESRIDM is theorised and applied in the context of nitrogen management in Australian Cotton production through a detailed case study and comparative analysis of three nitrogen fertiliser options. We find the ESRIDM is able to: (i) provide an overall assessment of the economic and environmental sustainability performance of each scenario modelled; (ii) increase the transparency of the production system to enable the identification of key issues which influence the likely suitability of each scenario; (iii) enable an analysis of probable outcomes under various alternatives; (iv) evaluate the feasibility of regulatory option modelled. While this paper offers a novel perspective to a specific context, ESRIDM is designed to be adapted to a variety of contexts with a view to contributing to the evolution of more sustainable systems of production more generally.

1. Introduction

A critical challenge facing humanity in response to climate change is how to adapt our production systems to fit within planetary boundaries. 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 (GHG) emissions (IPCC, 2019). In the specific case of cotton production in Australia, about 60% of farming related GHG emissions are attributable to nitrogen fertiliser (CRDC, 2024, p10), highlighting the salience of nitrogen management decision-making as critical to the emergence of a sustainable production system. Characteristics such as crop rotation strategy, commodity price volatility and fluctuating water supply require assessments and choices be made before the planting season. This implies that growers and other stakeholders need comprehensive systems that can provide timely evaluations based on factors such as 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 consider 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.

The challenge of adapting our production systems to operate within planetary boundaries is further exacerbated by limitations inherent in the design of typical production system performance evaluation systems (PSPES), with modelling typically falling short of comprehensively depicting the extent of environmental degradation (Brown, 2016; Pearce, 1976; Pham et al., 2020; Rambaud and Richard, 2015; Richard, 2012); and therefore are ill-equipped to explain the possible

* Corresponding author. UTS Business School, University of Technology Sydney, PO Box 123, Broadway, NSW, 2007, Australia.
E-mail addresses: Yujie.Liang@uts.edu.au (Y. Liang), Paul.J.Brown@uts.edu.au (P.J. Brown), Chris.bajada@uts.edu.au (C. Bajada), Hannah.Pham@uts.edu.au (H. Pham).

https://doi.org/10.1016/j.jclepro.2024.143917
Received 29 February 2024; Received in revised form 26 September 2024; Accepted 7 October 2024
Available online 16 October 2024

consequences caused by these systems, thus reducing our ability to identify what practices are most likely to be sustainable (Brown and Bajada, 2018; Cucurachi and Suh, 2017; Li, 2020; Pham et al., 2020). This challenge is particularly salient in the context of agriculture production system evaluation. While various models have been developed to shed light on specific aspects of performance, there is little integration. For instance, within the domain of nitrogen management sustainability practices, precision agriculture (Karunathilake et al., 2023; Sishodia et al., 2020) and site-specific management (Plant, 2001; Shaheb et al., 2022) 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.

As regulators respond to the unfolding climate emergency, understanding of the likely role regulatory options may play in supporting the emergence of sustainable production systems is interdependent with the evaluation of the production systems themselves (Jin et al., 2022; Sievert et al., 2022). That is, alternative production system designs will likely vary in their suitability for diffident regulatory options, each of which may take the form of a singular regulation or a set of regulations in combination (Garetti and Taisch, 2012).

Accordingly, the objective of this research is to explore and develop an integrated model to understand and evaluate the economic and environmental performance of alternative production systems, and associated GHG emission regulation options, in the context of agricultural cropping production system sustainability. Additionally, we evaluate the usefulness of the Environmentally Sustainable Residual Income Decomposition Modelling (ESRIDM) as a PSPES to support decision-making for various stakeholders through a case study. The focus of the case study is on-farm nitrogen management and relevant GHG emission regulation options for the Australian cotton industry.

This research offers several theoretical and practical contributions to literature. First, a clear integrated modelling conceptual framework is theorised with a view to assisting decision makers and other stakeholders on identification, measurement, management, and reporting production system sustainable value, including with respect to (i) specific short-term decisions at the organisation level, (ii) regulatory options to address externalities, and (iii) broader analysis of the likely long-term implications at the system level. Second, this research theorises the design of 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. Notably, to the authors knowledge, the theorised integrated modelling approach, ESRIDM, is the first to build on the Environmentally Sustainable Residual Income (ESRI) theory (Brown, 2016) and Pham et al.'s (2020) environmental performance measurement system; in combination with insights from Pearce (1976) and conceptually grounded in the principles of Life Cycle Assessment (LCA) and Life Cycle Costing (LCC). Notably, this approach diverges from the traditional step-by-step adherence to these methodologies by selectively integrating relevant and material elements from both LCA and LCC, adapting them within a broader, more holistic framework with a view to maximising decision usefulness and reducing the cost of estimation. Third, this study derives and articulates a decomposition model to enable stakeholders to better understand and explain performance in the model context, drawing inspiration from the approaches utilised in the evaluation and valuation of companies and other financial investments (Penman, 2003). Finally, to evaluate the validity of the modelling this paper gives a demonstration of the usefulness of the integrated modelling framework, using decisions about nitrogen application rates for cotton farms and one regulatory option, namely tree offsets as suggested by Australian regulators.

This paper is structured as follows: Section 2 provides a literature review and theory development; Section 3 describes the model design, mapping, and formula, followed by a case study to demonstrate how ESRIDM can be used in Section 4; Section 5 presents the findings and Section 6 concludes.

2. Theory development

In this section, we theorise the design of a conceptual framework for an ESRIDM integrated model dedicated to sustainable production, with its architecture grounded in and employing a synthesis of various guiding theoretical principles and extant modelling systems. Fig. 1 illustrates the theory structure of the developed ESRIDM, with key constructs theorised below. Simply put, the framework starts with identification of alternative production and regulation options, which are modelled together and translated into LCA and LCC data. This data is then available for analysis aided by Penman decomposition to create causally linked performance indicators at varying levels of decomposition, which culminate in a high-level indicator of the overall environmental and economic sustainability.

2.1. Sustainability framework

The departure point of the modelling is the principle that human activities should not exceed ecological safety thresholds and fundamental limits (e.g., Pearce, 1976). While we acknowledge that sustainability encompasses three integral facets: economic, social, and environmental development, we delimitate the modelling by primarily focusing on economic and environmental sustainability.

Arguably, sustainability requires adherence to at least two conditions. First, ecological sustainability (or strong sustainability if one so prefers), for it highlights the need to distinguish between natural and other capital resources; as over the long-term economic development should occur within the limits of natural resource availability and regenerative capacity (Hediger, 1999; Málovics et al., 2008; Pearce, 1998; Pham et al., 2020). Consequently, this implies an "ecological value principle" that recognises and quantifies the overall "value" of the diverse array of natural capital from an ecosystem perspective (Daly, 1991; Hediger, 1998; Pearce, 1996). This offers a theoretical framing that acknowledges the uniqueness and irreplaceability of natural capital, aligning with Pearce's (1976) proposition of a threshold for nature's resilience that is contingent of the characteristics of the specific capital under consideration.

