Modelling the Environment with Stakeholders

Theoretical Perspectives and Practical Applications for Transboundary River Management

Henry Daniel Amorocho Daza

MODELLING THE ENVIRONMENT WITH STAKEHOLDERS. THEORETICAL PERSPECTIVES AND PRACTICAL APPLICATIONS FOR TRANSBOUNDARY RIVER MANAGEMENT

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People commonly think that going through a PhD is a lonely journey. Such a belief is reinforced by the fact that PhD candidates are expected to defend their work alone—in conferences, yearly meetings, lunch seminars and, of course, against a doctoral committee at the end. I claim, from personal experience, that it is not possible to obtain a PhD in isolation; it is a collective effort.

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SUMMARY

Societal and scientific challenges

This PhD engages with a pressing environmental issue: managing water resources in transboundary basins. More than half of the world's freshwater flows are transboundary, posing significant challenges regarding water quantity and quality. This research focuses on transboundary water quality. If water is polluted upstream, contaminants may accumulate and be transported downstream. Such asymmetry creates management challenges in transboundary settings. The Lielupe River Basin, shared between Latvia and Lithuania, serves as a case study to illustrate the problem of transboundary river pollution.

We approach this problem using a systems perspective and contribute to two scholarly fields: nexus approaches and participatory systems modelling. Systems thinking and modelling synthesise the effects of interacting elements, while nexus approaches focus on synergies and trade-offs that exist among resource sectors, including Water-Energy-Food and Ecosystems (WEFE). Nexus perspectives are gaining ground in academic and policy circles, yet there is a gap between nexus *thinking* and *action*. Participatory modelling (PM) is concerned with the challenge of crafting environmental systems models with stakeholders. As a growing field, PM faces open research challenges concerning participation, ethics, uncertainty and policy relevance, amongst others.

This PhD offers a WEFE Nexus operationalisation using PM within the context of socio-environmental problems of the Lielupe River Basin. We engage with the challenge of connecting systems thinking to action, transitioning from theory to practice using a model-based participatory approach. In adopting a systems perspective, the dissertation 'zooms in and out' between theory and practice. Zooming in, the research is grounded in the case study's problematic situation, while zooming out facilitates reflection on the system tools used to make sense of the socio-environmental problems.

Structure of the dissertation

The dissertation is divided into two parts. Part I presents theoretical contributions, while Part II focuses on practical findings. Theoretically, we investigate ethical and participatory aspects in model development processes for socio-environmental problems. Practically, we explore the added value of implementing a participatory modelling approach that incorporates uncertainty in transboundary river management.

Main findings

Participatory considerations are relevant at every stage of an environmental modelling process. A suitable way to acknowledge them is through a modelling cycle perspective, one in which a model is conceived, developed and used in iterative phases. By integrating and aligning two relevant modelling cycles, one engaging with uncertainty, the other with participation, we

distinguish three general modelling phases: 1. Modelling foundations; 2. Model building and testing; 3. Model use and policy evaluation. The three phases capture the essence of a modelling endeavour, making explicit when and how stakeholders contribute to the model development process, moving from qualitative to increasingly quantitative modelling. This framework facilitates the analysis of interactions between participation and uncertainty across the phases, bringing conceptual clarity and practical tools to connect uncertainty and participation in model-based decision-making.

Complementary to the framework, we propose that participatory modelling has ethical implications. We do this through: (1) exploring the modelling cycle with an ethics lens; and (2) considering the realm of application in which PM happens. A practical way to explore ethical implications is through ethical questioning. We identify ethical questions relevant at each modelling stage as a structured entry point for ethical reflection. We propose ethical standpoints that are consistent with the practice of environmental participatory modelling. More specifically, we propose Sustainable Development and Human Rights as ethical standpoints for the participatory modelling practice, as they entail principles that should be considered in addressing socio-environmental problems. These standpoints serve to inspire the formulation of pertinent questions to be used in reflecting on the ethical implications of a specific PM endeavour.

Empirical contributions emerge from implementing a participatory modelling cycle in the Lielupe, a river basin facing challenges related to agricultural nutrient pollution. Applying the modelling cycle proposed in this thesis allowed us to shift attention from model features and validation towards the policy implications and limitations of simulation outputs in a local context. Results show that nutrient control policies are effective under ambitious land-use transitions. In implementing basin-scale solutions, exploratory analysis shows that nutrient control would reduce nitrogen concentration by about 30% with a 2% co-benefit of increasing vegetation stocks, yet at the cost of decreasing cereal production by 8%. Stakeholder-driven modelling highlights the importance of promoting active transboundary cooperation for water quality control. Results show that even highly ambitious *unilateral* action can delay the achievement of basin-wide quality objectives in the order of a decade, a critical finding for the Baltic region and the achievement of EU water quality objectives.

Finally, we provide empirical evidence of knowledge generation in the context of transboundary river management. Analysing how an environmental model evolves across a participatory process provides a rich picture of how learning takes place in a participatory process. We develop a structured approach to track model evolution and the learning of participants across the modelling cycle. Results illustrate that participatory modelling is a socially driven endeavour that fosters participants' learning across and beyond a modelling cycle. By tracking model evolution, we illustrate how early-stage stakeholder feedback lays the foundations that determine model capabilities. Results demonstrate participants' knowledge generation, albeit with temporal and structural differences between modeller and stakeholder groups. Yet, translating such knowledge into real-life and sustainable change remains an open academic and practical challenge.

Significance

Our theoretical perspectives contribute towards reflective, holistic, and transparent modelling practice, responding to the Good Modelling Practice paradigm. By breaking up a complex task, such as building an environmental model, into three phases, this dissertation offers researchers and practitioners a flexible framework for addressing socio-environmental issues (Chapter 2). This framework helps in structuring, rationalising and reporting complex participatory modelling endeavours in a straightforward way. By capturing the value that can be expected from implementing an iterative cycle, we facilitate identifying the missing steps in moving from decision support to policy implementation. As environmental modelling is not an objective endeavour, ethical considerations need to be discussed alongside the model's technical aspects (Chapter 3).

Empirical experience in the Lielupe River Basin constitutes an advance in nexus practice. We apply and reflect upon a participatory modelling process within an international, multi-stakeholder, resource nexus case. By interpreting model results within the socio-economic features and policy landscape of the local situation, we take a realistic, yet hopeful, approach towards relevant policy (Chapter 4). This approach can be taken as an invitation to environmental modellers to go beyond validated simulation results and explore policy implications. This thesis shows that modellers cannot do this alone. They need to be embedded within a broader inter- and transdisciplinary network of researchers and practitioners, through which model-based learning can be discussed and integrated into activities that may lead to local or regional change (Chapter 5). The thesis concludes with a call to modellers to consider how modelling can support broader participatory engagement processes rather than how participatory engagement can support modelling

SAMENVATTING

Maatschappelijke en wetenschappelijke uitdagingen

Dit promotieonderzoek richt zich op een urgent milieuvraagstuk: het waterbeheer van grensoverschrijdende stroomgebieden. Meer dan de helft van alle zoetwaterstromen ter wereld is grensoverschrijdend, wat aanzienlijke uitdagingen met zich meebrengt op het gebied van waterkwantiteit en -kwaliteit. Dit onderzoek richt zich op de waterkwaliteit in grensoverschrijdende gebieden. Als het water stroomopwaarts vervuild raakt, kunnen verontreinigende stoffen zich ophopen en stroomafwaarts worden meegevoerd. Een dergelijke asymmetrie zorgt voor uitdagingen in het beheer van grensoverschrijdende rivieren. Het stroomgebied van de Lielupe, dat door Letland en Litouwen wordt gedeeld, dient als casestudy om het probleem van grensoverschrijdende rivierverontreiniging te illustreren.

We benaderen dit probleem vanuit een systeemperspectief en leveren een bijdrage aan twee wetenschappelijke vakgebieden: nexusbenaderingen en participatief systeemmodelleren. Systeemdenken en modellering brengen de effecten van op elkaar inwerkende elementen in kaart, terwijl nexusbenaderingen zich richten op synergieën en afwegingen tussen verschillende sectoren, met name Water-Energie-Voedsel en Ecosystemen (WEFE). Nexusperspectieven winnen terrein in academische en beleidskringen, maar er is een kloof tussen nexus *denken* en *actie*. Participatief modelleren (PM) hoiudt zich bezig met de uitdaging om samen met belanghebbenden milieusysteemmodellen te ontwikkelen. Als groeiend vakgebied richt PM zich op openstaande onderzoeksvraagstukken met betrekking tot onder meer burgerparticipatie, ethiek, onzekerheid en beleidsrelevantie.

Dit proefschrift presenteert een operationalisering van de WEFE-nexus door middel van PM in de context van sociaal-ecologische problemen in het stroomgebied van de Lielupe. We gaan de uitdaging aan om systeemdenken te koppelen aan actie, waarbij we de overgang maken van theorie naar praktijk met behulp van een modelgebaseerde participatieve aanpak. Door een systeemperspectief te hanteren, ‘zoomt’ het proefschrift in en uit tussen theorie en praktijk. Bij het inzoomen is het onderzoek geworteld in de problematische situatie van de casestudy, terwijl het uitzoomen reflectie mogelijk maakt op de systeemhulpprogramma's die worden gebruikt om de sociaal-ecologische problemen te begrijpen.

Opbouw van het proefschrift

Het proefschrift is verdeeld in twee delen. Deel I presenteert theoretische bijdragen, terwijl Deel II zich richt op praktische bevindingen. Theoretisch onderzoeken we ethische en participatieve aspecten in modelontwikkelingsprocessen voor sociaal-ecologische problemen. Praktisch verkennen we de toegevoegde waarde van het implementeren van een participatieve modelleringsaanpak die onzekerheid in het grensoverschrijdende rivierbeheer meeneemt.

Belangrijkste bevindingen

Participatieve aspecten spelen in elke fase van een milieumodelleringsproces een rol. Een geschikte manier om hiermee rekening te houden is vanuit het perspectief van een modelleringscyclus, waarin een model in iteratieve fasen wordt bedacht, ontwikkeld en gebruikt. Door twee relevante modelleringscycli – de ene gericht op onzekerheid, de andere op participatie – te integreren en op elkaar af te stemmen, onderscheiden we drie algemene modelleringsfasen: 1. Grondslagen voor modellering; 2. Modelopbouw en -testen; 3. Modelgebruik en beleidsevaluatie. De drie fasen vatten de essentie van een modelleringsinspanning samen, waarbij expliciet wordt gemaakt wanneer en hoe belanghebbenden bijdragen aan het modelontwikkelingsproces, waarbij de overgang wordt gemaakt van kwalitatieve naar steeds meer kwantitatieve modellering. Dit raamwerk vergemakkelijkt de analyse van interacties tussen participatie en onzekerheid in de verschillende fasen, en biedt conceptuele duidelijkheid en praktische instrumenten om onzekerheid en participatie te koppelen aan modelgebaseerde besluitvorming.

Als aanvulling op het kader stellen wij dat participatieve modellering ethische implicaties heeft. Dit doen we door: (1) de modelleringscyclus vanuit ethisch perspectief te onderzoeken; en (2) stil te staan bij het toepassingsgebied waarin PM plaatsvindt. Een praktische manier om ethische implicaties te onderzoeken is door middel van ethische vraagstelling. We brengen ethische vragen in kaart die in elke modelleringsfase relevant zijn, als gestructureerd uitgangspunt voor ethische reflectie. We stellen ethische standpunten voor die aansluiten bij de praktijk van participatieve milieumodellering. Meer specifiek stellen we Duurzame Ontwikkeling en Mensenrechten voor als ethische standpunten voor de praktijk van participatieve modellering, aangezien deze principes in aanmerking moeten worden genomen bij het aanpakken van sociaal-ecologische problemen. Deze standpunten dienen als inspiratie voor het formuleren van relevante vragen die kunnen worden gebruikt bij het reflecteren op de ethische implicaties van een specifiek PM-project.

Het toepassen van een participatieve modelleringscyclus in het stroomgebied van de Lielupe, een riviergebied met nutriëntenverontreiniging door de landbouw, heeft geleid tot verschillende empirische bevindingen. Door de in dit proefschrift voorgestelde modelleringscyclus toe te passen, lukte het ons de aandacht te verleggen van modelkenmerken en validatie naar de beleidsimplicaties en beperkingen van simulatieresultaten in een lokale context. De resultaten tonen aan dat beleid voor nutriëntenbeheersing effectief is bij ambitieuze veranderingen in landgebruik. Bij de implementatie van oplossingen op stroomgebiedschaal blijkt uit verkennende analyses dat nutriëntenbeheersing de stikstofconcentratie met ongeveer 30% zou verminderen, met als bijkomend voordeel een toename van de vegetatievoorraden met 2%, maar ten koste van een daling van de graanproductie met 8%. Door belanghebbenden aangestuurde modellering bevordert de grensoverschrijdende samenwerking voor waterkwaliteitsbeheersing. De resultaten tonen ook aan dat zelfs zeer ambitieuze unilaterale maatregelen het bereiken van kwaliteitsdoelstellingen op stroomgebiedniveau met ongeveer tien jaar kunnen vertragen, een cruciale bevinding voor de Oostzeeregio en het behalen van de EU-doelstellingen voor waterkwaliteit.

Ten slotte leveren we empirisch bewijs voor kennisontwikkeling in de context van grensoverschrijdend rivierbeheer. Door te analyseren hoe een milieumodel zich tijdens een participatief proces ontwikkelt, krijgen we een goed beeld van hoe het leerproces in een dergelijk proces verloopt. We ontwikkelen een gestructureerde aanpak om de ontwikkeling van het model en het leerproces van de deelnemers gedurende de gehele modelleringscyclus te volgen. De resultaten illustreren dat participatief modelleren een sociaal gedreven onderneming is die het leren van deelnemers tijdens en na de modelleringscyclus bevordert. Door de modelontwikkeling te volgen, laten we zien hoe feedback van belanghebbenden in een vroeg stadium de basis legt voor de mogelijkheden van het model. De resultaten tonen de kennisontwikkeling van deelnemers aan, zij het met verschillen tussen modelleurs en groepen belanghebbenden. Het vertalen van deze kennis naar praktische en duurzame verandering blijft echter een open academische en praktische uitdaging.

Betekenis

Onze theoretische invalshoeken dragen bij aan een reflectieve, holistische en transparante modelleringspraktijk, in overeenstemming met het paradigma van een ‘Goede Modelleer Praktijk’. Door een complexe taak, zoals het bouwen van een milieumodel, op te splitsen in drie fasen, biedt dit proefschrift onderzoekers en praktijkmensen een flexibel kader voor het aanpakken van sociaal-ecologische vraagstukken (hoofdstuk 2). Dit kader helpt bij het op een overzichtelijke manier structureren, rationaliseren en rapporteren van complexe participatieve modelleringsprojecten. Door de waarde in kaart te brengen die kan worden verwacht van het implementeren van een iteratieve cyclus, maken we het gemakkelijker om de ontbrekende stappen te identificeren in de overgang van beslissingsondersteuning naar beleidsimplementatie. Aangezien milieumodellering geen objectieve onderneming is, moeten naast de technische aspecten van het model ook ethische overwegingen worden meegenomen (hoofdstuk 3).

De empirische bevindingen over het stroomgebied van de Lielupe betekenen een stap vooruit in de nexus-praktijk. We passen een participatief modelleringsproces toe en reflecteren hierop binnen een internationale casus met meerdere belanghebbenden van een nexus van hulpbronnen. Door modelresultaten te interpreteren binnen de sociaal-economische kenmerken en het beleidslandschap van de lokale situatie, hanteren we een realistische, maar hoopvolle benadering ten aanzien van relevant beleid (hoofdstuk 4). Deze benadering kan worden opgevat als een uitnodiging aan milieumodelleurs om verder te kijken dan gevalideerde simulatieresultaten en beleidsimplicaties te verkennen. Dit proefschrift laat zien dat modelleurs dit niet alleen kunnen. Ze moeten worden ingebed in een breder inter- en transdisciplinair netwerk van onderzoekers en praktijkmensen, waardoor modelgebaseerd leren kan worden besproken en geïntegreerd in activiteiten die kunnen leiden tot lokale of regionale verandering (hoofdstuk 5). Het proefschrift sluit af met een oproep aan modelleurs om na te denken over hoe het modelleren bredere participatieve betrokkenheidsprocessen kan ondersteunen, in plaats van hoe participatieve betrokkenheid het modelleren kan ondersteunen

CONTENTS

1

INTRODUCTION

1.1. BACKGROUND

The societal challenge of transboundary water management

This PhD research engages with a serious environmental issue: managing water resources in transboundary basins. Rivers are essential for humanity. They have shaped our civilization's history and continue to offer humans multiple resources and other intangible services. Rivers, as ecosystems, not only continue providing water (see Auerbach et al., 2014), but are intrinsically related to food production (crop irrigation, fisheries) and energy generation, for instance. They also provide navigation opportunities, recreation and cultural services for communities around the world. In their path from headwaters to the sea, rivers flow within and across countries, often crossing our current administrative boundaries. This is a pervasive challenge, as around 60% of global freshwater flows are transboundary, and these river basins host about 40% of the world's population (Munia et al., 2016). How upstream countries manage the rivers has downstream effects, both in terms of the quantity and quality of water. Evidently, if water is consumed upstream, it won't be available downstream. Besides, if water is polluted upstream, contaminants may accumulate and be transported downstream. Such asymmetry creates management challenges in international transboundary settings (van der Zaag, 2007). This research engages with the socio-environmental problem of water quality in transboundary river basins, by taking a case study approach to delve deeply into a specific transboundary river pollution problem in the Lielupe.

The Lielupe River Basin (17,788 sq.km) is shared between Latvia and Lithuania. The river basin is characterised by a flat landscape with fertile soils supporting agriculture (cereals, fodder crops) and dairy farming activities. Previous studies have identified a connection between the land-use patterns and multi-sectoral impacts (Sušnik et al., 2021). For instance, (i) the water quality is degraded by nutrient pollution of agricultural origin (Siksnane and Lagzdins, 2020), and (ii) the reduction in meadows and pastures owing to expanding cropland area puts pressure on local ecosystems (Melece and Shena, 2018). Given the transboundary nature of the basin, water quality is a 'wicked' problem—an ambiguous problematic situation with no single definition nor solution (see Rittel and Weber, 1973; Enserink et al., 2020; Ackerman, 2012), in which the upstream nutrient pollution (from Lithuania) affects water quality downstream (in Latvia).

The Lielupe River Basin (LRB) formed one of five case studies of the European Commission Horizon2020 NEXOGENESIS project (nexogenesis.eu) (refer to Part II for more details about this case study). To account for the complexity of resources and stakeholders in the basin, the project took a holistic approach known as the *nexus*. The nexus perspective pays attention to the synergies and trade-offs among resource sectors, often characterised by, but not limited to, Water, Energy, Food and Ecosystems (i.e. WEFE Nexus) (llouche, 2024; Liu et al., 2018; Lucca et al., 2025). This rather recent systemic approach connects with a long tradition of systems thinking in natural resources management (Susnik and Mellios, 2025). Regarding the stakeholders, a structured engagement process was employed in the project, following a co-creation approach (extensively reported by Avellán et al. 2025 and Huerker et al. 2022), that facilitated the formalisation of a nexus analysis using simulation models (see Chapters 4 and

5). This PhD research draws on the participatory modelling field to connect the analytical capabilities of simulation models and the implications of building and using such models in a participatory way, particularly in addressing the societal challenge of transboundary water management. The following section introduces the research field and key concepts of the thesis.

Research field: participatory modelling

A classical definition of a model is 'a simplification of reality'. This broad definition covers all sorts of physical, conceptual, mathematical and computational 'simplifications' that are commonly used to support our daily thinking and decision-making (see Ford, 2010). Chapter 2 presents a model-based policy analysis framework for socio-environmental systems—there, we use the term 'model' in a more specific sense. First, we understand it qualitatively. This entails systemically framing and conceptualising a socio-environmental problem as a 'map' of interactive elements that can be shared with a group of people. Put more simply, qualitative modelling provides an interpretable and communicable 'visual aid' to discuss problems in a systematic way. For instance, conceptual maps or causal loop diagrams may be used (see Chapter 2). Second, we understand it quantitatively. Taking the step beyond mapping of relations between elements, quantitative models use equations to formalise such relations mathematically.

Quantitative models are run on a computer and, ideally, after thorough validation protocols are performed, generate simulations that mimic 'reality'. Some quantitative models are termed 'virtual worlds' and 'microworlds' (see Morecroft,1988; Sterman, 1996, 2000). Virtual worlds facilitate learning about a problem using the logic of a 'flight simulator' (Sterman, 2000, 2006). They are virtual settings in which time and space can be compressed or dilated—allowing their users to experiment and improve decision-making intuition through near-real-time feedback, as an alternative to running slow and costly real-world experiments (Sterman, 2006). Computer simulation models allow for virtual experiments under conditions that are difficult or impossible to test in reality. They also allow testing of multiple (even millions) of scenarios quickly. In short, quantitative models, in the fashion of virtual worlds, are rather sophisticated simulation models that are crafted in a way that can facilitate learning about a complex issue.

Model-based participatory research studying systems in which people and nature are deeply entangled, often conceptualised as social-ecological systems (SES) (Fischer et al., 2015; Preiser et al., 2018), has evolved into an established field of research known as participatory modelling (PM) (Jordan et al., 2018; Sterling et al., 2019; Zellner, 2024). PM is defined as "a purposeful learning process for action that engages the implicit and explicit knowledge of stakeholders to create formalized and shared representations of reality" (Jordan et al., 2018, p. 1047). Such a definition acknowledges models as representations of reality but also emphasises the role of stakeholders in co-creating such models. Although PM scholars continue to use models (in the sense of 'virtual worlds'), this research field pays more deliberate attention to the model-building processes and how participants interact with each other and with the modelling tools, rather than focusing solely on the model as a final product (Grey et al., 2018).

PM is a growing research field with open research questions and challenges arising from the complexity of modelling the environment in systematic and participatory ways (Elsawah et al., 2020; Jordan et al., 2018). In addressing these questions and challenges, the Good Modelling Practice (GMP) paradigm is process-focused, encouraging modellers to disclose their model crafting rather than just presenting modelling results (Jakeman et al., 2024). This thesis makes both theoretical and practical contributions to good modelling practices in the field of participatory modelling. We do so by engaging with the open academic challenge of integrating participation, uncertainty and ethics in model development for socio-environmental problems. Empirical research operationalising such an integrative approach and exploring its added value and limitations is also lacking in the PM literature.

Research aims and questions

Within the field of PM, this dissertation employs a WEFE Nexus perspective towards the socio-environmental problems of the Lielupe River Basin. Recent literature calls for moving from nexus thinking to action (Simpson and Jewitt, 2019b; Susnik and Staddon, 2021). Here we engage with the research challenge of connecting nexus systems thinking to action, that is, transitioning from theory to practice in the Lielupe using a participatory modelling approach. Therefore, the PhD dissertation 'zooms in and out' between theory and practice (see Zellner, 2024). Zooming in, the research is grounded in the case study's specific problematic situation. Zooming out, conversely, allows reflection on participatory modelling practice and the systems tools needed to make sense of socio-environmental problems. This shift in focus therefore facilitates the doctoral research to engage with both theoretical and practical dimensions.

Theoretically, we investigate how ethical and participatory considerations can be incorporated in model development processes for socio-environmental problems, seeking to strengthen the methodological foundations of PM. Accordingly, we pose the following research questions:

1. How can participatory considerations be incorporated in model development processes for complex (dynamic and uncertain) socio-environmental problems? (Chapter 2)
2. What are the ethical implications of participatory modelling for socio-environmental problems? (Chapter 3)

Practically, the challenges lie in implementing the theoretical findings in a specific context, namely the Lielupe River Basin. Overall, we aim to explore the added value of implementing a PM approach that incorporates uncertainty in transboundary river management. Accordingly, we pose the following research questions:

3. Which policy insights emerge from the participatory development and application of a simulation model in transboundary river management (of the Lielupe River Basin)? (Chapter 4)
4. To what extent does participatory modelling promote participants' learning in the context of transboundary river management (in the Lielupe River Basin)? (Chapter 5)

Clearly, the added value is specified here as deriving from policy insights and the extent to which PM promotes participants' learning in the context of transboundary river management. We are aware that these are not the only effects or additional benefits of participatory modelling, but choose to focus attention on these aspects in the Lielupe River Basin case study.

Approach and structure of the thesis

The dissertation is divided into two parts. Part I deals with the theoretical aspects of participatory environmental modelling, consisting of Chapters 2 and 3. Part II, consisting of Chapters 4 and 5, deals with the empirical aspects of applying PM to the Lielupe River Basin case study, as part of the European Commission NEXOGENESIS project. Having a single case study offers the methodological benefit of in-depth analysis while enriching the empirical grounding in an academic field (Flyvbjerg, 2006), thereby facilitating testing our theoretical claims while offering wider insights for research and practice. In Part I, we develop structured modelling approaches aiming to support informed engagement in complex socio-environmental problems. We contribute to scholarship in the field of participatory modelling of socio-environmental systems, emphasising the importance of having structured environmental modelling approaches that promote dialogues in which experts and stakeholders work together in interdisciplinary and transdisciplinary ways. Overall, we rely on a modelling cycle perspective, one in which a model is conceived, developed and used in successive, iterative phases. This perspective identifies multiple qualitative and quantitative features and tools that contribute to using simulation models in informing real-life environmental policy. It also facilitates undertaking structured ethical questioning to reflect on the decisions that shape the modelling process.

In Part II, we explore the added value of implementing a PM approach that incorporates uncertainty in the transboundary river management of the Lielupe River Basin. In addition to providing an example of PM implementation, this section contributes to WEFE nexus scholarship and practice, as it applies and reflects upon a PM process within an international, multi-stakeholder, resource nexus case. The chapters in Part II exemplify the value of operationalising WEFE nexus systems thinking and analysis using the structured model-based approaches from Part I. Part I and Part II, therefore, are complementary in ensuring that the process of PM results in locally grounded environmental policy insights and that learning outcomes emerge from stakeholder engagement processes.

The title of this thesis, ‘Modelling the environment with stakeholders’, is inspired by merging the titles of two landmark contributions: ‘Modelling the environment’ (Ford, 1999, 2010) and ‘Modelling with stakeholders’ (Voinov and Bousquet, 2010). This choice not only honours the authors, but also responds to the fact that participatory environmental modelling is a problematic approach worth exploring in a PhD dissertation. Here, we argue extensively that the messy task of ‘modelling the environment with stakeholders’ underpins some of the more persistent challenges debated in the scholarly fields relevant to participatory environmental modelling.

This PhD research, therefore, contributes to an open academic and practical discussion about how models are co-developed with stakeholders as part of larger socio-environmental policy processes. We do so by engaging with the theoretical foundations of PM and their application in a WEFE Nexus case study. Chapter 2 sets the scene by developing a participatory modelling framework for socio-environmental problems. Chapter 3 discusses the ethical implications of modelling the environment in participatory settings. Chapter 4 presents a case study in which Chapter 2’s framework is applied. Chapter 5 is also practical, focusing on assessing the learning outcomes of the model-based intervention presented in the previous chapter. Chapter 6 concludes the dissertation by synthesising the contributions of the research to the PM field, discussing the limitations and pointing to avenues for further research.

1.2. RESEARCH APPROACH

This research adopts a systems perspective throughout. That is, thinking in systems connects our theoretical and practical contributions (Ackhof, 1994; Ison, 2008; Meadows, 2008). Among many systems modelling approaches (Kelly et al., 2013; Voinov et al., 2018), we adopt System Dynamics, an established modelling approach to simulate the behaviour of complex systems over time (Forrester, 1968; Meadows et al., 1970; Naugle et al., 2024). This modelling approach has been widely applied to environmental management problems (Ford, 1999; Meadows 2008; Moallemi et al., 2021) and is considered particularly suited to WEFE Nexus applications (Susnik and Mellios, 2025). We use a modelling cycle as a key analytical perspective throughout the thesis—as a structured and systemic way of approaching the task of creating a simulation model from concept to decision support tool (see Jakeman et al., 2024; Voinov et al., 2018; Videira et al., 2010).

Recent advances in modelling scholarship are concerned with crafting models in a more transparent, participatory and useful way. These considerations build on a long tradition and are synthesised as Good Modelling Practice (see Hamilton et al., 2022; Jakeman et al., 2024; Legasto et al., 1980). Here we contribute to Good Modelling Practice in PM in both Part I and Part II of this thesis. In Part I, we propose a modelling cycle that considers participatory and uncertainty aspects simultaneously. Additionally, we highlight the ethical dimension of modelling the environment via a modelling cycle approach. Part II, takes an empirical stance, it describes the implementation of the modelling cycle proposed in Part I, and emphasises the exploration of the added value of PM outcomes, taking a case study approach.

In using a single case study, this research responds to the practical benefits and constraints of contributing to a larger European Commission research project—NEXOGENESIS—via which this PhD was funded. A case study approach intrinsically offers knowledge depth over breadth. Quantitative and qualitative systems modelling tools are applied, while critical reflection is undertaken on the modelling process and its outcomes. In doing so, we offer practical insights that respond to the needs of the case study's stakeholders and we reflect on the learning outcomes during the project. In addition, our approach offers an example that may contribute to future Good Modelling Practice research within the PM field, relevant beyond the thesis case study.

1.3. Reading guide

This PhD dissertation is article-based. This implies that the chapters are self-standing and written for different research audiences—with their own introduction, methodological approach, results, discussion and conclusions. Nonetheless, the introduction section of this thesis (Chapter 1) sets the scene by organising the content chapters in a logical way, each of them responding to the PhD research questions. Likewise, a final synthesis section (Chapter 6) provides a critical reflection of the thesis findings. Keeping the systems perspective, the dissertation as a whole is more than the sum of its parts.

In contributing to theory and practice, the chapters of this dissertation read differently as they have a distinctive nature (See Figure 1.1). Chapter 1 has a descriptive style. Chapters 2 and 3 are essentially methodological contributions, yet Chapter 3 also comprises conceptual elements. Chapters 4 and 5 are empirical, the first model-focused, while the second is process-focused. Chapter 6 is critical and reflective in nature.

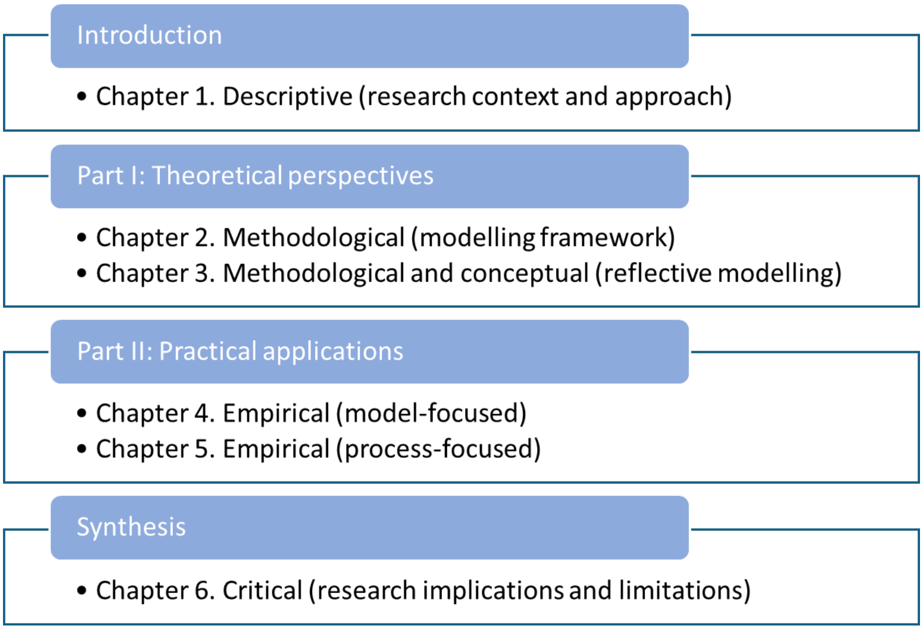

Figure 1.1. Nature of the contributions of each chapter in the dissertation.

The core content of the thesis is divided into two parts, Part I presenting the theoretical contributions, while Part II focuses on the practical findings. Each part comprises two chapters; each chapter responds to one of the four proposed research questions. That is, Part I, consisting of Chapters 2 and 3, constitutes an answer to research questions 1 and 2, respectively. Likewise, Part II, consisting of Chapters 4 and 5, explores research questions 3 and 4, respectively. A summary of the chapters is presented below.

Chapter 2. A framework to integrate uncertainty and participation in system dynamics modelling (based on Amorocho-Daza et al., 2025)

This chapter presents a participatory modelling framework that considers uncertainty in the context of socio-environmental systems analysis. This framework stems from integrating two existing modelling cycles that individually consider participation and uncertainty in SD modelling. A three-phase global modelling cycle and a set of tools to address the participation and uncertainty features in modelling problems within coupled social and ecological systems, termed social-ecological systems (SES), are distinguished. Such a framework is later operationalised in Chapters 4 and 5.

Chapter 3. Ethical considerations of modelling the environment with stakeholders (based on Amorocho-Daza et al., 2024)

This chapter discusses the ethical considerations of using System Dynamics to model social-ecological systems (SES). It identifies sustainable development and human rights as ethical standpoints for socio-environmental SD research and practice. A set of guiding ethical questions are identified and classified across an established participatory SD modelling cycle. This chapter can be read as an invitation to the System Dynamics research community to engage more deeply with ethical issues in their PM practice. Yet, as it acknowledges the ethical role of modellers in participatory settings, it also supported disclosing the modeller’s motivation and decision-making during the participatory modelling process presented in Chapter 5.

Chapter 4. Using a participatory system dynamics approach to assess transboundary nutrient pollution (based on Amorocho-Daza et al., 2026)

This chapter operationalises Chapter 2’s participatory modelling approach in the transboundary Lielupe river basin, shared between Latvia and Lithuania. Using a modelling cycle approach, it illustrates a stakeholder-driven pathway from generic and qualitative to increasingly quantitative system tools useful for basin-scale policy analysis. Stakeholders prioritised agricultural nutrient pollution as a critical nexus issue strongly linked to land-use. Results highlight the importance of promoting active transboundary cooperation for water quality control, as unilateral action hampers the effect of ambitious long-term policies. The chapter highlights the policy implications and limitations of the simulation outputs in the context of the Lielupe River Basin.

Chapter 5. Tracking model evolution and learning in environmental participatory modelling (based on Amorocho-Daza et al., under review)

Chapter 5 proposes a structured approach to track model evolution and the learning of participants across the modelling cycle phases presented in Chapter 2. Results illustrate that PM is a socially driven endeavour that can foster participants' learning across and beyond a modelling cycle. By tracking model evolution, we illustrate how early-stage stakeholder feedback lays the foundations that determine the capabilities of the final model. Our results evidence participants’ knowledge generation, yet with temporal and structural differences between the modeller and the stakeholder groups.

PART I

THEORETICAL PERSPECTIVE

2

A FRAMEWORK TO INTEGRATE UNCERTAINTY AND PARTICIPATION IN SYSTEM DYNAMICS MODELLING

Based on the published peer-reviewed article:

Amorocho-Daza, H., Sušnik, J., van der Zaag, P., & Slinger, J. H. (2025). A model-based policy analysis framework for social-ecological systems: Integrating uncertainty and participation in system dynamics modelling. Ecological Modelling, 499, 110943. https://doi.org/10.1016/j.ecolmodel.2024.110943

Abstract

Problems manifested within social-ecological systems (SES) exhibit dynamic complexity and hold implications for current and future human well-being and environmental sustainability. The complexity of these issues, the ever-present uncertainty inherent to SES, and the multi-stakeholder settings in which they are discussed call for participatory modelling to support decision-making on socio-environmental issues. Yet, this challenging endeavour requires a structured approach — a modelling cycle — to facilitate engagement with the implications of participation and uncertainty as focal points for Good Modelling Practice (GMP). Here we propose an integrated policy analysis framework for SES modelling using System Dynamics (SD). This framework stems from integrating two existing modelling cycles that individually consider participation and uncertainty in SD modelling. Three global modelling phases and a set of tools to address the participation and uncertainty features in SES modelling are distinguished. The framework contributes to mainstreaming GMP, offering a structured model-based approach to enhance the robustness and social acceptance of policies on critical socio-environmental issues.

2.1. INTRODUCTION

Human activities are driving multiple environmental changes at a planetary scale (Folke et al., 2021). Anthropogenic pressures are linked to global issues such as climate change, environmental deterioration, resource depletion, and loss of biodiversity (Díaz et al., 2019; Nelson et al., 2006; Rockström et al., 2009; van den Heuvel et al., 2020). As humanity continues to rely on natural resources, the natural resource base (e.g. water, land, fossil fuels, minerals) is changing on a global scale along with ecosystems (cf. Armstrong McKay et al. (2022)). In turn, fast-paced environmental change is compromising human well-being and access to basic resources (Gupta et al., 2023; Watts et al., 2021). These complex interactions between humans and the natural environment act across multiple scales and exhibit bi-directional influences.

Systems thinking offers a powerful approach to conceptualise complex human-nature interactions by focusing on the interaction of interdependent elements that form a whole rather than simply on the elements themselves (Ackoff, 1971, 1994; Ison, 2008; Ison, Maiteny, & Carr, 1997; Meadows, 2008; Mingers & White, 2010). Understanding human and natural elements as deeply intertwined is key to understanding the pressing socio-environmental challenges of our time (Folke et al., 2016). This idea has been articulated in the concept of social-ecological systems (SES) which can be defined as "interdependent and linked systems of people and nature" (Fischer et al., 2015, p. 145). SES are characterised as being nested across interacting scales (e.g. landscape, regional, and global) and embedded in the biosphere (Fischer et al., 2015; Folke et al., 2021). SES are complex systems as they are constituted relationally; adaptive; dynamic; open; contextually determined; and characterised by multiple causal pathways (Preiser, Biggs, De Vos, & Folke, 2018). These features imply that designing and implementing policies that deal with such systems is a non-trivial and complex task (de Gooyert et al., 2016; Kelly et al., 2013), likely with multiple feasible options.

Using models to represent SES is an essential part of exploring the potential impacts of socio-environmental policies. SES modelling aims "to characterise and explore complex socio-environmental issues in systematic and collaborative ways" (Elsawah et al., 2020, p. 1). This definition may be understood within a larger policy analysis framework (Mayer, van Daalen, & Bots, 2004, 2012). Among many systems approaches, policy analysis focuses on *analysing* a system to find ways to influence it towards desirable outcomes (van Daalen & Bots, 2010). This strongly relates to the *systematic* approach mentioned in the definition above. Likewise, developing a policy analysis approach in the context of SES has significant implications in terms of stakeholder participation (Amorocho-Daza, van der Zaag, & Sušnik, 2024; Bots & van Daalen, 2008; Clifford-Holmes et al., 2018). This relates to the challenge of developing models in *collaborative* ways. In short, SES modelling may be understood as a systems approach that takes an analytical perspective of a socio-environmental system, while recognising the criticality of having a subjective view of the problem situation at hand that arises from collaborative model building and use (van Daalen & Bots, 2010).

The ambition to engage with SES complexity in a participatory modelling setting is a formidable task for practitioners and researchers. Recent case studies (Bitterman & Webster, 2024; Mer, Vervoort, & Baethgen, 2020; Villamor et al., 2019) and reviews (Voinov et al., 2016; Whitley et al., 2024) illustrate the increasing attention toward participatory modelling approaches. Kelly et al. (2013) identified five modelling approaches that are suitable for integrating various SES processes in which stakeholders can explore, analyse, assess and communicate policy alternatives: System Dynamics, Bayesian networks, coupled component models, agent-based models and knowledge-based models. However, operationalising such modelling approaches in participatory settings remains a challenge. In a similar vein, Elsawah et al. (2020) recently identified eight grand challenges in SES modelling related to issues of epistemology, interdisciplinarity, uncertainty, scaling, and policy impact. This paper aims to explicitly engage with two of these grand challenges: (i) the integrated treatment of uncertainty in the modelling process; and (ii) the adoption of SES models to increase their impacts on policy.

The first challenge, the integrated treatment of uncertainty, recognises that uncertainty is ever-present in SES modelling (Ascough et al., 2008). Uncertainty, as defined by Walker et al.(2003, p. 8), is "any deviation from the unachievable ideal of completely deterministic knowledge of the relevant system". Brugnach et al. (2008, p. 5) extend this definition by including its *relational* dimension, as follows: "Uncertainty refers to the situation in which there is not a unique and complete understanding of the system to be managed". Therefore, uncertainty takes place across all modelling phases, at different levels (from determinism to total ignorance) and exhibits distinct *natures* (e.g. knowledge or epistemic uncertainty, variability or ontological uncertainty, and ambiguity) (Kwakkel, Walker, & Marchau, 2010). In contrast to this holistic view of uncertainty, SES modelling practices related to uncertainty have often been confined to quantitative approaches to data validity, model parameter sensitivity and structural testing (Maier et al., 2016). However, recent literature highlights that activities taking place at early modelling cycle stages (e.g. scoping and conceptualisation), while qualitative in nature, represent fundamental uncertainty sources (Nabavi, Daniell, & Najafi, 2017). The lack of integrated uncertainty assessment is also connected to the challenge of communicating uncertainty to stakeholders in model-based decision-making (Palmer, 2017). Better communication regarding uncertainty implies that stakeholders and modellers engage in dialogues to discuss both qualitative aspects, such as values, representation, prioritisation, and transparency, as well as quantitative aspects, including modelling outputs, scenarios, trade-offs, and risk, across the different stages of the modelling cycle (Elsawah et al., 2020).

The second challenge connects the need for participation with the expected policy impact deriving from the use of SES models (Elsawah et al., 2020). Participation can be understood as a process in which stakeholders "choose to take an active role in the decisions that affect them" (Reed, 2008, p. 2418). The need for participation can be justified via normative and pragmatic arguments (Reed, 2008). Normative arguments are often related to the democratic right to participation (Király & Miskolczi, 2019), but also can arise from the ethical implications of building models with stakeholders (Amorocho-Daza et al., 2024; Palmer, 2017). Pragmatic arguments are related to the expected benefits of engaging stakeholders in policy-making processes, in other words, a perspective in which participation is "a means to an end", such as enhancing the quality and durability of environmental decisions (Beierle, 2002; Reed, 2008). The latter aspect is the bridging factor between participation and policy impact. However, far from being a panacea, the successful delivery of participation "promises" is heavily context- and process-dependent (d'Hont & Slinger, 2022; Reed et al., 2017; Sarmiento et al., 2020). Narrowing down the aforementioned discussion, here we focus on the expected benefits of participation in model-based policy discussion settings around socio-environmental issues (Bots & van Daalen, 2008).

Participatory modelling settings are one of the instances in which stakeholders can take part in socio-environmental policy discussions. This generic terminology refers to the endeavour of modelling with stakeholders, very often in the context of socio-environmental issues (Videira, Antunes, Santos, & Lopes, 2010; Voinov & Bousquet, 2010). Building SES models in a participatory way is a co-creation process in which both researchers and stakeholders bring and put different perspectives and knowledge together in dialogue to improve the scope and purpose of the models (Bots & van Daalen, 2008; Norström et al., 2020; Slinger, 2023; Sterling et al., 2019; Voinov et al., 2014). Engaging stakeholders can enhance a shared understanding of complex, locally rooted, social and natural systems leading to the design of more comprehensive, locally relevant socio-environmental policies (Clifford-Holmes et al., 2018; d'Hont & Slinger, 2022; Slinger, Cunningham, & Kothuis, 2023). From a pragmatic perspective, an essential output of such dialogue is building knowledge and social capital that is reflected in the stakeholders' commitment towards crafting and implementing informed and effective socio-environmental policies (Sterling et al., 2019). Therefore, it is critical to understand the interlinkages between participation and policy impact in the context of the SES modelling cycle (Elsawah et al., 2020).

Integrating the dimensions of uncertainty and policy impact in SES participatory modelling has proven conceptually and practically challenging, but recent advances offer promising roads ahead. Regarding the first challenge, integrating uncertainty in SES modelling, Ascough et al. (2008) set the scene by categorising various typologies and sources of uncertainty in environmental decision-making. Some of these include how knowledge uncertainty is pervasive across the modelling cycle (across the process itself, in the model, and in the modelling outputs), or the importance of linguistic uncertainty, a social aspect related to language ambiguity and vagueness in a decision-making context. More recently, Maier et al. (2016) propose that integrated uncertainty modelling needs to consider multiple future scenarios, aiming to find alternatives with *robust* performance under many plausible futures, and aim for adaptive, flexible strategies. From the participation side, there is a growing recognition of policy-relevant modelling as closely aligned with transdisciplinary participation (Moallemi, Malekpour, et al., 2020). Recent reviews show how co-produced sustainability initiatives (e.g. making use of SES models) can serve multiple purposes (Chambers et al., 2021), mirroring previous calls for adaptive and flexible strategies (Pahl-Wostl, 2007). Despite some authors highlighting the importance of participation in dealing with modelling uncertainty (e.g. Ascough et al. (2008)), or the implications of uncertainty in participatory contexts (e.g. Barnhart et al. (2018); Martinez-Fernandez, Banos-Gonzalez, and Esteve-Selma (2021); Moallemi et al. (2023)), the academic literature lacks frameworks that conceptualise the integration of both uncertainty and participation across a SES modelling cycle.