Second, in addition to environmental aspects, sustainability should also incorporate economic and social dimensions to facilitate a more comprehensive approach (Alhaddi, 2015; Kleindorfer et al., 2005). At the economy level, economic sustainability refers to the capability of the

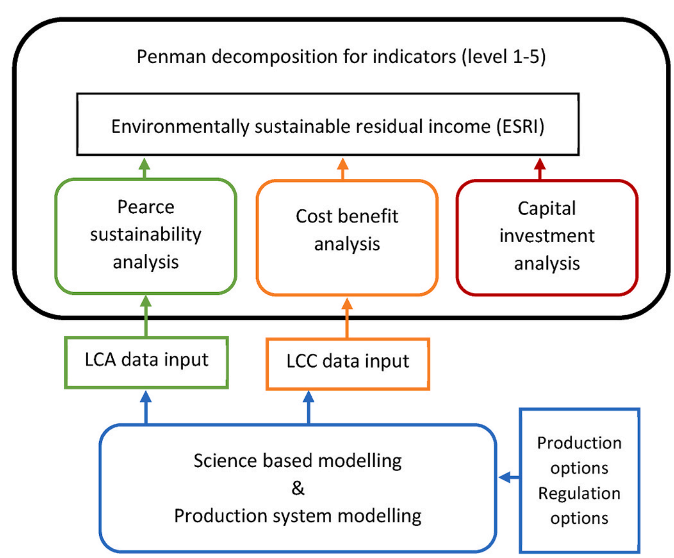

Fig. 1. Conceptual framework and theory structure for ESRIDM.

broader economic system to survive and evolve into the future to support current and future generations (Spangenberg, 2005). With respect to the organisational level, economic sustainability refers to the ability of the organisation to effectively manage resources to enable sufficient profitability to cover capital costs (Penman, 2003); and its capacity to adapt to evolving conditions such as fluctuating input and output prices as well as changes in policy landscapes (Doane and MacGillivray, 2001; Spangenberg, 2005). Penman (2003) explains that for financiers to assess overall economic sustainability, the performance indicator for profitability (ability to cover costs) should include a charge for opportunity costs of capital rather than solely profit. Where investments can create profit in excess of capital costs, they create Residual Income (otherwise known as Abnormal Earnings, Economic Value, or Economic Value Add), and those which do not are devalued.

While not the focus of this study, we note that social sustainability has been characterised as a society with effective feedback loops between stakeholders and decision-makers, where opinions, needs and improvements can be listened and respond to in a system characterized by inclusiveness, justice, and resilience (Missimer et al., 2017; Vallance et al., 2011; World Bank, 2020).

The ESRI theory offers a comprehensive perspective on the evaluation of critical economic and environmental sustainability dimensions by integrating assessments of the economic value of the environment and environmental degradation caused by economic activity (Brown, 2016). Brown (2016) defines ESRI as "a measure of whether an investment has the capacity to deliver a return on investment which is sufficient to cover the demands of financiers, as well as the costs of replacement or the cost of restoration of environmental functions" (p. 153). ESRI aims to create a more complete estimate of the economic value of a production system (e.g., an organisation's operations) by accounting for environmental costs and benefits, irrespective of whether the cost is externalised. In principle, by including environment sustaining costs, ESRI is informative about the value of a given production systems in the context of the system it is embedded in. The core formula of ESRI is:

$$ESRI = Net\ Profit - (R_e * Book\ Value\ of\ Owners\ Equity) - Environment\ sustaining\ cost \tag{1}$$

Where: net profit is the value remaining from revenue once expenses are deducted. Environment sustaining cost is defined as an "opportunity costs generated by the organisation's activities; being an estimate of the cost to replace or restore natural capital degraded in earning the income" (Brown, 2016, p. 153). R_e is an estimate of the required return of equity, or opportunity cost of equity.

ESRI is conceptualised as Residual Income (Net Profit subtracted by a capital charge to assess economic sustainability), less an estimate of environmental sustaining costs (Brown, 2016). Originating from the theory of sustainable national income (SNI), this approach emphasizes the integration of economic growth with environmental value assessment (Hueting, 1993). Richard (2012) innovatively merged this concept with the "triple bottom line", thereby pioneering advancements in the field of sustainable accounting (Rambaud and Richard, 2015; Richard, 2012). Brown (2016) expanded on Richard's (2012) framework and integrates Pearce (1976), factoring capital value into the concepts of environmentally sustainable national income (eSNI) in a more flexible way focused on decision-makers. This enhancement applies an opportunity cost to enable the analysis of environmental value changes and assess risk in the context of scarcity, bringing environmental burden into the realm of economic analysis (Hueting, 1993).

Pham et al. (2020) also recognised the value and necessity of environmental remediation and attempt to merge them into the integrated modelling system of Water and Economic Sustainability Performance Measurement (WESM). This study builds upon Pham et al. (2020) and extends 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. ESRIDM extents articulations to include nitrogen fertiliser and other material production emission and on-farm nitrogen management, alongside water management in Pham et al. (2020). In addition, as presented below, the ESRIDM decomposition model reflects ESRI value as the highest-level indicator instead of the efficiency-focused ratios in Pham et al. (2020).

2.2. Modularization thinking in production system sustainability evaluation

One of the theoretical constructs implicit in our ESRIDM is systems thinking, whereby the performance of a system is generated through the interactions among its components, rather than solely through the functions of individual parts (Hekkert et al., 2007; Malerba, 2002). Environmental and economic factors, alongside regulatory and decision-making processes, are crucial components that mutually influence one another as they exist interpedently as part of systems.[1] The design of the integrated model is thus aimed at enabling the observation of the interplay between these components and their collective impact on overall performance.

Modular thinking has gained widespread adoption in production design since Starr (1965), primarily as a strategic approach to improve efficiency while decomposing and organizing complex designs and processes (Salhieh and Kamrani, 2008). Despite the growing integration in facilitating sustainable improvement (Mesa et al., 2020; Mutingi et al., 2017; Sonego et al., 2018), there is little exploration of such designs within the framework of sustainable production system evaluation (Sonego et al., 2018).

Furthermore, for ESRIDM, the modular design allows users to selectively focus on specific modules, as each module has functionality within the scope of the analysis. This aspect is pivotal as it establishes a framework for the integration of economic analysis tools (LCC[2]) with environmental impact analysis tools (LCA[3]) within the ESRIDM approach, and facilities the inclusion of additional modules to suit the needs of users. The widespread adoption of both methods in analysis and measurement within the field of sustainable production has garnered significant interest in their integration (Heijungs et al., 2013; Hou et al., 2022; Santos et al., 2019), yet the actual implementation of this integration faces numerous challenges; such as disparities in the required data scope and depth, incongruities in the coupling and conversion of measurement units, and mismatches in the timing of production activities (De Luca et al., 2017; França et al., 2021; Heijungs et al., 2013). We invoke the use of a Penman (2003) styled decomposition method to address these challenges, as argued below.

In summary, we contend that it is desirable to enable flexible coupling between LCA and LCC utilising a decomposition method, as it

[1] The concept that the modular nature of elements within and between systems jointly construct the sustainability of a given society has some resonance with the concept of ecological civilisation, which implies a role for consideration of political factors (Chen and Shi, 2022; Wei et al., 2021; Xue et al., 2023). It is beyond the scope of the current paper to explore this aspect.

[2] The concept of LCC offers a structural basis for methods of cost management (Blanchard, 1978; Swarr et al., 2011) and is extensively applied within the realm of sustainability topics, particularly in the context of cost savings (cf. Gorjian et al., 2022; Woodward, 1997; Zou et al., 2019).

[3] LCA is governed by standards set by the International Organization for Standardization (ISO). A salient advantage of its design lies in the flexibility it offers for application, which empowers stakeholders to evaluate the environmental impacts associated with emerging technologies. In the agriculture sector, researchers have used LCA to analyse the new challenges brought by sustainability requirements (cf. Matos and 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., 2022), and provide decision-making information on sustainability (cf. De Backer et al., 2009; Hasler et al., 2015).

permits their interconnection or replacement as needed by users while maintaining the independence and tractability of the overall system. This approach underscores the benefits of coupling flexibility and modularization, circumventing the constraints imposed by the stringent one-to-one correspondence requirements of strong coupling.

2.3. System thinking and Penman decomposition method

One of the key methods developed in accounting and finance to help understand and explain the links between organisations and the broader systems they are situated within is the Penman (2003) decomposition method. While it was derived from the widely used DuPont decomposition model which identifies and explains the causal links between performance measures at different levels within an organisation, the Penman (2003) decomposition method was developed to enable the evaluation and valuation of companies and other financial investments. Further, its flexibility has enabled it to be extended in a variety of contexts (e.g., Pham et al.'s (2020) WESM cotton farm water management modelling which focuses on water sustainability) to support managers in better sustainability-related decision making and evaluation.