This article proposes a modelling cycle framework to support system dynamics SES modelling and policy evaluation in a stakeholder engagement context, accounting for both uncertainty assessment and participation. In addition, a range of tools and approaches are proposed at each stage in the framework to address these different facets. System Dynamics (SD) is selected as the modelling approach of choice in this context due to its analytic capabilities to simulate complex systems' behaviour, flexibility of application, and proven use in stakeholder participatory settings (Elsawah et al., 2017). These capabilities mean that SD is being applied across various SES fields, including agriculture and natural resource management (Turner et al., 2016), water resources management (Phan, Bertone, & Stewart, 2021; Zomorodian et al., 2018), environmental health (Currie, Smith, & Jagals, 2018), and in holistic public health approaches such as One Health (i.e. health of people, animals, and the environment) (Xie et al., 2017).

Multiple similar frameworks can be designed, none of them fully comprehensive, yet here we propose a framework that aims to be useful in mainstreaming Good Modelling Practice (GMP) in the context of complex SES modelling settings. In the introductory article of this Joint Special Issue, Jakeman et al. (2024) argue that transitioning toward the widespread adoption of GMP requires enhancing reflexivity and transparency in SES modelling practices. As recognised by the authors, reaching this vision requires a whole-cycle perspective that addresses uncertainty while promoting stakeholder participation as focal points of GMP. Here we aim to contribute to such a purpose with a framework that aligns the dimensions of uncertainty and stakeholder participation into a single modelling cycle using SD. This synthesis may be useful for SES modellers in devising a modelling roadmap that explicitly engages with the implications of uncertainty in already complex participatory settings. In addition, the present framework could facilitate transparent communication regarding the model's *crafting*— that is, *how* an SES model is built and used (Jakeman et al., 2024). In other words, researchers and practitioners can use the framework to better communicate the rationale behind the decision points that drive the modelling process. Examples of suitable modelling tools and approaches are included across the different modelling phases to offer concrete ways to operationalise the framework.

2.2. DEVELOPING A UNIFIED SD MODELLING FRAMEWORK FOR SES POLICY EVALUATION

This section presents the approach to integrating uncertainty and participation in an SD modelling framework. Section 2.1 justifies this endeavour. Section 2.2 presents two modelling frameworks that engage with uncertainty and participation aspects separately. In Section 2.3, we conceptually align these frameworks into a unified framework, distinguishing three main modelling phases. Section 2.4 describes the iterative revision cycle that facilitates applying the modelling framework. Section 2.5 presents a summary of the modelling tools and techniques that can be deployed at different stages of the modelling cycle.

2.2.1. Why integrate uncertainty and participation in an SD modelling framework?

SD is deeply rooted in systems thinking and possesses both qualitative and quantitative attributes, making it well-suited to address the implications of uncertainty and participation in complex socio-ecological systems (Lane, 2010). The qualitative SD stream has a long history of building systems models with stakeholders, e.g. group model building and participatory modelling (Király & Miskolczi, 2019). This literature not only focuses on the output of SD participation (e.g. SD models) but also on the complexities of the process itself (Freebairn et al., 2019; Hovmand et al., 2012; Vennix, 1999), as well as on the transformative social outputs that may derive (Hovmand, 2014; Luna-Reyes et al., 2018; Rouwette et al., 2011). The quantitative SD stream focuses on using models to make sense of complex policy questions through numerical simulation (Meadows & Robinson, 1985; Sterman, 2002). Quantitative SD models are flexible in accommodating quantitative uncertainties in the form of parameter variations, and scenario and sensitivity analyses to make sense of possible futures in a complex and rapidly changing world (Kwakkel & Pruyt, 2013b; Moallemi, Kwakkel et al., 2020). Despite a mixed qualitative and quantitative modelling approach lying at the foundation of SD (Lane, 2010; Sterman, 2002), current SD socio-environmental practice evidences a lack of such integration (Moallemi et al., 2021).

Participation and uncertainty can seem very distinct in the SD practice, yet are closely related, impacting each other. Qualitative approaches usually have a rich understanding of a problematic situation but do not benefit from the possibility of testing desired policies (under uncertainty) in a simulation environment, assessing their impacts under myriad futures. Quantitative SD that engages with uncertainty assessment offers a rich vision of the uncertain future but it can be out of context if not discussed with stakeholders who want to use the model to answer difficult questions. A recent review of SD in the context of sustainable development provides further insights on the level of integration and identifies two requisite improvements (Moallemi et al., 2021): (i) more stakeholder participation is necessary, as more than 70% of SD sustainability applications do not include any form of participation; and (ii) SD could benefit from incorporating interdisciplinary perspectives, particularly by developing robust models that can explicitly deal with deep uncertainties about the future. These are important gaps that could be bridged by better integration of the qualitative and quantitative capabilities of SD, more specifically by aligning participation and uncertainty across the modelling process (Moallemi et al., 2023).

Recent advances in integrating SD approaches with other problem-structuring methods represent a way forward for an integrated multi-method perspective for policy analysis and decision-making (Rouwette & Franco, 2024). Similarly, here we propose a framework that aims to articulate the seemingly opposite features of participation and uncertainty in SD modelling. Perhaps the common thread between the two is to understand *ambiguity* as an essential dimension of uncertainty (Brugnach et al., 2008). Ambiguity implies that uncertainty is not only an issue of objective knowledge, but mostly about *whose* knowledge is considered. In short, considering ambiguity implies that uncertainty is also an issue of knowledge plurality. This is particularly relevant for SES-related issues, in which people and the environment are so intertwined that local stakeholders should have a say in modelling endeavours that could be used to inform policies that affect them (Amorocho-Daza et al 2024).

Such a broader perspective on uncertainty bridges model-related uncertainty, which focuses on quantitative aspects like parameters and scenarios, with participatory uncertainty that arises from the deliberation on the problem and potential solutions. Uncertainty, therefore, manifests in both the quantitative and qualitative dimensions of SD modelling practice, yet it is often treated separately. This paper aims to align and integrate these two dimensions into a coherent modelling cycle as a GMP. More specifically, here we contribute to normalising GMP by exploring the interactions between uncertainty and participatory aspects in SD modelling and how they can be understood and structured in the context of a modelling framework.

2.2.2. Two SD modelling frameworks incorporating participation and uncertainty

SD practice conceptualises the different stages in a structured, iterative modelling process as a cycle. It is only recently that uncertainty has been considered across the SD modelling cycle. This research builds upon the SD modelling cycle proposed by Auping (2018) which consists of a five-step cycle modifying the steps of Sterman (2000): (1) Problem articulation; (2) Conceptualisation; (3) Formulation; (4) Evaluation; and (5) Policy Testing. A strength of this new approach lies in incorporating uncertainty throughout this cycle. Figure 2.1a depicts the inclusion of uncertainty in SD modelling.

At the same time, a comprehensive participatory SD modelling framework should involve stakeholders throughout. This paper builds on the framework of Videira et al. (2010), as a relevant SD-based participatory framework in the context of sustainability. The Videira et al. framework considers various phases, including: (1) Scoping and abstraction; (2) Envisioning and goal setting; (3) Model formulation and confidence-building, (4) Simulation and assessment, and (5) Evaluating and monitoring. The approach identifies how the outputs of each phase become inputs for the next phases. For example, conceptual models (Mirchi et al., 2012; Purwanto et al., 2019) from the “Scoping and abstraction” phase can be useful in the “Envisioning and goal setting” phase. Figure 2.1b summarises the SD participatory modelling framework.

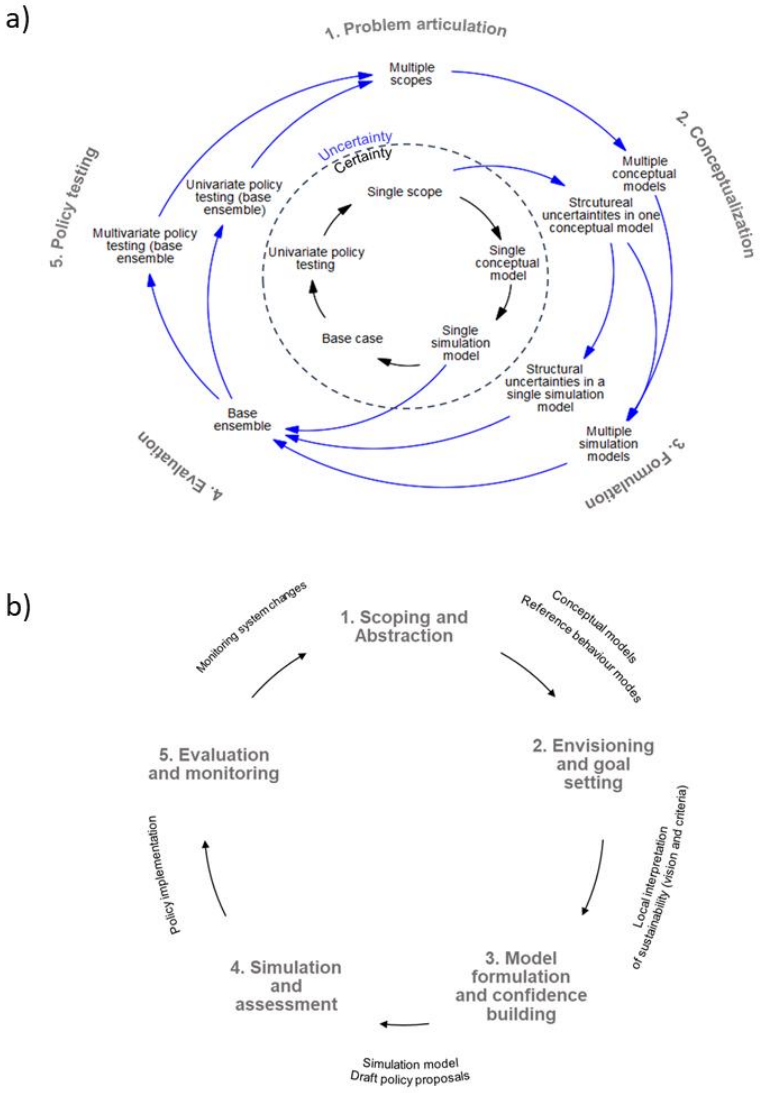

Figure 2.1 System Dynamics modelling frameworks for (a) considering uncertainty in the model development cycle (modified from Auping (2018)) and (b) implementing a participatory approach (modified from Videira et al., (2010))

This paper aligns the modelling cycle under uncertainty proposed by Auping (2018) with the participation stages identified by Videira et al. (2010). We integrate both approaches into a single modelling framework accounting for both participation and uncertainty analysis with two main variations. First, we adapt Auping's (2018) problem articulation phase by not only recognising multiple scopes in the modelling process but also aiming to integrate them into a negotiated scope that will guide the rest of the process. This aims to synthesise the various scopes that emerge within a multi-stakeholder setting (Ackermann, 2012; Rosenhead, 2006). Despite being challenging, this effort is consistent with a paradigm of dialogical learning in socio-environmental participatory modelling (Brugnach et al., 2011; Brugnach & Ingram, 2012). Second, we exclude the last stage of the Videira et al. (2010) cycle ("Evaluating and monitoring") as the current framework is oriented to policy decision analysis rather than the implementation and subsequent evaluation of policy actions in the environmental system itself. This simplification remains consistent with recognising the importance of linking model-based analysis with policy implementation, which becomes evident when viewing policy-making as a multi-stage iterative process (see Walker et al. (2003) and Sterman (1994; 2000, p. 88)). This article solely addresses the model-based policy cycle, which is nested within the broader policy implementation process.

2.2.3. A unified SD modelling framework aligning participation and uncertainty

A participatory modelling framework to integrate SD with other modelling tools to support SES policy decision-making under uncertainty is presented (Figure 2.2). By adapting and extending Auping's (2018) and Videira's (2010) approaches, this unified framework considers the implications of uncertainty and stakeholder participation in SES modelling. Figure 2.2 shows the unified modelling cycle and a suggestion of modelling tools that can be useful in each part of the cycle. The innermost part of Figure 2.2 highlights the structural implications of uncertainty (see Figure 2.1a). The traditional modelling cycle is shown in the inner ring. The outer ring represents the stages of stakeholder participation, mapped to both the modelling cycle and how uncertainty plays a role. The outer gears refer to potentially suitable tools, methods, and techniques that could be implemented at each stage. A detailed account of each of the cycle's phases follows. The dynamic interconnectedness between the (SD) modelling cycle and the stages of stakeholder participation is apparent in Figure 2.2, with each of the participation and modelling stages influencing and complementing each other.

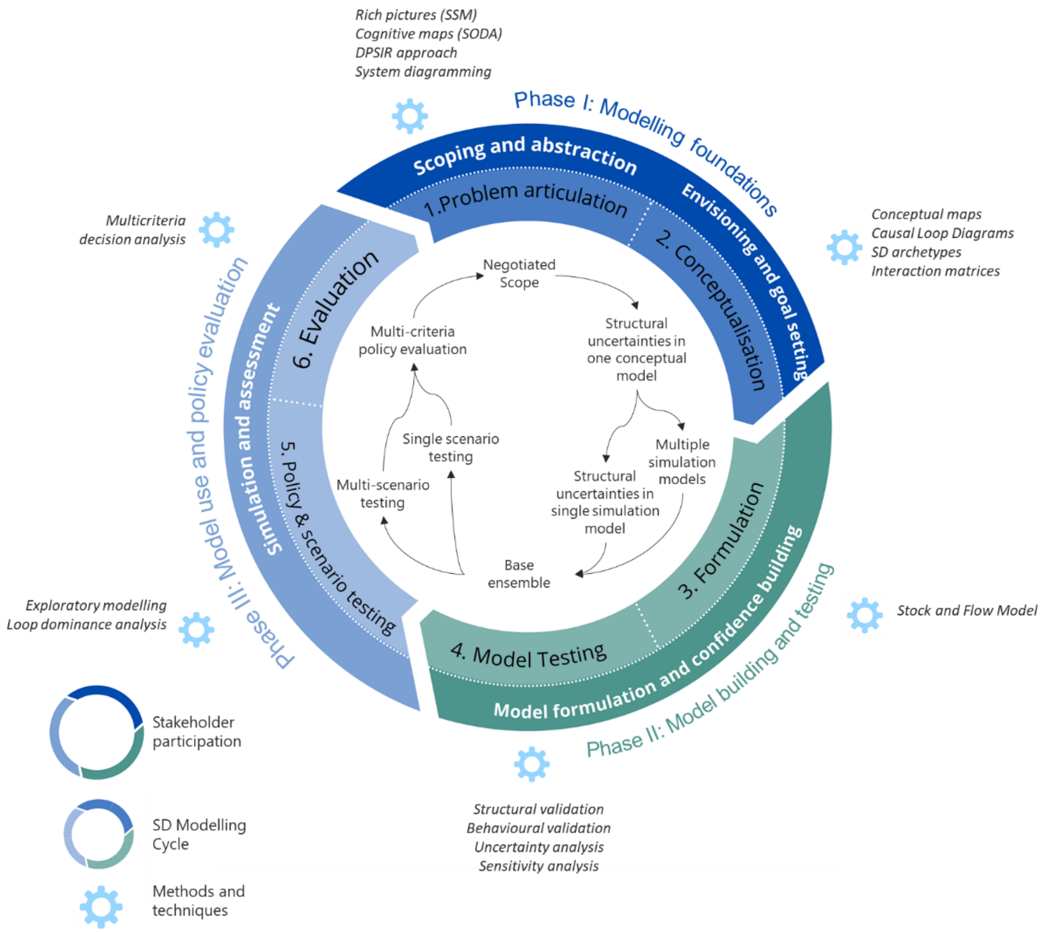

Figure 2.2. The unified SD modelling framework distinguishes three primary phases in the modelling cycle: Modelling foundations (dark blue), Model building and testing (green), Model use and policy evaluation (light blue) and comprises the modelling process cycle (inner circle) under uncertainty (circle centre) (adapted from Auping (2018), stakeholder participation (outer circle) (adapted from Videira et al. (2010)) and relevant modelling tools (outer 'gears').

Three distinct phases facilitate the analysis of the interactive nature of dealing with uncertainty and participation in SD. The first *modelling foundations* phase focuses on all the activities developed before starting a quantitative modelling exercise, such as defining and conceptualising the issues at hand, and defining desirable futures in which such issues are addressed. The second phase, *model building and testing*, deals with the intricacies of crafting and testing a quantitative model within the boundaries and purposes defined in phase one. The third and last phase*, model use and policy evaluation* involves the use of the quantitative model, particularly as a decision-support tool to test policy alternatives to deal with the issue identified in phase one. Below we present a detailed account of the activities in each phase.

The modelling cycle is an idealised version of logical, step-wise phases and stages that go from defining a problem to selecting a solution. A disclaimer is that implementing such a cycle will be *messy* in reality. In a practical case, we would expect feedback and reprocessing across and within the modelling stages. A second point of attention is that this cycle can only take place in a participatory setting where the interested parties (i.e., stakeholders and modellers) agree on the modelling process, an approach in line with what other authors have called a *dialogical learning strategy* (Brugnach et al., 2011). Agreeing on the process implies valuing the rationale behind the participatory modelling exercise; it does not mean agreeing on other specific aspects such as the problem definition or the key variables to be considered. However, not agreeing on the process also implies that parties who do not agree may need to withdraw from the modelling process and that engaging them will require other strategies (Brugnach et al., 2011). This practical implication demonstrates that a participatory modelling process occurs within a larger policy arena (see McEvoy, 2019, pp. 32-36).

A useful conceptualisation for the relevance of a participatory modelling process is to understand it as nested within a larger policy implementation cycle (Sterman, 2000; Walker et al., 2003). In other words, a policy recommendation arising from a modelling exercise can be ideally implemented and monitored, as part of a larger policy implementation cycle. SD researchers and practitioners have continuously pursued policy relevance since the field's inception (Buzogany, Kopainsky, & Gonçalves, 2024; Forrester, 1994, 2007; Ghaffarzadegan, Lyneis, & Richardson, 2010). Here we aim to contribute towards this overarching objective by proposing a structured approach to support socio-environmental policy-making. Yet, our proposed framework does not offer a guide for policy implementation, monitoring, and evaluation. Understanding this limitation is crucial for building bridges between policy evaluation and application. Hence, our proposed three-phase framework can better elucidate the essence of what a modelling exercise can offer within a larger socio-environmental policy setting.

Phase I: Modelling foundations

During phase I, the modelling foundations are established. Here we first describe the corresponding stages of the SD modelling cycle under uncertainty, followed by the parallel modelling stages using a participatory modelling perspective.

SD modelling under uncertainty (inner circle and centre in Figure 2.2)

Two stages are relevant for establishing modelling foundations using a modelling cycle under an uncertainty approach: problem articulation, and conceptualisation.

Problem articulation

The problem articulation phase defines the model's purpose (Sterman, 2000) or "aims at articulating the central problem which needs to be researched" (Auping, 2018, p. viii), and determines the next phases in the cycle. Auping (2018) highlights that as SD usually deals with *wicked* problems, multiple scopes (potentially leading to multiple models) could be suitable for studying a problem under uncertainty. The involvement of various stakeholders with differing perspectives in socio-environmental debates often results in multiple, sometimes conflicting, framings and scopes, leading to ambiguity (Dewulf et al., 2005), another dimension of uncertainty (Brugnach et al., 2008). Brugnach et al. (2011) identified various strategies that can be used to cope with such ambiguity: rational problem-solving, persuasion, dialogical learning, negotiation and opposition. From the previous list, a participatory modelling approach is primarily consistent with a dialogical learning strategy, as it assumes that stakeholders have a legitimate interest in co-producing a model in an active dialogue with their counterparts (Amorocho-Daza et al., 2024; Videira et al., 2010). Accordingly, a stakeholder dialogical strategy facilitates a transition from multiple problem scopes to a joint problem definition (i.e. negotiated scope) that is meaningful and relevant for the participants in the overarching process (Brugnach et al., 2011). In this paper, we consider that problem articulation under ambiguity (a dimension of uncertainty) involves integrating multiple scopes (i.e. Auping (2018)) rather than simply adopting a single scope (i.e. Sterman (2000)) into a negotiated scope arising from stakeholder dialogue (Ackermann, 2012; Rosenhead, 2006).

Conceptualisation

The conceptualisation phase's primary aim is to identify the main relations between key variables, which often build on the mental models of stakeholders and experts (Auping, 2018), demonstrating a clear link to stakeholder participation. Mental models are "internal representations of external reality that people use to interact with the world" (Jones et al.,2011, p. 1). During conceptualisation, mental models are translated into tangible conceptual representations. Tools like conceptual maps and causal loop diagrams help visualise and characterise system relationships (Ford, 2010; Voinov et al., 2018). However, collaborative conceptualisation is a complex process itself (Luna-Reyes et al., 2006). To cope with it, SD practitioners use *scripts*, as a "tool for helping facilitation teams visualize and solve problems in the design of group model building sessions" (Hovmand et al., 2012, p. 183). Hovmand et al. (2012) initiated a collaborative platform, *Scriptapedia*, to document and expand the group model-building practice in the SD community. An example of the deployment of a group model-building script in the context of a contested socio-environmental system is described in Purwanto et al. (2019).

Modelling scripts are helpful in structuring the use of complementary tools to support system conceptualisation with multiple stakeholders. Choosing among which tools to use varies on a case-to-case basis, but the display of such tools needs to happen in a structured and purposely crafted way. Tools and techniques useful to support the conceptualisation process include the use of SD archetypes (Mirchi et al., 2012); interaction (or causal) matrices (Sanò, Richards, & Medina, 2014); and more general methods such as participatory scenario planning and SWOT (Strengths, Weaknesses, Opportunities, Threats) analysis (Barnaud et al., 2013; Ritzema et al., 2010; Voinov et al., 2016).

The conceptualisation phase deals with uncertainty in different dimensions. First, as conceptualisation follows problem formulation, it is also subject to the ambiguity derived from the presence of multiple stakeholders, and therefore, multiple mental models (Brugnach et al., 2011). Thus, participatory system conceptualisation is not a deterministic task; rather, it depends on the participants involved, the facilitation method employed (e.g. scripts), and notably, the interaction among stakeholders throughout the process. Knowledge, or epistemic, uncertainty is also present due to the limited knowledge about the system components and their interactions. Conceptualisation is the first stage in which structural uncertainties (i.e. how model variables are connected/related) are identified to be assessed in later stages of the modelling cycle. This is evident in discussions about the polarity and global relevance of certain variable relations and feedback loops in causal loop diagrams (CLDs) (Sanò et al., 2014). This challenge can inspire early conversations on how to account for *scenarios* (e.g. changes in external influences, new variables, relations and influences to consider) from the beginning of the modelling cycle. In sum, embracing ambiguity and knowledge uncertainty implies the recognition of going beyond a *deterministic*, to a *structurally uncertain* conceptual model.

Stakeholder participation (outer circle, Figure 2.2)

Two stages are relevant for establishing modelling foundations using a participatory modelling approach: scoping and abstraction; and envisioning and goal setting.

Scoping and abstraction

The scoping and abstraction phase relates to the problem articulation and conceptualisation stages of the modelling cycle. Scoping or framing a problematic socio-environmental situation is not an objective exercise (Dewulf et al., 2005; Dewulf, Craps, & Dercon, 2004), as it deals with important value-related questions that will frame the rest of the modelling process (Amorocho-Daza et al., 2024; Gregory et al., 2020). Abstraction-related activities coincide with the conceptualisation stage in the modelling cycle, as they depart from a problem situation and aim to translate stakeholders' mental models of the problem into qualitative models based on a deliberation process (Vennix, 1999). At this stage, uncertainty is evidenced by the ambiguity arising from multiple problem frames, as well as by the limited knowledge regarding the variables (and their interaction) that are relevant to understanding the problem itself. Problem structuring methods (PSMs) are helpful to engage with the aforementioned complexities.

PSMs are qualitative approaches to making sense of ill-structured problems (Smith & Shaw, 2019), i.e. problems arising from "situations characterized by multiple actors, differing perspectives, partially conflicting interests, significant intangibles and perplexing uncertainties" (Rosenhead, 2006, p. 762). PSMs have been established in the management literature for over 40 years (Mingers, 2011; Smith & Shaw, 2019), and they remain relevant across several fields of research and practice (Gomes Júnior & Schramm, 2021; Mingers & White, 2010). The three most established PSMs are Soft Systems Methodology (SSM), Strategic Options Development and Analysis (SODA) and the Strategic Choice approach (Ackermann, 2012; Wright et al., 2019). However, other frameworks such as DPSIR (Drivers, Pressures, State, Impact and Response) (Bell, 2012) and systems diagramming (Enserink et al., 2022; van der Lei et al., 2011) can be deployed as PSMs. A recent review indicates that the primary applications of PSMs are found in the business management domain, while environmental applications are relatively scarce, comprising only 17% of the reported case studies in academic literature (Gomes Júnior & Schramm, 2021). Relevant applications of PSMs in complex socio-environmental problems have been reported using different methods: SSM (Bunch, 2003; Suriya & Mudgal, 2012), SODA (Elsawah et al., 2015; Hjortsø, 2004), and DPSIR (Gregory et al., 2013; Wantzen et al., 2019).

Scoping and abstraction activities often result in building qualitative system representations that give a rich understanding of the problem under consideration. Examples of these representations are the *rich pictures* of SSM (see Bunch (2003)) or the *collective cognitive maps* of SODA (see Elsawah et al., 2015) or the *systems diagrams* of policy analysis (Enserink et al., 2022; van der Lei et al., 2011). These representations can be integrated with other tools as part of a broader modelling cycle (Elsawah et al., 2015; Howick & Ackermann, 2011; Nijmeijer, 2018; Rodriguez-Ulloa & Paucar-Caceres, 2005). For instance, a PSM (e.g. SSM) can be incorporated into the SD modelling cycle in the shape of qualitative system representations such as causal loop diagrams (Paucar-Caceres & Rodriguez-Ulloa, 2007). Building such qualitative models has several benefits, including (a) adding rigor to the analysis and discussion; (b) scoping a concise and shared understanding of a problem; (c) serving as a group memory of participatory sessions (Vennix, 1999). However, both individual and group dimensions are sources of 'messy problems' in participatory modelling settings (Vennix, 1999). This is why successful participatory modelling sessions with stakeholders should have structured planning, often in the form of SD scripts (Andersen & Richardson, 1997; Hovmand et al., 2012; Luna-Reyes et al., 2006), and facilitators with the right set of attitudes and skills (Vennix, 1999) to arrive at an adequate scoping and abstraction of a socio-environmental problem in the form of a meaningful system representation (Sterling et al., 2019).

In summary, scoping and abstraction activities within a participatory process offer a meaningful approach to navigating the complexity of socio-environmental problems and the ambiguity arising from the multiple possible frames to define them. By engaging with a rich problem definition, abstraction follows as a way to a shared understanding in the form of a qualitative system representation. This highlights the essential connection between an active stakeholder dialogue and uncertainty (i.e. ambiguity), even before defining the first equation of a quantitative environmental model.

Envisioning and goal setting

After scoping a problem and advancing in its conceptualisation, the engagement of stakeholders in SES will focus on envisioning and goal setting. The purpose of this phase is to develop shared visions of the future of the system, and discuss sustainability criteria (Videira et al., 2010). Visioning engages with uncertainty by connecting current trends to (un)desired futures (Slinger et al., 2023). Conceptualisation can go beyond current issues by considering future trends. This activity can take place in stakeholder workshops and encourage the use of conceptual maps and CLDs as a common ground to discuss and set priorities for later stages of the modelling cycle (e.g. formulation and evaluation). This stage might be the starting point to ask questions related to simulation capabilities (e.g. what will the simulation model be able to measure?) and policy opportunities (e.g. which kind of policy interventions are associated with a desired future?). Discussions around the concept of sustainability may reflect stakeholders' values and preferences. A discussion around sustainability could help stakeholders to clarify "*what* they want to sustain and *for how long*" (Stave, 2010, p. 2765). Sustainability policies can have goals that reflect diverse values, such as integration, anticipation, precaution, participation, and equity (Gasparatos, El-Haram, & Horner, 2008). The deliberation should end with a context-specific interpretation of sustainability (Videira et al., 2010). This will be crucial for later stages of the modelling cycle (i.e. policy - scenario testing and evaluation).

Phase II: Model building and testing

During phase II the model is built and tested. Here we first describe the corresponding stages of the SD modelling cycle under uncertainty, followed by the parallel modelling stages using a participatory modelling perspective.

SD modelling cycle under uncertainty (inner circle and diagram, Figure 2.2)

Two stages are relevant for building and testing the model using a modelling cycle under uncertainty: formulation, and model testing.

Formulation

During model formulation, the model is specified mathematically and simulated using computational tools (Auping, 2018; Sterman, 2000). An SD simulation model is a quantitative representation of the variables and relations that were identified during conceptualisation with each variable and relation specified in terms of stocks, flows and associated parameters. The resulting system of non-linear first-order differential equations, visualised as a Stock and Flow Model (SFM), is solved numerically to yield graphs of the time varying dynamics of the complex system (Banks & Slinger, 2011; Ford, 2010 - Part I). Much of the SD literature covers the conceptual and mathematical foundations that underlie the relations between a system's stock structure and the dynamic behaviour emerging from it (Ford, 2010 - Part I; Meadows, 2008 - Part I; Naugle, Langarudi, & Clancy, 2024; Sterman, 2000 - Parts II-V). Conceptual models serve as a guide to formalising a quantitative model of a system. The translation process can be challenging because not every detail of the conceptualisation can be formally quantified. Therefore, model formulation requires balancing the complexity reflected in the CLD with the simplicity needed for quantification (Amorocho-Daza et al., 2024). Freebairn et al. (2019) propose a conceptual framework focused on this challenge.

Formulating an SD modelling exercise under uncertainty builds on the structural uncertainties identified during the first phase of the modelling cycle. More specifically, having a structurally uncertain CLD that reflects the complexity of the issue and the intrinsic ambiguity of its conceptualisation has implications for the formulation phase. Auping (2018) highlights that structural uncertainty can be captured in single or multiple models derived from a structurally uncertain CLD. A single quantitative simulation model can incorporate uncertainty in different ways. As the modelling exercise enters the quantitative realm, a quantitative assessment of uncertainty becomes possible, for example, in the form of statistical and scenario uncertainty. Statistical uncertainty may be captured by variations in models' parameters (Ford & Flynn, 2005; Kwakkel & Pruyt, 2013a). Scenario uncertainty uses scenarios to capture diverging trends or driving forces beyond the model's scope, such as climate change or socioeconomic pathways (Moss et al., 2010). Maier et al. (2016) highlight conceptual and practical aspects of considering statistical and scenario uncertainty in the context of modelling under deep uncertainty. Furthermore, considering multiple models might be an option when having a single simulation model is unfeasible or undesirable. This, however, comes at the cost of increasing the analysis complexity at later stages in the modelling cycle, such as when using the model for decision-making.

Model testing[1]

The model testing phase aims to increase stakeholder confidence in the model (Forrester & Senge, 1980), further demonstrating the link to stakeholder participation. During this phase, various tests are performed to check whether the model has been correctly constructed (i.e., verification) and whether it is fit for purpose (i.e., validation) (Auping, 2018). Not limited to statistical validation, structural and behavioural validation tools are suitable for this purpose (Forrester & Senge, 1980). After testing, the model is acknowledged to reproduce the general behaviour of the system. This is often done via a base modelling run representing the behaviour of the variables of interest over time.

Incorporating uncertainty during model testing involves moving beyond a base case, to a base ensemble. A base case results from a single (deterministic) run of the simulation model for specific variables of interest. In contrast, an uncertainty approach considers a base *ensemble*, a set of bundled simulations, that capture multiple modelling runs (from individual or multiple models) that represent a broad spectrum of system trajectories to understand the range and distribution of the output variables of interest when the parameters and structure of the model are considered stochastic (Auping, 2018; Bankes, 2002; Kwakkel & Pruyt, 2013a; Maier et al., 2016). Figure 2.3, illustrates how a system can have different trajectories in different scenarios, but there is statistical uncertainty associated with each of the trajectories, illustrated by the translucent bands (Figure 2.3). In sum, instead of simulating a single future, an ensemble of model runs (in different scenarios) is a relatively straightforward way to represent several possible futures based on a wide range of model outcomes and scenarios. In addition to the aforementioned uncertainty analysis, performing a sensitivity analysis can provide additional insights into the main factors (parameters) which drive the model's overall output uncertainty (Saltelli et al., 2019). Sophisticated methods for testing the sensitivity of the models' outputs to changes in (non-linear) graphical functions are also available (Eker, Slinger, van Daalen, & Yücel, 2014).

[1] A change in the terminology from Auping (2018) is proposed: "Model testing" is proposed instead of "Evaluation". This change is similar to the terminology originally proposed by Sterman (2000). The term "Evaluation" is used for the last stage in the modelling cycle in *Figure 2.2*.

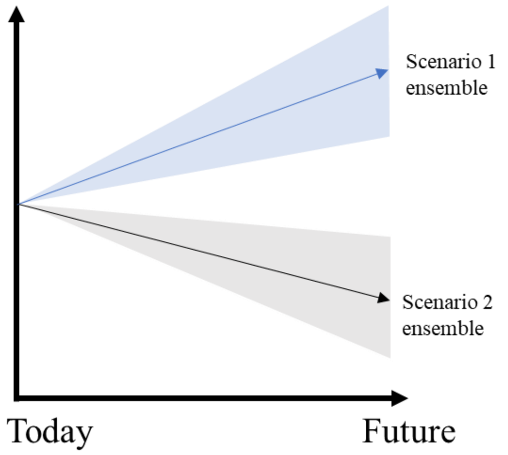

Figure 2.3. Conceptual representations of statistical and scenario uncertainty, the two trends represent diverging scenarios with increasing uncertainty over time (shaded area), modified from (Maier et al., 2016)

Having a range of plausible values for a variable of interest provides richer modelling results to stakeholders when compared with a single model-run approach. This might enable discussions to determine if such a range is consistent with real-world expectations or available data, thereby providing an opportunity for exploring empirical grounds for validating the modelling outputs. Nonetheless, this expected benefit comes at the cost of communicating quantitative uncertainty features to a diverse stakeholder audience, likely unfamiliar with such jargon and thinking (van der Bles et al., 2019). Another point of attention arises when only ranges are considered in the ensemble. In such a case, the system's intrinsic behaviour may be difficult to observe or may even be obscured. This practice may limit the observation of recurrent modes of dynamic behaviour (e.g. a system that exhibits oscillation vs. goal seeking), important in SD practice (Mirchi et al., 2012; Sterman, 2000).

Participatory modelling cycle (outer circle, Figure 2.2)

There is one relevant stage for building and testing the model using a participatory modelling approach: model formulation and confidence building.

Model formulation and confidence building

During the formulation and model testing phases, the qualitative models developed with stakeholders are translated into simulation models while gaining confidence in their capabilities (e.g. see model testing section above). Collaborative SD development may foster learning, co-production of knowledge and development of innovative solutions (Videira et al., 2010). However, this process involves a tension between complexity (the system described in causal loop diagrams) and simplicity (i.e. feasibility of creating a simulation model based on such diagrams) (Barreteau et al., 2014; Freebairn et al., 2019). Freebairn et al. (2019) propose a structured process to guide this translation process illustrated with a case study. In a context with uncertainty, stakeholders and analysts should agree on how to best estimate parametric uncertainty, as well as possibly formulate various simulation models that account for structural uncertainty.

Stakeholders can increase their confidence in the simulation model by providing input about a model's *structure* and its capabilities in representing the *behaviour* of the real system. Freebairn et al. (2019, p. 16) also highlight the importance of this phase and describe it as a process of "engaging with and communicating the model's results". Slinger (2023) provides a detailed and practical illustration of validation activities in an SES participatory modelling setting. The model testing process can go beyond a deterministic assessment and integrate the dimension of uncertainty by going from a base case to a base ensemble (described above). Stakeholders can discuss the quantitative effect of uncertainty in terms of output variables' ranges, but qualitative understanding can also be valuable, for example, by assessing possible changes in the system's modes of behaviour due to parametric variation in some of the simulations (Kruseman Aretz, 2023; Nava Guerrero, Schwarz, & Slinger, 2016).

Phase III: Model use and policy evaluation

During Phase III, the model is put to use and can support policy evaluation. First, we describe the corresponding stages of the SD modelling cycle under uncertainty, followed by the parallel modelling stages using a participatory modelling perspective.

SD modelling cycle under uncertainty (inner circle and diagram, Figure 2.2)

Two stages are relevant for model use and policy evaluation using a modelling cycle under an uncertainty approach: policy and scenario testing, and evaluation.

Policy and scenario testing[2]

At this stage, the model is used to test policies under different scenarios. Policies are intended changes to modify the system's performance (Sterman, 2000). Scenarios represent exogenous visions of the future, for example in terms of climate change and/or socio-economic development (Enserink et al., 2022; Moss et al., 2010). Scenario thinking is increasingly critical in the discussion around SES (O'Neill et al., 2020). Therefore, accounting for the performance of policies in various climate and societal pathways provides a better understanding of the system's exogenous sources of uncertainty (Wu, Elshorbagy, & Alam, 2022). Following the Maier et al. (2016) conceptualisation, engaging with both endogenous and exogenous sources of uncertainty requires estimating endogenous uncertainty on top of different scenarios (see Figure 2.3). This allows for estimating initial base ensembles (see Phase II, model testing). In other words, it is possible to consider various scenarios (exogenous uncertainty) and explore their associated parametrical variability (endogenous uncertainty).

At the policy testing stage, the base ensemble can be compared with a 'policy ensemble' to explore how activating policies may affect system behaviour compared to the base ensemble scenario. This relatively straightforward approach is known as the "*design of experiments*" in the exploratory modelling literature. However, the exploratory modelling perspective offers a repertoire of approaches that combine different strategies based on the assumptions made about the decision and uncertainty space (Moallemi, Kwakkel, et al., 2020). This perspective actively

[2] A change in the terminology from Auping (2018) is proposed: "Policy and scenario testing" is proposed instead of "Policy testing" to explicitly account for the interaction between policies and scenarios.

engages with exploring robust policies—those that perform well regardless of deep uncertainty conditions (Kwakkel, Walker, & Haasnoot, 2016; Moallemi, Kwakkel, et al., 2020).

Other analytical approaches can be useful at the policy testing stage. For example, system interventions, i.e., policies, can be designed based on the leverage points proposed by Meadows (1999). Policies can be designed to deal with shallow (at the level of parameters and feedback structures) or deep (at the level of design and intent) leverage points (Abson et al., 2017). Shallow points can be easily incorporated into an SD model. For example, some authors have suggested that policy testing can be made more systematic and automated by evaluating changes in multiple parameters to achieve a desired outcome (Auping, 2018; Moallemi, Kwakkel, et al., 2020). A more efficient approach according to Meadows would be to focus on dominant feedback loops that drive the system's behaviour. Tools such as "Loops that Matter" facilitate this identification to focus on policies that directly intervene in dominant loops to obtain the desired effects on the system (Schoenberg, Davidsen, & Eberlein, 2020). Deeper leverage points in the domain of design (e.g. rules of the system and information flows) are effective in creating wide system changes (Abson et al., 2017), and may potentially be incorporated in a quantitative SD model. However, despite the potentially transformative efficacy of the deepest leverage points, they remain difficult to capture in policies. Formulating a co-creation dialogue in which increasingly deeper leverage points are considered, and incorporated into SD model structures, would likely provide learning opportunities on ever-effective changes in SES.

Evaluation[3]

In the evaluation phase, policies are evaluated and ranked according to stakeholders' priorities. Contrary to Sterman (2000), the current framework decouples policy testing from policy evaluation. Practicality is one reason, as there is already substantial complexity in the stage of *policy and scenario testing* (under uncertainty). Second, the evaluation phase resembles a decision analysis or decision-making process that can be done outside the SD simulation environment (Slinger & van Daalen, 2003). The Multi-Criteria Decision Analysis (MCDA) approach is useful for addressing the challenge of structuring decision-making problems with multiple alternatives, indicators and objectives, particularly in complex socio-environmental settings (Amorocho-Daza et al., 2019; Giove et al., 2009; Lahdelma, Salminen, & Hokkanen, 2000). Indeed, the SD approach can provide a simulation environment where it is possible to go from *what if* policy exploration towards more structured decision-making (Phan et al., 2021). Coupling SD models with MCDA methods represents a research opportunity to expand SD capabilities in complex decision-making problems (Elsawah et al., 2017; Phan et al., 2021; Zomorodian et al., 2018). SD can be merged with MCDA by, for example, deriving indicators and scenarios directly from the SD models while estimating relevant criteria and their weights in discussion with stakeholders. Yet, relatively few case studies showcase this integration in fields such as water resources management (Elshorbagy, 2006; Momeni et al., 2021; Xi & Poh, 2014) and solid waste management (Gul & Haydar, 2024).

[3] This phase was not included in the framework of Auping (2018). The term "Evaluation" is inspired by Sterman (2000), who proposes to include "policy formulation and evaluation" in a single phase.

Performing the evaluation stage under uncertainty adds a layer of complexity to the decision-making process. While traditional MCDA assumes deterministic conditions to structure the decision-making process, recent advances examine how it is possible to consider uncertainty in the evaluation of policies (Durbach & Stewart, 2012). Some literature case studies illustrate how to incorporate quantitative uncertainty in MCDA using stochastic parameters (Scholten et al., 2015) and scenarios (Lienert et al., 2015; Ram, Montibeller, & Morton, 2011). This stage of the modelling cycle offers the opportunity to incorporate the estimated quantitative uncertainties from the SD model through both statistical and scenario analyses into a structured decision-making approach. It is noteworthy that when undertaken within the SD modelling cycle under uncertainty process, such a structured and computationally based decision analysis does not necessarily involve multiple stakeholders.

Stakeholder participation (outer circle, Figure 2.2)

There is one relevant stage for model use and policy evaluation using a participatory modelling approach: simulation and assessment.

Simulation and assessment

During the final phases of the participatory SD modelling cycle that concentrate on policy and scenario testing as well as evaluation, stakeholders are involved in assessing the outcomes of model simulations. This contrasts with the SD modelling cycle under uncertainty where the inclusion of stakeholders in the evaluation phase is optional. However, Lahdelma et al. (2000) consider assessing policies against multiple criteria a difficult task that requires structured approaches and stakeholder participation. Accordingly, in the simulation and assessment phase, SD modelling can be combined with other methods so that the performance of policy initiatives in complex socio-environmental settings can be evaluated effectively. As described above, multi-criteria decision analysis techniques help bring structure to the evaluation process (Videira et al., 2010). Likewise, considering long-term diverging system trajectories can help to account for structural uncertainty in policy evaluation (Moss et al., 2010). For instance, recent modelling efforts have assessed critical socio-environmental issues considering multiple socioeconomic and climate scenarios (Alizadeh, Adamowski, & Inam, 2022; Graham et al., 2020). Integrating scenarios in a MCDA support framework is an approach that promises to improve policy evaluation (Ram et al., 2011), and can help stakeholders to prioritise amongst many potentially feasible options.

Here, the criteria and priorities identified in the *Envisioning and goal-setting* stage are operationalised to serve as criteria for assessing the performance of the different policies considered. The engagement of stakeholders across the model-building process may increase their confidence in the model outcomes. This, in turn, is expected to increase the likelihood of using SD models as decision-support tools to engage in desired policy paths (Stave, 2010). Participation is therefore key for moving from concept to action in SD applications (Stave, 2002). Participatory modelling may enhance not only stakeholders' general understanding of the system but, more specifically, their awareness about the likely outcomes of policy changes that they help to design and test. Despite the achievement of these potential benefits being a desired outcome of the participatory process, engaging stakeholders up to this stage remains a challenging endeavour as it requires acknowledging and addressing both the methodological and social complexity of the participatory modelling process (Voinov & Bousquet, 2010).

By bringing people together to identify relevant criteria to assess policies, uncertainty appears again in the form of ambiguity. Here we move back into the realm of values and subjective priorities that dominate the early stages of the modelling cycle (Amorocho-Daza et al., 2024). Recent research suggests that enlarging the value and time spectrum (e.g. considering intra-intergenerational justice) in complex socio-environmental issues has several implications for model-based decision-making (Jafino, Kwakkel, & Taebi, 2021). Interestingly, even a single decision-maker can show inconsistency in ranking criteria and alternatives when they are framed distinctly (Tversky & Kahneman, 1981). Recent research shows that the framing effect also occurs when stakeholders establish priorities for environmental problems with a focus on future generations (Kuroda et al., 2021). Therefore, the estimated criteria weights after deploying an MCDA are inherently uncertain (Durbach & Stewart, 2012). Exploring policy ranking variation via sensitivity analysis over the weights and reframing questions of value (e.g., in terms of benefits vs. losses or present vs. future generations) can offer insights and trigger discussions about which policies seem to be more desirable and *robust* in the face of uncertainty.