Penman (2003) intricately connects the financial and accounting aspects of an organisation, employing a methodical deconstruction of accounting and financial values, ratios, and their foundational units at each level to analyse and elucidate the determinants of operating and financial performance, and its intrinsic value given the system level context. While Pham et al. (2020) incorporated the Penman decomposition method into sustainable production systems, their choice of resource use efficiency as the highest-level indicator exposes the method to the risks associated with a Jevons Paradox at a systems level (Siami and Winter, 2021).

The integration of the Penman decomposition method as a primary framework in ESRIDM is justified for several reasons: (i) It dissects an enterprise's operations based on production activities, aligning closely with the principles of LCC and LCA. (ii) the layered structure of the Penman decomposition method aligns with systemic thinking, where high-level indicators present an overarching view of performance metrics, while the associated lower-level indicators and formulas provide a comprehensive explanation of the causes and primary drivers of these high-level indicators.

Open or closed, systems thinking emphasizes the interconnectedness of various elements within a system (Arnold and Wade, 2015; Colchester, 2016; Schlüter et al., 2023). This encompasses the causal relationships and feedback loops between elements, and elements with the whole system, as well as the system's adaptability to continuous change (Folke et al., 2002; Monat and Gannon, 2015). Additionally, it involves the dynamic stability or equilibrium of the system as a whole (Arnold and Wade, 2015; Schlüter et al., 2023). These characteristics reflect the inherent complexity and challenge in predictability, necessitating ongoing learning and adaptation, as well as long-term planning (Arnold and Wade, 2015; Kopainsky et al., 2011; Monat and Gannon, 2015).

A significant illustration of the use of systemic thinking in the context of sustainable and production systems is the close relationship between circular economy, environmental management, and systems thinking (Seiffert and Loch, 2005; Williams et al., 2017; Zhang, 2019). Nevertheless, efforts to combine this approach with modular design and Penman's (2003) decomposition theory have been notably absent.

3. Model development

3.1. Model functions

In this section we develop a specific ESRIDM model based on the conceptual framework and theory structure developed in the previous section, as presented in Fig. 2. The decomposition model presented herein depicts a comprehensive overview of the potential interconnections between farm operating level activities and changes in high-level indicators. 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) and hence is the focus of the model structure.[4] The model serves as a tool for decision makers in identifying and tracking the specific steps or processes that have the greatest potential to make a significant impact to production system sustainability. ESRIDM formula for the top-level indicator is:

$$ESRI = NOPAT - ESC - (R_{wacc} \times NOA) \tag{2}$$

where NOPAT is net operation profit after tax, WACC is weighted average cost of capital[5] and NOA is net operating asset.

As illustrated in Fig. 2, ESRIDM separates the high-level indicators of ESRI into six sub-modules: (1) farm operation accounting module, (2) farm performance capital module, (3) nitrogen management module, (4) water management module, (5) environment sustaining cost (ESC) module, and (6) 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 emissions). The ESC module includes the LCA emission of nitrogen fertiliser production cost, farm operating pollution cost and any other environmental remediation cost.

Fig. 3 presents the decomposition, focusing on enabling the estimation and analysis of operational activities of a typical cotton farm and is designed to provide insights into economic and environmental performance in the context of cotton production.[6] Taking this approach may assist stakeholders to optimise their decision-making processes and improve overall farm performance from both economic and

[4] Emissions from other activities, such as use of diesel and lime, are assumed to be identical across the three scenarios of differing nitrogen fertiliser application levels. This is because diesel consumption depends on irrigation choices and farm size, while lime use is influenced by soil conditions and farm size, both of which are consistent across the scenarios in this case study. Previous research has identified nitrogen-related emissions as the primary differentiator, with other emissions (e.g. soil carbon and methane) being negligible in relation to nitrogen application decisions (Russell et al., 2009). Accordingly, these variables are held constant in this study and could, in principle, be excluded. However, they are included here to provide readers with a sense of the relative sustainability impact of the cropping systems. Future studies may wish to evaluate a broader range of environmental degradation factors.

[5] The formula for R_{wacc} is the WACC formula widely used in corporate finance, shown 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. 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.

[6] It is important to note that several agricultural practices have not been included in the modelling due to delimitation, such as crop rotation, drip and spray irrigation, pest control, and weed management. This is because the only one decision (the quantity of nitrogen fertiliser application) and one regulatory option are modelled. It is beyond the scope of this study to estimate all combinations of production system design, with similar delimitations reflected in other integrated modelling decision-making studies such as Leip et al. (2008), Pham et al. (2020) and Wolf et al. (2003). The modelling and example are readily adapted to a range of alternative specifications. For example, the effect of crop rotation can be added to provide a more complex farming system. Introducing measures like crop rotation at the preliminary analysis stage would complicate the analysis without decisively influencing the outcomes.

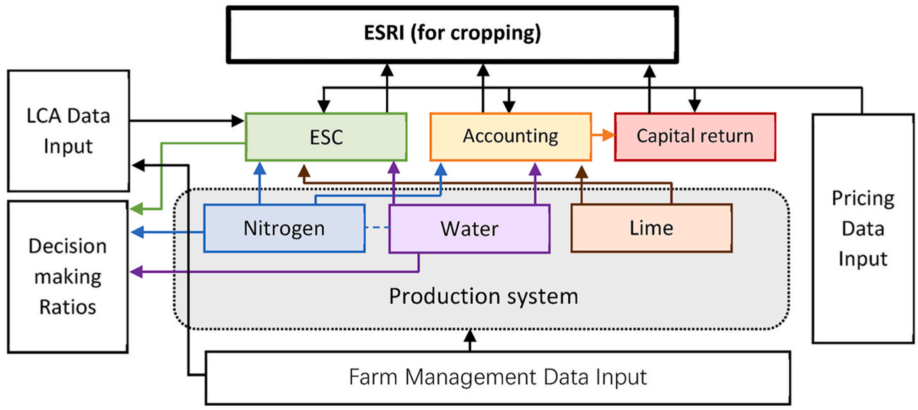

Fig. 2. Structure and scope for developed ESRIDM for cropping.

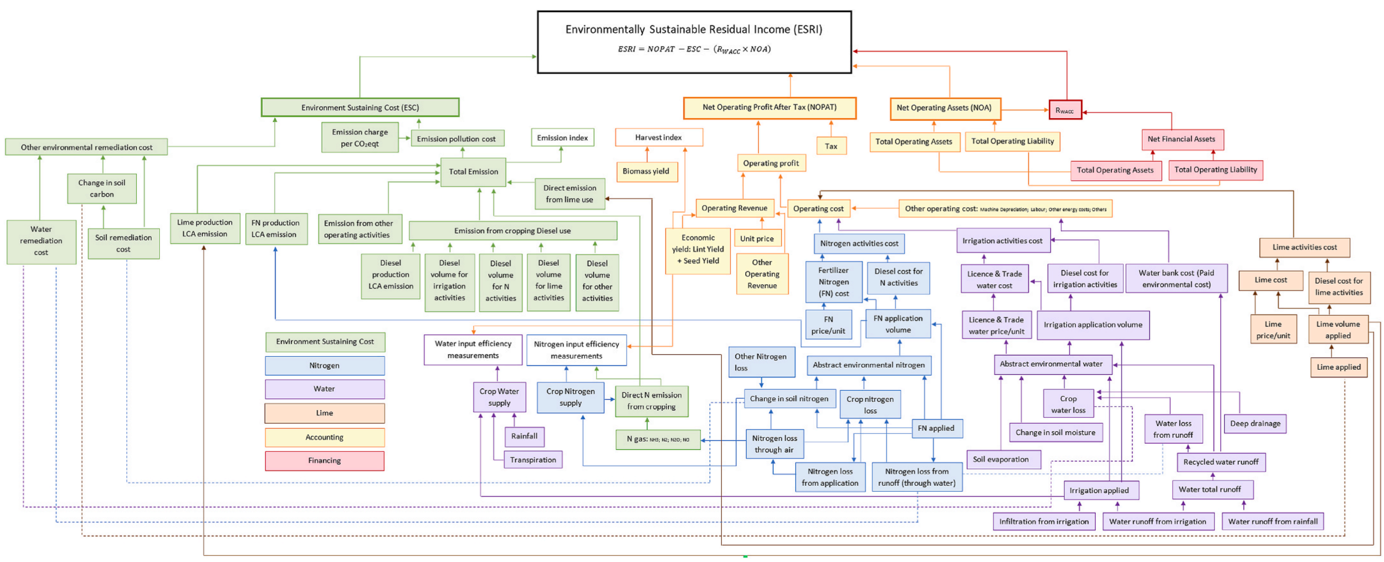

Fig. 3. Decomposition mapping of ESRIDM.