2.2.4. Iterative revision

The 6-step SD modelling cycle (inner circle, Figure 2.2) can itself be understood as an adaptive feedback cycle or iteration (Figure 2.4). In the first iteration of the cycle (blue arrows, Figure 2.4) all phases from problem articulation to evaluation are executed, after which a second iteration (dashed green arrows, Figure 2.4) can be considered as a refinement guided by stakeholders' feedback and evaluation. The main purpose of the initial cycle is to build and gain confidence in a model (or a set of differently structured models) that accurately represents the behaviour of the system. The revision iteration focuses on enhancing the capabilities of the model(s) as a decision-support tool. Therefore, the emphasis of the revision cycle is on adapting the simulation model(s) to test new policies or incorporate new insights of interest to the stakeholders. Often, the revision is quicker than the initial cycle, as it does not focus on conceptualisation nor on model testing. Instead, it is assumed that the changes proposed by the stakeholders can either be included fairly easily in a revision of the model(s) or can be simulated directly in the existing model(s). This is a plausible assumption as the overall structure of the complex social-ecological issue is already systemically captured in the simulation model(s). Likewise, an extensive model testing phase is not required because (i) each model has already been tested and (ii) the stakeholders' confidence in the capabilities of the model(s) is already established.

The iterative steps described above can be understood through the concept of *self-reference* (Hofstadter, 1979). That is, the modelling process evolves in reference to itself. This idea becomes clearer once we distinguish between a problem situation and a problem-solving system (van Daalen & Bots, 2010). For instance, a problem situation can be an SES issue; such as water pollution affecting human and environmental health. In contrast, a problem-solving system could be formed by actors that participate in a policy analysis process and how they interact among themselves and with models of the issue; for example, this could involve a riparian community, local public servants, and a group of modellers collaborating to develop a model for supporting decision-making about water quality and health concerns. The self-reference perspective helps to highlight that it is the people engaged throughout the modelling cycle who ultimately determine the process's outcomes. Here we show how such interactions can converge to support stakeholders to identify and commit towards ways that improve a socio-environmental problem.

Embracing a self-reference perspective is useful to clarify the close interaction between the *problem* and the *problem-solving system*. For instance, stakeholder interactions could have at least two main effects, one on the problem-solving system progress itself (i.e. modelling cycle), and the second on the *real-life* problem situation. This in turn may also change further interactions and discussions among stakeholders. There is therefore a mutual self-referencing between the *real-life* problematic situation and the concrete group of actors participating in a modelling cycle. According to Hofstadter (1979), understanding such complex self-referencing interactions can be aided by visual representations such as the M.C. Escher's paradoxes (e.g. *Print Gallery*, 1956 - Escher in het Palais (n.d.)).

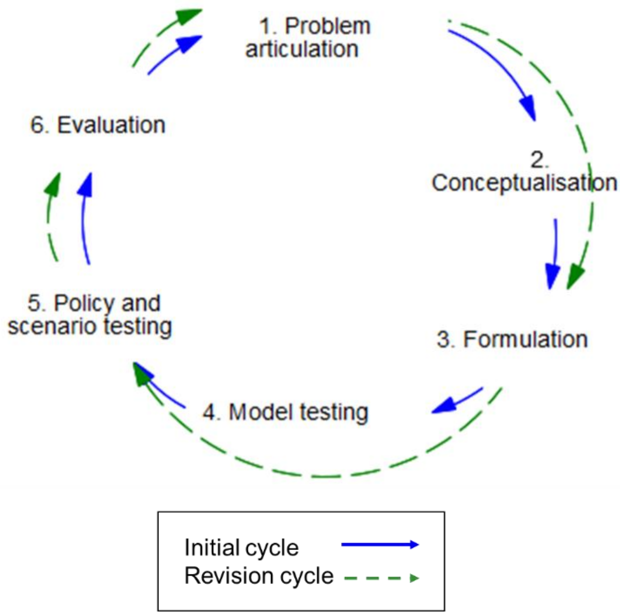

Figure 2.4 Initial and revision iterations of the SD modelling cycle

2.2.5. Methods and techniques (outer "gears", Figure 2.2) to support the process

In Table 2.1, we provide a detailed but non-comprehensive account of useful tools to support the modelling process, structured according to the three main modelling stages proposed in Section 2.3. The table provides specific information on where to implement the tools throughout the modelling stages, along with a brief description of each tool's features, as well as their benefits, limitations, and illustrative references.

Table 2.1 Modelling tools and techniques to support different phases of the unified SD modelling framework

Phase	**Methods/ techniques**	**Implementation stage**	**Description**	**General benefits**	**General limitations**	**Key references for applications**
I: Modelling foundations	Rich pictures (SSM)	- Stakeholder participation cycle: Scoping and abstraction. - SD modelling cycle: Problem definition	A *rich picture* is a pictorial overview that "portrays actors and elements in a problematic situation and indicates relationships among them" (Bunch, 2003)	- Promotes holistic thinking, as "pictures are a better medium than linear prose for expressing [multiple and interacting] relationships" (Checkland, 2000) - Rich pictures can be constantly upgraded based on the stakeholder's understanding of the problematic situation (Bunch, 2003)	- Making system pictures is a skill that comes naturally and easily to some people, while others may find it challenging (Checkland, 2000). Therefore, the role of facilitators is critical in developing system representations that reflect the *richness* of the problematic situation, including the perspectives of people with varying communication skills. - Getting from *messy* to *meaningful* system representations (i.e. rich pictures) and stakeholder discussion may require deploying other complementary SSM tools (see *root definition* and *CATWOE analysis* – Checkland (2000))	Bunch (2003) Suriya and Mudgal (2012)

Phase	Methods/ techniques	Implementation stage	Description	General benefits	General limitations	Key references for applications
	Cognitive maps (SODA)	- Stakeholder participation cycle: Scoping and abstraction. -SD modelling cycle: Problem definition	Cognitive maps are "a picture or visual aid in comprehending the mappers' understanding of particular, and selective, elements of the thoughts (rather than thinking) of an individual, group or organization" (Eden, 1992)	- Combining individual into collective cognitive maps is possible using specialised software tools (i.e. Decision Explorer) (Ackermann, 2012; Elsawah et al., 2015). - Collective maps offer the opportunity to observe collective convergent representations of a problem emerging from seemingly diverging points of view (Ackermann, 2012) - It is a transparent, and systematic approach helpful to connect qualitative to quantitative modelling approaches (Eden, 1988; Elsawah et al., 2015).	- Translating the mappers' narratives into maps can be overwhelming. This 'rich qualitative source' needs to be narrowed down according to the specific objectives of building the maps (Elsawah et al., 2015). - The maps are restricted by what people are willing to share. Inquiring about a controversial issue could generate resistance among the participants to explain their inner rationale and motivations (Elsawah et al., 2015).	Elsawah et al. (2015)
	DPSIR	- Stakeholder participation cycle: Scoping and abstraction.	It is a systems framework that explores the complex relationship between human and natural	- Facilitates a systemic understanding of the origins and consequences of	- Implementing the framework requires a deep and participatory understanding of the	Bell (2012) Gregory et al. (2013)

Phase	Methods/ techniques	Implementation stage	Description	General benefits	General limitations	Key references for applications
		- SD modelling cycle: Problem definition	systems through a conceptual understanding of interconnected Drivers, Pressures, State, Impact, and Responses (EEA, 1999).	environmental problems, explicitly incorporating feedback structures to do so. - Can be easily integrated with other methodologies to explore specific social-ecological problems in further detail. - Its simplicity facilitates co-creation and communication among various environmental stakeholders (e.g., policymakers, local communities, academics).	socio-environmental problem at hand. If multiple perspectives are not included, the analytical usefulness of the tool can be hindered. - The conceptual characterisation is flexible but might be insufficient to represent the complexity and cross-scale nature of environmental issues.	Wantzen et al. (2019) Zare et al. (2019)
	System diagrammin g	- Stakeholder participation cycle: Scoping and abstraction. - SD modelling cycle: Problem definition.	It is a simple system representation resulting from an iterative process of establishing a boundary and elucidating the following elements: external factors, internal factors and their relationships, means (or levers or steering factors), and measurable criteria (or objectives) (Enserink	- It focuses on analytical rigour, consistency and conceptual clarity - It is more easily transferable (e.g. to students and practitioners) than traditional PSMs. - Being easy to communicate and revise, "the approach	-It is consistent with a 'consultancy setting' with a clear problem owner. The allied XLRM framework has been used extensively in this way. The multi-actor version can become rather complex. -The approach is suited to demarcating an	Hidayatno, Rahman, and Muliadi (2015) Nijmeijer (2018)

Phase	Methods/ techniques	Implementation stage	Description	General benefits	General limitations	Key references for applications
			et al., 2022). The representation is built based on seven critical guiding questions (van der Lei et al., 2011). System diagramming is closely aligned with the XLRM framework (Lempert et al., 2003).	leads to an internally consistent systems model that represents the problem definition and delineation" (van der Lei et al., 2011, p. 1401)	appropriate system boundary, external factors, means and criteria, but must be deployed alongside other tools such as conceptual maps and CLDs to avoid a *black-box* model.	
	SD archetypes	- Stakeholder participation cycle: Envisioning and goal setting. - SD modelling cycle: Conceptualisation.	System archetypes are generic system structures showing common or generic patterns of behaviour over time (Mirchi et al., 2012; Wolstenholme, 2003).	- They synthesize "much qualitative and quantitative modelling effort cumulated over many years by many analysts", offering learning systemic opportunities in new problems and domains (Wolstenholme, 2003) - They represent a structured and *free-standing* way to understand the reason behind complex systems' counterintuitive behaviour (Wolstenholme, 2003)	- The archetype provides a starting assumption but it has to be empirically tested (Oberlack et al., 2019). - There might be specific conditions in which certain archetypes are applicable (or not) to a particular context (Magliocca et al., 2018)	Bahri (2020) Edwards et al. (2023) Gohari et al. (2013) Moallemi et al. (2022) Phelan et al. (2020)

Phase	Methods/ techniques	Implementation stage	Description	General benefits	General limitations	Key references for applications
	Causal matrices	SD modelling cycle: Conceptualisation.	Causal matrices help identify relationships and polarities among several variables (Sanò & Medina, 2012; Sanò et al., 2014).	- A causal matrix can be easily transformed into a causal loop diagram (Sanò et al., 2014) - Individual matrices can be aggregated into a *group* matrix that potentially leads to a shared causal loop diagram on the problem at hand (Sanò & Medina, 2012)	- A set of initial variables is a prerequisite of the method. These could be obtained with participatory scripts e.g. Nominal Group Technique (see Scriptapedia) - The method starts from a reductionist approach, so it might not help to develop a "system perspective" from the very beginning of the modelling process	Sanò and Medina (2012) Sanò et al. (2014)
	Conceptual maps	SD modelling cycle: Conceptualisation.	Conceptual maps are qualitative system representations that include key variables and describe how they are connected.	A comprehensive conceptual map can synthesize an important amount of information that can be very useful in later modelling stages. It provides a shared vision of the system as well as its main components and connections.	Building conceptual maps is a creative and open-ended process. Without facilitation, it can grow too complex to handle and connect with later modelling stages (Freebairn et al., 2019).	Purwanto et al. (2019) Sušnik et al. (2021)
	Causal Loop Diagrams (CLDs)	SD modelling cycle: Conceptualisation.	CLDs go beyond conceptual maps by characterising the causal relations among system variables. They are	CLDs provide important learning opportunities by allowing stakeholders to understand the	Just as with the conceptual maps, CLDs can easily grow too complex. Additionally, identifying polarities for every	Bahri (2020) Purwanto et al. (2019) Zhao et al. (2021)

Phase	Methods/ techniques	Implementation stage	Description	General benefits	General limitations	Key references for applications
			developed following a standard notation to describe the balancing or reinforcing influences between variable pairs, as well as feedback loops (Mirchi et al., 2012).	system as a whole and identify their key feedback relationships.	relationship adds a layer of complexity, as there are interlinkages where it is not easy, intuitive, or even possible to assign a polarity.	
II: Model building and testing	Stock and Flow Model	- Stakeholder participation cycle: Model formulation and confidence building. - SD modelling cycle: Formulation	Stock and flow models (SFMs; cf. Ford (2010)) aim to simulate a complex system. Often built based on CLDs, SFMs represent a quantitative effort to characterise the behaviour of complex systems.	SFMs can serve as platforms for policy experimentation to address complex issues. Stakeholders can use them to learn about complex systems (e.g., SES) and to further develop policy discussions around them.	SFMs are simpler than CLDs. Not all qualitative complexity can be integrated into a quantitative model. Additionally, some stakeholders may face difficulties when interpreting the quantitative results of a simulation model. Effectively communicating insights from quantitative models is a challenge for modellers.	Prasad et al. (2022) Turner et al. (2016)

Phase	Methods/ techniques	Implementation stage	Description	General benefits	General limitations	Key references for applications
	Structural validation	- Stakeholder participation cycle: Model formulation and confidence building. - SD modelling cycle: Model testing	It is a set of qualitative tests that aim to compare the structure of the real system with the structure of the model that represents it (Forrester & Senge, 1980).	Test various model features (see Barlas (1996)): - Parameter-confirmation: To establish the conceptual and numerical validity of model's the parameters (i.e. constants) when compared with the real system - Extreme condition: To anticipate the model's behaviour under extreme conditions (e.g. parameters and equations) and compare it with the expected real system's behaviour - Dimensional consistency: To check dimensional consistency in both sides of the model's equations	Identifying the 'real system' features can be problematic. For the parameter confirmation, available information about the system's variables might be limited, non-existent, or anecdotical. This also applies to extreme condition tests, for SES it can be simply too difficult to know what would be the real-life behaviour of a system if such extreme conditions have not been previously identified or measured.	Barlas (1996) Qudrat-Ullah and Seong (2010)
	Behavioural validation	- Stakeholder participation cycle: Model	It is a set of qualitative and quantitative tests to evaluate "the adequacy of	Test various model features (see Barlas (1996)):	The main limitation here is obtaining sufficient and reliable information that	Barlas (1996) Naderi et al. (2021)

Phase	Methods/ techniques	Implementation stage	Description	General benefits	General limitations	Key references for applications
		formulation and confidence building. - SD modelling cycle: Model testing	model structure through analysis of behaviour generated by the structure" (Forrester & Senge, 1980).	-Extreme condition: To compare the model's behaviour under extreme conditions (e.g. parameters and equations) and compare it with the expected real system's behaviour - Behaviour sensitivity: To identify changes in model's behaviour in response to changes in parameter values. - Behaviour reproduction. To check if the model is able to reproduce the real system's behaviour (e.g. symptom generation, frequency generation, multiple mode)	facilitates the comparison of the modelling results with the observed behaviour of the system.	

Phase	Methods/ techniques	Implementation stage	Description	General benefits	General limitations	Key references for applications
	Uncertainty analysis (UA)	- Stakeholder participation cycle: Model formulation and confidence building. - SD modelling cycle: Model testing	This umbrella term covers a set of tools that focus on characterising uncertainty in the model's output (Ghanem, Higdon, & Owhadi, 2017). Monte Carlo Methods (MCMs) are widely applied methods for model-based uncertainty analysis. MCMs can be used to do a quasi-random sampling of the parameter values using (often) pre-defined probability density functions (PDFs) to give a density distribution on the main outputs (Ford & Flynn, 2005).	SD simulation software already includes tools to perform uncertainty analysis. For example, quasi-random sampling is available on SD software (Stella, Vensim).	A good uncertainty analysis practice requires *global* analyses. That is, *global* uncertainty analysis methods test the output modelling uncertainty based on multiple simultaneous parameter variations. A local uncertainty analysis, in contrast, focuses on exploring a subset of factors or even parameters one at a time. This is a discouraged practice as it does not correctly represent models with non-linearities (Saltelli et al., 2019).	Ford and Flynn (2005) Kwakkel and Pruyt (2013b) Martinez-Fernandez et al. (2021)
	Sensitivity analysis (SA)	- Stakeholder participation cycle: Model formulation and confidence building. - SD modelling cycle: Model testing	This analytical tool aims to assess the impact of uncertain input factors on the overall uncertainty of the model's outputs (Saltelli, 2002). In other words, sensitivity analysis is useful to identify "which input factors	By performing sensitivity analysis, modellers can identify which factors (parameters) contribute the most or least to the model's overall uncertainty. This information might be useful to prioritise	Similar to the UA, SA should be performed using a *global* instead of a *local* approach. Global approaches consider the "factors' global influence in terms of their contribution to the variance of the model output, including the	Dai et al. (2024) Mai et al. (2020) Puy et al. (2021)

Phase	Methods/ techniques	Implementation stage	Description	General benefits	General limitations	Key references for applications
			contribute the most to model uncertainty". (Saltelli et al., 2019, p. 30).	resources to gather additional information that reduces uncertainty in the critical factors, while non-critical factors can have their values fixed (Saltelli et al., 2019)	effect of interactions among factors" (Saltelli et al., 2019, p. 31). In contrast, *local* approaches consider each factor individually using a 'one at a time' strategy. The latter approach is unsuitable for non-linear systems and under-explores the uncertainty space, particularly when several factors are considered.	
III: Model use and policy evaluation	Exploratory modelling	- Stakeholder participation cycle: Simulation and assessment. - SD modelling cycle: Policy and scenario testing	This approach offers various tools to operationalise modelling under deep uncertainty. To do so it follows a systematic approach to exploring the implications of the model's assumptions for decision-making. Not limited to quasi-random sampling, other experiments such as stress testing, worst-case scenario, and many objective optimisation can be integrated into a larger uncertainty-rich decision-	Exploratory modelling analyses can be done using open-source tools. The Exploratory Modelling Workbench is an open-source toolkit to develop exploratory modelling analyses (Kwakkel, 2017).	As more sophisticated analyses are developed, more effort and creativity are required for communication and promoting meaningful interactions between academic and non-academics.	de Haan et al. (2016) Kalra et al. (2015) Kwakkel et al. (2015) Moallemi et al. (2017)

Phase	Methods/ techniques	Implementation stage	Description	General benefits	General limitations	Key references for applications
			making framework (Moallemi, Kwakkel, et al., 2020).			
	Loop dominance analysis	- Stakeholder participation cycle: Simulation and assessment. - SD modelling cycle: Policy and scenario testing	Loop dominance evaluation assesses the relative importance of feedback loops in driving a system's behaviour (Ford, 1999). *Loops that matter* (LTM), is a recent numerical method for loops dominance evaluation. It defines loop scores to estimate each feedback loop's contribution to the model's behaviour, ranging from -1 to +1, also indicating the loop's polarity (Schoenberg et al., 2020; Schoenberg, Hayward, & Eberlein, 2023)	- Identifies the dominant feedback loops driving system behaviour. - Enables understanding of changes in feedback loop dominance over time. - Can highlight potential resistance or counterintuitive responses to new policies, aiding in better policy design and evaluation.	The loop dominance analysis adds complexity to the discussion of results. Explaining the related concepts and analysing the dominant feedback loops with stakeholders can likely generate further discussion. However, this exercise might be time-consuming and needs to be properly anticipated and scheduled.	Aboah and Enahoro (2022) Phaff et al. (2006)

Phase	Methods/ techniques	Implementation stage	Description	General benefits	General limitations	Key references for applications
	Multi-criteria decision analysis (MCDA)	- Stakeholder participation cycle: Simulation and assessment. - SD modelling cycle: Evaluation	MCDA methods support decision-making processes when several criteria and alternatives are considered (Lahdelma et al., 2000). They provide a systematic approach to assess the importance of criteria and how different policies perform against them (Hajkowicz & Higgins, 2008). Among others, widely adopted MCDA methods include Analytical Hierarchical Process (AHP), Technique for Order of Preference by Similarity to Ideal Solution (TOPSIS) and ÉLimination et Choix Traduisant la REalité (ELimination and Choice Translating REality) (ELECTRE).	- MCDA methods help evaluate alternatives considering multiple factors rather than a single metric/indicator. - They enhance stakeholder dialogue and confidence in policy selection as expert judgment is used to quantify the importance of different criteria.	- Different MCDA methods have varying approaches to weighing criteria. - As stakeholders are required again to give input at the final modelling stages, this may add to the stakeholder fatigue in a co-creation project - Using models under uncertainty requires adapting deterministic MCDA methods.	Antunes et al. (2006) Amorocho-Daza et al. (2019) Momeni et al. (2021)

2.3. CONCLUDING REMARKS

2.3.1. The dynamic relation between uncertainty and participation in model-based decision-making

Phase I: Modelling foundations

Dealing with a complex and problematic socio-environmental issue can be overwhelming, while making sense of intricate systems of people and nature, interacting at different scales remains an epistemological challenge. Trying to address the overwhelming knowledge uncertainty by acquiring more knowledge (e.g. more variables, measurements, and observations) is a strategy that can backfire. Various authors highlight the existence of an epistemology paradox: the more you know about a system the more complexity it exhibits (Brugnach et al., 2011; Brugnach et al., 2008; Dewulf et al., 2005). One way of dealing with this paradox comes from a clear definition of the problem using a systemic perspective. For instance, defining a relevant system boundary helps to better understand a socio-environmental problem (Nabavi et al., 2017). Defining what the issue is, in which geographical and time scale it operates, and who is part of it, are questions that can help to frame the problem to be explored. Addressing such questions is not an objective exercise. On the contrary, useful answers or approaches should be co-created with stakeholders relevant to the problem situation rather than derived by modellers only (Amorocho-Daza et al., 2024). Developing a participatory problem definition can help in tackling knowledge uncertainty.

Involving stakeholders in the modelling process can help in dealing with knowledge uncertainty, but inherently increases ambiguity, another dimension of uncertainty (Brugnach et al., 2008). Stakeholders that commit to a co-creation process agree to engage in a common process, but may continue to hold different perceptions, particularly concerning the problem definition and their system understanding. Indeed, ambiguity in perceptions can go beyond diversity in preferences regarding the system boundaries and extends deeply into the conceptualisation of the relations within such a system. Despite the diversity and range in perceptions associated with multiple stakeholders, expert knowledge and locally relevant experience from different stakeholders are useful at this stage to ensure that key variables and interlinkages are included and that the system boundary is well defined. Additionally, early participatory conceptualisation activities taking place after defining a relevant system boundary can help to lower ambiguity further by engaging again with knowledge about the system relations and structure. Interestingly, without participation, ambiguity is minimised but knowledge uncertainty can become overwhelming. With participation, despite ambiguity explicitly being present, more appropriate boundary conditions present opportunities to better deal with knowledge uncertainties about the relations within the system's boundaries.

Phase II: Model building and testing

It is primarily epistemological and ontological uncertainty that dominates this phase (Kwakkel, 2010; Wang, 2015). Epistemic uncertainty comes from the difficulty of translating the interlinkages found in conceptual models as mathematical equations in a simulation model. In this process, the modeller plays a crucial role, by making decisions about which simplifications, adaptations, and assumptions to apply based on the available data, system knowledge, physical constraints, and other factors. This translation process is a significant source of epistemic uncertainty. Ontological, or aleatory, uncertainty refers to the inherent variability observed in biophysical and social systems. For example, biophysical processes such as the nutrient and water cycles exhibit intrinsic stochastic behaviour. Socially driven trends can also be deeply uncertain, for example, socio-economic development pathways. Some of this variability can be quantified in the simulation model as statistical and scenario uncertainty.

Participation is central to the model testing stage. Here, key output variables of the model, in the form of scenario ensembles, can be estimated and compared with available field data and stakeholder knowledge. Quantitative estimation of uncertainty in the form of averages and ranges provides insight into the model's ability to accurately estimate the patterns, and trends observed in the real-world system. Even where information is available to quantitatively validate a model, stakeholders' input is essential for crosschecking whether the model behaviour aligns with their empirical knowledge about the system. In sum, testing activities help determine if the uncertainty inherent in the quantitative translation of the model is reasonable and appropriately captured for the socio-environmental problem at hand.

Phase III: Model use and policy evaluation

Towards the end of the first iteration of the modelling cycle, different sources of uncertainty have been accommodated into a simulation model. Utilising such a model to evaluate policies amidst uncertainty again necessitates the participation of stakeholders. The social use of the model as a decision support system means that ambiguity resurfaces. This occurs because pondering criteria and aggregating indicators brings forth value-based questions and links back to earlier modelling choices and stages such as envisioning. Moreover, human priorities are likely to fluctuate and be influenced by framing effects, such as focusing on gains versus avoiding losses, or prioritising present versus future generations (Jafino et al., 2021; Tversky & Kahneman, 1981). In short, the plurality of stakeholders' values and interests and the inherent bounded rationality of humans make ambiguity an inherent property of decision-making under uncertainty.

Paradoxically, the dialogue around the modelling outputs can help navigate ambiguity by using the model to help answer relatively simple *what if* policy questions, that is, performing *experiments* in a simulation environment (Zomorodian et al., 2018). Alternatively, the simulation model could also be integrated into a more sophisticated decision support system that helps to rank multiple policies against various performance criteria. Tools like MCDA offer the opportunity to integrate the simulation model outputs into a structured decision-making process that helps to rank the more desirable policies to pursue. However, considering uncertain modelling outputs adds complexity to the decision-making process and requires extending MCDA tools to handle uncertainty. This raises a further challenge, which is to communicate uncertainty to an audience of stakeholders and model users who may be unfamiliar with such concepts (van der Bles et al., 2019). Here it might be easier to start with a deterministic ranking of alternatives and later move on to explore how such ranking could alter in the face of uncertainty. This stepwise approach could offer opportunities to learn about the system's behaviour under diverse policies and changing scenario and parametric conditions.

2.3.2. Future directions for model-based policy analysis in SES

In this article we provide a framework in which we enunciate the main implications of uncertainty and participation for model-based decision-making in the context of SES. By integrating and aligning two SD relevant modelling cycles, one engaging with uncertainty and the other with participation aspects, we were able to distinguish three general modelling phases, namely: 1. Modelling foundations, 2. Model building and testing, 3. Model use and policy evaluation. Here we identify future research avenues that could emerge from deploying our unified SD modelling cycle.

For Phase I, there is potential to extend the use of problem structuring methods (PSMs) to enrich the problem definition in the modelling cycle of SES as a Good Modelling Practice (GMP). Although these methods are established in sectors such as business and health, environmental applications remain limited. Integrating PSM into co-creation methods is an active area of research (Cunningham et al., 2014; Slinger et al., 2023). Incorporating such activities into the SD modelling cycle could aid in articulating the problem more effectively and in building a richer problem definition (Rouwette & Franco, 2024). This would enhance the qualitative and quantitative models resting on such system understanding. Our focused attention on reviewing and articulating Phase I's methods is a practical contribution and an invitation for further research to address the currently largely deficient practice regarding problem scoping and participation in SES modelling (See Jakeman et al., 2024, Section 4, Points 1, 10, and 19).

While a holistic uncertainty approach is recognised as a topic for enhancing GMP (See Jakeman et al., 2024, Section 4, Points 4 and 6), doing so opens up new challenges for implementing Phase II. For instance, future research can help to validate whether quantifying uncertainty is helpful in the testing stage of SES simulation models. Here we hypothesize that this would be the case, as having a wide range of probable outputs for the variables of interest will yield more information about the range of variation that such variables exhibit in relation to existing field measurements or the experiential knowledge of stakeholders. However, a caveat is that presenting ensembles and ranges may obscure the observation of (dynamic and/or recurrent) modes of behaviour in these complex systems. More research is needed to elucidate the trade-offs in accounting for uncertainty at the model testing stage in practice. Future studies are needed to understand how to communicate such sophisticated numerical treatments of uncertainty to an audience that may be unfamiliar with the concepts and jargon of uncertainty assessment.

Despite progress in model-based policy evaluation (Phase III), the SES GMP literature can benefit from understanding how this process can be structured in practice in case studies that cover a variety of environmental issues at a diversity of spatial and temporal scales, and with variously composed stakeholder groups (See Jakeman et al., 2024, Section 4, Points 9 and 10). Moreover, we advocate further research on the integration of SD quantitative models into decision support systems, whether formal or informal process-based systems. These include applications linked to MCDA or multi-model systems combining different types of qualitative and quantitative modelling (see Slinger, 2023). Indeed, future studies could investigate the benefits and challenges of developing and applying environmental decision support tools that are deterministic versus explicitly accommodating of uncertainty, versus a combination of both.

Another potential benefit of our proposed framework is to facilitate SES modelling reporting and transparency as a GMP (see Jakeman et al., 2024, Section 4, Points 2, 13, and 17). Despite SES modelling case studies often exhibiting some of the generic modelling stages described in this article (e.g. scoping, envisioning, evaluation, etc), it is difficult to find applications that cover the whole modelling cycle. In fact, a recent participatory modelling review shows that not a single case study reported undertaking activities across all the modelling cycle stages (Voinov et al., 2016). This does not necessarily imply that these activities were not covered in practice, but may indicate that comprehensively reporting SES modelling activities in a single research article is extremely challenging. Our proposed 3-phase modelling cycle can aid with this scientific communication issue. Future case studies on SES modelling for policy evaluation could report in terms of the three modelling phases, describing the relevant activities undertaken in each phase, and making use of some of the proposed modelling tools or contributing other relevant tools. Moreover, the unified modelling framework can serve as a tool in designing a stakeholder and uncertainty-inclusive modelling process, that addresses each of the requisite activities in turn. Such future studies can also use the unified modelling framework to structure reporting on their challenges, lessons learned, and overall experience of employing a participatory modelling cycle (under uncertainty) to address a socio-environmental issue.

Nonetheless, the proposed framework has intrinsic limitations as it is tailored to modelling SES from an SD paradigm. Despite the extensive benefits that we have argued above, the SD approach has limitations as it is not spatially explicit and it focuses on modelling *wholes* rather than *individuals* or agents, both features being particularly relevant for ecological research (Vincenot et al., 2011). Therefore, adapting the proposed 3-phase framework to other SES modelling paradigms is an open avenue for future research. For instance, the framework can be adapted to model SES problems that are better addressed from an Agent-Based Modelling (ABM) perspective (e.g. see Bourceret et al. (2024)), or even, going a step forward, by integrating SD with Agent-Based-Modelling (ABM) in a single modelling cycle (Vincenot et al., 2011). Our framework can act as a baseline for such expansion and adaptation. Other modifications may come from applying the proposed SD-based framework in a more concrete type of SES (see Datola et al. (2022), for a relevant application in the context of urban resilience). This opens up ample possibilities for tailoring the overall SES framework to overarching themes, such as integrated resources management (Ghodsvali, Dane, & de Vries, 2022) and conservation science (Sala & Torchio, 2019); or to specific biophysical environments, such as coastal systems (Slinger, Taljaard, & d'Hont, 2020), river basins (Cabello et al., 2015) and forests (Fischer, 2018).

Finally, a synthesised SES policy evaluation modelling cycle opens the opportunity to connect better with policy application. Here we argue that an SES modelling exercise ideally consists of three global phases: it starts with a problem that is conceptualised, is later translated into a model that facilitates collective understanding, and is finally used to evaluate policies aimed at tackling the initially identified problem. This global understanding can simplify the dialogue between the stakeholders responsible for making SES policy decisions (or providing input) and those responsible for implementing and monitoring such policies (Nuno, Bunnefeld, & Milner-Gulland, 2014). More research is needed to capture the synergies and barriers that arise from a nested policy evaluation/implementation approach in dealing with socio-environmental problems. Case studies can illustrate what happens after a policy to tackle an SES issue has been selected via a participatory modelling process (see Slinger, 2023; Clifford-Holmes et al., 2018). Who are the decision-makers and implementers? Who is part of both groups, and how do they interact? How is the policy implemented and monitored? Does the implementation align with the recommended policy path supported by the modelling cycle? How does it differ? Does the monitoring feed back into another participatory policy evaluation cycle? Does the model suggest trends that became apparent in the real world after the implementation of policies? These questions open up an exciting avenue of research to understand the factors that support or hinder a fluid and collaborative SES policy evaluation and implementation process. A better understanding of these interactions can help to design and implement sound, concerted, and impactful policies to address the critical socio-environmental issues of our time.

3

ETHICAL IMPLICATIONS OF USING SD TO MODEL SOCIAL-ECOLOGICAL ISSUES

Based on the published peer-reviewed article:

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Abstract

The social-ecological systems (SES) approach elicits a broad understanding of some of the most pressing socio-natural challenges (e.g. resource scarcity, biodiversity loss, and climate change) and the responsibility that humans have in addressing them. System Dynamics has proven a powerful paradigm for dealing with complex SES-related issues. Here we discuss some ethical considerations of using System Dynamics (SD) to model SES, something that is often either overlooked or discussed as an isolated issue. Sustainable development and human rights are used as ethical standpoints across the modelling cycle, opening the discussion around guiding principles that need to be considered when modelling SES. Based on these, a set of guiding ethical questions are identified and classified across a participatory SD modelling cycle. This structured approach is a simple yet potentially useful tool for SD practitioners to examine the ethical implications of their modelling endeavours in the context of grand societal challenges.

3.1. INTRODUCTION

Questions of sustainability have been central to System Dynamics (SD) practice throughout the history of the field. World Dynamics (Forrester, 1971), and Limits to Growth (Meadows et al., 1972) were key in illustrating the impossibility of pursuing infinite economic growth and consumption on a finite planet (Meadows & Meadows, 2007; Randers, 2000), both of which used SD as the modelling paradigm. More than 50 years later, the sustainable development paradigm is now at the centre of the global political agenda (UN General Assembly, 2015), aiming to balance social, economic and environmental dimensions by taking an intergenerational perspective (Sachs, 2012; United Nations, 1987). SD continues to be relevant in addressing today's global sustainability challenges (Hjorth & Bagheri, 2006; Moallemi et al., 2021; Randers, 2000). For instance, a recent review identifies SD applications across all 17 United Nations Sustainable Development Goals (Moallemi et al., 2021). This reflects SD's strength to deal with a broad spectrum of "big issues" or "grand societal challenges" (Kwakkel & Pruyt, 2013; Lane, 2010), as varied as climate change, fisheries, biodiversity conservation, forest fires, water planning, air quality and waste management, among many others (Collins et al., 2013; Dudley, 2008; Fiddaman, 2002; Ford, 2010; Stave, 2010).

Efforts towards sustainable development should recognise the deeply intertwined nature of human and natural systems (Folke, Biggs, Norström, Reyers, & Rockström, 2016). This idea has been articulated in the concept of social-ecological systems (SES), understood as "interdependent and linked systems of people and nature" (Fischer et al., 2015, p. 145). SES are complex systems nested across multiple interacting scales (e.g. landscape, regional and global scales) embedded in the biosphere (Fischer et al., 2015; Folke et al., 2021; Preiser, Biggs, De Vos, & Folke, 2018). A sustainable development approach aligned with an SES perspective considers that "economic activities are part of the social domain, and both economic and social actions are constrained by the environment" (Wu, 2013, p. 1003). A SES approach is helpful to understand some of the most pressing current sustainability challenges, including climate change, environmental deterioration, and biodiversity loss, (Díaz et al., 2019; Folke et al., 2021; Nelson et al., 2006; Rockström et al., 2009). Designing and implementing policies that deal with such systems is a non-trivial and complex task (de Gooyert, Rouwette, van Kranenburg, Freeman, & van Breen, 2016; Kelly et al., 2013).

The long-term and holistic focus of SD is helpful in addressing SES-related issues (Elsawah et al., 2017; Kelly et al., 2013). SES interventions often rely on simplistic and short-term perspectives leading to policy choices that are ineffective in reaching their intended objectives, e.g. due to policy resistance; or may even have serious unintended consequences (Collins, de Neufville, Claro, Oliveira, & Pacheco, 2013; Pahl-Wostl, 2007; Sterman, 2006; Sterner et al., 2006). These issues stem from the human-bounded rationality to understand the feedback structures that drive complex systems' behaviour (Sterman, 2002). SD simulation tools offer an alternative to these practices by aiming to capture the systemic structure of an SES problem to subsequently explore its long-term pathways under diverse policy actions (Dixson-Declève et al., 2022; Ford, 2010; Sterman, 2000).

Despite the aforementioned capabilities, here we argue that using SD in the context of complex global socio-environmental challenges has implications in terms of participation and ethics. In this paper, we use an SES lens to explore these implications. The social element of SES calls for participation, and a participatory process opens up ethical concerns and dilemmas. Despite our analysis focuses on the interface between human and environmental systems, our insights remain relevant for SD applications in the context of wider issues (e.g. health, education, migration, climate change, etc.). Therefore, this article's approach of integrating participatory and ethical dimensions does not limit it to SES but can be extended to a wider set of grand societal challenges (e.g. achieving the SDGs).

The social dimension of SES highlights the importance of developing SD models in a participatory manner. Engaging stakeholders is imperative as they are either interested or affected parties in addressing an SES problem (Király & Miskolczi, 2019; Stave, 2002). SD participatory approaches, like group model building, have a long tradition in organisational contexts and may offer important insights in the context of SES (Hovmand, 2014; Luna-Reyes et al., 2006; Rouwette, Vennix, & Mullekom, 2002). Yet, SD participatory approaches in socio-environmental issues remain relatively sparse (Stave, 2002, 2010; Videira, Antunes, & Santos, 2009; Videira, Lopes, Antunes, Santos, & Casanova, 2012). For instance, a recent review shows that 70% of peer reviewed SD applications in the context of sustainable development do not report any form of participation (Moallemi et al., 2021). SD practitioners should address the issue of stakeholder participation as a general calling to socio-environmental modelling approaches (Voinov & Bousquet, 2010; Voinov et al., 2016). This sparseness reiterates our ambition of making the principles in this paper with its focus on SES applicable to other modelling foci.

Second, modelling people and the environment goes beyond technicalities and has ethical implications. The SD modelling cycle often gathers modelers and stakeholders with the task of translating "real-world" issues into conceptual and quantitative models to use them to support policy choices or discussions (Freebairn et al., 2019). This process is a complex social construction that is far from objective (Vennix, 1999). It involves activities of judgement, prioritisation, pondering, negotiation and simplification. Hence, SD models incorporate the values and worldviews of the persons that take an active role in their development (Palmer, 2017). This raises ethical implications that should be explicitly considered and discussed (Nabavi, Daniell, & Najafi, 2017; Pruyt & Kwakkel, 2007). Ethically transparent SD modelling approaches are therefore necessary in the context of SES and beyond.

This article, though using SES as a lens, aims to be useful as an entry point for SD modellers to consider participation and ethics in the context of grand societal challenges. More specifically, we focus on the often-disregarded ethical dimension of SD modelling, illustrated through a socio-environmental lens. We argue that a structured ethical reflection can take place in the context of any general SD participatory modelling process. We propose two ethical standpoints for SD applications in SES, namely sustainable development and human rights. These standpoints rely on important principles that are helpful to guide SD practice and can be operationalised in the form of ethical questions. In short, the practice of translating relevant guiding principles as a collection of ethical questions should be considered across any SD participatory process to explore the ethical implications of modelling the complex socio-environmental challenges of our time.

3.2. AN ETHICAL LENS TO MODEL SES

3.2.1. Ethics and System Dynamics

Ethics is a branch of philosophy that reflects on morals (Kirchschlaeger, 2021), being concerned with what is morally good and bad, right and wrong (Singer, 2022). In simpler words, ethics is "a general concept referring to the way we think about normative issues" (Ormerod & Ulrich, 2013, p. 293). Ethics questions what ought to be (i.e. to what end, on what grounds and why) "in a rational, logically coherent, methodological-reflective, and systematic way" (Kirchschlaeger, 2021, p. 31). Therefore, an ethical stance can be used to systematically scrutinize the values underlying human actions as well as their consequences (Ormerod & Ulrich, 2013; Rachels & Rachels, 2019). An applied ethics perspective is fundamental to discern the practical implications of current (and future) human endeavours, and to question and help to shape them (Kirchschlaeger, 2021). Applied ethics has been used in multiple problems and disciplines (Chadwick, 2012), yet few works have taken such perspective in SD and related fields (e.g. operational research) (Ormerod & Ulrich, 2013; Pruyt & Kwakkel, 2007; Rauschmayer, Kavathatzopoulos, Kunsch, & Le Menestrel, 2009). The present article takes an applied ethics perspective by looking at the practice of SD modelling from an ethical lens.

Ethics is pervasive across the SD practice. This is evident explicitly and implicitly in various ways: in SD theory and practice (e.g. due to SD pragmatic focus on what an issue "ought" to be (Nabavi et al., 2017)); in SD models (e.g. as constructs embedding values of their crafters (Palmer, 2017)); in the professional conduct of SD practitioners (e.g. System Dynamics Society Code of Conduct (System Dynamics Society, 2019)); and in SD institutions (e.g. System Dynamics Society's mission and vision (System Dynamics Society, 2023)). However, the discussion around the ethical considerations of SD remains limited and implicit. Few authors have raised concerns about this, and a more open discussion about the ethical dimension within SD is needed (Palmer, 2017; Pruyt & Kwakkel, 2007).

Perhaps one of the most important realizations to start deliberating about the ethical implications of SD models is to understand them as "engineered" artefacts (Olaya, 2014). As such, models are built with a purpose (Olaya, 2016) and are not neutral (Katz, 2011). They rather are ethically charged entities (Palmer, 2017), embedding the values and worldviews of their crafters. In this line, Palmer (2017) asserts that the moral value of an SD model is evident through the consequences of its practical use (e.g. policy design and implementation). Nevertheless, is important to realise that the ethical implications of an SD model would depend on the extension of the system boundary that it represents. A system boundary can define a fairly simple system that does not raise important ethical concerns, but as a boundary extends to consider socio-environmental elements, ethical considerations become more critical (Nabavi et al., 2017). The above raises important implications for any participatory approach, with both modellers and stakeholders taking an active and deliberative role during the model building process.

3.2.2. Ethical motivations for sustainable development and SES

There are various implicit or explicit motivations for taking a sustainable development perspective to manage SES. A first example can be understood as an ethic of survival. Early SD practitioners illustrate such concepts by suggesting that unsustainable resources exploitation might cause future population overshoot and collapse (Forrester, 1971; Meadows et al., 1972). More recently, leading earth scientists have raised similar arguments suggesting that large human-driven environmental changes are surpassing life supporting "planetary boundaries", posing an existential threat to human civilization (Folke et al., 2021; Rockström et al., 2009). Extreme scenarios such as these challenge both human and other life forms and raise ethical concerns regarding the responsibility and care that humans should have towards the preservation of life from an intergenerational perspective (Berti, 2014).

Beyond survival, various authors have highlighted that justice and human dignity need to be considered in the context of sustainable development. Leach et al. (2018) argue that a SES perspective demonstrates the intertwined nature of equity and sustainability. Along a similar line, Gupta et al. (2023) propose that planetary boundaries should consider justice and aim to reduce harm, increase basic resources access, and challenge inequalities from an intergenerational perspective. To do so, human rights should protect human dignity by setting a minimum level of access to critical resources for people now and in the future (Gupta et al., 2023; Kirchschlaeger, 2020). Nevertheless, ethical implications arising from considering the value of non-human nature need more attention and could also be considered in a larger framework of "biosphere" responsibility (Folke et al., 2016; Jax et al., 2013).

From the above follows that SES issues are not just "environmental", and that their "social" element has many ethical implications. Recent reports recognise this issue by pointing out that the deep drivers for environmental change lie in people's values and behaviours (Díaz et al., 2019). Donella Meadows already argued in a similar way by pointing out that deep levers to intervene in a system lie in the dimensions of design (i.e. social structures and institutions) and intent (i.e. values, goals and worldviews) (Abson et al., 2017; Meadows, 1999). Therefore, as Chan et al. (2020, p. 694) summarised: "transformative change towards sustainable pathways requires more than a simple scaling-up of sustainability initiatives—it entails addressing these levers and leverage points to change the fabric of legal, political, economic and other social systems".

Reflexive SD practice should engage more explicitly with such ethical motivations and deep drivers of change (i.e. values and behaviours) in the context of contemporary grand social challenges. Simulation models can become spaces where abstract ethical concepts (i.e. justice) are considered in a more tangible way. For example, considering the issue of intergenerational justice with respect to access to resources, an SD model can be an entry point to discuss questions such as how to avoid potential long-term policy maladaptation? or how to define "minimum levels of resources" for different groups and across generations? In that way, models can be used as deliberation spaces of desirable futures.

3.3. Two ethical standpoints in SD modelling of SES

In dealing with social-environmental challenges, a sustainable development paradigm is not value-neutral (Holden, Linnerud, & Banister, 2017). First, it relies on principles such as intra/intergenerational justice and the precautionary principle (Paterson, 2007; Spijkers, 2018). Hence, designing sustainability policies requires an open deliberation about these principles in order to operationalise them (Karlsson, 2007; McDermott, Mahanty, & Schreckenberg, 2013). Second, as humans are at the centre of sustainable development, human rights need to be considered in the context of sustainability (Kirchschlaeger, 2021). This is evident since the UN General Assembly (2022) recently recognised the human right to "a clean, healthy and sustainable environment", implying that every human being is a right holder, but also a duty-bearer towards a sustainable environment. Thus, an ethical lens is necessary to guide sustainability practice (de Vries, 2019; Holden et al., 2017; Leach et al., 2018). Here we propose sustainable development and human rights as general ethical standpoints for coping with SES, and particularly for using SD to model SES.

3.3.1. Sustainable development

Sustainable development has its ethical roots in various principles, most evident of which is arguably the principle of intra- and intergenerational justice which strive for equality or equal treatment of humans within and across generations (Kirchschlaeger, 2021). Intergenerational justice demands that each generation should consider succeeding generations "to satisfy their needs, to avoid serious harm and to have the opportunity to enjoy things of value" (Thompson, 2010). Reaching intergenerational justice implies addressing the issue of justice in the present generation (i.e. intragenerational justice) (de Vries, 2019; Sen, 2011) while focusing on today's children as a generational bridge with future generations (Berti, 2014; Thompson, 2010). This continuum is necessary to achieve transformational pathways of "equitable sustainability" (Leach et al., 2018). It is noteworthy that various authors have used the concept of equity, both intra- and intergenerational, as strongly related to justice in the context of sustainability (Leach et al., 2018). Notable applications can be found in the fields of conservation (Klein, McKinnon, Wright, Possingham, & Halpern, 2015; Schreckenberg, Franks, Martin, & Lang, 2016; Zafra-Calvo et al., 2017), payment for ecosystem services (McDermott et al., 2013) and resources management (van der Zaag, 2007).

Connected to intergenerational justice is the precautionary principle (Raffensperger & Tickner, 1999), as implying that "we should avoid activities that we have reason to believe could do serious harm to either present or future people" (Thompson, 2010). In the context of SES, the precautionary principle enables the adoption of preventive action to protect both humans and the environment when stakes are high in the face of uncertainty (Bourguignon, 2016; Kriebel et al., 2001). Such a broad definition makes the principle's operationalisation a matter of intense academic and even legal debate, usually held at the national and international spheres (Garnett & Parsons, 2017; Paterson, 2007). However, a more widespread implementation of the principle should start moving towards more local and specific contexts (European Environmental Agency, 2013). Likewise, the EEA (2013) warns that addressing current and future controversies around the precautionary principle should learn from past mistakes, as there is an already extensive list of cases where preventive action failed to protect human and environmental health.

The long-term focus of SD can be used to reflect on the implications of considering sustainability principles in the context of current grand societal challenges. Here the "umbrella" concept of sustainability considers many sectors and issues related with economic, social and environmental systems (e.g. SDGs). SD models can be used to assess how the policies of today may have irreversible impacts for future generations (e.g. loss of health, poverty traps, persistent pollution, species extinction). Specific simulation models (e.g. assessing the potential long-term health impacts of pollution) can add up to improve the contextual understanding of complex trade-offs of benefits and risks not only within but also across generations.

3.3.2. Human rights

Human rights set a minimum standard to protect human dignity (Kirchschlaeger, 2016, 2020), relying on the principles of freedom, equality and justice (Kirchschlaeger, 2013). It therefore has elements overlapping with those of sustainable development (i.e. justice). Article 1 of the Universal Declaration of Human Rights states that "All human beings are born free and equal in dignity and rights. They are endowed with reason and conscience and should act towards one another in a spirit of brotherhood". Among its characteristics, universality is perhaps human right's strongest attribute, as it "entails that humans are human rights holders and that their human rights need to be respected, protected and realized" (Kirchschlaeger, 2021, pp. 160-161). Human rights are therefore an ethical common ground for every human being and human endeavour (Kirchschlaeger, 2016).

Scientific progress encompasses human rights. As an essential part of human existence, human rights protect scientific enquiry, ensuring academic freedom and serving as a fundamental point of reference for scientific practice (Kirchschlaeger, 2013). SD literature is part of the wider context of scientific progress (Forrester, 1987) and therefore is subject to human rights considerations. System Dynamicists are expected to protect human rights by: (1) respecting human rights; (2) contributing to the realization of human rights; and (3) setting priorities according to human rights. These duties can take a negative or positive outlook: by doing or by omitting something in order to contribute to the realisation of human rights.

SD practice should consider explicitly the ethical standpoint of human rights, but this makes more sense on a case by case basis. SD practitioners should ask themselves how their SD project can be linked to a specific human right in a positive (e.g. models that help understand how to improve the quality of education) or negative outlook (e.g. models that help understand how to prevent biodiversity loss). Considering human rights in SD also implies an invitation to move away from the role of "neutral modeller" towards an "activist modeller". The latter role has an explicit ethical stance (e.g. based on human rights or sustainable development) regarding a particular issue and uses modelling as an analytical tool to convey a message to change the situation around the problem at hand (Voinov, Seppelt, Reis, Nabel, & Shokravi, 2014).

3.4. PARTICIPATORY SD MODELLING AND ETHICS IN SES

3.4.1. The need of SD participatory approaches to model SES

The social element of SES calls for promoting participatory modelling approaches. Participation is not a new concept in SD; on the contrary, it can be traced back to the discipline's early stages (Lane, 2010). However, traditional SD group model building often involves a "client" group, resembling a consultancy setting (Andersen & Richardson, 1997). In such a context, Vennix et al. (2000, p. 379) argues that participation can bring several benefits which include: (1) "to capture the required knowledge in the mental models of the client group"'; (2) "to increase the chances of implementation of the model results", and (3) "to enhance the client's learning process". However, a broader set of perspectives in favour of participation can be considered when dealing with socio-environmental issues (Norström et al., 2020; Voinov et al., 2014). For instance, recent research shows how participatory modelling can enrich the understanding of complex social interconnections in the context of local sustainability transitions (Szetey, Moallemi, & Bryan, 2023). Or how integrating local knowledge in formal SES assessments recognise the stakeholders' role in a socio-environmental issue and foster reflection about the impact of their actions, priorities and visions (Norström et al., 2020; Rodríguez, Reu, Bolívar-Santamaría, Cortés-Aguilar, & Buendía, 2023).

Current SD participatory practice promotes wider stakeholder participation in the context of social change and environmental management. This has been done by promoting democratic participation and social capital strengthening to favour transparency and deliberation in a multi-stakeholder debate around socio-environmental issues (Király & Miskolczi, 2019; Stave, 2002). However, adopting such an approach brings up ethical considerations arising from the interaction between modellers and stakeholders, for example, in terms of power, justice and knowledge (Jordan et al., 2018; Norström et al., 2020). Recent community-based SD initiatives have taken a proactive approach to empower marginalised communities in dealing with complex social problems (Gallagher et al., 2020; Hovmand, 2014; Király & Miskolczi, 2019; Trani, Ballard, Bakhshi, & Hovmand, 2016). Participatory SD modelling has been used in the context of transdisciplinary environmental management and policy (Stave, 2010). This article builds on the latter approach as it explicitly deals with the interaction between society and environment in the context of a public policy debate.

In this context, Videira et al. (2010) proposed a participatory modelling cycle aiming to be implemented in the context of environmental assessment and decision making. This framework involves the following phases: (1) Scoping and abstraction; (2) Envisioning and goal setting; (3) Model formulation and confidence-building; (4) Simulation and assessment and (5) Evaluating and monitoring. This SD based framework promotes continuous stakeholder participation to learn about SES and deliberate about policy alternatives to sustainability problems. Here we build on the Videira et al. (2010) framework and use it to explore the ethical implications of modelling SES, taking human rights and sustainable development as standpoints.

Ethical considerations can be identified all across a participatory modelling cycle. Starting from the scoping and abstraction phase, stakeholders need to be able to meaningfully engage in the process and be able to question and define the limits of the issues at hand. They should have a voice in envisioning desired future(s) for the system. Their role can be key in validating the model's structure and outputs. Towards the end of the modelling cycle, stakeholders should be able to use simulation outputs as a starting point for discussing policy options. The following section examines these considerations in more detail, while explicitly considering human rights and sustainable development.

3.4.2. Asking ethical questions in SD SES participatory processes

Asking ethical questions is a practical approach to integrate ethics into SD modelling practice. Ethical questions inquire about values and responsibility, particularly regarding conflicting notions of the good (Ormerod & Ulrich, 2013). Nabavi et al. (2017) point out that defining boundary conditions in SD requires ethical judgement as it not only deals with the question of what "is" but also of what "ought" to be. They also reflect that the latter should be explicitly done within an ethical framework (e.g. sustainability principles). Pruyt and Kwakkel (2007) and Palmer (2017) offer a set of ethical questions to guide the implementation of a System Dynamics assessment. However, it is important that these questions are asked following a logical order. To aid the aforementioned issues, this article proposes the classification of these questions across the SD participatory modelling cycle in the context of SES using two ethical standpoints.

Asking ethical questions is relevant for the SD stakeholder participation cycle across a broad range of modelling studies as they deal with matters that affect people and the environment. Here the ethical standpoints of human rights and sustainable development need to be explicitly considered in the context of SES.Table 3.1, shows some relevant ethical questions to be examined across SD applications in the context of SES while considering the two central standpoints in this paper. The proposed set of questions is general and therefore not exhaustive. Rather it is meant to be a starting point to promote a discussion about the ethical implications that emerge across the SD modelling cycle and for a range of SD modelling studies in areas beyond SES. More tailored questions will likely rise while discussing ethical concerns in specific SD applications around SES. The following sections offer a detailed discussion of the questions applicable within each phase of the participatory modelling cycle as proposed by Videira et al. (2010) .

Table 3.1 Guiding ethical questions across a System Dynamics framework. Questions in bold are newly proposed in this paper, with the remainder from various sources: Pruyt and Kwakkel (2007), Palmer (2017)**, (Stave, 2010)†*

Scoping and abstraction	**Envisioning and goal setting**	**Model formulation and confidence building**	**Simulation and assessment**
• **Who matters? *** • **What matters? *** • **What time horizon matters? *** • **What are the boundaries of the system/model to be considered? *** • **What is the time frame considered? *** • **Who participates? *** • **Whose world-view, value system, perspective, and interests are taken into consideration?** • **Who decides from what perspective? *** • **What is the role of the analyst? ***	• What dimensions are considered important? * • Do the participants/stakeholders determine the dimensions to be considered? * • **What is "sustainability" in the specific context for different stakeholders?** • What do the stakeholders want to "sustain" and for how long? † • **Is there agreement regarding vision of a desired "sustainable future" among stakeholders? If not, whose perspective is more visible? Why?** • **Who might be positively or negatively impacted if this vision is reached?** • **Are the visions of a "sustainable future" intra/intergenerationally just? How are future stakeholders (i.e. children, coming generations) considered in it?**	• Have the modeller/analyst made all possible input to the model as objective as possible?** • How have the modeller/analyst introduced bias into the model? ** • **How is the modeller/analyst communicating such bias to the stakeholder group and the general public?** • **How is the modeller/analyst reflecting about his/her own motivations, worldviews and goals are incorporated in the model?** • How accurate is the representation of society in the model? ** • **Does the model reflect the structure found in the real-world?** • What other design options are possible? **	• Will the model be used to develop policy? ** • What is the level of uncertainty (robustness)? ** • What will the policy do to society if the causal assumptions in the structure are wrong? ** • Have the modeller/analyst communicated the uncertainty to decision-makers? ** • Will the policy developed from the model create harm for society if the assumptions are indeed incorrect? ** • Does the policy produce the good for which It was intended? ** • Are there foreseeable unintended side effects? ** • Do the side effects of implemented policy indicate that the model design is inaccurate? **

Scoping and abstraction	Envisioning and goal setting	Model formulation and confidence building	Simulation and assessment
	• Do the visions of a "sustainable future" prevent potential harm? • Does the envisioning and goal setting potentially infringe human rights? • Does the envisioning and goal setting phase take active responsibility in human rights protection? How? - by respecting human rights? by contributing to the realization of human rights? by setting priorities according human rights?	• Does the model reflect the behaviour of the real-life problem/system based on a selected set of indicator values? • Do stakeholders agree to use the simulation model based on its capabilities to balance the complexity reflected in the conceptual maps and the simplicity required for quantification?	• How to weigh the criteria and assess the performance of policy options? (who selects the criteria? Why? To what end?) • How can stakeholders incorporate the insights of the "envisioning and goal setting" to inform their decision-making process?

Scoping and abstraction

Practical questions such as delimiting the problem or system in space and time requires ethical judgement (Nabavi et al., 2017), especially when dealing with complex SES, but also applicable to other systems. Likewise, determining who will participate and their motivations is necessary to have a wider understanding of the worldviews that will be embedded in the model. Yet, having a reflection of perspectives that are excluded from the modelling process is useful to be aware of the model's limitations. This reflection may highlight the need to include new participants. New stakeholders can be considered, for instance, based on human rights (e.g. people whose rights might be potentially affected by policy outcomes of the modelling process (Gallagher et al., 2020)) or sustainable development (key stakeholders who are potentially responsible for (un)sustainable outcomes in SES (Videira et al., 2012)). The analysts' self-reflection about their role and motivation is key in this process. Although neutrality is often a desired quality, modelling is not objective. That is why analysts should identify and be reflexive about their own motivations, worldviews and goals and question how they bring them into the modelling exercise (Ives, Freeth, & Fischer, 2020; West & Schill, 2022).

Envisioning and goal setting

This phase's ethical challenges relate to the definition of criteria and the system's vision(s) of the future, something critically important when dealing with SES and many broader global grand challenges, including climate change, biodiversity loss, and ecosystem changes. It is important to define the criteria that will be useful to assess the performance of future interventions. To this end, the umbrella concept of "sustainability" can be used to discuss and agree on a specific definition for each particular context (Videira et al., 2010). An active approach towards human rights protection should be the fundamental ethical ground for the discussion of desired futures. Similarly, to envision multiple desired futures (scenarios) will help stakeholders define more explicitly which futures they value the most and why. An in-depth enquiry regarding these aspects should consider principles related with sustainable development, such as intra/intergenerational justice and the precautionary principle. Likewise, it is important to have a vision of how the participatory process of discussing and agreeing would make certain criteria and visions of the future more visible while, almost inevitably, others become less visible.

Model formulation and confidence building

The role of the analyst is central as the main actor developing SD quantitative models. A conscious effort to craft a model that balances simplicity and complexity is key. Not every aspect from conceptual mapping can be quantified in a simulation model, yet the model needs to reflect the complexity and behaviour of the real system or issue at hand. The modellers should recognise themselves as a very likely source of bias and consider how they are actively looking to identify and minimise it, as well as trying to make explicit the remaining bias. This process can be made more transparent by involving stakeholders in the validation phase.

Simulation and assessment

If a model is used to support policy, its strengths and limitations must be openly discussed and recognised by stakeholders. An important limiting aspect of simulation models is uncertainty. Recognising and communicating uncertainty is therefore critical (Palmer, 2017). Assessing the model's robustness can help to discuss the risks of using models to support policy decisions by considering multiple scenarios (Moallemi, Kwakkel, de Haan, & Bryan, 2020). Finally, it might be valuable at this stage to re-examine the implications of choosing certain policy alternatives to reach a desired "sustainable future" accounting for human rights aspects, according to the "envisioning and goal setting" phase.

3.4.3. Promoting ethical exploration in SD modelling

A broader and deeper ethical discussion is necessary in SD and other modelling disciplines. The practice of asking ethical questions is relevant for any SD participatory process to explore the ethical implications of modelling society and the environment. Ethical questioning can be aligned with other processes and practices taking part in a modelling cycle. For instance, this approach may be a foundation for implementing future "ethical" scripts for SD group model building interventions (Andersen & Richardson, 1997; Hovmand et al., 2012; Luna-Reyes et al., 2006). Such scripts ideally would not only deal with a particular issue at hand, but would also facilitate the operationalisation of reflection and discussion around the principles underlying human rights and sustainable development in participatory settings. This is a hopeful direction, as facilitated group ethical discussion can help to enrich the skill of judgement, or "practical wisdom", to cope with contesting values, reach agreements and move towards action in complex settings (West & Schill, 2022).

In addition to the group ethical exploration, SD modellers would be able to better understand the ethical dimension of their practice by improving personal skills such as reflexivity, accountability and deliberation. These skills can be strengthened, for example, by having reading groups to broaden the knowledge about ethics and philosophy, or by recording personal video diaries as places to share questions, challenges and feelings related to the development of an SD project (West & Schill, 2022). Better individual ethical-related skills will very likely, in turn, enrich ethical reflection in a group environment. Exploring these and other novel ways to promote skills that facilitate ethical deliberation is a promising field for research that may benefit SD education and practice.

3.5. CONCLUDING REMARKS

SD practice has ethical implications, evident through: (1) exploring the whole SD participatory cycle with an ethics lens; and (2) considering SD applications, e.g. sustainable development, resource management (Pruyt & Kwakkel, 2007). An ethical perspective also allows to recognise SD models as entities that encapsulate various stakeholders' values and worldviews, especially when considering models of potential futures and how those futures might look like. Using social-ecological systems as a lens for analysis, this article provides a structured framework aiming to make explicit both the critical role of stakeholder participation in SD modelling, and the ethical implications within that modelling cycle, especially in the context of sustainable development.

Sustainable development and human rights were presented as ethical standpoints for SD across the modelling cycle. Sustainable development applications require an open discussion around the concepts of intra/intergenerational justice and the precautionary principle. Human rights protect System Dynamicists' freedom to develop research, but demand their responsibility towards human rights recognition and protection in relevant modelling studies.

SD practitioners and researchers should adhere to certain principles across the modelling cycle regardless of the field of study. For instance, the System Dynamics Society's code of conduct encourages its members to adhere to three main principles: (i) contribute to society and human well-being, (ii) prevent conflict of interests, and (iii) respect diversity and prevent discrimination (System Dynamics Society, 2019). Here we propose to consider some principles of special relevance for the SD community in addition to the aforementioned:

- The underlying principles of human rights (i.e. freedom, equality and justice) need to be a fundamental guide for the SD practice;
- SD modellers involved in sustainable development applications need to adhere to the principles of intra/intergenerational justice and precaution, particularly when dealing with socio-environmental issues;
- SD studies and applications must be transparent and explicit, especially regarding its assumptions and limitations in face of uncertainty. Palmer (2017) further emphasises the need for transparency across the SD modelling process.

This list, though far from exhaustive, highlights the importance to keep identifying and discussing ethical principles that are necessary to guide the SD practice. We acknowledge that our proposed principles interact with the SDS list and help to complement and enrich it. For example, a human rights perspective makes the principle of "contributing to society" more tangible and clearly fosters the respect of diversity and the prevention of discrimination. Transparency is also aligned with the prevention of conflict of interests. It is hoped that this article will contribute to making modelling processes, especially those with a strong stakeholder engagement and participatory component, more ethically transparent, and ultimately more relevant to an increasingly complex world in which policy is ever-more guided by simulation models and their outcomes.

This essay should be read as a starting point, and an invitation, to further discuss and address the ethical implications of SD applications in a participatory modelling context in general. It proposes a set of principles and ethical questions that need to be discussed along with stakeholders in the context of modelling projects dealing with complex issues. This structured 'questioning' approach is a simple yet potentially useful tool for SD practitioners to examine the ethical implications of their modelling endeavours in the context of grand societal challenges, including climate change, migration-related challenges, and the implications of ecosystems degradation. More ethical-aware approaches of operational modelling can build upon the above in the form of "ethical scripts" for future group model building initiatives (Andersen & Richardson, 1997; Hovmand et al., 2012; Luna-Reyes et al., 2006). These questions may inspire SD practitioners to take part in other practices that improve the reflexivity (e.g. reading group discussions about philosophy and ethics) and deliberation (e.g. personal video diaries) around their modelling endeavours (West & Schill, 2022).

Going forward, continued ethical deliberation is necessary both to prevent violations to important rights and principles but also for taking a pro-active approach to achieve the "good" in a "sustainable" future. As a first step, this process can start with the modellers' self-assessment as active ethical actors. SD practitioners make many choices throughout the modelling process, i.e. how you steer stakeholders to frame the problem; or what you emphasise in a conceptual diagram; what you include in, and exclude from, the simulation model; the variables/results that are chosen to be made visible and reported; and what potential interventions you test with your model, their implications, and the assumptions built into them. Our proposed ethical questions aim to help revealing these practices towards a more reflexive and transparent SD modelling practice.

The recognition of ethics as pervasive across the SD practice should hopefully lead to more widespread discussions among practitioners and experts. Operationalising ethics in SD requires reflecting on how abstract concepts (e.g. justice and precaution) take shape in the context of concrete case studies. Participatory modelling approaches allow the opportunity to discuss the implications of such practical ethical insights. SD is a powerful tool to support sustainable policy making, and as such should point to objectives that promote human dignity and protect the environment. An ethics lens can serve as a compass to guide this process.

PART II

PRACTICAL APPLICATIONS FOR TRANSBOUNDARY RIVER MANAGEMENT

4

USING A PARTICIPATORY SYSTEM DYNAMICS APPROACH TO ASSESS TRANSBOUNDARY NUTRIENT POLLUTION

Based on the published peer-reviewed article:

Amorocho-Daza, H., Sušnik, J., Slinger, J. H., & van der Zaag, P. (2026). A participatory system dynamics approach to assess transboundary nutrient pollution: modelling the water-energy-food-ecosystems nexus in the Lielupe River Basin, Lithuania and Latvia. Ecological Modelling, 513, 111417. https://doi.org/10.1016/j.ecolmodel.2025.111417

Abstract

Managing natural resources in transboundary river basins is a complex task in which societal needs and environmental impact are intertwined. The nexus paradigm engages with such a challenge by analysing synergies and trade-offs across Water-Energy-Food-Ecosystems (WEFE) sectors. We present a WEFE nexus operationalisation using a participatory modelling approach in the transboundary Lielupe river basin, shared between Latvia and Lithuania. Using a modelling cycle approach, we illustrate a stakeholder-driven pathway from generic and qualitative to increasingly quantitative system tools useful for basin-scale policy analysis. Stakeholders prioritised agricultural nutrient pollution as a critical nexus issue strongly linked to land-use. Three policy alternatives to address this issue were co-identified with stakeholders from both riparian countries: (i) implementing nature-based solutions; (ii) transitioning to organic agriculture; and (iii) promoting arable land-use transitions to former native landscapes. The long-term effect of such policies is explored using a System Dynamics simulation model. Results highlight the importance of promoting active transboundary cooperation for water quality control, as unilateral action hampers the effect of long-term ambitious policies. Even highly ambitious unilateral action can delay the achievement of river basin quality objectives in the order of a decade, a critical finding for the wider Baltic region and the achievement of EU water quality objectives. Based on an exploratory analysis, we found that implementing basin-scale solutions for nutrient control would reduce nitrogen concentration by around 30% with a 2% co-benefit of increasing vegetation stocks, yet at the cost of decreasing cereal production by 8%. This work illustrates the capabilities of a tailor-made simulation model crafted to answer locally relevant policy questions with a nexus perspective in a transboundary river basin. Developing and using a simulation model in a participatory way can explore policy futures while fostering dialogue among riparian stakeholders. This is a promising way to promote cooperation towards solving critical socio-environmental issues in transboundary rivers.

Keywords: WEFE Nexus, system dynamics, participatory modelling, transboundary cooperation, nutrient pollution, land-use change, nature-based solutions

4.1. INTRODUCTION

The nexus perspective is an integrated sustainability paradigm focusing on the cross-sectoral connections and management of critical resources for society (Albrecht, Crootof, & Scott, 2018; Liu et al., 2018). Global policy and academic discourses have long considered the importance of an integrated perspective on resource management, yet nexus approaches have raised interest in recent years (Mohtar & Lawford, 2016). As proposed by Hoff (2011), the nexus perspective highlights the interconnections between the Water-Energy-Food (WEF) resource sectors, as well as the synergies and trade-offs that exist among them (Sušnik, 2018) (see Box 1). Since its introduction, policy and academic groups concerned with natural resource management have rapidly adopted nexus thinking (Allouche, 2024). Despite such popularity, scholars highlight the current gap between nexus *thinking* and *action* (Simpson & Jewitt, 2019b). Nexus thinking is wide and generic, but its implementation is a highly contextual task (Liu et al., 2017; Sušnik & Staddon, 2021).

Many challenges and opportunities emerge when the nexus perspective is applied in different contexts and scales, starting with defining which sectors constitute the *nexus* (Sušnik & Staddon, 2021) (see Box 1). For instance, moving beyond a resource-focused perspective to recognise a wider environmental dimension in nexus assessments has promoted the use of *WEFE nexus* terminology, which explicitly incorporates *ecosystems* (Carmona-Moreno, Dondeynaz, & Biedler, 2019; Lucca et al., 2025; Sušnik & Staddon, 2021; van den Heuvel, Blicharska, Masia, Sušnik, & Teutschbein, 2020). Explicitly addressing ecosystems brings further complexities to nexus implementation as it extends the scope from resource management to a wider socio-environmental perspective (Ghodsvali, Dane, & de Vries, 2022; Lucca et al., 2025). Such a broader approach is suitable for exploring complex settings such as river basins (Gain et al., 2020).

In addition to biophysical processes, river basins exhibit high complexity as people live and develop economic activities in them (Bakhshianlamouki, Masia, Karimi, van der Zaag, & Susnik, 2020; Ravar, Zahraie, Sharifinejad, Gozini, & Jafari, 2020). Such complexity can be exacerbated as rivers often flow across country borders forming a transboundary river basin. Some 60% of global freshwater flows are transboundary, hosting about 40% of the world's population (Munia et al., 2016). Upstream and downstream countries are therefore connected through water and beyond (Zeitoun, Goulden, & Tickner, 2013). Due to their significance, considerable efforts have been made to understand how riparian countries interact about their shared water resources (Bernauer & Böhmelt, 2020; Yoffe, Wolf, & Giordano, 2003).

Such a complex socio-environmental setting calls for an integrated perspective to highlight the interactions across WEFE sectors in transboundary basins (De Strasser, Lipponen, Howells, Stec, & Bréthaut, 2016; Lawford et al., 2013). Abundant research has focused on understanding how decisions around water resources affect other sectors such as food (e.g. crop production) and energy (e.g. hydropower generation), generating important socio-economic costs and benefits for riparian countries (see Arjoon et al., 2016). For instance, technical-oriented approaches have been proposed to address complex water allocation problems (Kucukmehmetoglu & Guldmann, 2010; Scott, El-Naser, Hagan, & Hijazi, 2003) and foresee optimal operation of river infrastructure in transboundary settings (Digna et al., 2018;

Verhagen, van der Zaag, & Abraham, 2021). A nascent WEFE nexus approach on transboundary issues is bringing new perspectives around river basin planning and management.

Applying the WEFE Nexus at the transboundary river basin scale can be done by using qualitative, quantitative, or mixed approaches. Emerging qualitative frameworks raise the importance of engaging riparian stakeholders for actively debating and committing to addressing transboundary nexus challenges (De Strasser et al., 2016). These dialogues should involve multiple local stakeholders and can be facilitated by external experts coming from diverse organisations (e.g. multilateral, governmental, non-governmental and academic) (Armitage et al., 2015; Daher, Hannibal, Mohtar, & Portney, 2020; Tuler et al., 2023). Quantitative nexus approaches are often developed via integrated modelling tools (Endo et al., 2015; Endo et al., 2020; Kaddoura & El Khatib, 2017). Making use of models can be helpful to explore long-term policy questions in a simulation environment. In this way, using integrated models can facilitate learning by exploring the effect of policies in a 'safe' virtual space before implementing them in the field (Medema, Mayer, Adamowski, Wals, & Chew, 2019; Pereira Ramos, Kofinas, Sundin, Brouwer, & Laspidou, 2022; Sušnik et al., 2018). For instance, quantitative nexus approaches have been applied to explore cooperation strategies focused on water quantity in contested transboundary river basins (Elsayed, Djordjevic, Savic, Tsoukalas, & Makropoulos, 2022) and to identify leverage points for nexus-wide change in river basin planning (Coletta et al., 2025). Despite the high potential of integrated modelling for informing WEFE Nexus policies in transboundary rivers, its use remains limited (Bwire, Mohan, Karthe, Caucci, & Pu, 2023). One plausible cause of this is the difficulty of engaging riparian stakeholders and integrating their knowledge in a modelling endeavour; this calls for mixed nexus approaches.

Mixed nexus approaches make use of both qualitative and quantitative tools in the context of river basins and beyond. Recent frameworks can facilitate this integration by considering mixed methods across a modelling cycle in the context of complex socio-environmental settings, such as river basins (Amorocho-Daza, Sušnik, van der Zaag, & Slinger, 2025; Jakeman et al., 2024). In recent years, there have been important contributions with a mixed nexus approach at different scales. Sušnik et al. (2021) present the development of a System Dynamics quantitative approach engaging local stakeholders to evaluate national-level nexus policies in Latvia. Similarly, Roy et al. (2024) present a whole-cycle System Dynamics approach to explore the drought-food security nexus in Bangladesh yet without explicit stakeholder participation. On a more local scale, González-Rosell et al. (2020) illustrate a participatory System Dynamics approach to understanding WEF Nexus interactions and evaluating strategies for the region of Andalusia, Spain. Almulla et al. (2022) propose a mixed nexus approach integrating stakeholder dialogues across the development of a WEF model used to evaluate long-term policies for the Souss-Massa basin in Morocco. Likewise, recent participatory modelling research is adopting nexus thinking to understand agriculture-related challenges—both at local and river basin scale—focusing on long-term trends across water, ecosystems and food sectors (Pagano et al., 2025; Rashidian et al., 2025). Despite these advances, the academic literature still lacks research about mixed nexus approaches in the context of transboundary river basins, particularly with a focus on water quality.

This article presents the output of a participatory modelling (PM) experience aiming to explore the WEFE Nexus in an international river basin context. The approach is implemented in the Lielupe River Basin (LRB), an agriculture-intensive transboundary river basin shared between Latvia and Lithuania, as one of the case studies of the Horizon2020 "Facilitating the next generation of effective and intelligent water-related policies utilising artificial intelligence and reinforcement learning to assess the water-energy-food-ecosystem (WEFE) nexus"—NEXOGENESIS project (nexogenesis.eu). Here we present a WEFE Nexus application at a transboundary river basin showcasing both qualitative and quantitative modelling outcomes, following an integrated PM cycle. In this article, our focus lies on two products or outcomes of a PM intervention: the simulation model and the policy insights that can be derived from using it (Gray et al., 2018). More specifically, we show how the simulation model capabilities enable the exploration of long-term outcomes of locally relevant policies amid uncertainty.

Box 1: Nexus approaches

Among many sustainability paradigms, the nexus perspective brings attention to the synergies and trade-offs among resource sectors (Allouche, 2024; Liu et al., 2018). Synergies exist where interventions in one sector have a positive effect on other(s); for example, the relation between water availability and hydropower generation. In contrast, trade-offs occur when interventions in a certain sector negatively affect other(s); for example, by developing intensive agriculture, river water quality is negatively affected via diffuse nutrient pollution. Nexus research therefore strives to reach a holistic understanding of the resource interdependencies to promote synergies and minimise trade-offs across resource sectors (Simpson & Jewitt, 2019a).

Despite the evident importance of the WEF sectors for society, researchers and practitioners promptly pointed out the need to include other sectors in the Nexus. For example, various scholars proposed including land and climate (WEF-CL) in specific case studies (Laspidou, Mellios, & Kofinas, 2019; Sušnik et al., 2021). Others also pointed out that explicitly considering the ecosystems in the nexus was of critical importance as they underlie the resource sectors and beyond (Folke et al., 2021; Rockström et al., 2009; Sušnik & Staddon, 2021; van den Heuvel et al., 2020). For instance, the Water-Energy-Food-Ecosystems (WEFE) Nexus is emerging as a flexible approach integrating research and policy perspectives toward sustainable resource management (Carmona-Moreno et al., 2019; Lucca et al., 2025). While nexus terminology and focus are evolving (WEF, WEF-CL, WEFE, and others), the field remains focused on promoting an integrated perspective toward sustainable resource planning and management.

4.2. METHODS

Our methodological approach begins with an overview of our framework of choice, a policy analysis framework based on a modelling cycle (Section 2.1). We then present the case study and the participatory setting in which the framework is applied (Section 2.2). The subsequent sections (2.3-2.5) explain the global phases of the modelling cycle as implemented in the case study. We intentionally present the model's development and characteristics as part of the methods rather than the results. This decision is based on two main reasons: (1) our chosen framework focuses on policy analysis, taking a step further from model building and testing;

and (2) literature on participatory modelling (PM) emphasises that the value of a PM intervention lies not only in the model itself but also in the policy insights it generates (Gray et al., 2018). Accordingly, our methods section focuses on the modelling process as a means to derive those insights, which are reported in the results (Section 3).

4.2.1. Policy analysis framework — A participatory modelling cycle under uncertainty

We adopted Amorocho-Daza et al.'s (2025) framework as a structured policy analysis approach to formulate a socio-environmental System Dynamics (SD) model with stakeholders while considering uncertainty throughout the process. Each of the three global modelling phases, namely (1) modelling foundations, (2) model-building and testing, and (3) model use and policy evaluation, was followed in developing the SD model for exploring the WEFE Nexus in the context of a transboundary basin (Figure 4.1). For each modelling phase, we implemented specific tools that contributed to making a model useful for policy analysis ('gears' in Figure 4.1).

In Phase I, the modelling foundations are established. Here, modellers and stakeholders worked together to scope the issue and begin its conceptualisation. Scoping activities helped to focus on the problem of interest, particularly regarding the definition of a locally rooted vision of sustainability (Nabavi, Daniell, & Najafi, 2017). For the conceptualisation, we developed a qualitative system representation of the defined issue with stakeholders. This formed the foundation for developing a quantitative model (Freebairn et al., 2019).

During Phase II, we built and tested a quantitative simulation model. We used SD as the modelling paradigm. A quantitative SD model takes the form of a Stock and Flow Diagram (SFD) (Naugle, Langarudi, & Clancy, 2024). Here, the conceptualised relationships were operationalised as a network of parameters, flows, and stocks that can be mathematically represented as coupled differential equations (Ford, 2010; Sterman, 2000). The model was verified in terms of structure and behaviour with the help of stakeholders and pre-existing/historical data.

Phase III focuses on model use, particularly to evaluate socio-environmental policy alternatives. This phase relates to the scoping activities of Phase I by using the simulation model to test policies that can potentially contribute to resolving the identified socio-environmental issue(s). At this stage, stakeholders used the model as an experimentation and learning tool and provided feedback for its potential later use in decision-support settings.

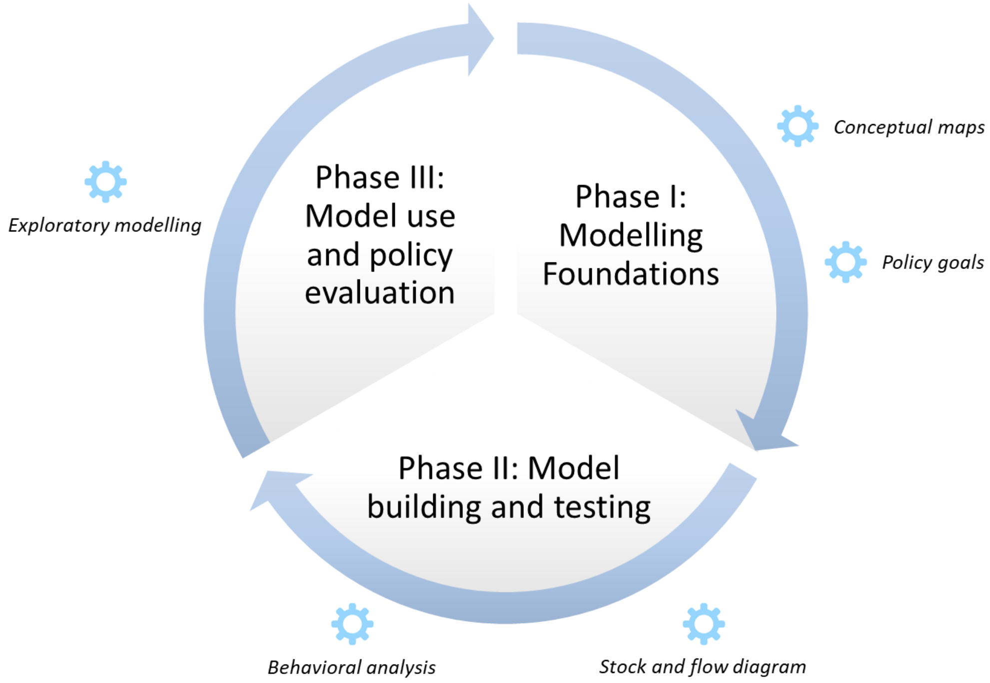

Figure 4.1. The modified three-phase modelling framework (after Amorocho-Daza et al. 2025)

4.2.2. Case study

The modelling framework is applied to explore WEFE policies in the Lielupe transboundary river basin (Figure 4.2). The Lielupe River Basin (LRB) is one of the transboundary river basins shared between Latvia and Lithuania (Figure 4.2). It has an area of ca. 17,800 km^2, distributed almost equally across Latvia (8,850 km^2) and Lithuania (8940 km^2) (NEXOGENESIS, 2022). The LRB is one of the main river basins shared between Latvia and Lithuania, occupying around 14% and 16% of the countries' areas, respectively (FAO, 2016). Lithuania, the upstream country, contributes about 56% of the river basin's annual flow. The rivers Mūša (Lithuania) and Nemunėlis (at the Lithuanian-Latvian border) merge into the Lielupe River in the city of Bauska (Latvia) (European Regional Development Fund, 2019). From this point on, it flows north for 119 km into the Gulf of Riga, with a mean flow of 3,540 Mm^3/year (112 m^3/s) (FAO, 2016).

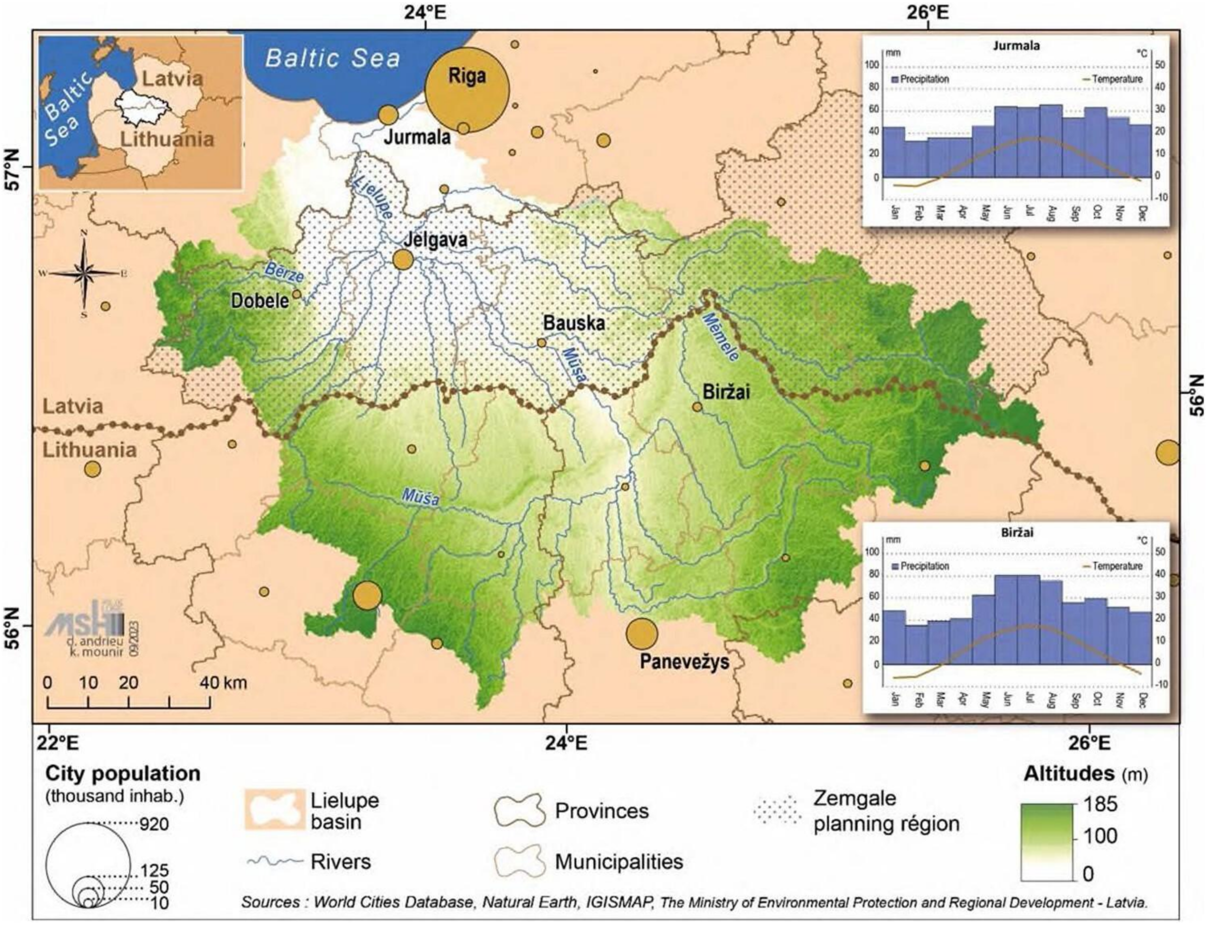

Figure 4.2. Lielupe River Basin (Mooren et al., 2024) Note: CC BY-NC-ND 4.0

The socio-economic activities in the LRB affect local ecosystems. The river basin is home to ca. 12% of the Latvian population and 11% of the Lithuanian population, in total ca. 800,000 inhabitants, half in urban areas (NEXOGENESIS, 2022). The main economic activities of the LRB are related to the services and agriculture sectors. Regarding land use, the main land uses of the basins correspond to arable land and forests, each one accounting for 43% of the total basin's area (See Appendix A, Table 3). Agricultural land use competes with the natural grassland habitats of the region. Productive activities, particularly agriculture, have caused high nutrient concentrations in the river for decades (Siksnane & Lagzdins, 2020; Stålnacke, Grimvall, Libiseller, Laznik, & Kokorite, 2003). According to FAO estimates, more than 70% of the total nitrogen and more than 40% of the total phosphorus inland load is caused by human activities (FAO, 2016). The main source of nitrogen pollution is related to agriculture, while the main contributors of phosphorus are municipal and industrial wastewater. Controlling nutrient pollution in the basin is required to reduce existing negative pressures on aquatic ecosystems, not only in the river itself but also in the Baltic Sea basin as a whole (Limburg, Breitburg, Swaney, & Jacinto, 2020). These river basin issues, therefore, resonate with a larger 'wicked' problem connecting agricultural landscapes to water bodies across the European Union (Wiering et al., 2020).

Stakeholder participation

Stakeholders of the LRB are engaged in a WEFE Nexus policy co-creation initiative, including a strong focus on participatory modelling, following a structured approach as reported in Hüesker et al. (2022). Further details about the stakeholder engagement strategy in the Lielupe and the rest of the NEXOGENESIS case studies, including stakeholder selection, modes of engagement and overall outcomes of the participatory process, can be found in Avellán et al. (2025). Local case study partners —BEF Latvia (an environmental NGO focusing on the Baltic Region)— identified and engaged stakeholders that either have an "interest in the application of project results and products" and/or are "directly engaged in the project implementation and/or outcomes". A summary of stakeholders' affiliations is shown in Table 4.1.

Table 4.1. Summary of the stakeholders' affiliation (LV =Latvia; LT Lithuania)

Stakeholders' main classification	Organisations
National authorities	Ministry of Agriculture (LV), Ministry of Environment (LT), Latvian Environment Geology and Meteorology Centre (LEGMC) (LV) Lithuanian Hydrometeorological Service (LT)
Regional planning authorities	Zemgale Planning Region (LV)
Local municipalities in Latvia and Lithuania	Jelgava (LV), Bauska (LV), Panevėžys (LT) Biržai (LT)
Research institutions/universities	Latvia University of Life Sciences and Technologies (LV) University of Latvia (LV)
Other associations	Farmer groups: NGO Association "Farmers' Parliament", LPKS "LATRAPS" Environmental NGOs: Green Liberty (LV), Salgale rural support association (LV), Center for Environmental Policy (AAPC) (LT)

Stakeholders took part in six workshops across three years, as summarised in Table 4.2. For each workshop, a purpose aligned with the proposed phase of the modelling framework was identified. Additionally, the table presents the facilitation approach implemented in each workshop, as well as the main inputs and outputs. An online summary of each workshop is available via links in Table 4.2 for interested readers.

Table 4.2. Summary of stakeholder workshops

Workshop	1	2	3	4	5	6
Date	10/02/2022	02/11/2022	15/06/2023	06/02/2024	02/10/2024	27/05/2025
Location	Online	Riga, LV	Vilnius, LT	Riga, LV	Riga, LV	Riga, LV
Phase of the modelling framework	Modelling foundations	Modelling foundations	Modelling foundations	Model building and use	Model use and policy evaluation	Model use and policy evaluation
Number of participants	10	10	10	18	11	17
Workshop purpose	Identification of main nexus issues in the basin	Discussion about the current state of the basin and the required policies to improve it	Nexus policies prioritisation for the basin	Presentation of preliminary results of a simulation model with policies for the basin	Stakeholder interaction with a web-based decision support system	Discussing the practical implications of policy alternatives in the context of the local/transboundary governance roadmap
Facilitation approach	Small group discussions	Collective brainstorm on policy alternatives for Nexus sectors	World Café (WEFE sectors) on policy instruments Dot voting on policies	Plenary discussion about preliminary results of the model Q&A session - modelling capabilities,	Supported testing of the tool in groups (+task) Feedback (plenary) to improve the model and tool functionalities	Plenary discussion on policy package validation Dot voting for characterising the needed activities to

Workshop	1	2	3	4	5	6
			prioritisation per WEFE sector	assumptions and limitations		reach the basin's policy goals
Inputs (modelling cycle tools)	Early conceptual map	Early conceptual map	Draft policies by sector	Simulation model results	Decision support system (with the model in the background)	A set of policy tools and their expected (modelled) performance
Outputs (modelling cycle tools)	Identification of the key nexus issues in the basin	Revised conceptual map	Prioritised nexus policy goals	Stakeholder feedback and requests for: updating the model and designing a DSS to use it	Stakeholder feedback about: the DSS functionalities and improving the precision of some key modelling outputs	Stakeholder feedback about: Opportunities and limitations of taking the proposed policy packages into action.
Online summary	Link	Link	Link	Link	Link	Link

4.2.3. Phase I

The most relevant outputs of the stakeholder workshops 1 to 3 in terms of the modelling cycle tools were the conceptual map and the prioritised policy goals for the basin. These are important inputs for developing a simulation model.