sustainability perspectives, by providing the final output ERSI and step-by-step outputs in different modules showing the causal links between inputs, activities and summary performance indicators. Detailed modelling explanations, and formulas are contained in the supplementary material A.

3.2. Modelling explanation

Table 1 gives an overview of the levels and indicators in ESRIDM, with the highest level 1 indicator being ESRI. 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 at a given level.

4. Case demonstration

4.1. Case context and method

The case study provides an opportunity to evaluate the model's validity by estimating one decision option and one regulatory option utilising publicly available data. Validity in this context is the extent to which the model corresponds to what would be observed in a practical setting (Pham et al., 2020). The case study is of an average sized irrigated cotton farm of 467 ha (Cotton Australia, 2022). The grower faces the choice between three scenarios, with varying amounts of nitrogen fertiliser (urea) application, namely, 200 kgN/ha (scenario 1), 250 kgN/ha (scenario 2) and 300 kgN/ha (scenario 3) (Rochester, 2011). The regulatory option modelled is based on the grower fully offsetting their incremental GHG emissions through a tree planting project. This regulatory option is inspired by one of the preferred options proposed in the Australian Carbon Credits (Carbon Farming Initiative) Act 2011, namely the CO2 Australia blue eucalyptus planting project.[7]

4.2. Data and assumptions

Data is sourced from published studies and authors' estimation. The key sources are from CRDC (2019), Department of Climate Change (2023), Kumar et al. (2021), Laveglia et al. (2022), Macdonald et al. (2016), Pham et al. (2020), Rochester (2003), Rochester and Bange (2016), Sevenster et al. (2024) and Yang et al. (2017). The value of the farm is estimated as 13,793.10 $/ha based on a review of typical cotton farm market price, WACC is assumed as 6% (ABARES, 2023).

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

Table 1
Summary of key elements at each level of the decomposition mapping.

Decomposition Level	Key elements for each level			
1	**Estimated ESRI value**			
2	ESC	NOPAT	NOA	Rwacc
3	Emissions pollution cost Other environmental remediation cost	Net operating profit after tax Total operating assets Total operating liabilities	Total operating assets Total operating liabilities	
4	Total emissions Other pollution	Operating revenue Other operating revenue Operating cost Other operating cost		
5	Emissions from operating activities (include production emission) Emissions from other operating activities (include production emission)	Economic yield Cost of Energy Nitrogen Application (N) Cost of Energy Water (W) Cost of Energy Lime (L) Cost of Nitrogen Fertiliser (N) Seed cost Irrigation cost Lime cost		

Note: (i) For NOA in this case study, to simplify the calculation, an assumption of cotton farm market value is used instead of a direct measurement. (ii) In addition, additional ratios in ESRIDM are designed to evaluate bespoke performance dimensions. Thus, the ratios may be treated independently of levels and maybe be adjusted to fit a variety of contexts.

As the water and irrigation module is adapted from Pham et al. (2020), in this case study the water module is simplified; focusing 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 et al., 2020), diesel usage volume (Sevenster et al., 2024), and water licence market price (ABARES, 2022). Additional details and working are presented in the supplementary material section B.

4.3. Key findings of the case study

Table 2 presents the estimated values for each key indicator by level.

The value of the top-level indicator (the ESRI value) for all three scenarios are negative. This indicates the net economic value (NOPAT) generated from production is insufficient to cover the cost of environmental degradation caused (ESC) under the conditions modelled. Accordingly, it is apparent that the existing agricultural cropping practices presented in Table 2 are not sustainable, particularly when considering the emissions stemming from the supply of raw materials and the generation of production waste, such as nitrogen leakage and pollution. An absence of surplus value from cotton production would imply that environmental remediation is unlikely to be easily resolved without regulatory intervention of the production systems to reduce emissions.

To extract further insights into the key drivers (e.g., lower-level indicators) of ESRI, the second level of analysis is presented in Table 2. While NOPAT component provides a cost analysis from an accounting perspective, the ESC is elevated due to various underlying factors (e.g., emissions from operating activities, emissions from fertiliser production and emission offset cost).

When comparing the scenarios, NOPAT is highest under scenario 3, with \$1395.47/ha being a 47% higher profitability than scenario 1, reflecting the higher yield from additional nitrogen fertiliser application. While ESRI suggests that scenario 1 is the most sustainable under the choice set, focusing on performance indicators which excludes externalised costs favours the scenario which is least sustainable, highlighting the necessity for more comprehensive performance measurement of the performance of production systems.

Notably, the increase in ESC is substantial, representing an increase of 204%, to \$8829.35 for scenario 3. 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 offsets, 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. Remediation costs represent our estimate of the cost associated with the tree planting regulatory option modelled, and the results indicate there is insufficient surplus value from the typical cotton farm to cover these costs. This suggests that such a regulatory option is unlikely to be successful 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 are additional challenges associated with this regulatory option, such as the quantum of land available for production because of tree planting for remediation (see supplementary material B.2).

The lower-level indicators provide additional insights into the drivers and magnitude of overall differences in performance. For example, Level 4 in Table 2 presents total revenue (total economic yield) and total emissions, which are used to estimate the emission index (presented in the Additional ratios in Table 2). The emission index shows noticeable differences between scenario 1, scenario 2 and scenario 3 (0.19 CO_2ekg/\$, 0.24 CO_2ekg/\$ and 0.52 CO_2ekg/\$ respectively. This highlights a turning point in the diminishing marginal returns of ESC, where similar economic yields (\$330/ha between scenario 1, scenario 2, and \$385/ha between scenario 2, scenario 3) are achieved at a greater ESC (\$997.13/ha between scenario 1, scenario 2, and \$4930.83/ha between scenario 2, scenario 3).

To maximise the mitigation of GHG emissions, it is necessary to consider not only the project's design but also the multiple indicators, stakeholders, and the external system level contexts. Furthermore, within this case study, several concerning findings have emerged that warrant further investigation. Notably, 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. Modelling allows for the estimation of a break-even emission price for growers (estimated as NOPAT/Total emissions), which ranges between \$0.38 to \$0.78/CO_2ekg (see supplement information Table A.7). If the market emission charge falls within this price range, it will advantage those growers contributing lower levels of emissions (for instance, a grower applying 200 kgN/ha would generate less ESC compared to one applying 300 kgN/ha, though the latter presents a higher NOPAT currently). Consequently, inadvertently rewarding those who pollute more and penalising those who pollute less. In this 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 a 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 efficiently.

In Table 2, the production of nitrogen fertiliser results in emissions of 310.43, 388.04, and 465.65 CO_2ekg/ha respectively (level 5, Table 2), which are a substantial component of total emissions. Accordingly,

Table 2
ESRI value for tree offsets plan.