Conceptual map

The conceptual map represents an effort to account for the interlinkages and causal relations among the variables of the nexus system in the basin. Stakeholders were introduced to an early conceptual map of the WEFE Nexus system in Latvia, originally proposed by Sušnik et al. (2021), during Workshops 1 and 2. This served first as a discussion starter to prioritise the main issues of the basin from a nexus perspective, and later to adapt it to the scale and priorities of the Lielupe (see Table 4.2). Stakeholders were able to debate whether the pre-identified links were relevant or not and to add new ones to the map. Figure 4.3a presents the WEFE Nexus conceptual map for the Lielupe river basin after incorporating stakeholder feedback. It shows a deeply interconnected and complex system consisting of the six nexus sectors identified as relevant for the river basin: *Water*, *energy*, *food*, *ecosystems*, *land*, and *climate*.

The conceptual map highlights how the identified variables affect one or more sectors in the LRB. Despite the myriad processes and interactions in the system map, it is important to highlight some key sectors and interactions. For instance, the *land* sector was identified as a central sector that affects many other sectors via processes that happen on land. Land use is closely connected to food and renewable energy production and drives trade-offs such as water quality deterioration and/or reduction of natural landscapes. Given the predominance of agriculture, the *food* sector shows several connections and impacts with other sectors. The causal relationships across nexus sectors are further explored and formalised in Phase II of the modelling cycle (see Section 2.4). A final version of the map presents an extension considering the countries' interactions in the basin. That is, the conceptual system map explicitly considers the two riparian countries independently yet connected via water flowing from the Lithuanian to the Latvian side of the basin (Figure 4.3b).

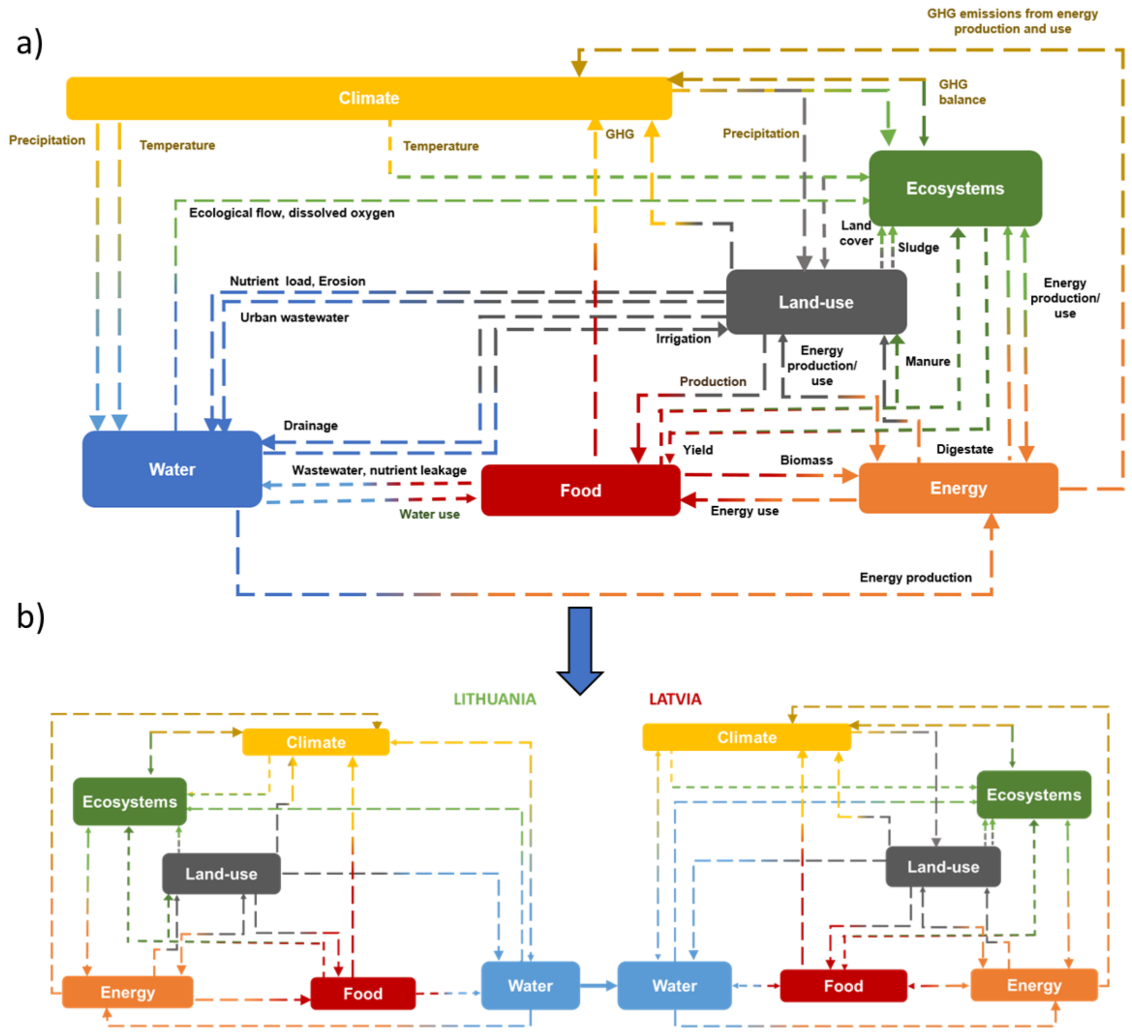

Figure 4.3. a) Conceptual system map of the nexus sectors in the Lielupe River Basin b) conceptual representation of upstream and downstream nexus system connected by water in a transboundary setting

Policy goals

The policy goals represent priority points for improving the current state of the LRB in accordance with a local vision of sustainability. As part of workshop 3, stakeholders were introduced to a list of policy alternatives potentially relevant to address local WEFE goals—based on a screening of national objectives, both in Latvia and Lithuania. Using the World Café approach (Löhr et al., 2020), in a first iteration, stakeholders revised, discussed, modified, and even proposed new policy alternatives for each WEFE sector in small groups. In a second iteration, stakeholders prioritised the policy alternatives that were more relevant to achieve river basin goals—making use of dots to vote for their preferred options. In the absence of a clear WEFE policy landscape with quantifiable river basin objectives (Mooren et al. 2024), the feedback from stakeholders provided a grounded perspective to connect means (policy alternatives) to ends (policy goals) in the participatory modelling exercise. In short, the workshop outcomes helped the NEXOGENESIS team narrow down a list of policy goals that could potentially be impacted by implementing relevant policy alternatives in the Lielupe and could be evaluated in a simulation environment. The final list of goals focused on aspects related to water quality and protection of natural ecosystems in the LRB (Table 4.3). These goals were subsequently incorporated within the SDM to assess the potential nexus-wide impacts of their implementation and to highlight possible policy synergies and trade-offs.

Table 4.3. Policy goals for the Lielupe basin

Goal	Description	Indicator	Year	Target
Goal 1. Reduce the nitrogen concentration in Lithuania by 15% in 2050	Reduce the nitrogen concentration in the Lielupe River Basin (Lithuania) by 15% in 2050	Percentage of nitrogen concentration reduction compared with the baseline (2015)	2049	-15%
Goal 2. Reduce the nitrogen concentration in Latvia by 20% in 2050	Reduce the nitrogen concentration in the Lielupe River Basin by 20% (Latvia) in 2050	Percentage of nitrogen concentration reduction compared with the baseline (2015)	2049	-20%
Goal 3. Equitable contribution from Lithuania to control transboundary nutrient pollution	Lithuania contributes (in proportion to its catchment area) to control nutrient pollution in the basin	Lithuania's contribution to control nutrient pollution in the basin	2015-2049	53%
Goal 4. Equitable contribution from Latvia to control transboundary nutrient pollution	Latvia contributes (in proportion to its catchment area) to control nutrient pollution in the basin	Latvia's contribution to control nutrient pollution in the basin	2015-2049	47%
Goal 5. Increase bird biodiversity by 20% in 2027.	Increase bird biodiversity (species richness) in the Lielupe River Basin compared with the baseline (2015)	Bird biodiversity	2027	+20%
Goal 6. Promote organic farming in Lithuania	Develop organic farming in 13% of agricultural land by 2028 in Lithuania	Fraction of arable land with organic farming in Lithuania	2028	13%
Goal 7. Promote organic farming in Latvia	Develop organic farming in 25% of agricultural land by 2030 in Latvia	Fraction of arable land with organic farming in Latvia	2030	25%

4.2.4. Phase II

This phase focuses on the development of a quantitative simulation model based on the Phase I foundations.

Stock and flow diagram - overview

A Stock and Flow model was developed based on the foundations identified above (Figure 4.3 and Table 4.3). Figure 4.4 presents an overview of the main stocks and variables considered in the model. The dotted lines identify the WEFE sectors in the model (i.e. land, water, food, ecosystems). Here is important to mention that not all the sectors characterised in the conceptual map were included in the simulation model (i.e. climate and energy); this evidences a transition from a generic systems perspective to a more focused analysis of a local problem from a systemic perspective. For instance, as the issue of nutrient pollution gained prominence in the stakeholder discussions, the energy sector remained relatively isolated and out of the model scope; similarly, the climate sector was less emphasised yet considered as a driver of biophysical changes in the basin. As recognised from the conceptual map, the land sector is central to the nexus in the basin. Four important land stocks are identified as Forests, Grasslands, Arable land, and Arable land with nutrient treatment. Arable land is the current dominant land use in the basin (ca. 43% of the land area), a condition that requires transformation according to the stakeholders' local sustainability vision. Hence, three policy levers are considered as future interventions in the basin (highlighted in purple): 1) conversion of arable land to grasslands, 2) implementation of nutrient reduction options, and 3) transition to organic agriculture. Depending on the extent of such policies, cascading effects are expected on the ecosystems, water, and food sectors.

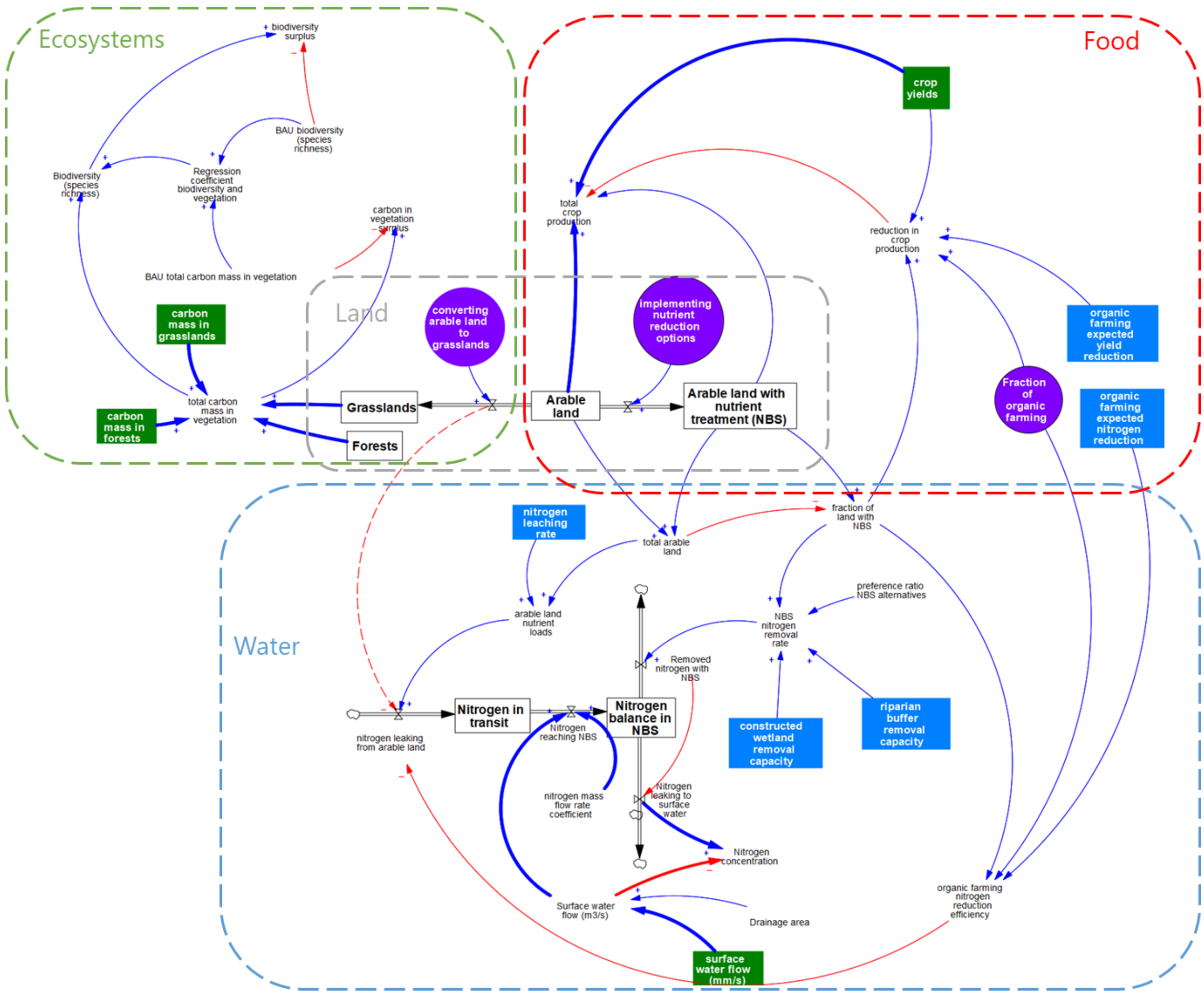

Figure 4.4. Stock and flow diagram of the Lielupe River Basin

For the *Ecosystem* features, transitioning back to a condition of more *natural* land use in the basin has important effects. Considering the basin's main land uses gives a proxy of the total carbon mass stored in vegetation in the basin. Natural land cover features such as grasslands and forests are the main contributors to the total vegetation stocks in the basin (see Figure 4.4, Ecosystems, interlinkages in bold); in contrast, the contribution of arable land to this stock is negligible (see Appendix A, Figure 10 and 11). Vegetation stocks are related in complex ways to other forms of biodiversity, such as animal species. Here we explore the effect of river basin vegetation stocks on animal biodiversity in the basin.

The *Food* sector is modelled from a food production perspective, focusing on the dominant crops of the basin (see Appendix A Table 4.2) (see Figure 4.4, Food, interlinkages in bold). Food production is estimated as the product of arable land and crop yield. To capture long-term uncertainty in crop production, crop yield is modelled as an exogenous variable responding to climate change scenarios. We also incorporated a probabilistic change in productivity based on the implementation of organic farming practices. Although this approach can capture a wide range of variability in food production and respond to endogenous land use changes, it has limitations. The approach does not endogenously model the impact of soil quality on food production, a modelling approach thoroughly illustrated by Rashidian et al. (2025).

Notwithstanding, our relatively straightforward modelling approach emerged from the participatory process, where stakeholder discussion did not focus on accurately representing food production but rather on assessing food-related trade-offs with other sectors, particularly water quality.

For the w*ater* sector, intensive agricultural activities export nutrient loads into surface water. Nutrients of importance are in the form of nitrogen (nitrate – NO_3^-) and phosphorus (phosphate – PO_4^{3-}); here we focus on nitrates as they are of special concern for stakeholders for two reasons: first, as they constitute the dominant contributor to total N; and, second, due to their persistently high concentration, not only in the LRB but in other Baltic basins (Siksnane & Lagzdins, 2020; Stålnacke et al., 2003). The nitrogen leaching from arable land is highly mobile from the soil to the water systems (e.g. groundwater, rivers, and estuaries). High nitrogen loads and concentrations in the basin are associated with eutrophication in the Gulf of Riga (Baltic Sea), the discharge point of the Lielupe River (Finni, Kononen, Olsonen, & Wallström, 2001; Lundberg, 2005; Murray et al., 2019). The model focuses on capturing nitrogen accumulation and transport processes, from soil to rivers and from rivers to estuaries, in a conceptual and aggregated way. Surface water flow is considered as a model parameter that drives the nitrogen leaching rate, and that can be used in combination with the nutrient loads to estimate river nutrient concentration (see Figure 4.4, water, interlinkages in bold; Appendix A, Section 1.3 for a detailed explanation). This high-level understanding of the nutrient movement across the river basin facilitates exploring the effect of different policies at a river basin scale to reduce, intercept and remove nitrogen before it enters the surface water system.

Various input variables, marked with green in Figure 4.4, are considered as long-term time series estimated via climate change projections following the CMIP6 runs (See Appendix A, Table 10). Most of the biophysical variables were obtained via the third phase of the Inter-Sectoral Impact Model Intercomparison Project (ISIMIP3b), a cross-sectoral framework for climate change projections (Frieler et al., 2017; Warszawski et al., 2014). Biodiversity-related variables are estimated with the aid of the Globio 4 model, a global biodiversity model (Schipper et al., 2020). Their application to the Lielupe River Basin and other case studies of the NEXOGENESIS project are further described by Trabucco et al. (2024). Similarly, variables identified with blue in Figure 4.4 are stochastic parameters derived from the academic literature (See Appendix A, Table 11).

A more detailed account of each sector's modelling strategy and equations is available in Section 1 of the Supplementary Information.

Transboundary interactions

Modelling nutrient flow across the basin offers an opportunity to characterise the transboundary nitrate mass flow. As proposed in the conceptual map (Figure 4.3b), the stock and flow diagram presented in Figure 4.4 can be transformed to account for upstream and downstream interactions. Such a change emerged from stakeholder feedback during Workshop 4, as they pointed out that assuming policies are implemented across the whole basin in the simulation model does not reflect Lielupe's transboundary reality, in which lack of coordination and unilateral actions regarding water quality management currently predominates (see Mooren et al. 2024). In response to this feedback, Figure 4.5 shows the modelling strategy to decouple the upstream and downstream riparian countries. Such a structure explicitly indicates that the Lithuanian nutrient load outflow is an input for Latvia, which adds to the diffuse nutrient pollution generated downstream. In other words, both countries are connected by the water flow, which transports nutrients from upstream to downstream.

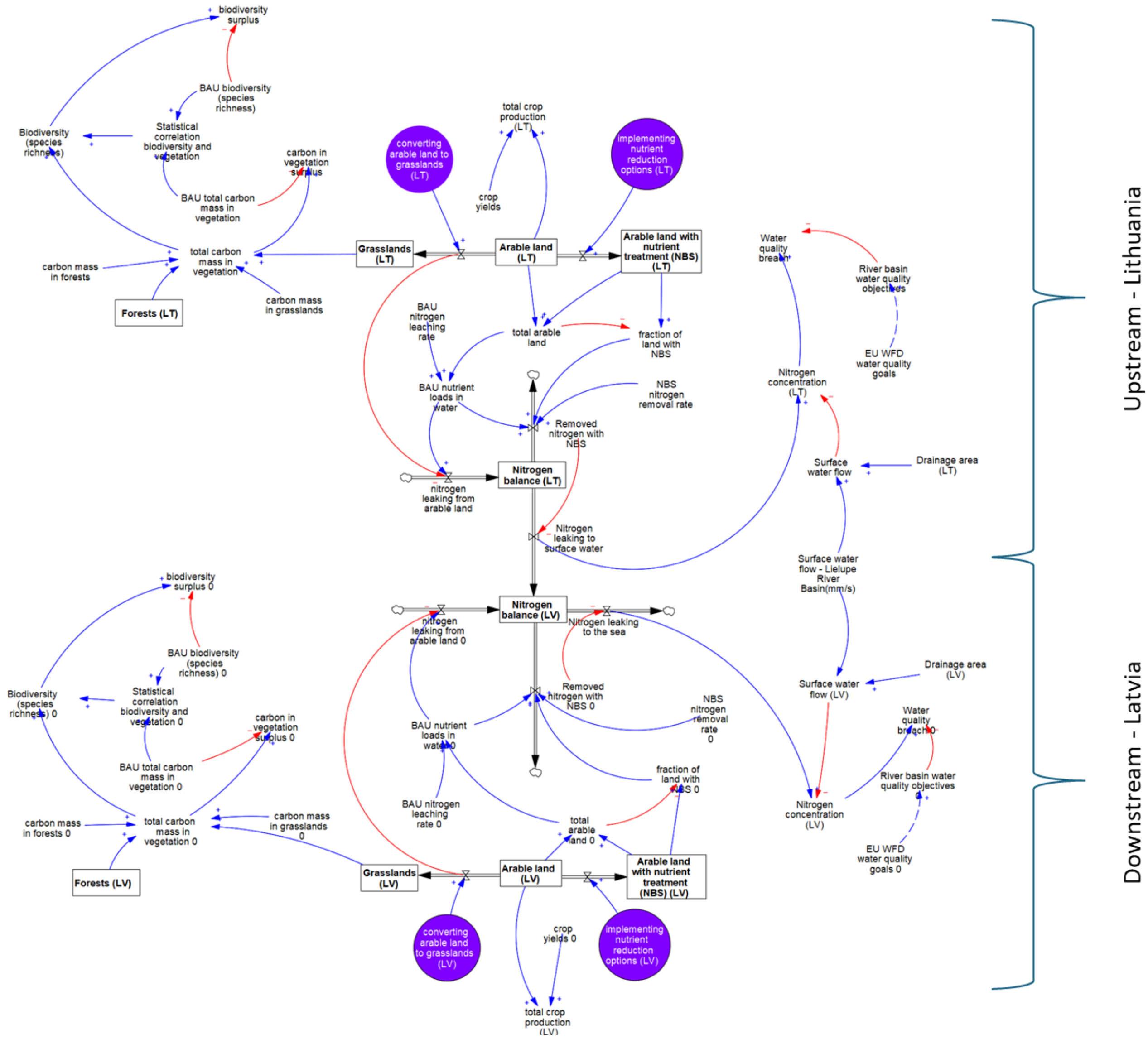

Figure 4.5. Stock and flow diagram reflecting the Upstream-downstream interaction in the basin

This model structure builds on the conceptual map but focuses on water quality rather than quantity (which is not presently a major issue in the basin). Such a model structure explicitly shows the asymmetry of a transboundary setting, offering flexibility to simulate the likely effect of implementing unilateral or bilateral environmental policies in the basin (Elsayed et al., 2022; van der Zaag, 2007).

Model behavioural analysis

For each nexus sector, a key variable is selected to explore the simulated behaviour over time without including policies. In this analysis, the main land stocks remain static and, therefore, the behaviour of the key variables is explained solely by exogenous projections (see Figure 4.4, variables marked in green; Section 2.4.1 for more details). The model uses a monthly timestep over 35 years (420 months), covering the 2015-2050 period. The chosen timestep allows for harmonising data input from multiple datasets and global models (see Appendix A, Table 10), also allowing for capturing seasonal dynamics of key variables (e.g. water quality). The simulation timespan reflects a medium-term planning horizon that allows system trends to be captured while giving time for corrective action to be taken; yet it is not so long that uncertainty dominates the narratives. Its initial and final year also reflect the data harmonisation across multiple input data sources under common climate change projections (i.e. CMIP6 – RCP2.6 and RCP8.5 scenarios), as detailed in the Appendix A. The behaviour of each selected variable is shown using a 90% dynamic confidence interval based on 1000 simulations of the SD model under the RCP2.6 scenario (see Figures 4.6-4.8) and using a Sobol sequence sampling with the stochastic parameters as summarised in Appendix A, Tables 10 and 11.

For the food sector, we selected cereals' total monthly crop production (combining summer and winter wheat, and maize). This decision is based on the fact that wheat and maize crop areas only account for two-thirds of the basin’s total arable land, and the crops are comparable in yield terms (see Appendix A, Table 2 and Figure 1). An important disclaimer is that, although crop yield typically is relevant at the seasonal or yearly time scale, we chose to estimate a monthly and uniform equivalent value of food production to facilitate comparison with other variables at every simulation timestep. Figure 4.6 shows the dynamic confidence interval for cereal production. The variable exhibits fluctuations in crop yield across the model timespan (see Appendix A, Figure 1). However, as the median, minimum, and maximum values remain fairly consistent over time, the monthly crop production can be described by a uniform distribution with a mean value of ca. 200,000 ton/month (115,000-283,000; 90%CI).

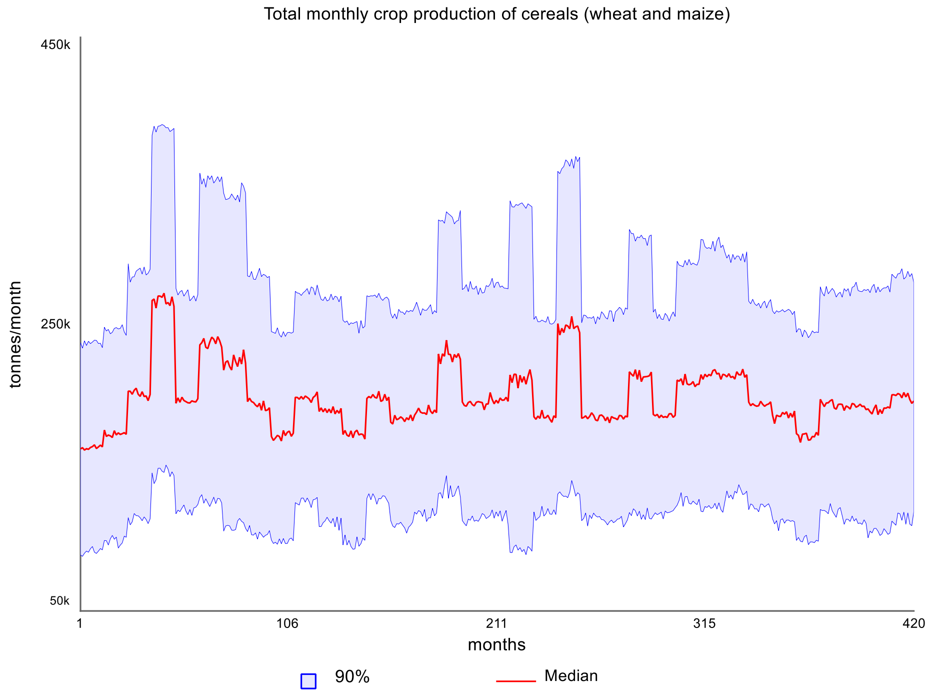

Figure 4.6. Dynamic 90% confidence interval for total monthly crop production of cereals.

For the ecosystems sector, we selected the total carbon mass in vegetation. This variable is selected as it is a common feature of different land uses and can be aggregated at the river basin level. Figure 4.7 shows the dynamic confidence interval for the total carbon vegetation stocks. The variable exhibits two different behaviours across the simulation. During the first half of the timespan, the carbon mass in vegetation increases linearly within a relatively narrow range. In contrast, during the second half of the simulation, the median value tends to decrease and exhibits larger uncertainty. This two-stage behaviour resembles the forecasts of average carbon mass density in vegetation, particularly carbon density in forests (see Appendix A, Figure 7). Vegetation stocks start at 67M ton C (66-68; 90% CI) and increase to a maximum of 80M ton C (78-81; 90% CI), to later decline to 75M ton C (72-78, 90% CI).

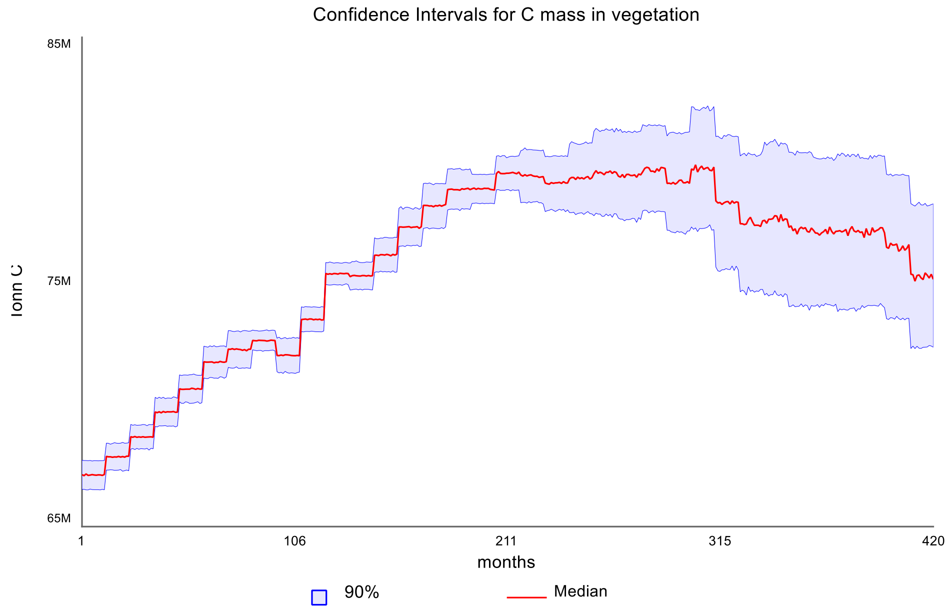

Figure 4.7. Dynamic 90% confidence interval for carbon mass in vegetation

For the water sector, nitrogen concentration was selected as a key variable because local authorities periodically measure it as a water quality proxy. Figure 4.8 shows the dynamic confidence interval for nitrogen concentration at the endpoint of the basin. As presented in SI's Eq. 18, nitrogen concentration is estimated as the ratio of nitrogen mass flow to water flow. The variable exhibits strong fluctuations that resemble seasonal flow variations (see Appendix A, Figure 3). The low values of concentration have a median value of 0.3 mg/L (0.2-4.2; 90% CI). Mean values of concentration exhibit a median value of 1.6 mg/L (0.8-7.4; 90% CI). High values of concentration exhibit a median value of 5.9 mg/L (3.3-9.4; 90% CI). The magnitude of these values and their seasonal behaviour are consistent with previously reported long-term behaviour of nitrogen concentration in the basin (see Stålnacke et al., 2003).

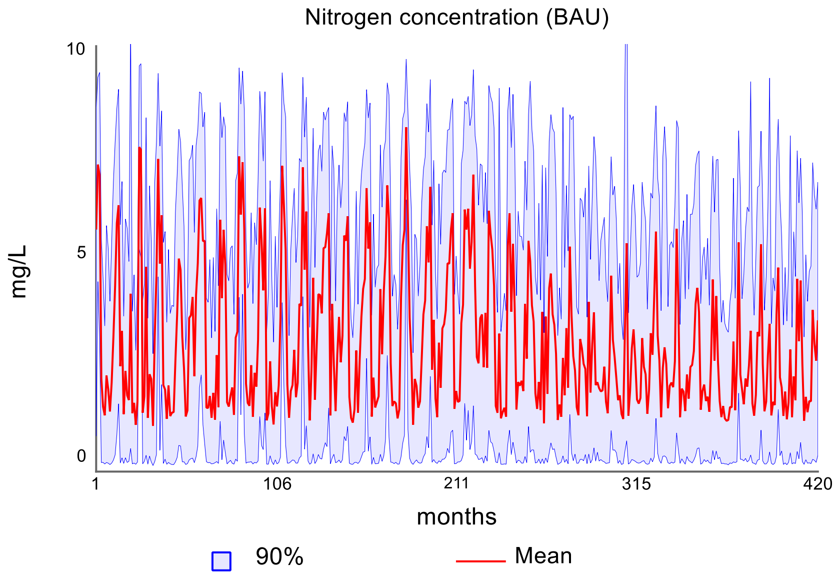

Figure 4.8. Dynamic 90% confidence interval for the nitrate concentration

Table 4.4 presents numerical values of the median and 90% confidence interval for the variables discussed above at discrete points in time. Four ‘snapshots’ are considered at the beginning of every decade in the period 2020-2050 as a proxy of the variables' change over time. These values are later used as a reference baseline for the key variables, useful in comparing the effects of policies in the exploratory modelling stage.

Table 4.4. Key variables’ model baseline with a 90% CI across four decades.

Key variables	Percentile	2020	2030	2040	2050
Total cereal production (1,000 ton/month)	5%	120	119	120	117
	50%	194	189	185	201
	95%	272	257	254	283
Total carbon mass in vegetation (Mton C)	5%	70.1	77.2	77.0	72.4
	50%	70.6	78.1	79.0	75.3
	95%	71.2	79.0	81.1	78.1
Nitrogen concentration (mg/L)	5%	0.5	0.8	0.8	0.4
	50%	5.4	6.1	5.2	3.8
	95%	8.1	8.5	7.9	6.9

Stakeholders were particularly interested in exploring and improving the model's capabilities to represent water quality in the basin, particularly during workshops 4 and 5. To validate the model, we used the same data source as for the calibration of the water module (see Appendix A, Table 3). However, we only used the most recent two years of observations, as they were not included in the calibration. Figure 4.9 presents water quality monitoring monthly observations in the period 2018-2020 alongside three time series of model outputs (5%, median and 95% of 1000 model instances) over the same period.

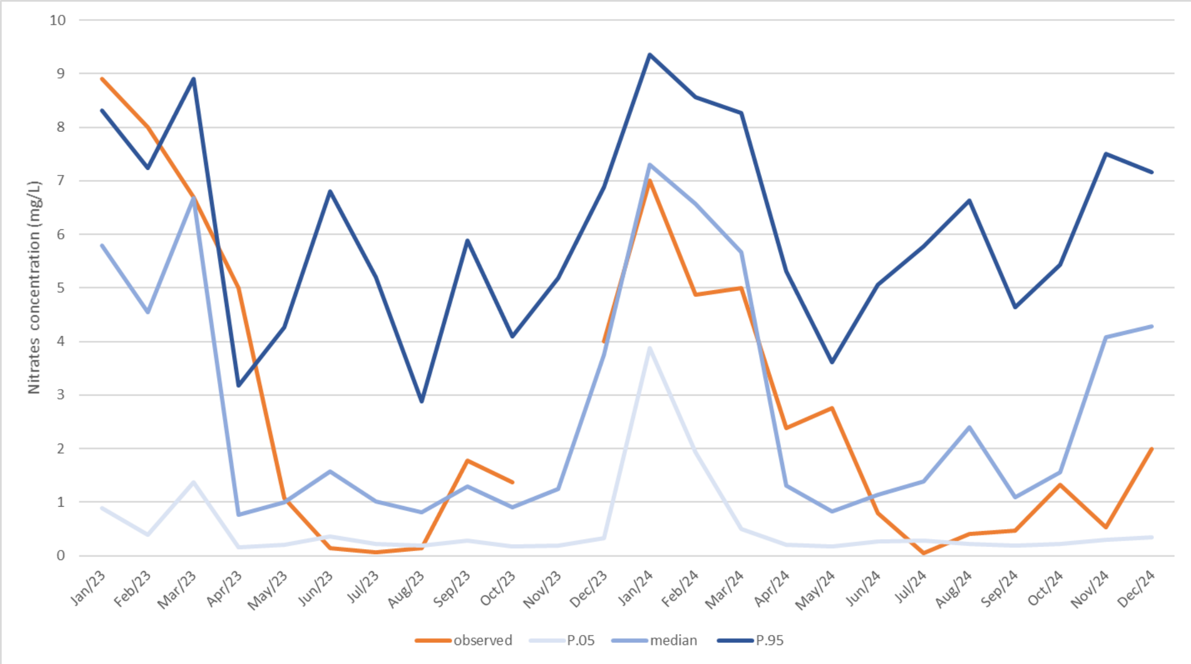

Figure 4.9. Observed vs. model outputs - nitrogen concentration.

A regression analysis between the median response and the observed nitrate concentration in the river was performed (See Appendix A, Table 6 and Figure 7). The regression coefficient is a significant predictor of the observed data (P=4.06E-05), able to account for more than half of the observed variance (Adj-R2=0.54). The coefficient value is estimated as 0.92, with an estimated value in the range of 0.55-1.3. As the value of 1 lies in the confidence interval, there is statistical confidence that the proposed model accurately captures the nitrogen concentration behaviour in the Lielupe. The approach explained above explains the nutrient concentration dynamics based on a biophysical variable—surface flow—which is also modelled stochastically as an exogenous variable (See Appendix A, Figure 3). This means that the nitrogen concentration baseline is exogenously, not endogenously, driven. This is a simple yet robust mechanism of modelling nutrient transport (See Appendix A, Section 1.3),

After establishing an exogenous and biophysically driven baseline behaviour for the three proposed variables, the following section focuses on assessing the expected effects of land-use change as an endogenous and policy-driven exploration. When implementing policies, the model stocks are not static but exhibit a dynamic behaviour that 'activates' most of the identified interlinkages as a cascading effect of long-term land-use change in the basin (see Figure 4.4, non-highlighted interlinkages). For instance, the proposed land-use policy levers (see Figure 4.4, variables marked in purple) can be tested to evaluate nitrogen reduction, interception and removal. Similarly, although cereal production is driven by crop yield, and the basin's total carbon in vegetation depends on vegetation types, these variables also respond endogenously to land-related policies. For instance, cereal production is a function of arable land, and carbon mass in vegetation is affected by the extension of grasslands. In short, although our modelling approach uses exogenous factors to estimate the behaviour of the key model's variables, the core model insights emerge from analysing the endogenous effect of policies on such variables. The following section details our exploratory perspective for policy evaluation.

4.2.5. Phase III

We proposed an exploratory modelling approach to use the model for policy evaluation. More specifically, we considered two perspectives, one analysing a limited but structured set of policy futures, and the second using an open, wider policy exploration.

Comparing policy ambition and transboundary cooperation

For the exploratory analysis of policies, we compared the change occasioned by implementing a policy relative to the previously defined reference baseline (Table 4.4). Two interacting criteria are considered for the analysis: Transboundary cooperation and policy ambition. For the first criterion, we consider levels of cooperation: bilateral, unilateral (Lithuania), and unilateral (Latvia). For the second, we consider the ambition level of the deployed policies as null, low, medium and high, based on the policy combinations as described in Table 4.5. Unilateral scenarios assume that while one riparian country deploys the policies to some degree (e.g. low to high policy ambition), the other country does not make any contribution (i.e. null policy ambition). The interacting criteria result in a 3x3 results matrix (see Table 4.7) to be compared with the baseline, as presented in Table 4.4

Table 4.5 List of policies categorised by policy ambition

Policies	Description	Policy ambition			
		Null	Low	Medium	High
1. Implementation of nutrient reduction options in arable land	Implementing nutrient reduction options in a fraction of the total arable land (fraction of land with nutrient reduction) by 2050	0	0.25	0.5	0.75
2. Implementing riparian buffers	Implementing riparian buffers to intercept 50% of the agricultural runoff of the arable land with nutrient reduction (Along with policy 1)	No	Yes	Yes	Yes
3. Implementing constructed wetlands	Implementing constructed wetlands to intercept 50% of the agricultural runoff of the arable land with nutrient reduction (Along with policy 1)	No	Yes	Yes	Yes
4. Implementing organic agriculture	Implementing organic agriculture in a fraction of the	0	0.25	0.5	0.75

Policies	Description	Policy ambition			
		Null	**Low**	**Medium**	**High**
	arable land with nutrient reduction				
5. Conversion of arable land to grasslands	Converting 10% of the total arable land into grasslands by 2030	No	Yes	Yes	Yes

Open policy exploration

Although the approach above enables a structured in-depth analysis, it considers only a limited number of policy combinations (i.e. 9 instances). Therefore, instead of exploring a well-defined and deterministic policy package (Table 4.5), here we explore the broader expected effects of the proposed policies in both riparian countries using a large and stochastic decision space following an exploratory modelling paradigm (Auping, 2018; Moallemi, Kwakkel, de Haan, & Bryan, 2020). Table 4.6 summarises the probability distribution and parameters considered for each of the policies that were introduced in Table 4.5. Therefore, one instance of the analysis will imply assigning a random value for each of the policies listed below. By simulating multiple runs of the simulation model, each one with stochastic instances of the policy variables, it is possible to explore the effect of the policies on the WEFE system under deep uncertainty (Moallemi et al., 2020). In other words, by performing an open policy exploration, it is possible to estimate the effect of testing a large set of policy combinations in both riparian countries.

Table 4.6. Policies and stochastic ranges considered for an open exploration.

Policies	Description	Range in Lithuania	Range in Latvia
1. Implementation of nutrient reduction options in arable land	Implementing nutrient reduction options in a fraction of the total arable land (fraction of land with nutrient reduction) by 2050	UNIF (0-0.75)	UNIF (0-0.75)
2. Implementing riparian buffers	Implementing riparian buffers to intercept a fraction x of the agricultural runoff of the arable land with nutrient reduction (Along with policy 1)	UNIF (0-1)	UNIF (0-1)
3. Implementing constructed wetlands	Implementing constructed wetlands to intercept a fraction y of the agricultural runoff of the arable land with nutrient reduction (Along with policy 1). Note that the sum of the fractions of riparian buffers (x) and wetlands (y) is equal to 1.	UNIF (0-1)	UNIF (0-1)

Policies	Description	Range in Lithuania	Range in Latvia
4. Implementing organic agriculture	Implementing organic agriculture in a fraction of the arable land with nutrient reduction	UNIF (0-0.75)	UNIF (0-0.75)
5. Conversion of arable land to grasslands	Converting 10% of the total arable land into grasslands by 2030	[0,1]	[0,1]

4.3. RESULTS

Here we focus on the policy insights as a product of the PM intervention (Gray et al., 2018). They are derived from implementing Phase III of the PM cycle, that is, using the SD model for policy evaluation. Policy insights arise from two analyses, the first exploring the effect of policy ambition in a transboundary context, and the second, an open policy exploration.

Interested readers and potential users can also explore the model and implement policies using the Nexus Assessment Policy Assessment Tool (NEPAT) (nepat-dev.nexogenesis.eu), a web-based decision support tool that was developed to foster policy dialogues with riparian stakeholders (Echevarría, Dkouk, & Nievas, 2024; Mooren et al., 2025).

4.3.1. Comparing policy ambition and transboundary cooperation

Combining multiple levels of policy ambition alongside levels of transboundary cooperation offers a rich picture of the policy futures for the Lielupe. The results reported in every grid of the matrix are the key variables' estimates after applying every combination of levels of policy ambition along with the different levels of transboundary cooperation (Table 4.5). The median and 90% CI for the key variables were estimated in 1000 simulations of the SD model (see Appendix A, Table 12), using a Sobol sequence sampling with the stochastic parameters as summarised in Appendix A, Table 10 and 11.

To ease the analysis, Table 4.7 presents the simulated relative changes and 90% CI of the key variables for every combination of the distinct levels of transboundary cooperation and policy ambition, compared with the baseline scenario. The table is colour-coded based on the desirability (green- desirable, red – undesirable) of each variable to increase or decrease: for instance, decreasing cereal production is undesirable (red), decreasing nitrogen concentration is desirable (green), and increasing vegetation stocks is desirable (green). The colour code is consistent across the 9 combinations of each variable, showing darker (lighter) colours based on the values with higher (lower) magnitudes. The following paragraphs report on the results, analysing the matrix by rows (level of transboundary cooperation).

Table 4.7. Relative change of key variables for every combination of transboundary cooperation and policy ambition, compared with a baseline scenario.