Decom-position level	Indicator or element	Data Type	Scenario 1 (200kgN/ha)	Scenario 2 (250kgN/ha)	Scenario 3 (300kgN/ha)	Units
1	ESRI	C	−2777.89	−3573.45	−8261.47	$/ha
2	ESC	C	2901.38	3898.52	8829.35	$/ha
	NOPAT	C	951.08	1152.65	1395.47	$/ha
	Cost of Capital (NOA * Rwacc)	C	827.59	827.59	827.59	$/ha
3	Emission pollution cost	C	2901.38	3898.52	8829.35	$/ha
	Other environmental remediation cost	A	–	–	–	$/ha
	NOP	C	1268.11	1536.87	1860.62	$/ha
	Tax (25% tax rate)	C	317.03	384.22	465.16	$/ha
	Total operating assets	C	13,793.10	13,793.10	13,793.10	$/ha
	Total operating liabilities	A	–	–	–	$/ha
4	Total emission	C	1218.04	1636.65	3706.67	CO_2ekg/ha
	Other pollution	A	–	–	–	CO_2ekg/ha
	Operating revenue	C	6367.24	6697.24	7082.24	$/ha
	Other operating revenue	A	–	–	–	$/ha
	Operating cost	C	2182.82	2244.06	2305.31	$/ha
	Other operating cost	C	2916.31	2916.31	2916.31	$/ha
5	Fertiliser nitrogen production emission	M	310.43	388.04	465.65	CO_2ekg/ha
	Lime production emission	M	47.84	47.84	47.84	CO_2ekg/ha
	Diesel production emission	M	100.17	100.17	100.17	CO_2ekg/ha
	Direct emission from cropping (nitrogen fertiliser and lime)	M	349.04	690.04	2682.45	CO_2ekg/ha
	Direct emission from cropping (diesel use)	M	410.72	410.72	410.72	CO_2ekg/ha
	Emission from other operating activities	A	–	–	–	CO_2ekg/ha
	Economic yield	C	6367.24	6697.24	7082.24	$/ha
	Lint yield	M	2550	2700	2875	kg/ha
	Seed yield	M	2704.42	2704.42	2704.42	kg/ha
	Total fertiliser nitrogen cost	M	244.98	306.22	367.47	$/ha
	Total water cost	M	751.41	751.41	751.41	$/ha
	Lime cost	M	663.30	663.30	663.30	$/ha
	Energy (diesel) cost	M	394.13	394.13	394.13	$/ha
	Seed cost	M	129	129	129	$/ha
Additional ratios estimated	Emission index	C	0.19	0.24	0.52	CO_2ekg/$
	Nitrogen input efficiency	C	51.65%	51.04%	49.50%	Rate

Note: i) ESRI (level 1), ESC, NOPAT, Cost of Capital (level 2). Rwacc is assumed as 6%. (ii) In Table 2, 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 data of this case study is utilised from publicly available official sources or published journals. See detailed data source references and calculations in the supplementary material B. (iii) The decomposition level represents the level of granularity, starting from the high-level indicator ESRI at level 1, which is decomposed into ESC, NOPAT and Cost of Capital at level 2, and with increasing level of decomposition through to level 5. Two additional ratios are estimated to reflect indicators which do not easily fit into a liner decomposition. (iv) In Table 2, the Emission index = Total emissions/Economic yield and Nitrogen input efficiency = Nitrogen uptake by crops/Nitrogen fertiliser applied.

higher application rates of nitrogen lead to greater negative externalities through its supply chain through production. This implies that producers bear the responsibility of considering both their level of inputs and the choice of their suppliers.

In addition, the utilisation of ratios such as the nitrogen input efficiency rate and the water input efficiency rate present certain limitations in this context, where scenario 3 has similar nitrogen input efficiency (49.5%) than the other two scenarios (51.65% and 51.04% for scenario 1 and 2 respectively) but significantly higher nitrogen emissions (3706.67 CO_2ekg/ha) in Table 2 (1218.04 and 1636.65 CO_2ekg/ha for scenarios 1 and 2). These ratios overlook the aspect of scale and primarily indicate the level of efficiency within an organisation, without providing more insights into underlying mechanisms that drive such efficiency levels or establishing connections to other components of the system, particularly within the agricultural sector. This highlights the value of the Penman decomposition approach of presenting salient indicators which are interconnected to provide a more holistic perspective.

5. Discussion

In this section, we provide a discussion on potential usage of ESRIDM and decision making for sustainable production system.

5.1. Economic analysis of environmental externalities

Fig. 4 is a simplified illustration of the relation between farm management outcomes and corresponding environmental cost changes.

In Fig. 4, 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 corresponding to the same fertiliser application level with Q1. The environmental costs include nitrogen oxide emissions, fertiliser production emissions, water eutrophication caused by nitrogen loss, soil quality decline and/or salinisation caused by excessive fertilisation. When the application of nitrogen in Fig. 1 increases from 200 to 300 kgN/ha (c0 to c8), yield increases, so that cost per output for farmers decreases. Increasing the application from 330 to 350 kgN/ha, the nitrogen absorption capacity of crops reaches saturation. In

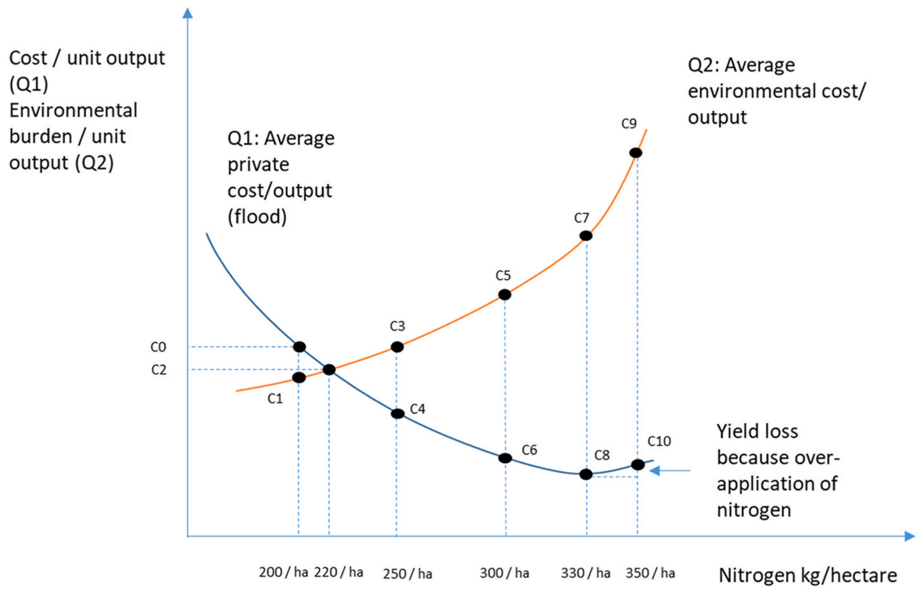

Fig. 4. Motivation of nitrogen fertiliser application and over application.

addition, overapplication may cause problems, such as late ripening of cotton balls and damage to fibre quality, shown by increasing costs (c8 to c10).

However, cotton farms choose to apply over 350 kgN/ha, even up to 500 kgN/ha because the costs of nitrogen fertiliser and water licenses are relatively low, the damage to yield and fibre is minimal, and farms are not required to remediate the externalised environmental degradation (difference from c9 to c10) resulting in significant environmental externalities. As these damages accumulate from c1 to c9, the net growth continues to accumulate at a faster rate with the increasing application of fertiliser (Pearce, 1976)

Fig. 5 shows how farmland annual output (curve F) and consumed assimilative capacity (total) (curve S) changes over time. In the modelling of emissions from IPCC (2021, p. 13), the emission of nitrous oxide is expected to increase, or in the most optimistic forecasts, result in a slight decrease.