Level of transboundary cooperation	Variables	Perc.	Level of policy ambition											
			Low				**Medium**				**High**			
			2020	2030	2040	2050	2020	2030	2040	2050	2020	2030	2040	2050
Bilateral	Total cereal production	5%	-2.4%	-13.9%	-12.5%	-11.8%	-8.1%	-13.5%	-18.2%	-22.4%	-8.3%	-20.1%	-24.8%	-22.1%
		50%	-2.2%	-11.0%	-9.8%	-12.2%	-3.2%	-13.6%	-15.9%	-18.2%	-7.5%	-19.5%	-21.9%	-26.3%
		95%	-4.2%	-8.2%	-10.7%	-10.6%	-4.4%	-13.0%	-15.7%	-16.5%	-8.0%	-16.5%	-22.7%	-25.6%
	Total carbon mass in vegetation	5%	1.1%	2.6%	2.6%	2.8%	1.2%	2.9%	2.9%	2.9%	1.3%	3.0%	3.1%	3.2%
		50%	1.2%	2.7%	2.7%	2.6%	1.1%	2.9%	2.9%	2.7%	1.3%	3.1%	3.1%	3.0%
		95%	1.1%	2.9%	2.5%	2.7%	1.1%	3.1%	2.8%	3.0%	1.2%	3.2%	3.0%	3.1%
	Nitrogen concentration	5%	9.3%	-14.9%	-6.3%	-19.8%	-7.3%	-31.9%	-25.6%	-34.8%	-17.6%	-27.1%	-52.2%	-52.7%
		50%	-3.4%	-10.3%	-19.9%	-22.4%	-7.1%	-20.7%	-35.9%	-35.6%	-11.5%	-30.5%	-55.9%	-56.2%
		95%	-0.9%	-9.6%	-18.0%	-16.5%	-4.4%	-18.1%	-31.7%	-33.7%	-11.2%	-29.1%	-47.6%	-49.0%
Unilateral (Lithuania)	Total cereal production	5%	-5.4%	-9.1%	-7.5%	-8.3%	-3.2%	-6.5%	-11.1%	-13.1%	1.1%	-7.6%	-13.6%	-12.2%
		50%	-2.4%	-5.1%	-5.8%	-6.3%	-1.7%	-6.7%	-8.3%	-9.5%	-1.9%	-9.1%	-9.7%	-12.2%
		95%	-2.3%	-3.4%	-5.7%	-6.1%	-3.0%	-4.9%	-6.8%	-7.0%	-1.9%	-7.5%	-9.7%	-10.8%
	Total carbon mass in vegetation	5%	0.6%	1.6%	1.6%	1.5%	0.6%	1.7%	1.7%	1.7%	0.7%	1.7%	1.7%	1.8%
		50%	0.6%	1.6%	1.6%	1.4%	0.7%	1.7%	1.6%	1.6%	0.7%	1.7%	1.8%	2.0%
		95%	0.6%	1.7%	1.5%	1.6%	0.6%	1.8%	1.5%	1.6%	0.6%	1.9%	1.7%	1.8%

	Variables	Perc.	Level of policy ambition											
			Low				**Medium**				**High**			
			2020	2030	2040	2050	2020	2030	2040	2050	2020	2030	2040	2050
Unilateral (Latvia)	Nitrogen concentration	5%	17.7%	-19.9%	-4.9%	-3.5%	16.6%	-16.6%	-9.0%	-19.8%	-5.9%	-27.7%	-13.3%	-26.0%
		50%	-1.0%	-4.7%	-11.7%	-10.8%	-5.1%	-9.1%	-19.6%	-18.8%	-6.0%	-16.3%	-29.0%	-28.7%
		95%	-0.2%	-5.1%	-11.2%	-8.2%	-2.5%	-8.4%	-18.0%	-18.9%	-4.9%	-14.1%	-25.7%	-26.4%
	Total cereal production	5%	-3.3%	-5.9%	-6.0%	-9.1%	0.4%	-6.9%	-6.8%	-7.3%	-0.4%	-6.3%	-10.1%	-12.0%
		50%	-1.3%	-5.6%	-3.3%	-5.7%	-1.4%	-4.5%	-4.6%	-5.8%	-0.5%	-6.2%	-4.8%	-8.7%
		95%	-1.2%	-4.0%	-4.9%	-6.2%	-3.2%	-5.4%	-6.4%	-5.8%	-3.9%	-6.2%	-6.3%	-7.2%
	Total carbon mass in vegetation	5%	0.5%	1.2%	1.1%	1.2%	0.5%	1.2%	1.3%	1.2%	0.5%	1.3%	1.4%	1.5%
		50%	0.5%	1.2%	1.1%	1.0%	0.5%	1.3%	1.2%	1.0%	0.5%	1.3%	1.3%	1.3%
		95%	0.5%	1.3%	1.2%	1.2%	0.5%	1.3%	1.2%	1.3%	0.5%	1.4%	1.3%	1.6%
	Nitrogen concentration	5%	5.6%	-17.8%	10.6%	-7.5%	-11.8%	-22.2%	-19.8%	-11.5%	-13.1%	-16.4%	-22.2%	-25.7%
		50%	-2.2%	-5.5%	-8.6%	-7.1%	-5.2%	-7.8%	-14.3%	-13.1%	-2.3%	-10.5%	-22.7%	-25.7%
		95%	-1.1%	-3.9%	-9.1%	-6.0%	-0.3%	-6.6%	-12.6%	-14.4%	-4.3%	-10.2%	-20.0%	-19.3%

A visual inspection of Table 4.7 shows that bilateral cooperation brings the most marked changes in the variables under consideration. Results exhibit a trade-off between the food sector at the expense of improvements in the water and ecosystems sectors, broadly consistent across a 90% confidence interval. The total cereal production shows a significant drop in the transition to the decade 2030 for every level of policy ambition. This is due to the policy of grassland expansion, which considers a 10% grassland transition from arable land by 2030. Additionally, as organic agriculture expands (medium and high levels of ambition), the expected drop in total crop production is steeper, ranging from 12 to 26% by 2050 (due to the reduced crop yield of organic agriculture, see Appendix A, Table 9). The grasslands policy implies a consistent but relatively low increment of the basin's carbon mass in vegetation of ca. 3%. More ambitious policies show only marginal increments in the basin's vegetation stocks. Nutrient concentration exhibits a gradual decrease as the fraction of land with nutrient control increases. The expected decrease in nutrient concentration exhibits a wide range between -56 to -22% by implementing (high to low) ambition policies by 2050.

Unilateral actions exhibit the same trends as described in the scenario of bilateral cooperation, yet with differences in magnitude in each riparian country. Considering unilateral actions by Lithuania, the expected drop in cereal production at the basin level will be in the range of -13 to -8% by implementing (high to low ambition) policies by 2050. Similarly, total vegetation stocks are expected to slightly increase in the range of 1-2% by implementing (low to high ambition) policies by 2050. Nutrient concentration by Lithuanian action is expected to have a reduction in the range of -29 to -11% by implementing (high to low) ambition policies by 2050.

Unilateral actions in Latvia show a relatively lower effect compared to Lithuanian unilateral action. If Latvia operates in isolation, the expected drop in cereal production at the basin level will be in the range of -12 to -7% by implementing (high to low ambition) policies by 2050. Similarly, total vegetation stocks are expected to slightly increase by ca. 1% by implementing (low to high ambition) policies by 2050. Nutrient concentration by only Latvian action is expected to have a reduction in the range of -26 to -6% by implementing (high to low) ambition policies by 2050.

The simulated changes associated with unilateral actions by either Latvia or Lithuania can be traced to the current land use in the riparian countries. As evidenced in Appendix A, Table 1, Lithuania accounts for roughly two-thirds of the basin's arable land. This explains that policies have a more marked effect if taken only in Lithuania compared with Latvia.

Results also indicate that unilateral action hampers the effect of long-term ambitious policies in the basin. This is evident by exploring the case of ambitious unilateral action of the riparian countries in the long term. By taking such an approach, Latvia and Lithuania could reach the target of 20% nitrogen reduction by 2040. In contrast, a scenario of medium ambition and bilateral action would reach the same target in 2030. This suggests that choosing bilateral cooperation over unilateral action can lower the individual burden of riparian actions and achieve water quality objectives faster, in the order of decades.

4.3.2. Open policy exploration

Results of the open policy exploration are compared with the reference baseline. The median and 90% CI for key variables are presented in Appendix A, Table 13. These values were estimated based on1000 simulations of the SD model, using a Sobol sequence sampling with the stochastic policies (Table 4.6) and parameters (see Appendix A, Table 10 and 11). To ease the analysis, Table 4.8 presents the expected relative changes and 90% CI of the key variables.

Table 4.8. Relative change of key variables for an open policy exploration compared with a baseline scenario.

Variables	Perc.	2020	2030	2040	2050
Total cereal production	5%	-1.2%	-8.2%	-10.0%	-12.6%
	50%	-2.3%	-5.2%	-7.1%	-8.4%
	95%	-3.1%	-4.7%	-7.0%	-9.0%
Total carbon mass in vegetation	5%	0.3%	0.4%	0.7%	0.9%
	50%	0.6%	1.5%	1.6%	1.4%
	95%	1.0%	2.6%	2.1%	2.2%
Nitrogen concentration	5%	4.3%	-31.1%	-17.7%	-37.1%
	50%	-5.2%	-14.1%	-26.4%	-28.6%
	95%	-3.8%	-12.2%	-20.5%	-20.6%

The results of Table 4.8 are congruent with the observed behaviour of policies as presented in the previous section (Table 4.7). That is, even by deploying the policies in a large decision space, water and ecosystem variables show dynamic improvement at the expense of reducing food production in the long term. By deploying the policies presented above, nitrogen concentration is expected to decrease by 29%, and total carbon mass in vegetation is expected to increase slightly by ca. 2%. In contrast, total cereal production is expected to drop by ca. 8%.

However, some differences of magnitude are worth noting in comparison with the previous section. First, the fall in total food production is not as sharp as presented in Table 4.7 (26% for ambitious bilateral cooperation). This might be due to considering a wide range of organic agriculture implementation, not forcing it to be as high as presented at the ambitious policy level (see Table 4.7). Second, a reduction of 29% in nitrogen concentration for 2050 aligns with the scenario of medium bilateral cooperation in 2040.

To complement the previous analysis, which focused on tracking the temporal evolution of the key variables, Table 4.9 shows summary statistics to track the effect of policies considering the whole simulation timespan. These results are overall consistent with previous analyses (see Tables 4.7 and 4.8), highlighting the synergies between nitrogen concentration and carbon mass in vegetation, with a trade-off for total cereal production. Yet, new insights emerge from analysing the standard deviation of the key variables. For cereal production, there is a slight reduction in variability, possibly linked to reductions in arable land. For carbon mass in vegetation, the variability increases; this can be explained from policies exploring the substitution of arable land—a land use that does not contribute to this variable—for grasslands and NbS (e.g. wetlands and riparian buffers)—land uses that bring (stochastic) benefits in terms of carbon in vegetation. Notably, there is also a significant decrease in nitrogen concentration. This is an important finding, as it shows that the policies under consideration—which are strongly focused on nutrient control—would be effective not only to decrease nutrient concentration but also to reduce its long-term variability. Our exploratory analysis, therefore, demonstrates that deploying policies 1 to 5 (Table 4.6) offers significant and quantifiable benefits to improve long-term water quality in the basin, even under deep uncertainty.

Table 4.9. Summary statistics for the model's key variables over the simulation timespan, taking the median values of 1000 model simulations in two scenarios: baseline and open exploration.

Summary statistics	**Total cereal production (1,000 ton/month)**			***Total carbon mass in vegetation (Mton C)***			***Nitrogen concentration (mg/L)***		
	BL	*OE*	*Relative change*	BL	*OE*	*Relative change*	BL	*OE*	*Relative change*
Mean	199.3	185.8	-7.3%	75.8	76.7	1.2%	2.9	2.4	-18.4%
Median	194.7	179.8	-8.3%	77.2	78.3	1.5%	2.0	1.6	-26.7%
Standard Deviation	22.1	21.5	-2.4%	3.7	4.0	8.4%	2.1	1.8	-12.2%
Minimum	160.1	154.3	-3.8%	67.1	67.1	0.1%	0.8	0.6	-30.4%
Maximum	274.5	264.2	-3.9%	79.7	81.0	1.6%	8.2	7.7	-6.5%

BL=Baseline, OE=Open exploration

4.4. DISCUSSION

4.4.1. Nexus modelling for the Lielupe

This research contributes to ongoing efforts to develop nexus modelling in a participatory way (Pagano et al., 2025; Hurtado et al., 2024) and so responds to recent research calls for a transition towards stakeholder-driven and interdisciplinary nexus research and practice (Sušnik & Staddon, 2021). Using a modelling cycle approach and adopting a nexus perspective, we illustrate a stakeholder-driven pathway from qualitative to increasingly quantitative system tools in exploring plausible futures for the Lielupe river basin. Although we focus on using a simulation model and its results, our structured approach illustrates how a quantitative model can evolve according to stakeholder priorities. Our research experience in the Lielupe followed a path that started from a generic WEFE systems representation and culminated in a more focused nexus problem. This resonates with a basic, but often neglected principle in System Dynamics, in which the epistemic purpose of modelling is not to represent a system *per se*, but rather a problem from a systemic perspective (Sterman, 2000, pp. 89). The participatory modelling approach led to the exploration of policies to address the wicked problem of nutrient pollution by focusing on assessing synergies and trade-offs across three main nexus sectors: water, food and ecosystems—with a strong emphasis on water quality.

Our results show both synergies and trade-offs across nexus sectors in the Lielupe river basin. Generally, alternatives that favour nutrient concentration reduction (water sector) have a minor benefit in the basin's vegetation stocks (ecosystems sector), yet reduce food production (food sector). This means that environmental benefits come at a cost, a common-sense economic finding. We found that implementing solutions for nutrient control (NBS, organic farming, and arable land reduction) reduces crop production. This comes from two factors: first, from the expected drop in crop yield from organic farming (Meemken & Qaim, 2018), and second, from the land-use trade-offs required to build the NBS or to transition to other landscapes such as grasslands (Trodahl, Jackson, Deslippe, & Metherell, 2017). It is worth noting that the first factor is limited by our approach of modelling crop yields exogenously rather than endogenously, which would imply estimating yield as a function of soil quality. Other research has found that considering soil quality allows for a counterfactual scenario in which organic practices promote the preservation of a relatively high yield versus a scenario of long-term yield reduction due to soil degradation (Rashidian et al., 2025). Despite this limitation, our model covers a wide range of climate change-driven crop yield projections, an essential feature for performing exploratory analysis. The second factor of land-use trade-offs is modelled endogenously; such a modelling feature provides wide analytic capabilities and further insights about the scale of change that is needed to achieve local sustainability goals.

Overall, our results illustrate nexus synergies between water and ecosystems, but also a trade-off —a cost — that will likely affect the food sector. Yet, such a cost is relatively minor compared to its expected benefits. Deploying an open policy exploration analysis on the nutrient control options shows that achieving a nutrient concentration reduction of 30% would imply a reduction of less than 10% in food production, with a 1.5% increase in vegetation stocks. These findings are in line with previous research highlighting the role of wetlands in addressing the trade-off between water quality and crop production (Cheng, Van Meter, Byrnes, & Basu, 2020; Matsuzaki et al., 2019) and with recent modelling experiences developed in Latvia highlighting trade-offs across food and ecosystems sectors (Sušnik et al., 2021). Although we consider a trade-off between organic farming practices and crop yield based on authoritative meta-analyses (Meemken & Qaim, 2018; Tuomisto et al., 2012), recent evidence suggests that implementing organic farming-related practices may maintain crop yield over the long term whilst providing biodiversity benefits (Berger et al., 2025). As this is a complex and contextual problem (Seufert & Ramankutty, 2017), future studies—both empirical (e.g. Berger et al., 2025) and model-based (e.g. Paturu & Varadarajan, 2025; Rashidian et al., 2025) —are necessary to better understand the potential long-term synergies and co-benefits of practising organic farming in terms of crop yield and biodiversity in the Lielupe and in the wider Baltic Region. Significantly reducing nitrogen concentration in the Lielupe requires large-scale and long-term cooperation in the basin. By cooperating, basin-scale benefits are achieved a decade faster and with 25% less individual effort compared to a scenario in which one riparian country acts and the other remains idle. Limits to scale are related to the nitrate reduction efficiency of the NBS systems, which is about 50%. This means that even by controlling all agricultural runoff using NBS as end-of-pipe treatment, aiming for nitrate reductions above 50% would require focusing on reducing nitrogen inflows, instead of only outflows (Galloway, Bleeker, & Erisman, 2021). Such alternatives include lowering the use of fertilisers (e.g., organic agriculture) or transitioning to other land uses (e.g., grasslands). Additionally, a long-term perspective is needed due to the NBS project's construction lead times and the basin's natural nitrogen accumulations. Nietch et al. (2024) recently reported that the design and construction of an advanced constructed wetland system of 55 ha took 11 years. Likewise, natural basin accumulations that delay nitrogen transport from fields to water bodies are in the range of 4-20 years (Dessirier et al., 2023; Melland, Fenton, & Jordan, 2018; van Meter & Basu, 2017; Vervloet, Binning, Borgesen, & Hojberg, 2018). This research engages with modelling and communicating these uncertainties and delays to stakeholders. We therefore contribute to helping prevent unrealistic and short-term expectations that can dominate nutrient policies at the river basin, national, and regional scales (Baltic Sea Centre, 2024; Basu et al., 2022; Meadows, 2008; Petersen, Blicher-Mathiesen, Rolighed, Andersen, & Kronvang, 2021).

Remarkably, intercepting a high proportion of the total agricultural runoff using NBS is not a land-intensive alternative. For instance, Nietch et al. (2024) recently reported a wetland to drainage area ratio of 1% for a wetland treatment system in the US, an estimate in line with other modelling exercises (Castellano, Archontoulis, Helmers, Poffenbarger, & Six, 2019). Our results suggest that significant long-term reductions in nitrogen concentration (ca. 35%) can be reached if half of the total basin's agricultural runoff has nutrient control. This means, for instance, that the effective land devoted to constructed wetlands would be roughly equivalent to 0.5% of the current Lielupe's arable land area (ca. 40,000 ha). Coming from an SD approach, these figures are aggregated and not spatially explicit. Thus, results may not be interpreted as if a large-single nutrient control intervention should be the only way forward to improve water quality in the Lielupe. On the contrary, significant water quality improvements at the river basin level are likely to be realistically achievable as the sum of many small-scale initiatives across the Lielupe's arable land area (for instance, across half of the farms in the basin) (Jacobsen, Anker, & Baaner, 2017).

4.4.2. Beyond the model: local trends and perspectives on a more sustainable future for the Lielupe

Local advances in the basin signal that this might be a plausible future in the basin. A first wave of constructed wetland pilots has already been built and is constantly monitored to assess the system's nutrient reduction efficiency (Grinberga, 2022; Lagzdiņš, 2025). Very recently, in April 2025, the Latvian Ministry of Agriculture announced the allocation of a 4M Euro budget to build new constructed wetlands (Latvijas Sabiedriskais medijs, 2025). According to a local expert, this budget may allow for building around 40 new wetlands (ca. 100k Euro per wetland) (Lagzdiņš, 2025). Following the budget allocation, deciding how to distribute such land and the design of the NBS is not a trivial task. It would require the involvement of multi-stakeholders (e.g. farmers, landowners, government officials, academics and NGO representatives) and should be informed by multiple fields of knowledge (e.g., landscape architecture, civil and environmental engineering, economics, management, law, sociology, and others).

An important dimension that can inform the decision-making after establishing land requirements for NBS is to develop an economic evaluation of nutrient control policies. A first approach would imply developing a whole life-cycle economic evaluation of the NBS transition, including capital costs (e.g., cost of purchasing plots to build NBS and construction costs) and maintenance costs discounted to present value (Chairat & Gheewala, 2024; Nietch et al., 2024). Further, a more comprehensive evaluation can include environmental benefits alongside financial costs (Alshehri, Harbottle, Sapsford, Beames, & Cleall, 2023). Despite this being a complex and fuzzy task, it can be done using simulation models, like the one proposed in this article, that dynamically consider locally relevant environmental variables and their response to policy alternatives (e.g., nitrogen concentration and carbon mass in vegetation) (Alshehri et al., 2023; Chairat & Gheewala, 2024; Sušnik, Masia, Kravčík, Pokorný, & Hesslerová, 2022). Other options to help quantify environmental benefits include preventing the payment of economic penalties due to unmet water quality objectives under international agreements, such as the Water Framework Directive (Kallis & Butler, 2001; Martin-Ortega, 2012). Likewise, increasing vegetation stocks and restoring fluvial and delta ecosystems can be associated with a broad set of services, such as provisioning, regulating, and cultural (Maseyk, Mackay, Possingham, Dominati, & Buckley, 2016; Riis et al., 2020) and even be connected to financial schemes such as payment for ecosystem services (Salzman, Bennett, Carroll, Goldstein, & Jenkins, 2018), though care would need to be taken so as not to ‘commodify’ ecosystems (Wunder et al., 2020).

Despite this being a promising way forward, it is one likely to face resistance. All across Europe, powerful farmer groups, often representing the interests of large-scale actors, have been actively opposing environmental policies in recent years (van der Ploeg, 2020). A steep increase in large-scale farming in Latvia at the expense of reducing native grasslands might be an indicator of this situation also taking place in the Lielupe over the 21st century (Melece & Shena, 2018). In such a contested situation, it is safe to assume that farmers and other actors (e.g. academics and government officials) frame agro-environmental issues differently. According to Brugnach et al. (2011), a possible way forward in this context might be to take a negotiation strategy. Following that path could mean that progressive policies and compensation mechanisms are implemented to secure farmers’ livelihoods, as they are required to make landscape changes (e.g. implement NBS in their farms) or reduce crop yields (e.g. transition to organic farming) as part of improving the basin’s environmental status.

In the scenario of more dialogue taking place, more cooperative strategies could be deployed, such as developing interdisciplinary projects with a strong co-creation focus (see Mooren et al. 2025). Participatory modelling products, such as the one presented in this article—including not only a model but its associated policy insights—can therefore inform local and regional policy dialogues (NEXOGENESIS, 2024, 2025). However, beyond using models to inform policy, PM settings offer other social outcomes. Further research is needed to explore, for instance, how PM settings could promote stakeholder learning and strengthen relationships that potentially inform and facilitate policy transitions in the Lielupe and other basins facing similar issues. Likewise, there is a need for more research into the PM processes that underpinned the development of the products presented in this paper. A deeper understanding of how to develop successful participatory approaches that not only inform but also facilitate sustainability transitions in river policy represents a promising avenue for both researchers and policymakers.

Implementing such transitions becomes increasingly complex in a transboundary setting. Yet here we showcase some of the benefits of taking a cooperative approach to improve water quality in the basin. Active cooperation can, therefore, be considered an incentive for both riparian countries to achieve currently unmet WFD objectives (Albiac, Calvo, & Esteban, 2024). Establishing cooperation mechanisms from relatively small technical scales to a high political level is needed to promote equitable achievement of water quality objectives in the basin (Milman et al., 2020; Schmeier & Shubber, 2018). Technical cooperation can be done via multiple initiatives, such as piloting NBS in both countries and even by establishing a joint water quality monitoring programme in the basin. Likewise, political cooperation could be done by creating an international river basin organisation for the Lielupe, with Latvia and Lithuania as new participants of the UN International Watercourses Convention (Gupta, 2016; McCaffrey, 2008). This organisation could coordinate the multi-stakeholder dialogues needed to implement a large-scale and long-term nutrient control strategy in the transboundary basin. Despite such broad potential, the Lielupe and other agrarian transboundary river basins have yet to reap the long-term benefits of such wide and cross-level international cooperation.

4.5. CONCLUSION

In this article, we present a model-based operationalisation of the Water-Energy-Food-Ecosystems (WEFE) Nexus approach in a transboundary river basin. The application is illustrated in the context of a transboundary and collaborative stakeholder setting to explore locally relevant policies for the Lielupe, a river basin shared between Latvia and Lithuania. By applying a model-based policy analysis framework, we showed how various stakeholder-driven milestones contributed to developing a System Dynamics simulation model that helps explore long-term WEFE policy alternatives in the basin. The results of the model offered insights regarding the long-term effects of land-use transitions in the Lielupe River basin from two analytic perspectives: the first, by exploring the effect of policy ambition in a transboundary context; and the second, via an open policy exploration. More specifically, here we illustrated three policy levers useful to understand the long-term impact and cascading effects of transitioning from an intensive agriculture landscape to a more *natural* land use in the basin: implementing nature-based solutions (NBS) to control nutrient diffusion in the basin; reducing arable land to extend native grasslands; and implementing organic agriculture.

Transitioning to an agricultural landscape that uses NBS to control nutrient pollution is a promising alternative, provided it is implemented at a large, basin scale. The larger the arable land fraction with nutrient control alternatives, the higher the expected reduction of nitrogen concentration in the basin. Yet, this comes with a relatively minor trade-off in food production —a finding which further empirical and model-based research on food production could quantify further. Our modelling results imply that the basin-scale effect of implementing NBS alternatives to reduce nitrogen concentration depends on both technical and socio-political factors. On the technical side, we explored how the intrinsic variation of nitrogen removal efficiency of the NBS systems, as well as future river flow variability, propagate into uncertainty in reaching nitrogen reduction objectives in the basin. Despite such uncertainty, results suggest that large-scale use of NBS to control nutrient pollution is a policy that exhibits robustness in meeting objectives to improve water quality in the basin in the long term. From a socio-political perspective, we showed that taking unilateral actions to improve water quality in the basin is insufficient. In contrast, by taking cooperative actions, each country enhances the efforts of its counterpart, which is translated into lower individual investments and faster achievement of objectives compared to a scenario of unilateral action.

A more radical alternative in terms of land-use change is reducing arable land to increase grasslands in the basin. Reducing arable land can lead to a long-term reduction in agricultural nitrogen loads in surface water, as the nitrogen stock in the soil will begin to deplete due to the absence of further fertiliser application. Replacing arable land with grasslands also offers the benefit of increasing carbon vegetation stocks, with potential wider biodiversity benefits in the basin. Changing land-use in this way leverages synergies across the *Ecosystems* and *Water* sectors, yet with important trade-offs in the *Food* sector. Agriculture degrowth has direct benefits in terms of water quality and ecosystems. However, this is a costly alternative as productive lands are expensive. Just as with implementing NBS, both biophysical and socio-political factors play a role in achieving the intended benefits derived from increasing grasslands in the basin. From a biophysical perspective, an important point to consider is the long-lasting nutrient leaching from agricultural fields, which persists for years after cropping activities have stopped. We also demonstrated high uncertainty in trying to account for the potential effects of increasing vegetation stocks and animal biodiversity. Likewise, socio-political factors drive key future decisions regarding the extent of agricultural degrowth and whether such initiatives happen unilaterally or in cooperation with other riparian countries.

We present a broad account of the model capabilities and results of a simulation model for a transboundary setting. The model integrates WEFE nexus sectors in the basin but keeps a distinction between upstream and downstream countries, accounting for their asymmetrical relationship. This modelling strategy, exhibiting both joint and decoupled features, offers opportunities for future research in the field of modelling and policy analysis. Future SD models can adapt this strategy to other transboundary basins, aiming to reflect challenges that were not accounted for in this article (e.g. water scarcity and floods). In the field of policy analysis, new studies can explore how this model, or similar participatory modelling applications, might be useful in enhancing and providing learning opportunities for stakeholders in the context of transboundary WEFE policy dialogues. Here we show how integrating biophysical and socio-political dimensions provides analytic opportunities to enhance transboundary resource dialogues. By exploiting these capabilities, riparian stakeholders may use co-created simulation models to facilitate dialogue and even negotiate their role in solving resource nexus challenges in transboundary river basins.

5

TRACKING MODEL EVOLUTION AND LEARNING IN ENVIRONMENTAL PARTICIPATORY MODELLING

Based on the article :

Amorocho-Daza, H., Avellán, T., Sušnik, J., Brēmere, I., Indriksone, D., Ryfisch, S., van der Zaag., P. & Jill Slinger. (under review). Opening the black box: tracking model evolution and learning in environmental participatory modelling. Under review in Systems Research and Behavioural Science

Abstract

Participatory modelling (PM) enables experts and stakeholders to co-create models that facilitate learning about an environmental problem. PM applications often focus on outcomes, for instance describing final models and the insights derived from their use. Yet, there is a general lack of knowledge about how models evolve while participants learn in a PM setting. By conceptualising modelling activities as nested within a wider stakeholder engagement process, here we develop and apply a structured approach to track model evolution and participants' learning. Model evolution is analysed using Meadows' leverage points. Knowledge generation is used as a proxy for learning and assessed for both modellers and stakeholders using quantitative and qualitative tools. The methods are applied in an international and interdisciplinary resource nexus project, on the Lielupe River Basin (Latvia and Lithuania). Our results illustrate that PM is a socially driven endeavour that can foster participants' learning across and beyond a modelling cycle. By tracking model evolution, we illustrate how early-stage stakeholder feedback lays 'deep' foundations that determine the final model capabilities. Our results evidence participants' knowledge generation, yet track temporal and structural differences between the modeller and the stakeholders. This research contributes to improving transparency and replicability in PM research and practice. Despite environmental PM promoting stakeholder learning, the translation of insights into real-life and sustainable change remains an open academic and practical challenge.

Keywords: participatory modelling, stakeholder engagement, learning, model evolution, leverage points, river basin management, environmental policy

5.1. INTRODUCTION

Background

Using models can support informed decisions in addressing wicked environmental issues (Kirschke and Newig, 2017; Sušnik and Mellios, 2025). They can be used as tools that support stakeholder groups (e.g. citizens and policymakers) in understanding the nature of complex socio-environmental systems while exploring effective interventions towards a more sustainable and desired future (Amorocho-Daza et al., 2025; Ford, 2010; Meadows, 2008; Meadows & Robinson, 1985). Although various modelling approaches are suitable for this task, each with its own capabilities and limitations (Kelly et al., 2013, Voinov et al., 2018), here we focus on System Dynamics (SD) as the modelling method.

Selecting who takes part in the modelling process is an important decision. Simply put, the model can be built only by a small group of 'experts' (e.g. modellers, environmental scientists) or also including a larger group of stakeholders (e.g. policy makers and citizens) (Voinov and Bousquet, 2010). Recent reviews point out that expert-driven approaches pre-dominate over participatory approaches in SD sustainability research (Moallemi et al., 2021). Various authors argue that the balance should change towards participation, as human society is at the centre of socio-environmental issues (Amorocho-Daza et al., 2024, Moallemi et al., 2021). In other words, modelling the environment should involve stakeholders in the model development process. The stakeholders who need to be involved include the people who will likely take an active role in implementing a solution supported by the model, and those potentially affected by how the model could be used in local decision-making processes (Palmer, 2017).

Group-Model Building (GMB) has a long-standing SD tradition of modelling with stakeholders (or so-called client groups) at the firm level (Andersen and Richardson, 1997; Vennix, 1999). GMB focuses on the processes and techniques that support the task of building SD models with stakeholders (Andersen and Richardson, 1997; Luna-Reyes et al., 2006) as well as on exploring the outcomes of deploying such interventions (Rouwette et al., 2002; Scott et al., 2016) and the causal mechanisms that drive them (Rouwette et al 2011). In parallel to GMB efforts, other SD researchers working on environmental problems have coined the term participatory SD to explore the model-building process in a different setting, that is, multi-stakeholder, public policy-oriented debates (Stave 2002, 2010; Videira et al., 2010, 2012). Participatory SD forms part of a larger participatory environmental modelling paradigm that is gaining momentum across multiple research fields, conveniently clustered as participatory modelling (Basco-Carrera et al. 2017; Voinov & Bousquet, 2010; Voinov et al., 2016).

Participatory modelling (PM) is characterised by the integration of a variety of qualitative and quantitative methods structured within a wider stakeholder engagement process to support environmental decision-making (Videira et al. 2010; Voinov et al., 2018; Gray et al. 2018). Here we employ Jordan et al. 's (2018, p. 1047) definition of PM "*as a purposeful learning process for action that engages the implicit and explicit knowledge of stakeholders to create formalized and shared representations of reality*". In general, stakeholders provide wide and deep contextual knowledge into PM discussions, as they are experts on local issues, both from their lived experience and their own domain knowledge (Jordan et al., 2018; Voinov & Bousquet, 2010; d'Hont and Slinger, 2022). In contrast, what modellers bring to a PM intervention is often a set of generic skills and tools (e.g. system thinking and modelling) as well as specific knowledge of certain domains (e.g. STEM fields), which can facilitate the systemic understanding of a problematic situation (Elsawah et al., 2024). By leveraging the interaction of these actors, PM has been used to address socio-environmental problems related to agriculture, water management, and biodiversity conservation, among many others (Brown Gaddis, Vladich, & Voinov, 2007; Gray et al., 2018; Videira, Antunes, & Santos, 2009; Videira et al. 2012).

Advancing Participatory Modelling

Building on Jordan et al.'s (2018) definition of PM, this article proposes structured approaches to characterise two outcomes across a long-term PM project: model evolution and participants' learning. There are abundant examples showing PM models as final products, focusing on their capabilities and the policy recommendations that emerge from using them (see Amorocho-Daza et al., 2026; Gray et al., 2018; Elsawah et al., 2017). Rather than focusing on the SD model as a final product, here we use a different analytical approach to track how an SD model *evolves* across a participatory process. By taking such a perspective, a simulation model is no longer perceived to emerge from a black box (for instance, a PM project). On the contrary, it facilitates studying the SD model as a highly dynamic and co-created artefact. In parallel to model evolution, we also propose an approach to track participants' learning in PM.

Despite recent literature calling for new ways to measure social learning in PM, tracking participants' learning at different stages of a PM project is rarely reported (Zellner, 2024; Jordan et al., 2018). Here, we use both quantitative and qualitative tools to characterise participants' learning, explicitly including both stakeholders and modellers, across a PM project. Quantitative assessment of learning is done via self-reported surveys exploring knowledge generation across various domains and types of knowledge (See Brand et al 2013). For the qualitative assessment, an interview with the modeller opens reflections about the modellers' learning process and decision-making rationale, a practice in line with recent literature that calls for reflective modelling (Amorocho-Daza et al., 2024; Ter Horst, 2025). The article also explores features that emerge from the interaction of model evolution and participants' learning.

In this article, we report a structured approach tracking model evolution and participants' learning as products of an environmental PM project. Our aim is to contribute to improving transparency, structured evaluation, and replicability in PM research and practice. Our overall approach relies on conceptualising a PM cycle as part of a larger stakeholder engagement process. The methods section explains our analytical approach to assess model evolution, knowledge generation, as well as their interaction, in PM (Section 2). Our results are later presented and discussed (Sections 3 and 4). The article closes with a conclusion (Section 6).

5.1.1. Case study

PM was applied within the EU Horizon 2020 NEXOGENESIS (NXG) project, which addressed the complex interconnections within the Water-Energy-Food-Ecosystems (WEFE) nexus using modelling, policy analysis, and governance tools across five case studies in Europe and South Africa (Mooren et al. 2025b, Sušnik, 2024a, 2024b). More specifically, we use the transboundary Lielupe River Basin as a case study (CS) (17,788 sq.km).

The Latvia-Lithuania transboundary Lielupe River Basin is characterised by a flat landscape with fertile soils supporting agriculture (cereals, fodder crops) and dairy farming activities. Previous studies have identified a connection between land-use patterns in the CS area with multi-sectoral impacts (Sušnik et al 2021). For instance, (i) the nutrient pollution of agricultural origin deteriorates the water quality (Siksnane and Lagzdins 2020), and (ii) increased area of cropland comes at the expense of reducing meadows and pastures, potentially causing pressure on local ecosystems (Melece and Shena 2018). Given the transboundary nature of the basin, water quality is a wicked problem in the Lielupe River Basin, in which upstream pollution (from Lithuania) affects water quality downstream (in Latvia).

One of the project's core outcomes was to co-develop simulation models and a decision support tool in which policy issues could be explored and discussed with and by local stakeholders using a WEFE Nexus perspective. To fulfil this purpose, a system dynamics simulation model was co-created between a modeller (this article's lead author, and external to the river basin), the CS leads (co-authors in this article, and local to the river basin), and a group of stakeholders of the basin, which aimed at investigating the implications of long-term transboundary policies related to WEFE sectors in the basin. The model development, participatory setting and results are reported in Amorocho-Daza et al. (2026). The model was used to build a web-based decision support system, the Nexus Policy Assessment tool[4] (NEPAT), to facilitate a user-friendly stakeholder interaction to test policy scenarios and support international stakeholder dialogue (Sušnik, 2024a, 2024b).

Six structured engagement workshops provided an iterative setting for a participatory modelling approach within a larger stakeholder engagement strategy, as summarised in Figure 5.1 and detailed in Appendix B, Table 1 (Avellán et al. 2025, Amorocho-Daza et al. 2026). The engaged stakeholders came from municipalities, national and regional authorities, research, business, and NGOs. A summary of the organisations that took part in the workshops can be found in Appendix B, Table 2. A total of 28 organisations were represented, while 16 of those took part in more than two or more events. These efforts were complemented by two international focus groups for deeper engagement with the modelling tool, as summarised in Appendix B, Table 3.

[4] https://nepat-dev.nexogenesis.eu/

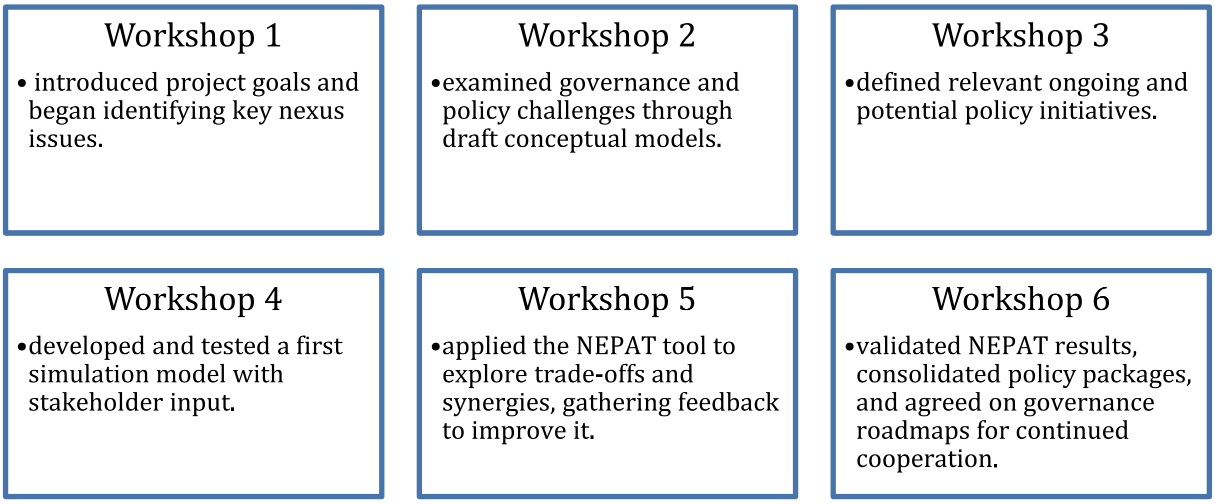

Figure 5.1 Overview of workshops in the Lielupe River Basin Case Study and their main aims

5.2. Methods

This section explains our operational approach to assess (i) model evolution, (ii) knowledge generation, and (iii) how these dimensions might interact across a PM intervention.

The assessment takes place in a participatory modelling setting that can be characterised via two closely related conceptualisations, one concerning participatory modelling activities (as proposed by Amorocho-Daza et al. (2025)) and the other focusing on wider stakeholder engagement (as proposed by Avellán et al. (2025)); each consisting of three generic phases, as described in Table 5.1. An important clarification is that participatory modelling happens within a wider stakeholder engagement process; that is, the modelling cycle is nested within the engagement cycle (see Figure 5.2). Despite the phases of stakeholder engagement closely aligning with the modelling cycle, the modelling foundations phase covers not only the co-exploration but also the start of the co-design stage. This comes from the fact that essential activities of co-design, such as using conceptual maps or discussing policy goals, can be considered as foundations for the development of a quantitative model (i.e. model building and testing phase).

Table 5.1 Summary of the three main phases in a participatory modelling cycle and stakeholder engagement strategy as proposed by Amorocho-Daza et al. 2025 and Avellán et al. 2025.

Phases	Stakeholder engagement (Avellán et al., 2025)	Modelling cycle (Amorocho-Daza et al., 2025)
I	Co-exploration: this is the information phase, its focus lies on contextual understanding and raising stakeholder awareness.	Modelling foundations: This phase is qualitative, focusing on defining the model's purpose in alignment with the problem or issue to be analysed. This is followed by early systems conceptualisation of the issue.

Phases	Stakeholder engagement (Avellán et al., 2025)	Modelling cycle (Amorocho-Daza et al., 2025)
II	Co-design: this phase unfolds through consultation and involvement, enabling reciprocal knowledge exchange in which stakeholders provide input, feedback, and deliberation to articulate their concerns and aspirations.	Model building and testing: This phase is increasingly quantitative as it focuses on formalising the foundations of a simulation model that can structurally and behaviourally represent the problem of interest.
III	Co-development: this phase advances to collaboration and empowerment, where stakeholders jointly create solutions, negotiate transformation pathways, and, in cases of empowerment, take ownership of the decision-making process.	Model use and policy evaluation: The third and final phase focuses on using the simulation model to test and evaluate policy alternatives in line with the objectives defined in Phase I.

Here we aim to evaluate both knowledge generation and model evolution across a PM intervention, with stakeholder workshops playing a central role in the analysis, as depicted in Figure 5.2. Here we suggest that stakeholder workshops are key events for both modelling and stakeholder engagement cycles as they help shape the model, validate its structure as well as its outputs, and generate opportunities for mutual learning. Therefore, the analytical elements are connected to specific workshops and are framed in accordance with the stakeholder engagement and modelling cycle phases. The learning of both stakeholders and the modeller is assessed using quantitative (i.e. surveys) and qualitative (i.e. interviews) tools. We use the term 'knowledge generation' as a proxy of learning, based on the three dimensions of knowledge generation described by Brandt et al. (2013) and five 'types of analysis' (concerning the generic domains that are relevant but contested in the case study) as proposed by Avellán et al. (2022, 2025). The model evolution is analysed using the leverage points proposed by Meadows (1999, 2008). Finally, we explore connections across model evolution and knowledge generation.

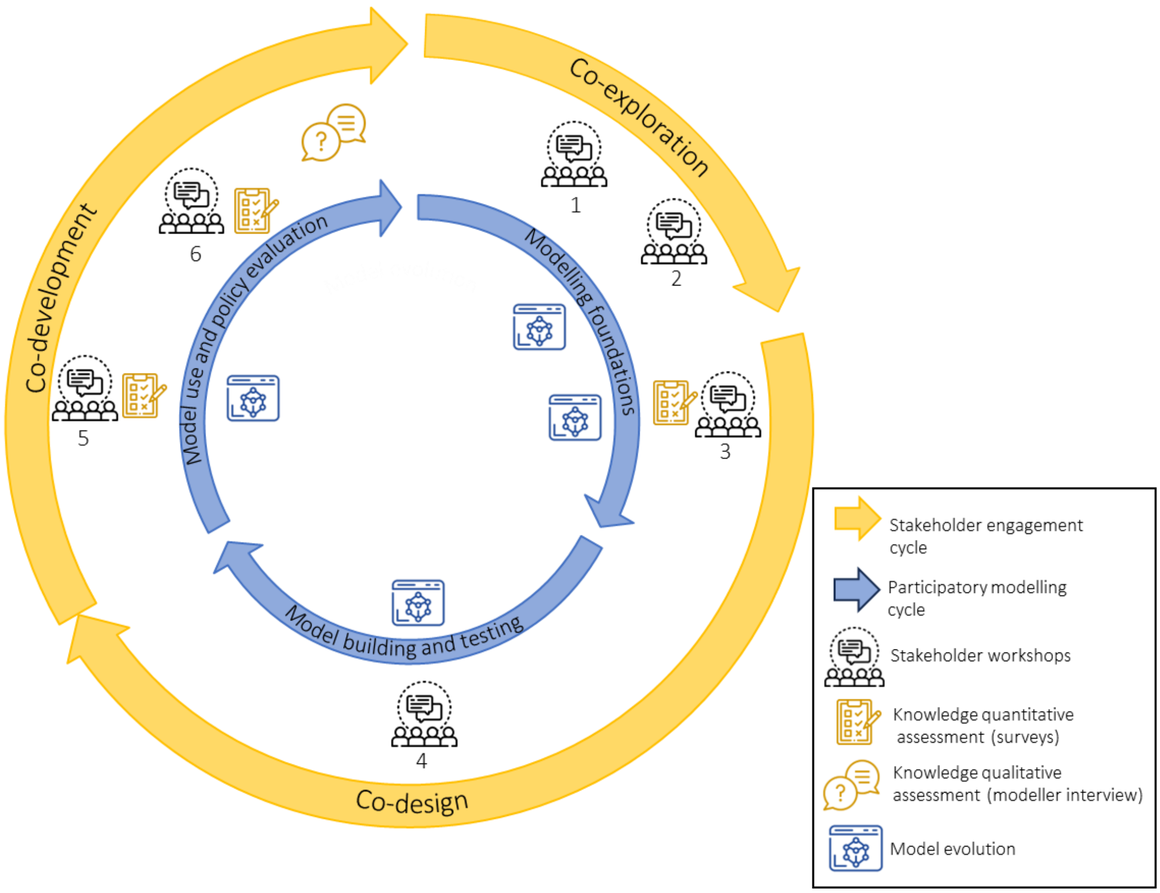

Figure 5.2 Methodological approach to assessing model evolution and learning in participatory environmental modelling.

5.2.1. Assessing model evolution

We used a structured approach to track the model's evolution over time as a PM product utilising Meadows' (1999, 2008) system leverage points hierarchy to characterise a set of model changes after considering stakeholder feedback. Meadows (1999, 2008) identifies twelve intervention 'points' in a system and suggests that the most effective way to drive systemic change lies in 'deep' leverage points, as opposed to 'shallow' leverage points. Despite the list originally focusing on places to intervene in real-world systems, here we use it in the context of crafting a model of a real-world issue. This is feasible as Meadows uses systems modelling language to characterise the leverage points (e.g. parameters, stocks, delays, feedback loops, etc), and lays a challenge to conventional quantitative SD modelling to test 'deeper' policy levers beyond parameters and feedbacks (e.g. power, paradigms and goals) (Amorocho-Daza et al., 2025).

As a complementary analytical feature, we follow Abson et al.'s (2016) terminology around Meadow's leverage points. Abson et al. (2016) revised Meadows' work and proposed clustering the twelve original points into four system characteristics, organised from 'shallow' to 'deep', as follows: *parameters*, *feedbacks*, *design* and *intent* (see Figure 5.3). In the context of this research, we propose that a 'deeper' change would mean that the model fundamentally changed after incorporating stakeholder feedback — in the dimension of *design* and *intent*; while a "shallower" change means that the stakeholders' observations lead to relatively minor changes—in the dimension of *parameters* and *feedbacks*. In sum, the model's depth of change can be classified according to the scale proposed by Meadows (1999, 2008), but the clustering of Abson et al. (2016) offers a useful complementary terminology.