Traditionally, agriculture practices (A0 and the area to its left in Fig. 5) represent slow yield growth and minimal environmental burden. With the introduction of chemical fertilisers and mechanical tools, yields begin to increase rapidly from A0 to A1, but so does the environmental burden due to higher chemical and fossil fuel usage. From A1 to A2, crop absorption capacity nears saturation, and further increases in fertiliser use lead to diminishing returns in yield and exacerbated environmental problems such as soil salinisation and water pollution.

As environmental damage becomes apparent and costs rise (A2 to A3), regulations and public pressure for sustainable practices increase. Some growers start adopting emission reduction measures, though economic incentives still favour pollutive practices unless regulated.

Left unchecked, severe environmental degradation leads to a decline in land quality and agricultural viability, pushing growers towards more

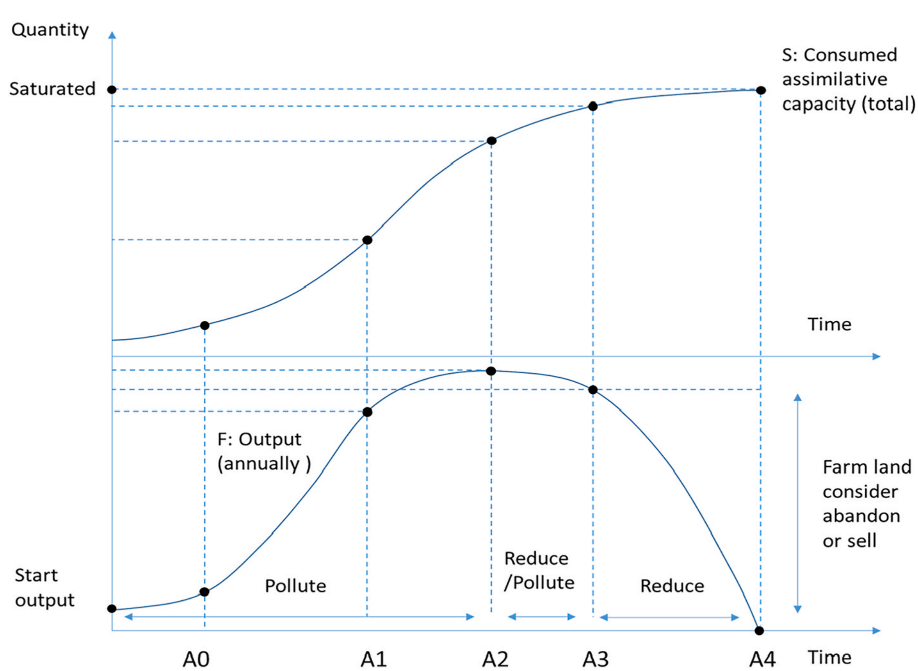

Fig. 5. Projection of relation between traditional agriculture production and consumed assimilative capacity (total) over the long-term.

sustainable practices (A3 to A4). However, dramatic recovery efforts are contingent on breakthroughs in agricultural technology or practices that can adapt to deteriorated conditions.

Theoretically, the best opportunity to reduce waste/pollution is from A0 to A1 before any significant damages have been realised. However, since the economic margins are high (from A0 to A1), the reduction/offset option is more likely to take off from A1 to A2 and is primarily accepted by audiences from A2 to A3.

5.2. To what extent is the model generalisable?

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 and Bryden, 2013). Modules in this modelling may be applied to different cropping systems with different nitrogen level input and irrigation choice.

Second, the nitrogen and water modulus are based on scientifically sound models published in reputable journals (e. g., Rochester, 2003, 2011), which are linked with accounting and other data to provide a more comprehensive depiction of the production system. This characteristic provides a robust data framework for analysis and can illuminate the connections and links between various modules. This feature reflects and elucidates the causal relationship between management decisions made and output delivered. As an illustration, in the case study, varying amounts of nitrogen application led to distinct emission liabilities, consequently yielding divergent offset costs. In this research, the model function is demonstrated through a case study that sources its data from literature and is calculated using formulas. However, to fully reflect the complex application decision, emission and yield output relationship, data input to the integrated modelling system should be provided through field tests or cropping simulation software, to ensure data consistency and comparability.

6. Conclusions

This research theorises a PSPES, the ESRIDM integrated modelling framework, elucidating its design and implementation. To evaluate the capacity for decision usefulness and validity, a case study is estimated focusing on three distinct nitrogen application scenarios, the efficacy of the ESRIDM integrated model is showcased.

ESRIDM contributes to sustainability modelling in several ways. First, it integrates science based and economic models with environmental metrics, offering a novel approach to assessing the sustainability of nitrogen management within agricultural production systems. This integration not only spans economic and environmental dimensions but also allows for a detailed analysis of how different management strategies specifically affect sustainability by decomposing ESRI value.

In addition, the ESRIDM may function as a decision-support tool that is adaptable at the system level, providing policymakers and agricultural producers with a framework to quantify and evaluate the environmental and economic outcomes of production system and regulation design choices. This model supports an interdisciplinary decision-making process, facilitating dialogue between scientific research and practical policy implementation.

By combining frameworks from economics, science, and management accounting, the ESRIDM demonstrates how multidisciplinary collaboration can address complex environmental issues. This multidisciplinary approach offers new perspectives and tools for sustainable development research, especially critical in the context of the identification and implementation of more sustainable agricultural practices.

There are several delimitations and limitations of the modelling, which hold promise for future research. Several factors such as variation in crop rotations and pest control strategies are not included. These are relevant to crop yield and emissions, such as pest control may be affected by activities such as the reduction of usage of chemicals and lime application. Further, only one regulatory option was modelled. A further limitation is the reliance on publicly available data and the constrained scope, which only encompasses activities until the harvest stage, excluding costs and pollution incurred after harvest. There is room for data improvements to include additional information of environmental degradation and its remediation, such as broader set of carbon and methane emissions and alternative regulatory options. The framework has been sufficiently explicated to enable future research to adapt it to these and a much wider set of contexts.

CRediT authorship contribution statement

Yujie Liang: Writing – review & editing, Writing – original draft, Validation, Methodology, Formal analysis, Conceptualization. **Paul J. Brown:** Writing – review & editing, Validation, Supervision, Methodology, Formal analysis, Conceptualization. **Christopher Bajada:** Writing – review & editing, Validation, Supervision, Methodology, Formal analysis, Conceptualization. **Hannah Pham:** Writing – review & editing, Validation, Supervision, Methodology, Formal analysis, Conceptualization.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jclepro.2024.143917.

Data availability

Data will be made available on request.

References

ABARES, 2022. Water Market Outlook retrieved from. https://www.agriculture.gov.au/abares/research-topics/water/water-market-outlook. viewed 22 September 2022.
ABARES, 2023. Financial Performance of Cropping Farms 2020–21 to 2022–23. Retrieved from. https://www.agriculture.gov.au/abares/research-topics/surveys/cropping#daff-page-main, 2023/07/25.
Alhaddi, H., 2015. Triple bottom line and sustainability: a literature review. Business and Management studies 1 (2), 6–10.
Arnold, R.D., Wade, J.P., 2015. A definition of systems thinking: a systems approach. Procedia Computer Science 44, 669–678.
Blanchard, B., 1978. Design and Manage to Life Cycle Cost. M/A Press, Portland.
Brock, P., Madden, P., Schwenke, G., Herridge, D., 2012. Greenhouse gas emissions profile for 1 tonne of wheat produced in Central Zone (East) New South Wales: a life cycle assessment approach. Crop Pasture Sci. 63 (4), 319–329.
Brown, P.J., 2016. Calculation of environmentally sustainable residual income (eSRI) from IFRS financial statements: an extension of Richard (2012). In: Richard, J., Rambaud, A. (Eds.), IFRS in a Global World: International and Critical Perspectives on Accounting. Routledge, pp. 141–157.
Brown, P.J., Bajada, C., 2018. An economic model of circular supply network dynamics: toward an understanding of performance measurement in the context of multiple stakeholders. Bus. Strat. Environ. 27 (5), 643–655.
Chen, P., Shi, X., 2022. Dynamic evaluation of China's ecological civilization construction based on target correlation degree and coupling coordination degree. Environ. Impact Assess. Rev. 93, 106734.
Colchester, J.J., 2016. Systems+ Complexity an Overview: an Accessible Introduction to the New Area of Complex Systems. NY: Create Space Independent Publishing Platform.
Cotton Australia, 2022. Industry Overview. https://cottonaustralia.com.au/industry-overview. viewed 28 Jan 2022.
CRDC, 2019. The Australian Cotton Comparative Analysis 2018. Cotton Research and Development Corporation, CRDC.
CRDC, 2024. Australian Cotton Sustainability Update 2023. Cotton Research and Development Corporation (CRDC). https://crdc.com.au/sites/default/files/pdf/2023/20Sustainability/20Update.pdf.
Cucurachi, S., Suh, S., 2017. Cause-effect analysis for sustainable development policy. Environ. Rev. 25 (3), 358–379.