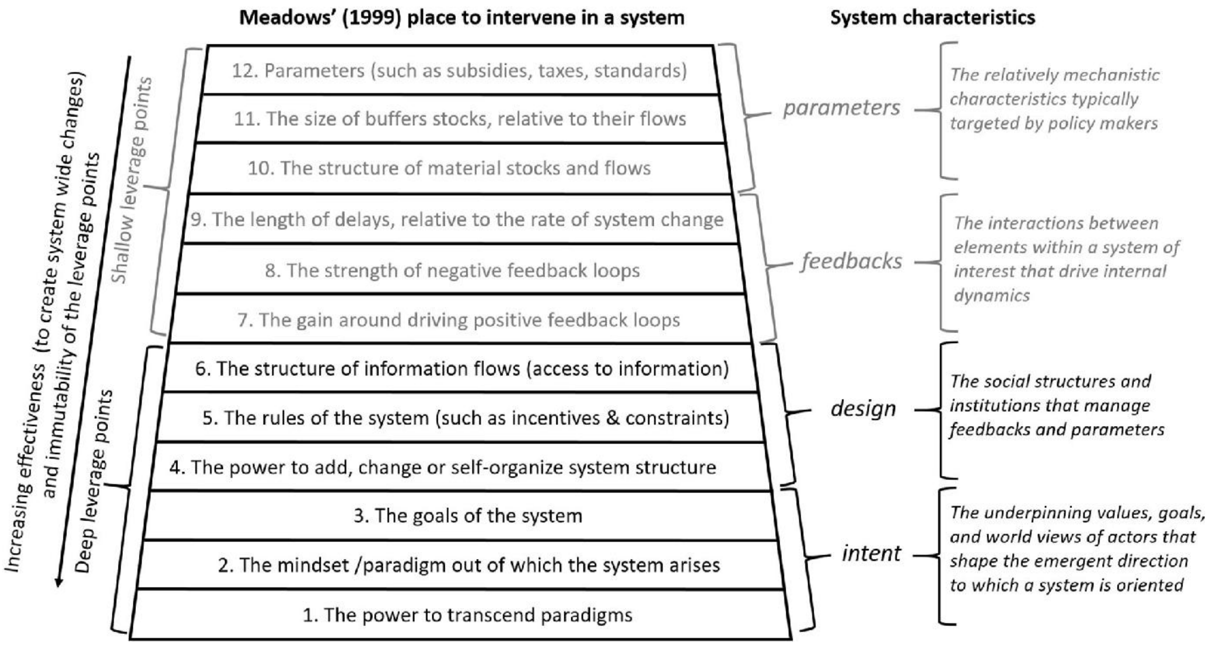

Figure 5.3 Summary of Meadows' leverage points as clustered and interpreted by Abson et al. (2016)

We propose a protocol to categorise model changes using Meadows' leverage points. The first two steps of the protocol concern in describing model changes over time in connection with stakeholder feedback (Figure 5.4). Having identified the changes, the following task is to categorise them according to the Meadows leverage points, following Figure 5.4's third and fourth steps. To do so, we propose asking a guiding question in a structured way: *Is this change consistent with leverage point Number XX proposed by Meadows (1999, 2008)?* This question can be asked starting from shallow to deep leverage points, iterating over the leverage points until a positive answer is found. To validate consistency, the analysis can be repeated from deep to shallow leverage points. If the same leverage point is identified after performing the two rounds of analysis (i.e. shallow-to-deep, and deep-to -shallow), the leverage point is selected, and a justification can be added for transparency. If there is a discrepancy in the leverage points, a second analyst can be invited to deliberate about the inconsistency. If agreement is reached, the justification is added; if not, the model change might not fit in any of the leverage points for multiple reasons, so no leverage point is identified, and a justification is documented.

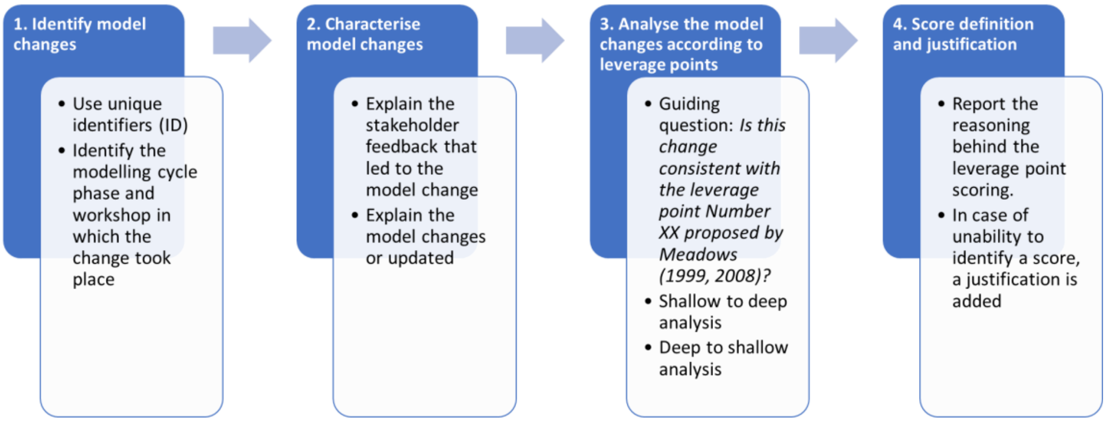

Figure 5.4 Protocol to categorise model changes using Meadows (1999, 2008) leverage points

5.2.2. Assessing knowledge generation

Similar to Meadow's framework of leverage points, the concept of knowledge generation is also concerned with the purpose of achieving systemic change. Brand et al 2013, proposes that systemic change requires three elements: understanding of the current situation (system), envisioning future desired states (target), and pathways connecting present to future (transformational). Knowing the degree to which knowledge is being generated in a resource nexus project was thus considered crucial. Assessing knowledge generation was operationalised in a survey that assessed perception in the three types of knowledge (system, target, and transformational, as proposed by Brand et al 2013) and in five 'types of analysis' (that is, broad knowledge domains of high relevance to the project needs, as proposed by Avellán et al 2022) (Figure 5.5 and Appendix B, Figure 1). Full documentation on the survey features and design as part of a wider stakeholder engagement strategy from which the information in this article is derived, is documented in Avellan et al. (2025).

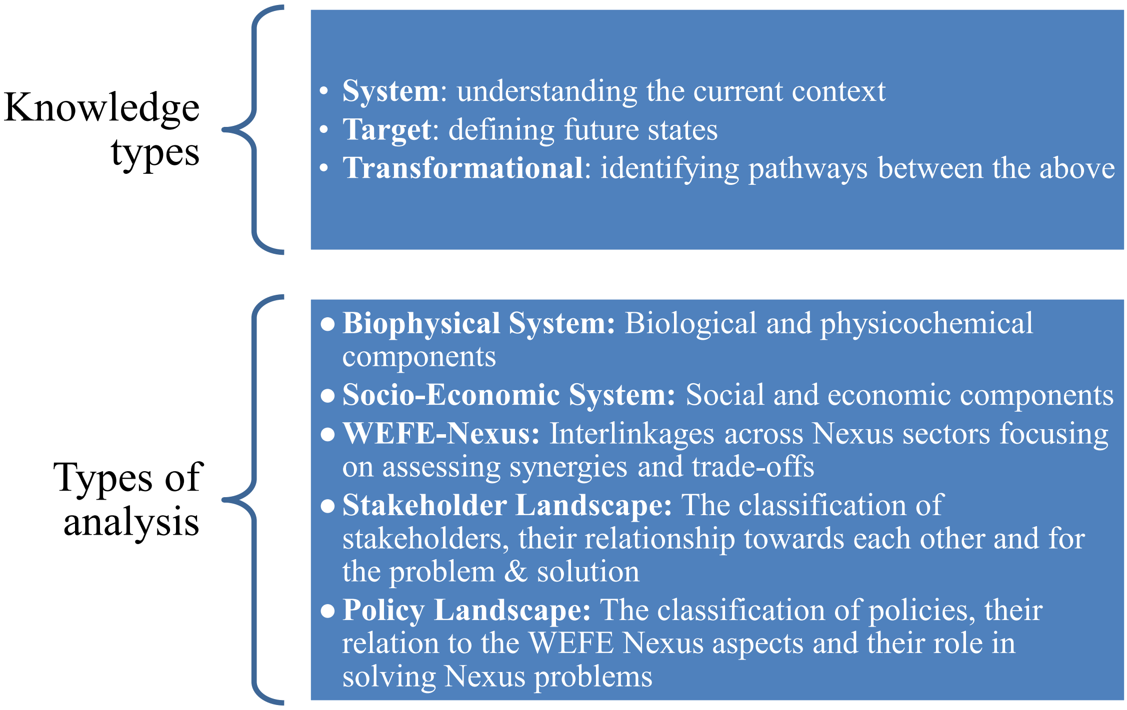

Figure 5.5 Knowledge generation operationalisation as two interactive factors: knowledge types (as proposed by Brandt et al 2013) and types of analysis (as proposed by Avellán et al 2025).

The knowledge generation framework (Figure 5.5) drove both the quantitative and qualitative assessments of perceived knowledge generation by the modeller and the stakeholders across the evolution of the project.

Quantitative assessment

At workshops 3, 4, 5, and 6 (Appendix B, Table 1), participants were asked to rate their perceived knowledge generation at the respective workshop they had just attended, based on a 7-point Likert scale, as per (a) the level of knowledge generated (system, target or transformational) and (b) the type of analysis (biophysical system, socio-economic system, WEFE Nexus, stakeholder landscape, or policy landscape). The survey is available in Appendix B, Figure 1. Survey questions were administered in local languages (Latvian, Lithuanian, and English). Data was downloaded from the online platform, translated to English, if needed, and manipulated further in Excel.

The survey was provided to the modeller (also the first author of this paper) to self-assess his perceived degree of knowledge in the respective levels and types of analyses. In that sense, this assessment is slightly different from the ones conducted by the stakeholders, as the modeller is *not* assessing his perceived knowledge acquired at the respective workshop but rather his level of knowledge at that point in time. Knowledge might have been generated through other means than the workshop, which, however, is also true to stakeholders to a certain extent.

Qualitative assessment

A 90-minute semi-structured interview was conducted between this article's second author (in charge of the stakeholder engagement process in NXG) and the first author (the modeller) in August 2025 to understand more deeply the modeller's perception of knowledge generation. The interview used the responses given by the modeller to the quantitative survey and discussed the modeller's perceived changes in knowledge. The main lines of questioning were:

1. The temporal evolution of the modelers perceived knowledge gains, with a particular emphasis on the role of the workshop (activities).
2. The changes of perceived knowledge gains in the different knowledge types and the challenges and opportunities in acquiring further knowledge in each.
3. The perceived difference in knowledge gains across the types of analysis and the role interactions with stakeholders and the CS team had.
4. The role of context, i.e. background of the modeller, previous contact with the sites/culture/country, team composition also with respect to gender and cultural background, etc., in acquiring or biasing knowledge gains.

The interview intended to better understand the perceived reasons for changes in knowledge and the potential avenues of knowledge generation of the modeler throughout the project. This was deemed critical as the modeler is the prime architect of the model structure which was used in the NEPAT and which SHs in turn used to explore the effects of potential policy decisions. The way the modeler therefore took up, interacted, felt and reacted to the various interactions fundamentally influenced model design, and in turn the potential impact of the project.

The transcript of the interview was used as the basis for the assessment. ChatGPT 4 was used to distil these perceptions from the transcript. The following prompt was used: *"Please provide a summary of the text that I will provide to you now highlighting the ways in which the modeller looks at system, target and transformation knowledge (a) in each of the two phases assessed (i.e. Phase I and Phase III), (b) for each of the five dimensions of knowledge created (biophysical system, socio-economic system, WEFE nexus, stakeholder landscape, and policy landscape) and (c) how it contrasts to the perception of the knowledge generated by the stakeholders in each of the phases, each of the generated knowledge areas, and each of the dimensions of knowledge. Single out quotes of the modeller that fit the items above*". The prompt output was revised and edited by the co-authors to improve precision and clarity, highlighting insights and quotes concerning knowledge generation and model evolution.

5.2.3. Contrasting the model evolution with knowledge generation

Here we conceptualise a dynamic relationship between the model's evolution and the knowledge generation across a PM cycle. Jakeman et al. (2024) suggest that early scoping and conceptualisation activities (Modelling foundations - Phase I) strongly rely on stakeholder input, followed by a transition to model-supported activities (Model building and testing – Phase II), and later to a more interactive transition in which stakeholders use the model (Model use and policy evaluation – Phase III). Based on those generic stakeholder-model dynamical interactions, we analyse the relationship between knowledge level and model evolution in each of the phases.

5.3. RESULTS

5.3.1. Model evolution

Table 5.2 is built based on an operationalisation of the protocol proposed above (see Figure 5.4) to assess model evolution in the participatory modelling case study of the Lielupe River Basin. The leverage point scores were determined by the CS modeller (lead author) and revised by the stakeholder engagement lead (2nd author). Stakeholders provided their feedback across multiple workshops, mostly in the form of plenary discussion and Q&A sessions (see Facilitation approach - Appendix B, Table 1) leading to model changes or updates developed by the modeller after the participatory sessions. The content of Table 5.2 was performed as an ex-post analysis, that is, the analysis took place after the workshops and subsequent model development.

A bird's eye view of the table shows that stakeholder input shaped the modelling process distinctively at different phases: from setting objectives and rules (Phase I), through testing and refining a first quantitative model (Phase II), to applying it for policy evaluation via a decision support tool (i.e. NEPAT) (Phase III). Early stages involved 'deep' changes to goals and rules, while later stages focused on refining parameters and making local minor adjustments (shallower changes). By Phase III, convergence was reached, with stakeholders validating the model's usefulness for exploring basin policies. Meadows' leverage points were used to classify the nature of the model changes, and color-coded darker to lighter corresponding to deeper to shallower leverage points. The last column indicates the state of the model in each modelling phase, as described by Amorocho-Daza et al. (2026), allowing tracking of the model's evolution across the PM process.

During Phase I, stakeholders provided feedback in two workshops (WS) (2 and 3). The feedback guided the model's purpose or intent by establishing clear model objectives (i.e., goals), even at a qualitative modelling stage. Some model design features (i.e., rules) started to emerge by linking activities to consequences (e.g., agricultural activities to water pollution), opening the way to later test policy options in the modelling cycle. WS 3 offered opportunities to define more operational model features. Overall, Phase I provided a 'deep' foundation for the rest of the modelling exercise.

In Phase II, a first quantitative version of the model was developed, and stakeholders provided their feedback during WS 4. After stakeholders checked preliminary results from the simulation model, some critical assumptions (e.g. a single river basin approach was a desirable approach) and limitations (e.g. the model's lack of capability to account for transboundary interactions) became evident. Stakeholder feedback resulted in another round of 'deep' model changes (i.e., goals and rules) to improve the model's capabilities (i.e., simulate transboundary interactions). They provided local information resulting in refining some model's parameters. Overall, this phase provided stakeholders with a tangible opportunity to test the model capabilities and ask for improvements, both ‘deep’ and ‘shallow’.

In Phase III, the model was updated and integrated into a web-based decision support tool—the NEPAT. In WS5, stakeholders had the opportunity to track key variables across baseline scenarios and test the expected outcomes of implementing long-term river basin policies. Based on the feedback that came from this exploration, the model was slightly refined to represent local conditions more accurately. However, no major or ‘deep’ changes were requested at this stage. This is an indicator of convergence in the modelling process.

Table 5.2 Summary of model evolution across the phases of the modelling cycle per workshop (WS). Operationalisation of the protocol to categorise model changes using Meadows' leverage points in the Lieupe River Basin participatory modelling experience. Blue-shaded numbers correspond to the hierarchy of leverage points proposed by Meadows, with darker values representing deeper leverage points.

Phase	WS	ID	Stakeholder input/feedback	Changes or updates to the model	Classification of changes using Meadows' system leverage points (1999, 2008)		Justification of the score	State of the model
Phase I	WS2	1	The most critical water parameter is nitrogen concentration (in terms of nitrates).	Modelling the nitrogen as a dynamic parameter that responds to WEFE Nexus policies	Goals	3	The main goal of the model is defined as assessing water quality responses to policy interventions	Conceptual map
		2	Agricultural nitrogen pollution drives total nitrogen pollution in the river	Prioritise a more detailed modelling of agricultural pollution over other sources, such as domestic wastewater	Rules	5	By rule, water quality is driven by agricultural activities	
	WS3	3	There is interest in bird conservation in the basin with ambitious targets	Tracking bird biodiversity and modelling it in connection with the basin's vegetation coverage	Goals	3	A secondary goal of the model is defined as assessing biodiversity responses to policy interventions	Stock and flow Diagram (in development)
		4	Nature-based solutions such as riparian buffers and constructed wetlands are gaining interest	Incorporating these NBS alternatives into the model	Rules	5	By rule, water quality can be improved via NBS	

Phase	WS	ID	Stakeholder input/feedback	Changes or updates to the model	Classification of changes using Meadows' system leverage points (1999, 2008)		Justification of the score	State of the model
			in the government's environmental policy					
		5	Large-scale transformation of land is slow	Incorporating long-term strategic objectives for land-use change to calculate the flow rates that affect the land stocks	Rules	5	By rule, water quality and biodiversity can only be impacted via long-term land-use changes	
		6	Land-use change drives river basin objectives	Model the basin's land uses using various interdependent stocks	Stock-and - flow structures	10	Land-use types modelled as stocks and transitions among them as flows	
		7	There is considerable uncertainty regarding the efficiency of NBS in improving water quality.	Modelling the efficiency of the NBS as a stochastic function based on pertinent academic literature meta-analysis	Parameters	12	A feasible parameter range is assumed and defined (for NBS removal efficiency)	
Phase II	WS4	8	Modelling the basin as a whole does not allow for establishing the effect of the inaction of the upstream country	Decoupling upstream and downstream countries to account for transboundary interactions	Goals	3	A new goal of the model is defined (as assessing water quality upstream-downstream interactions)	Stock and flow Diagram (completed)

Phase	WS	ID	Stakeholder input/feedback	Changes or updates to the model	Classification of changes using Meadows' system leverage points (1999, 2008)		Justification of the score	State of the model
		9	Latvian stakeholders already receive nutrient-rich water	Tracking each riparian country's contribution to the nitrogen concentration of the river (at the river mouth).	Rules	5	By rule, a transboundary setting implies that the activities of each riparian country are cumulative in terms of nutrients pollution	
		10	Lithuanian stakeholders have fewer incentives to have a better water quality than Latvian (which also has the delta)	Decoupling upstream and downstream policies, to be able to test non-cooperative and cooperative scenarios	Rules	5	By rule, countries do not necessarily cooperate, so flexibility is needed to test multiple scenarios	
		11	Forest land doesn't change much because it is protected	Model Forest land is a stock that does not change over time	Stock-and - flow structures	10	An static stock is assumed and defined (Forests)	
		12	Providing estimates for design parameters for constructed wetlands (area to catchment area) and riparian buffers (width)	Incorporating these parameters into the model	Parameters	12	A feasible parameter range is assumed and defined (for NBS design parameters)	

Phase	WS	ID	Stakeholder input/feedback	Changes or updates to the model	Classification of changes using Meadows' system leverage points (1999, 2008)		Justification of the score	State of the model
Phase III	WS5	13	The water quality in the river responds very slowly to policy interventions in the basin	Modelling system delays by considering a stock of nutrients in transit	Delays	9	Water quality responds slowly to management intervention and is modelled as a first-order delay structure	Web-based decision support tool (SFD in the background)
		14	Implementing nature-based solutions occupy space that needs to be accounted for (often a trade-off for arable land)	Modelling NBS as stocks connected to arable land	Stock-and - flow structures	10	A new stock is assumed and defined (Area of NBS)	
		15	Wetlands can provide benefits for biodiversity	Modelling the carbon mass in vegetation of wetlands and riparian buffers	Parameters	12	A feasible parameter range is assumed and defined (for carbon mass in vegetation of NBS)	

5.3.2. Knowledge generation

Quantitative assessment

Table 5.3 shows the perceived knowledge generation scores across as measured from WSs 3, 5, and 6 for the stakeholders and around WSs 3 and 5/6 for the modeller. Responses from WS4 are not considered representative (4 respondents), as they only cover 28% of workshop participants. Knowledge scores are presented in a matrix comparing two factors: Type of analysis (columns) and type of knowledge (rows). Overall estimates are calculated as the average value over rows and columns. Respondents are not necessarily consistent over the workshops. Consistency is difficult to validate as many stakeholders choose to remain anonymous at the time of filling the survey. Yet the proposed assessment reveals the average knowledge generation of a particular group of stakeholders representing the organisations described in Appendix B, Table 1 during the workshops 3, 5 and 6.

Table 5.3 Average knowledge survey scores classified by multiple factors (type of analysis, knowledge type, and participant type) and compared across different workshops (WS). Survey scores are colour coded (from red to green according to incremental knowledge generation).

		Types of analysis					
Participants	**Type of knowledge**	Biophysical system	Socio-economic system	WEFE Nexus	Stakeholder landscape	Policy landscape	Overall
Stakeholders (WS3)	System knowledge	3.1	3.6	5.0	5.3	4.6	4.3
	Target knowledge	3.3	3.8	4.8	5.2	5.3	4.5
	Transformation knowledge	3.7	4.3	4.9	4.9	4.4	4.4
	Overall	3.4	3.9	4.9	5.1	4.8	4.4
Stakeholders (WS5)	System knowledge	5.4	3.5	4.9	4.6	5.9	4.9
	Target knowledge	5.4	3.7	5.1	5.4	6.0	5.1
	Transformation knowledge	5.5	3.7	5.1	4.9	5.6	5.0
	Overall	5.4	3.6	5.0	5.0	5.8	5.0
Stakeholders (WS6)	System knowledge	4.3	4.1	5.4	5.5	5.1	4.9
	Target knowledge	4.8	4.6	5.6	5.6	5.3	5.2
	Transformation knowledge	5.3	4.4	5.4	5.3	5.0	5.1
	Overall	4.8	4.4	5.5	5.5	5.2	5.1
Modeler (WS3)	System knowledge	4.0	3.0	5.0	5.0	5.0	4.4
	Target knowledge	6.0	5.0	6.0	5.0	6.0	5.6
	Transformation knowledge	6.0	3.0	5.0	4.0	5.0	4.6
	Overall	5.3	3.7	5.3	4.7	5.3	4.9
Modeler (WS5/6)	System knowledge	6.0	5.0	7.0	5.0	6.0	5.8
	Target knowledge	7.0	6.0	7.0	6.0	6.0	6.4
	Transformation knowledge	6.0	5.0	6.0	5.0	5.0	5.4
	Overall	6.3	5.3	6.7	5.3	5.7	5.9

An overview of the results brings various insights. Comparing knowledge scores from participants shows that stakeholders' scores are, in most cases, lower than the modellers' scores over the workshops and across either dimension. Workshop 3 exhibits the lowest scores for both modeller and stakeholders, while WS 5 and 6 present higher yet similar outcomes, an indication of higher knowledge generation happening at final stages of the participatory modelling process. When analysing the variation of scores over the rows and columns, it is evident that knowledge scores strongly vary across types of analysis, but are rather homogeneous across types of knowledge.

Stakeholders in WS5 self-reported high scores in the biophysical and policy landscape dimensions compared to WS3. This latter workshop included activities of model structural and behavioural explanation (aiming to address transformation knowledge) and using the NEPAT to explore trade-offs and synergies across policies (aiming to address target knowledge; see Appendix B, Table 1 and Table 4). For the biophysical dimension, scores did not vary across types of knowledge whereas in the policy landscape, the knowledge scores were slightly higher in the system and target knowledge types. For the rest of the types of analysis, WS5 scores were almost the same compared to WS3. This responds to the fact that the discussion centred on using the model to test transboundary policies, cross-sectoral WEFE Nexus and stakeholder landscape dimensions did not take part in the discussion. A lower knowledge score in the socio-economic dimension may have come from stakeholders discussing the model limitations about the costs of implementation.

At WS6, the final project's workshop, stakeholders self-reported knowledge gains across every type of analysis compared to WS3. It is worth mentioning that in contrast to WS5, the role of modelling was limited in WS6. Its focus was towards formulating a policy roadmap based on the policies identified and discussed in WS5 and moving towards reaching stakeholder agreements for continued cooperation beyond the project. These activities had a strong focus on transformation knowledge (Appendix B, Table 4). A detailed analysis shows marked knowledge gains in the biophysical and nexus types of analysis. For the biophysical analyses, knowledge was more marked in the transformation and target knowledge, whereas in the WEFE Nexus the knowledge gains were stronger in the target and transformation knowledge types. Minor gains are evidenced in the socio-economic, stakeholder and policy dimensions. Yet, target knowledge was relatively important in socio-economic and stakeholder knowledge. Results provide evidence of knowledge generation spillovers, as WS6 brought broad knowledge generation both in terms of types of analysis and knowledge types, despite its initial focus being intended to address mostly stakeholder and policy landscapes with a transformational focus.

According to the modeller's self-reported learning, his higher knowledge gains are related to the socio-economic and the WEFE Nexus dimensions. For the socio-economic dimension, gains are high because his starting knowledge was lower compared to the stakeholders'. Regarding the WEFE Nexus, knowledge generation was high due to the model building process and from the depth of discussion across the workshops, both framed in a WEFE Nexus perspective. Biophysical and policy knowledge gains are minor because the baseline was already high by the time of WS3. This evidences a possible lag between the modeller and the stakeholders' learning: while the modeller perceives a high level of knowledge in WS3, this knowledge is only 'shared' with the stakeholders via the model development and workshop activities developed later in time (i.e. WS5). Finally, stakeholder landscape knowledge was limited, perhaps because the modeller was not involved in WS6 planning—a workshop that brought learning for stakeholders in this dimension.

Qualitative assessment

The interview analysis shows that the modeller's experience demonstrates the interplay between knowledge acquisition, participatory engagement, and modelling outcomes in a complex transboundary river basin. Table 5.4 summarises key insights and quotes from the interview with the modeller, classified into two broad themes: Knowledge types and modelling choices. For the first theme, the modeller shows how interactions with stakeholders (or lack thereof) heavily influence his own knowledge in the socio-economic, stakeholder landscape and policy landscape dimensions. According to the interviewee, high learning outcomes in the biophysical and WEFE Nexus dimensions are better explained from an integral process involving not only stakeholder input but also engagement with academic literature and developing modelling tools. For the second theme, the modeller opens the thinking process that resulted in establishing some key model features. It is worth mentioning that the discussed modelling choices emerged from social interactions with stakeholders and other colleagues rather than from individual reflections of the modeller.

Table 5.4 Summary of the interview with the modeller

Theme	Dimension /features	Interview insights	Quote
Types of analysis	Biophysical	The modeller reported significant learning in the biophysical system, particularly regarding nutrient transport and water quality dynamics. This knowledge emerged from a combination of academic literature, stakeholder discussions, and professional dialogues, and was incorporated into the model.	"The process of making sense and modelling nutrient transport, particularly its long delays, offered me a huge learning opportunity. And that learning was incorporated in the model so that it explicitly accounts for these delays in nutrient transport."
	Socio-economic	Socio-economic system knowledge was more limited. Both the modeller and stakeholders perceived lower learning in this dimension due to the absence of local socio-economic expertise and the abstract nature of global socio-economic variables.	"We didn't have these kind of [socio-economic] discussions in the room, simply because the people that were there didn't have this knowledge. In the NEXOGENESIS group, we have the economics knowledge, but from a very abstract and high-level perspective that didn't connect with the local systems"

Theme	Dimension /features	Interview insights	Quote
	WEFE Nexus	Nexus learning was integral to the whole PM process from the modeller's perspective, as it connected interaction with stakeholders, academic literature and developing modelling tools.	"I was able to go to the field (…), to talk with people, read a lot of academic literature, and then try to make sense of the situation from a nexus perspective using modelling tools. That's why I reported a maximum learning [in the WEFE Nexus dimension]…I'm wondering if someone can invest that kind of time… [I think] very few can."
	Stakeholder landscape	Knowledge of stakeholders and institutions was essential for contextualizing the model. The modeller emphasized that understanding actors, their priorities, and transboundary dynamics informed both model logic and scenario development.	"It was really important to know the actors and the institutions that are behind an issue… I learned a lot through their thinking, through their explanations, through their viewing, I learned a lot about the river basin itself, but more specifically about the [local] issues."
	Policy landscape	The modeller gained understanding of potential interventions and policy measures, particularly those highlighted by stakeholders. While some of these interventions were already aligned with best practices, stakeholder discussions refined and contextualised them for integration into the model.	"Workshop 3 was kind of a key moment… It was an excellent brainstorming about policies and that triggered a lot of discussions that connected to how to make these policies in the field…It provided a very rich picture of the limitations and complexities of these policies…Then I did the reflections to incorporate them into the model."
Modelling choices	Identify policy levers	The modeller operationalised socio-economic factors as policy levers for experimentation, e.g. using land use change in lieu of direct evidence of GDP change due to agricultural shifts, rather than attempting to model underlying processes endogenously.	"These large-scale [land use] changes are dominated by socio-economic processes...and of course, we didn't have this knowledge. It lacked from the discussion. So what I did was, ok, let's model those land [use] changes as a place of experimentation. For instance, to test what will happen if I change [arable

Theme	Dimension /features	Interview insights	Quote
			land] by 1% compared to 90%. So instead of trying to understand the systems that dominate that change, what I did was to take the model as a place of experimentation, and that was a very personal choice.”
	Aligning model structure and policy goals	Collaboration with colleagues in Latvia, the case study leads and co-authors here, helped translate these discussions into the model structure and align them with real world policy goals.	“I told them [my colleagues], ok, I do have this small structure...this is more or less what the model can do as it is conceptualised. And what they did was to propose the policy goals, also aligning them with that model structure...so they helped me a lot.”
	Adding new modelling features	The modeller translated stakeholder concerns into actionable modelling scenarios, such as considering the consequences of non-cooperation between upstream and downstream countries.	“When they said we would like to see what if-scenarios if Lithuania doesn’t cooperate?… for me, that was not wishful thinking, but [a statement] very rooted in issues that they face. And then I thought, I can do something with this model to represent that.”

5.3.3. Synthesising model evolution and generated knowledge

Table 5.5 presents our approach to contrast participants’ knowledge level and the model evolution, by analysing how knowledge level and model evolution occur across the three global phases (Amorocho-Daza et al. 2025). Jakeman et al. (2024) posit that a modelling cycle is supported by stakeholder engagement processes and modelling tools at different degrees over the cycle’s phases, these are the ‘supporting dimensions’ presented in the table. Two types of analysis are considered to illustrate knowledge generation: biophysical and policy landscape, which showed the strongest knowledge gains between WS3 and WS5 (see Table 5.3). To track knowledge evolution, we use the metric of counting how many changes happened during the phases classified as either shallow or deep.

Phase I is stakeholder-driven; the modeller's knowledge gain is high from getting acquainted with a new problem situation. Stakeholders, on the contrary, are already experts on the matter, so they deliberate, but their knowledge gain is relatively low. Model changes during this phase are ‘deep’, laying model foundations and capabilities for the rest of the cycle. In Phase II, analysis is limited as there is no knowledge assessment. Therefore, it is not possible to assess the potential role of model validation in participants' learning. Here, model evolution exhibits both deep and shallow changes, yet in lower quantity than in the previous phase, a signal of convergence to a final model. Phase III is supported by both stakeholders and the model. The modeller's learning is relatively low as the model is (soon to be) finalised. Stakeholders by contrast have new learning opportunities from interacting with the model and debating its results. Regarding model evolution, only three shallow changes are registered, as the most critical stakeholder feedback has already been considered.

Table 5.5 Tracking participants' knowledge level and model evolution across the PM phases for the Lielupe case study

Phase	Supporting dimensions	Participant's knowledge generation	Model evolution
I	Stakeholder (WS1-WS3)	Biophysical: ● Stakeholders (3.4) ● Modeller (5.3) Policy landscape: ● Stakeholders (4.8) ● Modeller (5.3)	● 5 deep changes ● 2 shallow changes
II	Model (WS4)	Not available	● 3 deep changes ● 2 shallow changes
III	Mixed (WS5)	Biophysical: ● Stakeholders (5.4) ● Modeller (6.3) Policy landscape: ● Stakeholders (5.8) ● Modeller (5.7)	● 3 shallow changes

5.4. DISCUSSION

5.4.1. Model evolution

We apply a method for assessing model evolution using Meadows' leverage points. This is a novel contribution to the systems dynamics modelling field, as it offers a structured approach that not only opens the model structure to critical analysis (see Capelo et al., 2024; Kopainsky et al., 2024), but also the process by which a model evolves in response to stakeholder feedback in a PM setting. This structured approach moves towards more reflective and transparent modelling practice and shifts the emphasis from the 'final' model to the modelling process (Forrester, 1985), highlighting the cumulative nature of changes that lead to a final product (Amorocho et al., 2024). Our approach highlights the deep sources of change in modelling (e.g., goals and rules), in contrast to the often-enthusiastic focus on "shallow" dimensions (e.g., parameters and feedback structures) in quantitative modelling.

Results illustrate the model's evolution in response to a participatory process. These findings complement the modelling outputs for the Lielupe case study (Amorocho-Daza et al. 2026), which explains the modelling features in detail, pointing out that the modelling cycle started from a qualitative and systems generic perspective in the early stages, to become increasingly quantitative and problem-oriented—focusing on quantifying the cascading impacts of land-use change across nexus sectors. This paper shows that the insights from stakeholder inputs during Phase I of the modelling cycle were mostly 'deep' and therefore essential to scope the issue and define the modelling features. Amorocho-Daza et al. (2026) mention that specific modelling features (such as modelling transboundary water quality interactions) emerged from stakeholder requests during Phase II of the modelling cycle. In contrast, changes across Phase III, despite requiring significant modelling efforts (as presented in the quantitative model description of Amorocho-Daza et al. (2026)), did not bring any structural change to the model and remained in the 'shallow' dimension of change.

Considering Meadows' leverage points in participatory modelling opens exciting research avenues. Further empirical research can build on our methods to structurally explore model evolution across a broad set of participatory modelling settings. Such studies can investigate whether our observations of a converging deep-to-shallow model evolution pattern are present in other PM interventions. Another avenue of research lies in incorporating the leverage points as part of modelling workshops in the form of 'scripts', a widely recognised practice in participatory System Dynamics (Andersen and Richardson, 1997; Hovmand et al., 2012; Luna-Reyes et al., 2006).

5.4.2. Learning

In this article, learning was considered an emerging outcome of the PM cycle and stakeholder engagement process. As a growing field, research on learning in PM has focused on analysing the stakeholders' interactions that occur in workshop settings; more specifically, studying the role of model use (i.e. ABM and GIS) in shaping stakeholders' insights and discussions (Hoch et al., 2015; Radinsky et al., 2017; McEvoy, 2019). Here, we take a different modelling and methodological approach—System Dynamics, by proposing a proxy of learning via assessing knowledge generation across the phases of engagement. Our approach offers a knowledge assessment generation that is suitable to track both stakeholders' and modellers' learning across a PM intervention. This research is therefore a contribution to a gap in the PM literature on the 'learning' dimensions as identified by Jordan et al. (2019).

This research approaches the question of learning using the concept of knowledge generation which emerged from the sustainability science literature, yet there are other valid entry points for this task. Systems based research offers alternative conceptualisations around learning. Richardson et al. (1994) and Sterman (1994), for instance, consider the concept of mental models as central to learning, because change in behaviour ultimately relies on changes in the way we think about a problem. According to Richardson et al. (1994), mental models can be of three types: means, ends, and ends-to-means models. In line with this perspective, Rouwette (2016) proposes that learning in facilitated SD settings relates to the capability of having new insights about a problem of interest, a condition for deploying real-life system changes. Future research about learning in the field of sustainability can further engage with the concept of 'mental models' (as proposed by Jones et al 2011). Likewise, systems-based research can explore how the knowledge types that are broadly relevant to the field of sustainability science, do or do not resemble the foundational types of mental models identified by Richardson et al. (1994).

Stakeholder learning

Based on our assessment of stakeholder learning, results indicate that knowledge gains can be highly dependent on the agenda of a workshop yet not fully responsive to it. For instance, WS5 illustrates that knowledge generation was clustered into specific 'types of analysis', or knowledge domains, based on the workshop activities. Interestingly, results from WS6 evidence a more widespread knowledge generation, often beyond the knowledge type intended to be addressed. These findings come from the inherently social nature of a workshop setting. Despite careful planning invested in the design of the workshops, stakeholders respond and learn from them in unexpected ways (Hoch et al., 2015, Radinsky et al., 2017, Zellner et al., 2012, McEvoy, 2019). WS6 illustrates that knowledge generation 'spillovers' can be expected to happen beyond the modelling cycle. This indicates a need for further research to explore which factors, methods and tools (PM or SH engagement activities, or others) explain the knowledge generation in model-based participatory environmental projects (McEvoy, 2019). However, a remaining challenge is to find reliable yet practical ways of measuring learning in PM.

In this study, we opted for self-assessment as a way of assessing learning among participants in a PM intervention. This is the most common practice in measuring the outcomes of participatory SD, particularly learning (Rouwette al. 2002, Scott et al 2016, Rouwette 2016). Yet, empirical research in psychology points out that people have a limited ability to report the extent and causes of their own learning (Nisbett and Wilson, 1977; Rouwette, 2016). Other related research suggests the existence of dissonance between experienced and observed cognitive change in participatory modelling settings (De Gooyert et al. 2022). Based on these limitations, an alternative way to evaluate the effectiveness of participatory modelling to promote learning (e.g. cognitive change) is a pre-test post-test design that evaluates both experienced and observed cognitive learning (de Gooyert et al., 2022, Rouwette, 2016, McEvoy, 2019). However, doing this in practice is a challenging endeavour in projects that operate across long-term multi-stakeholder platforms.

While pre-test surveys and more comprehensive evaluation tools were planned and implemented at the start of NEXOGENESIS, the initial survey did not return a meaningful number of responses from the stakeholders as they considered its design too challenging and cumbersome. This situation called for a redesign of the survey, which was later implemented from WS3 onwards. In the end, the learning assessment reported in this article is considered a workable approach for the researchers and the stakeholders. It may therefore be a good starting point to structure future evaluations in resource management projects using PM approaches. Additionally, our mixed method approach illustrates a way forward in widening the vision of learning in PM, by including qualitative approaches as a complement to quantitative assessments. These may include but are not limited to interviews, focus groups and story-telling in reporting on both the individual and collective learning outcomes of PM participants (See Avellán et al., 2025).

Modeller learning

Contrasting stakeholder and modeller learning across the PM process brings important insights. For some of the analytical categories (e.g. biophysical, WEFE, and policy landscape), the knowledge of the modeller is relatively higher than the average reported from the stakeholders. This might reflect the scoping process of the modeller in which the interaction with stakeholders leads to boundary setting and inspires potential model development and improvements. During workshops, stakeholders share their contextual and rich questions and priorities about a wicked environmental problem. Sharing such concerns and uncertainties might be considered from the stakeholders' perspective as a lack of knowledge, however from the modeller perspective, such inputs are essential to lay the foundations on which a model can be built. This is particularly evident with the knowledge types of transformation and target, which emerge in early stakeholder discussions and from which a modeller can anticipate future model capabilities.

Regarding the interview with the modeller, this showed that the participatory modelling process enhanced understanding of both the biophysical and socio-economic dimensions of the river basin system, as well as the interconnections captured through the WEFE Nexus lens. While initially facing limitations due to sparse prior knowledge of local context and dynamics, he leveraged workshop discussions, field visits, and literature to gradually build a comprehensive understanding. The process of integrating policy instruments into the model, guided by colleagues familiar with local policy goals, allowed him to operationalise socio-economic levers despite incomplete knowledge of the underlying economic processes. Simultaneously, stakeholder interactions provided essential context, highlighted transboundary dynamics, and enabled validation and refinement of model assumptions, enhancing both the usability and credibility of the outputs.

Our experience also illustrates the challenge of achieving interdisciplinary insights among the researchers that, directly and indirectly, contribute to a PM project. Regarding the modeller learning, we observe that some types of analysis (e.g. stakeholder and policy landscape) remained rather limited, highlighting the need to have broader spaces of collaboration among modellers and other researchers—such as social scientists—dealing with the stakeholder and governance dimensions in an interdisciplinary research consortium. This requires going beyond achieving specific project tasks to fostering spaces for inter- and transdisciplinary learning which leverage knowledge generation within the PM intervention and beyond. The limited socio-economic features of the model, imply that existing siloed knowledge (multidisciplinarity) did not transcend to the research group (interdisciplinarity) or the wider stakeholder group (transdisciplinarity)—an experience that resonates with previously reported research (Podestá et al. 2013).

5.4.3. Bias in modelling and stakeholder representation

Bias remains a critical issue in participatory modelling and cannot be assumed to be resolved through stakeholder engagement alone. The literature consistently shows that both modellers and stakeholders shape how systems are defined, which variables are included, and how causal relationships are represented (Amorocho Daza et al., 2025, Enserink et al., 2022, Rouwette et al., 2011, Slinger, 2023). Modellers, in particular, are not neutral; their methodological preferences, disciplinary backgrounds, and prior experience influence model design and process structure (Voinov et al., 2018). Participatory settings involving multiple stakeholders may improve knowledge integration but can also introduce new biases if not managed reflexively, affecting the credibility of the resulting models.

In this study, we sought to make the modeller's positionality explicit. Through a targeted interview, we identified limitations in contextual knowledge of the case study, thereby surfacing a potential source of bias that often remains implicit in modelling practice. This aligns with calls in the literature for greater reflexivity in participatory modelling, where the assumptions and knowledge base of the modellers themselves are critically examined (Voinov et al., 2018, Amorocho-Daza et al., 2024, Ter Horst, 2025). While this does not remove bias, it enhances transparency and allows for a more informed interpretation of model structures and outputs.

Another issue concerns stakeholder representation. The composition of participants is known to strongly influence participatory outcomes, with inclusiveness and diversity of perspectives being critical design features (Newig et al., 2018). In our case, not all WEFE nexus sectors were consistently represented across workshops, with the food sector in particular being underrepresented. This imbalance has implications for both knowledge generation and model development, as certain sectoral perspectives, trade-offs, and priorities may be less visible in the co-produced model. As a result, the apparent integration of knowledge across the nexus should be interpreted with caution, as it may reflect the perspectives of more strongly represented sectors.

Taken together, these findings reinforce that participatory modelling is inherently shaped by both who participates and who models. While the PM process enabled substantial knowledge co-production and mutual learning, it also remained subject to structural constraints related to expertise and representation. Rather than eliminating bias, the overall approach developed here contributes to enhancing transparency. In other words, it helps to open a black box, so other researchers and practitioners can critically engage with the rather messy process of model evolution and learning in participatory modelling in a systematic way.

5.4.4. Scaling participatory modelling

Our conceptualisation portrays the modelling cycle as contained within the stakeholder engagement cycle, with a close, yet non-complete alignment among their phases. This understanding gives a wider sense of the temporality and the activities that support the development of a participatory model. This is particularly important for projects aiming to address *wicked* public environmental problems (see Rittel and Webber 1973) in which there are no clear 'owners' but a constellation of stakeholders with contested interests and power levels—yet with enough common ground to engage in dialogues that foster learning (i.e. a *dialogical learning strategy*, as proposed by Brugnach et al 2011). One of the ways in which such a dialogue can be operationalised is via a model-based approach such as participatory modelling, as explained by Amorocho-Daza et al. (2025).

Such an approach resonates with the major challenge of scaling participatory modelling for policy relevance. The activities of the final project's workshop (WS6) illustrated the need to go beyond PM if policy impact is desired. Once participatory modelling achieves its purpose—defining a meaningful and agreed way forward towards addressing a challenging environmental problem—other activities are needed to answer emerging questions, such as '*how* a certain policy (or set of policies) should be implemented, by *whom* and *when*'. That is, even if the PM setting is successful in creating spaces where shared policy futures are identified, a different set of strategic planning tools is required to actually implement them (Miedzinski et al., 2022, Slinger 2023). Other research developed as part of the NEXOGENESIS project proposes the development of governance roadmaps as tools that facilitate connecting the participatory modelling insights with local policy actions (Avellán et al., 2025b, Mooren et al., 2025a). Such understanding contributes to the theme of 'scaling' participatory modelling, as an existing major knowledge gap identified by Jordan et al. (2019). Future empirical research is needed to expand on how PM products can be fed into policy implementation cycles.

5.5. CONCLUSION

This research describes a structured approach applied in assessing model evolution and participants' learning in the context of an environmental participatory modelling (PM) project. Our approach conceives participatory modelling as happening within a wider stakeholder engagement process—that is, the modelling cycle is nested within the engagement cycle. Model evolution is tracked by analysing key stakeholder-driven model changes based on their 'depth', following the leverage points proposed by Meadows (1999, 2008). We distinguish different types of knowledge each responding to multiple domains (i.e. types of analysis) and show that modelling changes evolve from deep to shallow in time.