Daly, H.E., 1991. Elements of environmental macroeconomics. Ecol. Econ.: The Science and Management of Sustainability 32–46.
De Backer, E., Aertsens, J., Vergucht, S., Steurbaut, W., 2009. Assessing the ecological soundness of organic and conventional agriculture by means of life cycle assessment (LCA): a case study of leek production. Br. Food J. 111 (10), 1028–1061.
De Luca, A.I., Iofrida, N., Leskinen, P., Stillitano, T., Falcone, G., Strano, A., Gulisano, G., 2017. Life cycle tools combined with multi-criteria and participatory methods for agricultural sustainability: insights from a systematic and critical review. Sci. Total Environ. 595, 352–370.
Department of Climate Change, 2023. Energy, the Environment and Water. In: Australian National Greenhouse Accounts Factors Workbook 2023. Commonwealth of Australia.
Doane, D., MacGillivray, A., 2001. Economic Sustainability: the Business of Staying in Business. New Economics Foundation, pp. 1–52.
Folke, C., Carpenter, S., Elmqvist, T., Gunderson, L., Holling, C.S., Walker, B., 2002. Resilience and sustainable development: building adaptive capacity in a world of transformations. AMBIO A J. Hum. Environ. 31 (5), 437–440.
França, W.T., Barros, M.V., Salvador, R., de Francisco, A.C., Moreira, M.T., Piekarski, C. M., 2021. Integrating life cycle assessment and life cycle cost: a review of environmental-economic studies. Int. J. Life Cycle Assess. 26, 244–274.
Garetti, M., Taisch, M., 2012. Sustainable manufacturing: trends and research challenges. Prod. Plann. Control 23 (2–3), 83–104.
Gorjian, S., Bousi, E., Özdemir, Ö.E., Trommsdorff, M., Kumar, N.M., Anand, A., et al., 2022. Progress and challenges of crop production and electricity generation in agrivoltaic systems using semi-transparent photovoltaic technology. Renew. Sustain. Energy Rev. 158, 112126.
Hanafiah, M.M., Hasan, M., Razman, K., Harun, S.N., Sakawi, Z., 2022. Life cycle assessment of Laser-Induced maize production: adoption of sustainable agriculture practices. Appl. Sci. 12 (22), 11779.
Hasler, K., Bröring, S., Omta, S.W.F., Olfs, H.W., 2015. Life cycle assessment (LCA) of different fertilizer product types. Eur. J. Agron. 69, 41–51.
Hediger, W., 1998. Ecosystem management and sustainability: an ecological-economic model. Life Science Dimensions: Ecological Economics and Sustainable Use. Filander Verlag, Fürth, Germany 133–156.
Hediger, W., 1999. Reconciling "weak" and "strong" sustainability. Int. J. Soc. Econ.
Heijungs, R., Settanni, E., Guinée, J., 2013. Toward a computational structure for life cycle sustainability analysis: unifying LCA and LCC. Int. J. Life Cycle Assess. 18, 1722–1733.
Hekkert, M.P., Suurs, R.A., Negro, S.O., Kuhlmann, S., Smits, R.E., 2007. Functions of innovation systems: a new approach for analysing technological change. Technol. Forecast. Soc. Change 74 (4), 413–432.
Hou, Y., Qian, X., Zhang, R., Gu, F., Feng, P., 2022. Study on an integrated LCA-LCC model for assessment of Highway Engineering Technical Schemes. Buildings 12 (7), 1050.
Hueting, R., 1993. Calculating a sustainable national income: a practical solution for a theoretical dilemma. In: Approaches to Environmental Accounting: Proceedings of the IARIW Conference on Environmental Accounting, Baden (Near Vienna), Austria, 27–29 May 1991. Physica-Verlag HD, Heidelberg, pp. 39–69.
IPCC, 2019. Climate Change and Land. Intergovernmental Panel on Climate Change.
IPCC, 2021. Climate Change 2021: the Physical Science Basis. https://www.ipcc.ch/report/sixth-assessment-report-working-group-i/.
Jin, C., Tsai, F.S., Gu, Q., Wu, B., 2022. Does the porter hypothesis work well in the emission trading schema pilot? Exploring moderating effects of institutional settings. Res. Int. Bus. Finance 62, 101732.
Karunathilake, E.M.B.M., Le, A.T., Heo, S., Chung, Y.S., Mansoor, S., 2023. The path to smart farming: Innovations and opportunities in precision agriculture. Agriculture 13 (8), 1593.
Kleindorfer, P.R., Singhal, K., Van Wassenhove, L.N., 2005. Sustainable operations management. Prod. Oper. Manag. 14 (4), 482–492.
Kopainsky, B., Alessi, S.M., Davidsen, P.I., 2011. Measuring knowledge acquisition in dynamic decision-making tasks. In: The 29th International Conference of the System Dynamics Society. System Dynamics Society, Albany, NY, pp. 1–31.
Kumar, P., Verma, S., Gupta, A., Paul, A.R., Jain, A., Haque, N., 2021. Life cycle analysis for the production of urea through syngas. In: IOP Conference Series: Earth and Environmental Science, vol. 795. IOP Publishing, 012031, 1.
Laveglia, A., Sambataro, L., Ukrainczyk, N., De Belie, N., Koenders, E., 2022. Hydrated lime life-cycle assessment: current and future scenarios in four EU countries. J. Clean. Prod. 369, 133224.
Leip, A., Marchi, G., Koeble, R., Kempen, M., Britz, W., Li, C., 2008. Linking an economic model for European agriculture with a mechanistic model to estimate nitrogen and carbon losses from arable soils in Europe. Biogeosciences 5 (1), 73–94.
Li, Z., 2020. PBCLM: a top-down causal modeling framework for soil standards and global sustainable agriculture. Environmental Pollution 263, 114404.
Macdonald, B.C.T., Chang, Y.F., Nadelko, A., Tuomi, S., Glover, M., 2016. Tracking fertiliser and soil nitrogen in irrigated cotton: uptake, losses and the soil N stock. Soil Res. 55 (3), 264–272.
Malerba, F., 2002. Sectoral systems of innovation and production. Res. Pol. 31 (2), 247–264.
Málovics, G., Csigéné, N.N., Kraus, S., 2008. The role of corporate social responsibility in strong sustainability. The Journal of Socio-Economics 37 (3), 907–918.
Matos, S., Hall, J., 2007. Integrating sustainable development in the supply chain: the case of life cycle assessment in oil and gas and agricultural biotechnology. J. Oper. Manag. 25 (6), 1083–1102.
Mesa, J.A., Esparragoza, I., Maury, H., 2020. Modular architecture principles–MAPs: a key factor in the development of sustainable open architecture products. Int. J. Sustain. Eng. 13 (2), 108–122.
Missimer, M., Robèrt, K.H., Broman, G., 2017. A strategic approach to social sustainability–Part 1: exploring the social system. J. Clean. Prod. 140, 32–41.
Monat, J.P., Gannon, T.F., 2015. What is systems thinking? A review of selected literature plus recommendations. Am. J. Syst. Sci. 4 (1), 11–26.
Muth Jr, D.J., Bryden, K.M., 2013. An integrated model for assessment of sustainable agricultural residue removal limits for bioenergy systems. Env. Model. Software 39, 50–69.
Mutingi, M., Dube, P., Mbohwa, C., 2017. A modular product design approach for sustainable manufacturing in a fuzzy environment. Procedia Manuf. 8, 471–478.
Pearce, D., 1976. The limits of cost-benefit analysis as a guide to environmental policy. Kyklos 29 (1), 97–112.
Pearce, D., 1996. Economic valuation and health damage from air pollution in the developing world. Energy Pol. 24 (7), 627–630.
Pearce, D., 1998. Cost benefit analysis and environmental policy. Oxf. Rev. Econ. Pol. 14 (4), 84–100.
Penman, S.H., 2003. In: Financial Statement Analysis and Security Valuation, second ed. McGraw-Hill, New York, USA.
Pham, H., Sutton, B.G., Brown, P.J., Brown, D.A., 2020. Moving towards sustainability: a theoretical design of environmental performance measurement systems. J. Clean. Prod. 269, 122273.
Plant, R.E., 2001. Site-specific management: the application of information technology to crop production. Comput. Electron. Agric. 30 (1–3), 9–29.
Rambaud, A., Richard, J., 2015. The "triple Depreciation line" instead of the "triple bottom line": towards a genuine integrated reporting. Crit. Perspect. Account. 33, 92–116.
Richard, J., 2012. Comptabilite' et de'veloppement durable. Economica, Paris.
Rochester, I.J., 2003. Estimating nitrous oxide emissions from flood-irrigated alkaline grey clays. Soil Res. 41, 197–206.
Rochester, I.J., 2011. Assessing internal crop nitrogen use efficiency in high-yielding irrigated cotton. Nutrient Cycl. Agroecosyst. 90 (1), 147–156.
Rochester, I.J., Bange, M., 2016. Nitrogen fertiliser requirements of high-yielding irrigated transgenic cotton. Crop Pasture Sci. 67 (6), 641–648.
Roy, P., Nei, D., Orikasa, T., Xu, Q., Okadome, H., Nakamura, N., Shiina, T., 2009. A review of life cycle assessment (LCA) on some food products. J. Food Eng. 90 (1), 1–10.
Russell, A.E., Cambardella, C.A., Laird, D.A., Jaynes, D.B., Meek, D.W., 2009. Nitrogen fertilizer effects on soil carbon balances in Midwestern US agricultural systems. Ecol. Appl. 19 (5), 1102–1113.
Salhieh, S.E.M., Kamrani, A.K., 2008. Modular design. Collaborative Engineering: Theory and Practice 207–226.
Santos, R., Costa, A.A., Silvestre, J.D., Pyl, L., 2019. Integration of LCA and LCC analysis within a BIM-based environment. Autom. ConStruct. 103, 127–149.
Schlüter, L., Kørnøv, L., Mortensen, L., Løkke, S., Storrs, K., Lyhne, I., Nors, B., 2023. Sustainable business model innovation: design guidelines for integrating systems thinking principles in tools for early-stage sustainability assessment. J. Clean. Prod. 387, 135776.
Seiffert, M.E.B., Loch, C., 2005. Systemic thinking in environmental management: support for sustainable development. J. Clean. Prod. 13 (12), 1197–1202.
Sevenster, M., Bell, L., Anderson, B., Jamali, H., Horan, H., Simmons, A., et al., 2022. Australian grains baseline and mitigation assessment. Grains Reseaarch Update 16.
Sevenster, M., Islam, N., Grant, T., Jamali, H., Antille, D., Macdonald, B., Austin, J., Weaver, T., 2024. Greenhouse gas baseline and accounts for the Australian cotton sector. Methodology Report. CSIRO, Australia.
Shaheb, M.R., Sarker, A., Shearer, S.A., 2022. Precision agriculture for sustainable soil and crop management. In: Soil Science-Emerging Technologies, Global Perspectives and Applications. IntechOpen.
Siami, N., Winter, R.A., 2021. Jevons' paradox revisited: implications for climate change. Econ. Lett. 206, 109955.
Sievert, K., Chen, V., Voisin, R., Johnson, H., Parker, C., Lawrence, M., Baker, P., 2022. Meat production and consumption for a healthy and sustainable Australian food system: policy options and political dimensions. Sustain. Prod. Consum.
Sishodia, R.P., Ray, R.L., Singh, S.K., 2020. Applications of remote sensing in precision agriculture: a review. Rem. Sens. 12 (19), 3136.
Sonego, M., Echeveste, M.E.S., Debarba, H.G., 2018. The role of modularity in sustainable design: a systematic review. J. Clean. Prod. 176, 196–209.
Spangenberg, J.H., 2005. Economic sustainability of the economy: concepts and indicators. Int. J. Sustain. Dev. 8 (1–2), 47–64.
Starr, M.K., 1965. Modular production-a new concept. Harv. Bus. Rev. 131–142.
Swarr, T.E., Hunkeler, D., Klöpffer, W., Pesonen, H.L., Ciroth, A., Brent, A.C., Pagan, R., 2011. Environmental life-cycle costing: a code of practice. Int. J. Life Cycle Assess. 16, 389–391.
Tricase, C., Lamonaca, E., Ingrao, C., Bacenetti, J., Giudice, A.L., 2018. A comparative Life Cycle Assessment between organic and conventional barley cultivation for sustainable agriculture pathways. J. Clean. Prod. 172, 3747–3759.
Vallance, S., Perkins, H.C., Dixon, J.E., 2011. What is social sustainability? A clarification of concepts. Geoforum 42 (3), 342–348.
Wei, F., Cui, S., Liu, N., Chang, J., Ping, X., Ma, T., et al., 2021. Ecological civilization: China's effort to build a shared future for all life on earth. Natl. Sci. Rev. 8 (7), nwaa279.
Williams, A., Kennedy, S., Philipp, F., Whiteman, G., 2017. Systems thinking: a review of sustainability management research. J. Clean. Prod. 148, 866–881.
Wolf, J., Beusen, A.H.W., Groenendijk, P., Kroon, T., Rötter, R., Van Zeijts, H., 2003. The integrated modeling system STONE for calculating nutrient emissions from agriculture in The Netherlands. Environ. Model. Software 18 (7), 597–617.
Woodward, D.G., 1997. Life cycle costing—theory, information acquisition and application. Int. J. Proj. Manag. 15 (6), 335–344.

World Bank, 2020. Five Things You Need to Know about Social Sustainability and Inclusion. Retrieved from. https://www.worldbank.org/en/news/feature/2020/09/02/five-things-about-social-sustainability-and-inclusion.
Xue, B., Han, B., Li, H., Gou, X., Yang, H., Thomas, H., Stückrad, S., 2023. Understanding ecological civilization in China: from political context to science. Ambio 1–15.
Yang, R., Sun, Q., Zhao, H., Zou, G., Liu, B., Li, L., 2017. Precision application of biogas slurry and its environmental effects in paddy fields. Journal of Agro-Environment Science 36 (8), 1566–1572. CRDC (2019), The Australian Cotton Comparative Analysis 2018, Cotton Research and Development Corporation (CRDC).
Zhang, H., 2019. Understanding the linkages: a dynamic sustainability assessment method and decision making in manufacturing systems. Procedia CIRP 80, 233–238.
Zou, T., Zeng, H., Zhou, Z., Xiao, X., 2019. A three-dimensional model featuring material flow, value flow and organization for environmental management accounting. J. Clean. Prod. 228, 619–633.