The participatory modelling setting is provided by NEXOGENESIS, a European Commission Research Project, more specifically by one of its case studies exploring socio-environmental problems in the Lielupe River Basin from a resource nexus perspective. By tracking model evolution, we are able to illustrate how stakeholder feedback in early stages of the modelling cycle lays ‘deep’ foundations that determine the model capabilities of later stages—converging to ‘shallow’ refinements. Our results illustrate that stakeholder learning is complex and partially responds to workshop design, though we evidence knowledge spillovers across knowledge types and types of analysis. Modeller learning comes first from the understanding of a new problematic situation and later from the challenge of formalising it using system tools. In our study, the modeller learns the most in the early stages whereas stakeholders learn more towards the end of the modelling cycle Our results show that overall, PM outcomes (e.g. model and learning) emerge from participants' interactions, in which they jointly co-create a shared understanding of a problematic situation via system tools. Our open ‘black box’ illustrates that participatory modelling is a socially driven endeavour that can foster participants' learning via model building.

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6

SYNTHESIS

6.1. RESEARCH CONTRIBUTIONS AND IMPLICATIONS

This dissertation contributes to the theory and practice of modelling the environment with stakeholders. Our theoretical contributions, in Part I of the thesis, clarify that environmental modelling is a participatory and value-laden process that requires to be addressed and disclosed via structured approaches—also referred to as good modelling practice. Our main theoretical contributions comprise: (1) a flexible 3-phase modelling framework for addressing socio-environmental issues; (2) a structured set of ethical questions when engaging in participatory modelling.

Part II presents the application of our modelling framework in a resource nexus case study dealing with the issue of transboundary nutrient pollution. Our empirical research contributes to nexus action in the Lielupe River Basin, and provides insights that are potentially transferable to other cases. The co-created simulation model demonstrates that the efficiency of nutrient control policies in transboundary settings is highly dependent on both land-use transitions and bilateral cooperation. The most successful policies exhibit strong cooperation while requiring less ambition in land-use transition. The experience in the Lielupe illustrates that model evolution and learning are socially driven across a participatory modelling process.

This thesis therefore contributes towards holistic, reflective and transparent modelling practice. It provides environmental modellers with practical tools for modelling with stakeholders. Ultimately, the dissertation presents a call to consider how modelling can support broader participatory engagement processes rather than how participatory engagement can support modelling.

Part I - Theoretical perspectives

The first part of this PhD dissertation offers environmental modellers, more specifically SD researchers and practitioners, structured modelling frameworks aiming to support informed engagement in wicked socio-environmental problems. More specifically, this research unpacks the implications of modelling the environment in terms of incorporating elements of uncertainty analysis, participation and ethics. A modelling cycle perspective is adopted—in which a model is conceived, developed and used in successive, iterative phases—and embedded in an analysis of a phenomenon of interest using a SES perspective.

Chapter 2 illustrates the dynamic interaction between stakeholder participation and uncertainty across a modelling cycle, in which three generic modelling phases are identified: (I) Modelling foundations, (II) Model building and testing, and (III) Model use and policy evaluation. Here, a wide set of tools and techniques is identified and classified across the three phases to address uncertainty and participation features in socio-environmental modelling settings. This contribution provides methodological clarity on how to link and position both qualitative and quantitative tools within a single modelling framework. The work, therefore, bridges concepts and methods in two seemingly disparate sets of SD practices, one with a quantitative-policy focus (see Kwakkel and Pruyt 2015, Auping 2018), and the other emphasising qualitative and participation dimensions (see Vennix 2000, Luna-Reyes et al. 2006, Andersen et al. 2007). Our work echoes early SD ideas aiming to distinguish and integrate the ‘model versus a modelling process' (Forrester, 1985) and other literature calling for an integration of *soft* and *hard* approaches in the broader field of operations research (Westcombe et al., 2006; Johnson et al., 2018).

This chapter, therefore, promotes an integrated and interdisciplinary approach to environmental SD modelling. More specifically, it proposes a structured integration of various established approaches in different academic fields into the SD modelling cycle, such as Problem Structuring Methods (PSM) (Phase I), Exploratory Modelling (EM) (Phase II and III), and Decision Analysis (DA) (Phase III). Qualitative SD methods align strongly with the tradition of PSM, while quantitative methods are connected with EM and DA practices. While a multidisciplinary phased setup—allocating tasks by expertise across the modelling cycle—might seem feasible in principle, such a message does not derive from the work. On the contrary, the chapter emphasises the deeply intertwined nature of quantitative and qualitative modelling, particularly in their joint use at the end of the modelling cycle—when the model is employed for policy evaluation. Using a model in decision support requires (re-)engaging with the framing and priorities set in the cycle's early stages. Evidently, neither qualitative nor quantitative SD expertise alone is sufficient to deploy a whole modelling cycle to address an environmental problem. Instead, these approaches go hand in hand (as also pointed out by Jakeman et al. 2024). Moreover, the full range of complementary tools and methods required to implement the iterative modelling cycle from beginning to end extends beyond SD expertise alone (see Chapter 2 and 5). This implies that working with experts from other disciplines, i.e. an interdisciplinary approach, is imperative when designing an SD project with a socio-environmental focus.

Likewise, by emphasising the need to model with stakeholders—and aiming to support real-life policy interventions—the proposed modelling framework raises the importance of moving towards transdisciplinary approaches in environmental SD. Other research has recently pointed out that SD modellers need a set of competences that suit the complexity that working in participatory settings entails (Elsawah et al., 2023). This implies that SD modellers need to gain transdisciplinary skills to effectively co-create with experts and stakeholders alike (see Slinger and Kothuis 2022, Slinger et al. 2023). We identify several tools and techniques that can facilitate this process from an operational perspective. Our modelling framework also shows how stakeholders influence the modelling cycle in multiple ways. This implies that they are not only there as specific input sources to scope the issue or for model validation—their active contribution is transversal to the modelling process (as also highlighted by Jakeman et al., 2024). By pinpointing this, we provide researchers and practitioners with a structured approach to designing model-based participatory interventions. Our framework encourages stakeholder participation, not as a mere requisite, but as a potentially transformative feature of environmental modelling—one that is essential in allowing modelling outcomes to inform real-world problem solving. Chapter 3 explores the broader implications of modelling with stakeholders.

In Chapter 3, we identify ethical standpoints for socio-environmental SD research and practice, as well as guiding ethical questions to be asked at different stages throughout the modelling cycle. Therefore, we offer SD researchers and practitioners a structured entry point to explore the messy yet ever-present ethical dimension of socio-environmental problems. This research connects with Chapter 2, as many ethical questions emerge from the uncertainty and participation dimensions of environmental modelling. For instance, defining who participates, when and how is not merely a procedural but essentially an ethical question. Similarly, deciding how to frame and communicate uncertainty and its implications for decision-making extends beyond the technical domain and enters the ethical realm. Recognising the ethical dimension as pervasive in a modelling cycle is a contribution to the SD field and an invitation to revisit its philosophical foundations, to explore the alignment with moral objectives such as promoting human dignity and protecting the environment. It is an invitation to draw ethical discussion from the periphery to the centre of SD theory and practice.

Overall, Chapters 2 and 3 promote reflective, holistic, and transparent modelling practices—contributing to a Good Modelling Practice (GMP) paradigm (see Jakeman et al. 2024 and Hamilton et al. 2022). By breaking a complex task, such as building an environmental model, into three essential phases, Chapter 2 provides researchers and practitioners with a flexible framework that may also be suitable for application to environmental issues beyond the social-ecological focus presented here. This framework helps in structuring, rationalising and reporting complex participatory modelling endeavours in a straightforward way (see a similar proposal by Gray et al. 2018). By capturing the value that can be expected from implementing an iterative modelling cycle, we aim to facilitate identifying the missing steps in moving from decision support to actual policy implementation. In Chapter 3, we argue that as environmental modelling is not an objective endeavour, ethical considerations should be discussed alongside the model's technical aspects. Therefore, we encourage researchers and modellers to be reflective and transparent about their choices and provide ethical questioning as a way to debate and disclose modellers' decisions (as similarly proposed by critical perspectives in the field of Operations Research, see Ulrich 1987). The following section illustrates two ways in which the Chapter 2 framework was operationalised for a specific socio-environmental problem situation involving multiple competing interests and stakeholders— the Lielupe River Basin case study.

Part II - Practical applications for transboundary river management

The second part of this PhD dissertation aims to explore the added value of implementing Chapter 2's modelling framework, in the context of a transboundary river management problem. Both chapters in this section make use of the case study of the Lielupe River Basin (Lithuania-Latvia) and its nexus-related and transboundary issues, as part of the EC research project NEXOGENESIS. Chapter 4 illustrates the deployment of the 3-phase modelling cycle to explore the issue of transboundary nutrient pollution in the Lielupe River. It shows how the model evolved as part of an inclusive participatory process yet focuses on using the model to derive locally relevant policy insights. In a complementary fashion, Chapter 5 makes use of the modelling framework phases to characterise stakeholder-driven model evolution and assess how learning took place across the participatory process—for both modeller and stakeholders.

Chapter 4 engages with the challenge of presenting a whole-modelling cycle in a single research paper—an unconventional but desirable approach (see Voinov et al. 2014). This is achieved by a subtle methodological choice, that is, considering the 3-phase modelling cycle as the method itself. By doing so, the results are not limited to the model or to the model validation but can be extended to the policy insights emerging from its use. This allowed us to shift our attention to the policy implications and limitations of the simulation outputs in the context of the Lielupe River Basin. This methodological approach is a contribution to the field of environmental management and modelling, as it presents a whole modelling cycle approach, including the participatory process, yet focusing on the key policy messages that aim to inform a broader, grounded policy process.

Our methodological contributions connect to calls in the literature to report research experiences developing and using ‘fit-for-purpose’ models as part of good modelling practice (Hamilton et al., 2022). According to Hamilton and colleagues, ‘fit-for-purpose’ models are: “*useful*, addressing the needs of the end user; *reliable*, obtaining an adequate level of certainty or trust; and *feasible*, within practical constraints of the project” (2022, p. 1). Chapter 3 reports how the three modelling phases contributed to achieving these model features. The model's ‘usefulness’ stems from the modelling foundations phase, in which the model is scoped with stakeholder participation to answer locally relevant questions. Arriving at this end is not straightforward, as this is an intensive experience—half of the project’s stakeholder workshops and time were devoted to this phase. This corresponds with the theoretical expectations proposed by Jakeman et al. 2024. The ‘reliability’ dimension came later at the model-building and testing phase. Here, the validation process is reported using locally relevant data provided by stakeholders. The feasibility dimension is addressed indirectly by showing the development of the final phase—model use and policy evaluation—which shows how the model is integrated into subsequent project activities, including developing decision support tools and stakeholder dialogues.

Likewise, by focusing on the policy analysis dimension, Chapter 4 illustrates a step forward in connecting modelling results with local policy processes. By interpreting the model results in accordance with the socio-economic features and the policy landscape constraints of the local situation, we take a more realistic, still hopeful, approach towards policy relevance. This chapter’s approach therefore can be taken as a precedent and an invitation to environmental modellers not to stop once simulation results are validated, but to go beyond to explore their policy implications. Modellers cannot do this alone; they need to be part of a larger inter- and transdisciplinary network of researchers and practitioners through which modelling insights can be brought into further dialogues and included in activities that potentially lead to local or regional change. Chapter 4 illustrates how this interaction (or lack thereof) took place in the Lielupe River Basin. This is possible by taking a perspective wider than the modelling cycle and considering it as nested within a stakeholder engagement process in which the dimensions of stakeholder interaction and policy dialogues are prevalent.

Chapter 5 is an invitation to open the black box of PM by exploring how models evolve while participants learn in a PM setting. This chapter shows that PM participants in the Lielupe (modeller and stakeholders) participated and interacted across several workshops, a process that promoted learning and led to the incremental co-creation of an environmental simulation model focused on a locally-relevant problem. Therefore, the chapter contributes a new methodological perspective and a case study to assess learning and model evolution in PM. Future studies can tailor and apply our methods to continue exploring the features of PM products in a structured and replicable way (see Hubacek et al., 2017). Our approach offers an opportunity for researchers to compare potentially distinctive and desired PM product features—for instance, broad social learning, fit-for-purpose models—with the outcomes of using conventional expert-led modelling. Interestingly, our analytic approach can be extended beyond quantitative model-based settings, to wider environmental participatory interventions such as Problem and Game Structuring Methods (Ackerman, 2012; Rosenhead, 2006; Westcombe et al., 2006; Cunningham et al., 2014; Slinger et al., 2023). In short, our approach is potentially helpful in capturing and comparing the 'value' of doing PM versus deploying conventional environmental modelling or Problem and Game Structuring Methods.

Despite our research contributing to the research avenues mentioned above, perhaps our most important contribution is to point out the intrinsic limits of PM in influencing environmental policies on the ground. Chapter 3 and 4 illustrate that PM alone is limited by its own scope, tools and knowledge in achieving policy relevance. Here we have another call to incorporate inter- and transdisciplinary perspectives in environmental modelling, and consider how modelling can support broader participatory engagement processes rather than how participatory engagement can support modelling. These, and more limitations and future research opportunities, are discussed further in Section 6.3.

6.1. RESPONDING TO THE RESEARCH QUESTIONS

1. How can participatory considerations be incorporated in model development processes for complex (dynamic and uncertain) socio-environmental problems?

Participatory considerations are relevant at every stage of an environmental modelling process. A suitable way to acknowledge them is through a modelling cycle perspective—one in which a model is conceived, developed and used in successive, iterative phases with stakeholders. By integrating and aligning two relevant modelling cycles, one engaging with uncertainty and the other with participation aspects, we were able to distinguish three generic modelling phases, namely: 1. Modelling foundations, 2. Model building and testing, 3. Model use and policy evaluation. The three generic modelling phases capture the essence of a modelling endeavour, making explicit when and how stakeholders can contribute along the model development process, moving from qualitative to increasingly quantitative modelling tools. This modelling cycle perspective facilitates the analysis of interactions between participation and uncertainty across the phases. Such a framework brings conceptual clarity and practical tools to approach the dynamic relation between uncertainty and participation in model-based decision-making.

2. What are the ethical implications of participatory modelling for socio-environmental problems?

Participatory modelling has ethical implications, evident through: (1) exploring the modelling cycle with an ethics lens; and (2) considering the realm of application in which PM happens, for instance, in the context of resource management. Regarding the first point, a practical way in which ethical implications can be explored is through ethical questioning. We identify ethical questions relevant to different modelling cycle stages as a structured entry point for ethical reflection. For the second point, we take a different approach, and consider ethical standpoints that are consistent with the realm of applications of environmental PM. More specifically, we propose Sustainable Development and Human Rights as ethical standpoints for PM practice, as they entail a set of principles that need to be considered for addressing socio-environmental problems. These standpoints, in turn, can inspire the formulation of pertinent questions for reflecting on the ethical implications of a specific PM endeavour.

3. Which policy insights emerge from the participatory development and application of a simulation model in transboundary river management (of the Lielupe River Basin)?

Implementing a participatory modelling cycle in the Lielupe River Basin led to various policy insights. By applying the 3-phase modelling cycle proposed in this thesis (see Research Question 1, Chapter 2) as a method, attention shifted from the model features and model validation to the policy implications and limitations of the simulation outputs in the local context. Modelling results for the Lielupe overall show that nutrient control policies are effective under ambitious land-use transitions. By implementing basin-scale solutions, exploratory analysis shows that nutrient control would reduce nitrogen concentration by around 30% with a 2% co-benefit of increasing vegetation stocks, yet at the cost of decreasing cereal production by 8%. However, stakeholder-driven modelling features highlight the importance of promoting active transboundary cooperation for water quality control, as unilateral action hampers the effect of long-term ambitious policies. Results show that even highly ambitious *unilateral* action can delay the achievement of river basin quality objectives in the order of a decade, a critical finding for the wider Baltic region and the achievement of EU water quality objectives. Moreover, the extensive stakeholder discussions within the participatory modelling setting facilitated discussion and analysis of the implications and real-world plausibility of achieving such policy futures.

4. To what extent does participatory modelling promote participants' learning in the context of transboundary river management (in the Lielupe River Basin)?

Learning is considered one of the key outcomes of participatory modelling. This dissertation evaluates empirical evidence of knowledge generation and model evolution in the context of transboundary river management in the Lielupe River Basin. In doing so, we highlight that learning assessment should not be limited to stakeholders but also include the modellers as key participants of a PM project. Additionally, analysing how an environmental model evolves across a participatory process provides a richer picture of how learning takes place in a participatory process. We propose a structured approach to track model evolution and the learning of participants across a modelling cycle. Our results illustrate that PM is a socially driven endeavour that can foster participants' learning across and beyond a modelling cycle. By tracking model evolution, we illustrate how early-stage stakeholder feedback lays the foundations that determine the final model capabilities. Our results evidence participants' multi-dimensional knowledge generation, yet with temporal and structural differences between the modeller and the stakeholder groups. Although we found evidence that environmental PM promotes stakeholder learning, the translation of insights into real-life and sustainable change remains an open academic and practical challenge.

6.2. LIMITATIONS AND FURTHER RESEARCH

Part I - Theoretical perspectives

Perhaps the most evident limitation of this PhD dissertation is that the ethical modelling framework was only proposed but not operationalised. This has a practical reason; a structured participatory process was already being designed and deployed before the ideas of Chapter 2 were even conceived (see Hüesker et al. 2022, Avellán et al. 2025). In other words, while my own ethical journey was beginning, the stakeholder engagement strategy in the NEXOGENESIS project was already well underway. However, some of the practices and ideas proposed in Chapter 2 helped me navigate my own PhD journey. First, and perhaps not explicitly evident in the chapters, ethical reflection has helped me to see myself not as an objective scientist but as a subjective systems thinker. This positionality helped me to deeply value my interaction with other researchers and stakeholders in the PM process. Some of my own thinking, reasoning, and agency throughout the process are disclosed in Chapter 5—where the interview with the modeller is presented. I tried to follow my own advice by aiming for transparency in reporting the PM experience. This is explicitly evidenced in the way articles in Part II of the dissertation deliberately focus on opening up the modelling process (the 'how')—disclosing assumptions, decisions and limitations along the PM project trajectory—rather than just on the modelling products alone (the 'what'). Although this is not as deep and structured as the ethical questioning proposed in the research paper—which is an invitation for a whole research community—it provides an example for environmental modellers on how to deliberate about their own positionality and report their results in a more transparent way.

Surprisingly, ethical reflection in environmental modelling is in the early stages. After the publication of our article, other authors have cited our work and joined the academic conversation around ethical modelling (for instance, Gordon et al. 2024, McAlister et al. 2026, Szetey et al. 2025, ter Horst 2025). This marks an exciting trend towards more reflective modelling in the SD community and beyond. Future methodological work can focus on enhancing our cycle-based ethical questioning by considering other established philosophical frameworks in the broader field of systems thinking and operations research (Ulrich, 1987; Ulrich and Reynolds, 2010). Ulrich's Critical System Heuristics (CSH) provides a set of 12 ethical questions categorised over relevant dimensions that can be further analysed and integrated with the questions we postulate in Chapter 3. Pathways for further empirical research are broad and may consider developing and reporting on strategies to operationalise wide and universal ethical standpoints to concrete and specific socio-environmental problems in the context of PM applications. Challenging yet exciting research could report on strategies adopted and lessons learned while promoting ethical reflection on stakeholder workshops—for instance, providing insights about how to balance deliberation and reflection without losing scope in a time-constrained setting.

Recent research highlights that underpinning environmental co-creation initiatives such as PM is a deep commitment towards 'democratising' expertise while 'expertising' democracy (Liberatore and Funtowicz, 2003; Zellner, 2024; Berg and Lidskog, 2018; d'Hont and Slinger, 2022); that is, to promote deliberation spaces in which both experts and citizens actively work together towards addressing environmental public policy challenges. Future research could focus on unpacking the implications of having such 'democratic' commitment through an ethical lens. This inspires questions such as '*What are the values that underpin democratising knowledge in environmental modelling settings and beyond?*' And '*How to articulate them as part of the PM processes?*'. Answering these questions implies crossing disciplinary boundaries in the fields of environmental modelling, public policy and philosophy, bordering ongoing, highly influential, efforts in the field of ecosystem services and biodiversity protection (see Luque-Lora 2024, Chan et al. 2015, Diaz et al. 2016).

Regarding our proposed modelling framework in Chapter 2, it is important to highlight that it emerged from analysing the dimensions of uncertainty and participation in socio-environmental SD modelling. Therefore, it is formally limited to this domain and highly applicable here. Our framework is one of many possible frameworks, yet is useful for addressing two of the eight SES grand modelling challenges identified by Elsawah et al. (2020). These are related to combining qualitative and quantitative methods and integrated uncertainty assessment. However, our proposed global modelling phases resonate with wider environmental modelling practice. Interestingly, the structure and logic of the modelling cycle display a certain 'isomorphism' when compared with a parallel modelling framework proposed by Jakeman et al. (2024)—which aims to be relevant to the whole field of socio-environmental modelling. Additionally, the article has been embraced by the environmental modelling community, obtaining 18 citations in Google Scholar (13 in Scopus), just one year after its publication. Future research could undertake an in-depth analysis of the similarities, differences and complementarity of the framework proposed in Chapter 2 and the one proposed by Jakeman, et al. (2024).

Likewise, proposing new frameworks that connect modelling and policy implementation cycles would be a valuable contribution to guide researchers and practitioners in advancing the field of SES modelling towards increased policy relevance. Further research addressing such a task may consider building on existing conceptualisations and extensive empirical work exploring the connection between stakeholder learning and action (i.e. behavioural change) in the SD field of Group Model Building (GMB) (Rouwette, 2016). More reflections about the challenge of the real-world application of PM are provided in the following section.

Part II - Practical applications for transboundary river management

In Part II, we provide an in depth report on some of the most important outcomes of the PM process developed in the Lielupe case study—including the participatory simulation model and the policy insights that emerge from its use (Chapter 4), as well as the participants' learning over the course of the project activities (Chapter 5). Although the chapters in Part II serve as an example for future empirical research engaging with the paradigm of Good Modelling Practice (GMP), the PM literature might benefit by including critical perspectives in analysing the crafting and decision-making processes that occur along the modelling cycle. This thesis proposes exploring ethics in modelling practice (see Chapter 3) whilst recognising that including other lenses, such as power and politics, can be valuable in studying issues such as bias and representation in PM (see Ter Horst 2025; Ter Horst et al., 2023).

Generalisability is limited in Part II of the thesis. This emerges from the methodological choice of using a single case study offering analytic depth over breadth (Flyvbjerg, 2006). For instance, Chapter 4's insights about the relative importance of bilateral cooperation over individual land-use change ambitions of the riparian countries in transboundary river management derive from the Lielupe. Future research could explore to what degree these local findings extend to other transboundary basins, particularly those dominated by agrarian land use. Similarly, Chapter 5's findings—evidencing participants' learning and characterising model evolution in PM—are also highly contextual. Future empirical research can compare our results with other PM case studies to evaluate whether similar outcomes arise (or not) in other settings.

That being said, it is worth mentioning that obtaining these outcomes required significant costs in terms of financial and human resources. The PM setting in which Part II takes place was a European Commission-funded project – NEXOGENESIS – with a budget of 5M Euro and a duration of 4 years (2021-2025). More specifically, the Lielupe case study was one of the 5 case studies of the project. This meant that, roughly, the project had a total of 1M Euro (or 250k EUR/year) per case study to accomplish its goals (not limited to the PM). Approximately 70 people contributed to the overall project, spanning the 17 consortium members. More specifically, a group of 2 or 3 senior environmental experts and 1 PhD expert in System Dynamics (myself) exclusively worked on the Lielupe case study over the project lifespan, with many other experts from various disciplines (e.g. climate modelling, governance and IT) contributing less regularly at different project stages. These simple and broad indicators highlight the magnitude of the challenge in developing a PM project. Surprisingly, a recent review shows that, despite their criticality, cost considerations are rarely discussed in PM research articles (Hedelin et al., 2021). Yet, as PM projects have high demands in terms of time and skills, it is clear that funding requirements are not issues exclusive to NEXOGENSIS but practical limitations to successfully implementing a PM endeavour. Put simply, gathering highly skilled people over a long period is expensive.

This raises immediate questions, such as *Is developing a PM project worthwhile in terms of its costs?* Or *What is the value that PM brings to promote a real-world environmental change?* Other researchers have asked similar questions and framed them as critical for the research field (Zellner, 2024; Hedelin et al., 2021). There are various ways forward for the field to gain policy relevance; for instance, by considering *means*, *ends*, and connecting *means to ends*. A way to address the *means* is to consider critical factors that might foster real-world impact during the planning phase of a PM project. Hedelin and collaborators (2007, 2015, 2021) proposed structured yet flexible instruments to support PM planning in considering policy impact. Research about the *ends* is also rather scarce, as pointed out in Chapter 5. Examples of *ends* can be PM products such as network development or promoting social learning—a topic to which we contribute in this dissertation (Chapter 5)—or even a change in participants' beliefs (Smagl and Ward, 2015). However, more ambitious *ends* can be set by aiming to induce specific real-world policy changes. This is hardly addressed in PM literature, yet perhaps constitutes a practice to be desired. Interestingly, the NEXOGENESIS project did set objectives in this regard in the form of stakeholder agreements to be reached at the end of the project, for each case study (Sušnik, 2024b). Research about the nature of the agreements and their implications for local policy settings is under development.The real challenge, and an open research avenue, lies in connecting the dots, *means to ends*. Both empirical and theoretical research is needed to bring light to this knowledge frontier, that is, to understand which features of PM promote policy impact. Research that provides deep accounts about how to deploy PM features, what does (not) work and in which contexts, while distilling more generalisable insights from case studies is needed. This knowledge frontier is acknowledged and extensively documented in other participatory SD fields (see Rouwette, 2016). PM researchers do not need to start from scratch here, but can learn from transdisciplinary studies that draw on problem and game structuring methods (see Slinger et al. 2022**,** Slinger et al. 2023) and a number of scientific disciplines whose paradigm is gravitating towards practical applications (see Spruijt et al. 2014, Wigboldus et al. 2016, Albros et al. 2020). Future research exploring the policy impact of PM may use concepts such as Meadows' leverage points (see Chapter 5), as a promising way forward in analysing and promoting high-leverage strategies for real-world change, as already applied in the field of sustainability transitions (Abson et al., 2016; Chan et al., 2020; Fischer and Riechers, 2018; Leventon et al., 2021).

PM is an interdisciplinary field, where various knowledge fields—systems thinking, modelling, stakeholder engagement and environmental science—merge and find a place. However, to become policy relevant, PM researchers and practitioners need to work with experts from other seemingly distant fields, including, but not limited to, public policy and administration, governance, project management, and even sociologists, economists and lawyers. That being said, by leveraging relevant products and taking bolder interdisciplinary steps, PM could become better positioned for shaping future environmental policies in inclusive and systematic ways.

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APPENDIX A

Supplementary Information: A participatory system dynamics approach to assess transboundary nutrient pollution: modelling the water-energy-food-ecosystems nexus in the Lielupe River Basin, Lithuania and Latvia

Section 1 of this document presents the detailed assumptions, equations, parameters, and baseline outputs of the System Dynamics model presented in Section 2.4 of the main article. Model files using the RCP2.6 and RCP8.5 climate scenarios are available using the following DOI: 10.4121/4be86f5e-7f92-44b3-9b7b-6627478bb6c0.

Section 2 shows supplementary tables to the results (Section 3 of the main article).

1. STOCK AND FLOW MODEL

1.1 Land sector

The land sector consists of land uses represented as stocks. The stocks considered in the simulation model are *arable land*, *grasslands,* and *forests.* Current stock values were estimated using remote sensing tools and are summarised in Table 1.

Table A1. Areas in the LRB by land use type, including riparian countries' areas and relative areas.

Land Use	Total Area (ha) - LRB	Relative area (%) - LRB	Total area - (ha) - Lithuanian side	Relative area (%) - Lithuanian side	Total area - (ha) - Latvian side	Relative area (%) - Latvian side
Arable land	751590	43%	479301	55%	272289	32%
Forest	737767	43%	282214	32%	455554	53%
Grasslands	239240	14%	114651	13%	124589	15%
Artificial and other	290	0%	180	0%	110	0%
Total	1728888	100%	876346	100%	852542	100%

To model policies in the basin, we include an additional stock of $Arable\ land_{with\ nutrient\ treatment}$ that accounts for the arable land where nutrients reach an installation with any sort of nutrient treatment before entering surface water. Nature-based Solutions (NBS) and organic agriculture are part of the Best Management Practices (BMPs) that can be used to control nutrient pollution using natural processes to retain nutrients in soil and plants before reaching the water system (Lintern, McPhillips, Winfrey, Duncan, & Grady, 2020; Nsenga Kumwimba et al., 2023; Ricci, D'Ambrosio, De Girolamo, & Gentile, 2022). Important to highlight is that $Arable\ land_{with\ nutrient\ treatment}$ is not the area devoted to a particular NBS (e.g. constructed wetlands), but represents the agriculture area that drains to an NBS in aggregate. Therefore, the total arable land will be equivalent to the sum of the arable land with and without treatment (Eq. 1).

$$Total\ arable\ land = Arable\ land_{with\ nutrient\ treatment} + Arable\ land_{no\ nutrient\ treatment} \text{ (Eq. 1)}$$

The rate of transitioning from regular to treated arable land is modelled as a *policy lever* or policy entry point. That is, it is a long-term policy defined by the user as a model input. Eq. 2 describes a differential equation that includes the policy lever of interest.

$$\frac{dArable\ land_{no\ nutrient\ treatment}}{dt} = -Arable\ land_{no\ nutrient\ treatment} * rate_{nutrient\ reduction\ implementation} \text{ (Eq. 2)}$$

As the aforementioned rate may be difficult to estimate, we ask users to provide the long-term fraction of land with NBS that they would like to explore in the model. For example, a fraction of 0.5 would mean that the user would like to explore the effect of transforming half of the arable land into arable land with nutrient treatment by the end of the simulation period. The $Fraction\ of\ arable\ land\ with\ nutrients\ treatment$ is defined by Eq. 3.

$$Fraction\ of\ arable\ land\ with\ nutrients\ treatment = \frac{Arable\ land_{with\ nutrient\ treatment}}{total\ arable\ land} \text{ (Eq. 3)}$$

An expression for $rate_{nutrient\ reduction\ implementation}$ in terms of $Fraction\ of\ arable\ land\ with\ nutrients\ treatment$ is presented below:

$$rate_{nutrient\ reduction\ implementation} = -(1 - Fraction\ of\ arable\ land\ with\ nutrients\ treatment)^{\frac{1}{t_{final}}} - 1 \text{ (Eq. 4)}$$

Another important land use change in the basin is the transition from arable land to grasslands. Existing policies encourage reducing arable land by a certain percentage (e.g. 10%) in a definite period t_{final}. This rate is modelled as a policy entry point or policy switch in the form of a binary variable that activates or deactivates the policy.

$$\frac{dGrasslands}{dt} = rate_{arable\ to\ grasslands} = percentage\ reduction * \frac{total\ arable\ land}{t_{final}} * policy\ switch. \text{ (Eq. 5)}$$

An important stock in the LRB is forests. However, we consider them as static in the current model. That is, no increase or decline is considered for this land use as they are currently protected. Despite their ecological importance, forestry policies did not appear in the stakeholder discussions.

1.1.1 Derivation of Eq. 4

An analytical solution of Eq. 2 can be estimated by a generic equation of geometrical growth, as follows.

$$Arable\ land_{no\ nutrient\ treatment,t=final} = -Arable\ land_{no\ nutrient\ treatment,t=0} * \left(1 - rate_{nutrient\ reduction\ implementation}\right)^{t} \text{ (Eq. S1)}$$

It is possible to re-write Eq. S1 assuming that $arable\ land_{no\ nutrient\ treatment,t=0} = total\ arable\ land$ and incorporating Eq. 2.

$$\frac{Arable\ land_{no\ nutrient\ treatment,t=final}}{total\ arable\ land} = \frac{total\ arable\ land - Arable\ land_{with\ nutrient\ treatment,t=final}}{total\ arable\ land} = -\left(1 - rate_{nutrient\ reduction\ implementation}\right)^{t} \text{(Eq. S2)}$$

By replacing the fraction of arable land with nutrient treatment (Eq. 3) and replacing it in Eq.S3 Appendix A it is possible to solve for $rate_{nutrient\ reduction\ implementation}$.

$$rate_{nutrient\ reduction\ implementation} = -\left(1 - \frac{Arable\ land_{with\ nutrient\ treatment,t=final}}{Total\ arable\ land}\right)^{\frac{1}{t_{final}}} - 1 = -(1 - Fraction\ of\ arable\ land\ with\ nutrients\ treatment)^{\frac{1}{t_{final}}} - 1 \text{(Eq. S3)}$$

1.2 Food Sector

LRB's dominant land use is arable land dedicated to crop production. LRB produces various crop types, the main being wheat (summer and winter), rapeseed, maize, and dry pulses. Remote sensing data indicates that these four crops account for approximately 90% of the total cultivated area in the basin (Table 2).

Table A2. Main crops of the LRB

Crop type	**% cultivated area (of total arable land)**
Common wheat	63%
Rapeseed	13%
Dry pulses	10%
Maize	4%
Other crops	10%
Total	100%

These crops are therefore representative for estimating crop production in the basin. Long-term estimates of crop yield in the basin were taken from four impact-models of from ISIMIP2b, as described by Trabucco et al. (2024). Figures 1 and 2 summarise the crop yield time series confidence interval (CI) for the four main crops in the basin (Table 2) in two climate change scenarios (i.e. RCP 2.6 and RCP 8.5). Crop production can be estimated as cultivated area times crop yield (Eq. 6). It is worth mentioning that by modelling crop yield as an exogenous probabilistic variable, our approach does not incorporate soil quality as a driving factor of yield and, therefore, food production.

$$Crop\ production_{crop\ type} = Relative\ cultivated\ area_{crop\ type} * arable\ land * crop\ yield_{crop\ type}\ \forall\ crop\ type \quad \text{(Eq. 6)}$$

Current policies of the riparian countries encourage a transition from intensive farming towards organic farming practices. A literature meta-analysis has reported that organic agriculture affects crop yield compared with conventional agriculture, often reducing crop yield (Tuomisto, Hodge, Riordan, & Macdonald, 2012). The relative crop yield change ranges of organic farming are summarised in Table 10 for the crop types under consideration in the LRB. The model considers the effect of organic agriculture on food production by modifying Eq. 7 as follows.

$$crop\ production_{crop\ type} = relative\ cultivated\ area_{crop\ type} * crop\ yield_{crop\ type} * \left(Arable\ land_{traditional} + Arable\ land_{organic} * relative\ organic\ crop\ yield_{crop\ type}\right)\ \forall\ crop\ type \quad \text{(Eq. 7)}$$

Figure A1. Dynamic confidence intervals for cereal crops - RCP 2.6 and RCP 8.5

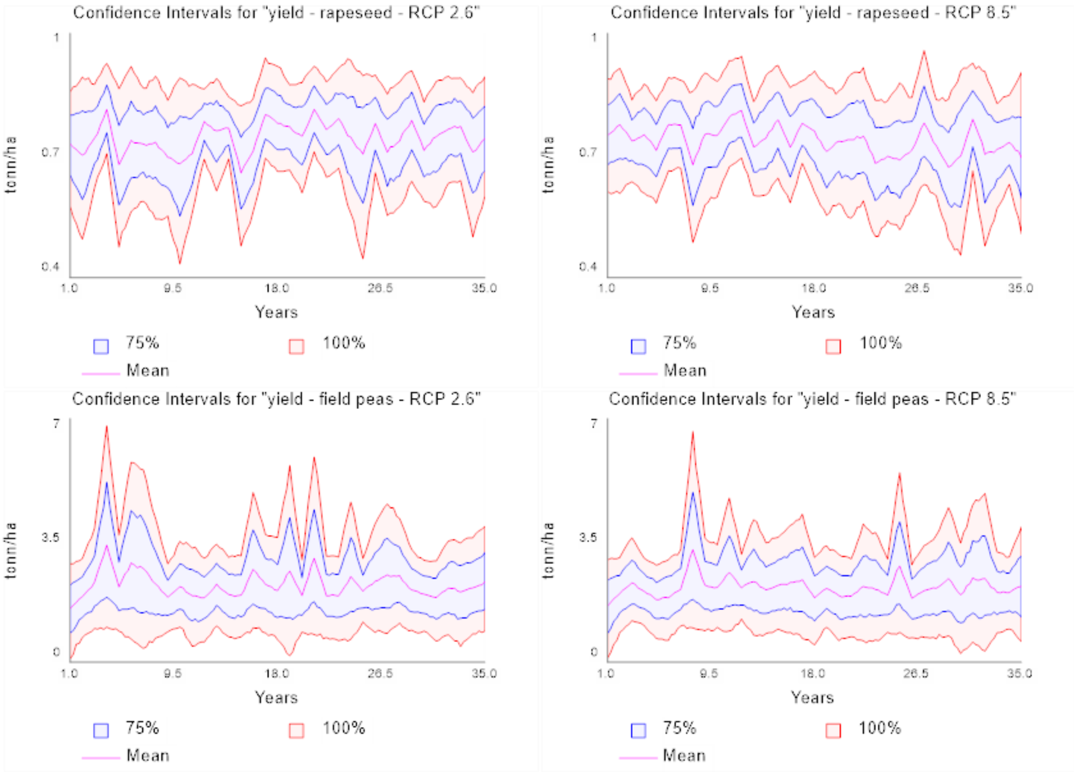

Figure A2. Dynamic confidence intervals for rapeseed and field peas - RCP 2.6 and RCP 8.5

1.3 Water sector

The water sector focus lies on water quality. Long-term estimates of surface water runoff in the basin were estimated using eight impact-models of ISIMIP2b, as described by Trabucco et al. (2024). For estimating surface water in terms of volumetric flow, we relied on the aforementioned data source, providing surface flow in terms of depth, and multiplied it by the drainage area of the LRB. Figure 3 summarises the time series' confidence interval for water runoff in two climate change scenarios (i.e. RCP 2.6 and RCP 8.5).

$surface\ water\ flow\ (volumetric) = \ surface\ flow\ (depth) * drainage\ area$ (Eq. 8)

For modelling water quality, we relied on a first-order delay structure with two connecting stocks to model the nutrient mass flow in the basin (Meadows, 2008; Sterman, 2000).

The first is $Nitrogen\ in\ transit$. This stock accounts for the nitrates that accumulate in the soil after leaving arable land but before reaching surface water bodies. The stock increases by the nitrate mass leaching from arable land, as a result of in-field fertilisation. Water quality monitoring records in the basin suggest that this is an important stock as even when agricultural activities ceased, the basin's nitrate concentration remained high and stable for at least a decade (Stålnacke, Grimvall, Libiseller, Laznik, & Kokorite, 2003). From this evidence, it is reasonable to infer the existence of a large nitrate accumulation in the basin causing delays in efforts to improve water quality in the basin. Such behaviour has been reported and modelled in multiple river basins (Melland, Fenton, & Jordan, 2018; van Meter & Basu, 2017; Vervloet, Binning, Borgesen, & Hojberg, 2018). To model such accumulation, Eq. 9 presents the mathematical formulation for the stock behaviour over time. For the stock's initial value, we estimated a wide range of 50,000 to 250,000 tonnes of Nitrate, corresponding to the cumulative leached nitrates in a period of 4-20 years, a reported catchment response time interval to nutrient control measures (Melland et al., 2018).

$\frac{dNitrogen\ in\ Transit}{dt} = nitrogen\ leaching\ from\ arable\ land - nitrogen\ reaching\ NBS$ (Eq. 9)

To characterise the inflow rate in Eq. 9 (Eq. 10), $nitrogen\ leaking\ from\ arable\ land$, two factors are needed, presented in Eq. 10: (i) the arable land; and (ii) the leaching rate corresponding to such area. Current arable land estimates are presented in Table 1. For estimating the leaching rate, we relied on the Zhou & Butterbach-Bahl (2014) meta-study of the academic literature providing median values and ranges for cropping systems of cereals, which roughly account for two-thirds of the land use in the basin (Table 10). Here we consider the expected leaching ranges for wheat, the dominant crop in the basin, are 29 kg NO3 ha^{-1} $year^{-1}$(4.2 – 54.4, 95% CI), a median value and range consistent with recent reporting on long-term nutrient leaching across the Baltic region (Motarjemi, Rosenbom, Iversen, & Plauborg, 2021; Siksnane & Lagzdins, 2020).

$nitrate\ leaking\ from\ arable\ land = total\ arable\ land * nitrate\ leaching\ rate$ (Eq. 10)

To define Eq. 9's outflow (Eq. 11) we relied on statistical analysis of water quality data provided by the Latvian Environment, Geology Meteorology Centre, at Kalnciems monitoring station in the period 2015-2024, using the first eight years for model calibration and leaving the rest of observations for validation. The measurements from the station represent an adequate proxy for the Lielupe's downstream indicators, as it is located 40 km from the river's mouth. We present the time series of water flow and nitrate concentration and further estimated nutrient loads by multiplying these terms (see Figure 4, Table 3). We found that nutrient load and concentration are significantly and linearly correlated with water flow at values larger than 30 m^3/s, with a coefficient of 23.6 (19.5-27.7, 95% CI, p=1.04E-16, Adj-R^2=0.7), and a significant value for the regression constant (see Figure 5, Table 4). Below this threshold, no significant statistical trend was identified (see Figure 6, Table 5). Therefore, for values smaller than the cutting point, we modelled the nitrates mass flow as a triangular function with parameters corresponding to the 5th, median and 95th percentiles of the observed data (see Table 7). The overall function for the nitrogen reaching NBS is defined as the maximum value between the regression function and a random triangular function (note that for small flow values, the regression function takes negative values).

$$nitrogen\ reaching\ NBS = \max\left(23.6 * surface\ water\ flow - 837.3, TRIANGULAR(1.1, 14.3, 103.5)\right) \quad \text{(Eq. 11)}$$

We considered a second stock to account for the *Nitrogen balance in NBS*. The purpose of this stock is to account for the nitrogen removal that occurs due to installing NBS in the basin and the subsequent nitrogen that finally leaks to the surface water. Given the limited empirical information regarding this stock, we assume that no accumulation of Nitrogen takes place in the NBS itself (See Eq. 12). However, a more complete conceptualisation of the real system might consider the *removed nitrogen* flow as an input to another stock, for instance, new vegetation, which is an accumulation that is not explicitly considered in the current modelling exercise.

$$\frac{dNitrogen\ balance\ in\ NBS}{dt} = 0 = nitrogen\ reaching\ NBS - removed\ nitrogen - nitrogen\ leaking\ to\ surface\ water \quad \text{(Eq. 12)}$$

A recent review shows that nitrogen removal capacity exhibits a wide range of efficiency and is dependent on the installed NBS system (Zhu et al., 2019). For the current model, we considered two NBS alternatives that are widely acknowledged to reduce diffuse nutrient pollution: (i) constructed wetlands; and (ii) riparian buffers (Castellano, Archontoulis, Helmers, Poffenbarger, & Six, 2019; Zhu et al., 2019). For the constructed wetland treatment system, we considered the set-up proposed by Castellano et al (2019), which consists of a two-stage treatment starting with a nitrification bioreactor followed by a constructed wetland. Riparian buffers, in contrast, are modelled as a single-stage treatment. The total nitrogen removed by NBS can be modelled as presented in Eq. 13.

$$removed\ nitrogen = nitrogen\ reaching\ NBS * \left(removal\ capacity_{contructed\ wetland\ system} + removal\ capacity_{riparian\ buffer}\right) \quad \text{(Eq. 13)}$$

To estimate $removal\ capacity_{contructed\ wetland\ system}$, we used reported nitrogen reduction efficiency ranges for the bioreactor ($\epsilon_{bioreactor}$) (Christianson, Bhandari, & Helmers, 2012) and for the constructed wetland ($\epsilon_{constructed\ wetland}$) as inputs (Cassman, Dobermann, & Walters, 2002). For riparian buffers, $removal\ capacity_{riparian\ buffer}$, we consider the nitrogen reduction efficiency ($\epsilon_{riparian\ buffers}$) ranges as reported in a recent metanalysis of the academic literature (Zhu et al., 2019) (see Table 10). It is important to mention that the overall removal capacity of both systems is constrained by the proportion to the fraction of arable land that drains to an NBS (i.e. *fraction of land with nutrient treatment* – Eq. 3) and the extension of the use of the alternative in relation to the other NBS treatment (i.e. *constructed wetland relative use extension* or *riparian buffer relative use extension* – Eq. 14).

$fraction\ of\ land\ drained\ by\ riparian\ buffers + fraction\ of\ land\ drained\ by\ constructed\ wetland = 1$ (Eq. 14)

$removal\ capacity_{riparian\ buffer} = fraction\ of\ land\ with\ nutrient\ treatment * \epsilon_{riparian\ buffer} * fraction\ of\ land\ drained\ by\ riparian\ buffers$ (Eq. 15)

$removal\ capacity_{contructed\ wetland\ system} = fraction\ of\ land\ with\ NBS * (\epsilon_{bioreactor} + \epsilon_{wetland} - \epsilon_{bioreactor} * \epsilon_{wetland}) * fraction\ of\ land\ drained\ by\ constructed\ wetland$ (Eq. 16)

To illustrate the use of Eq. 15, take the case in which all the arable land is drained to riparian buffers. In this case: $fraction\ of\ land\ with\ NBS = 1$ $fraction\ of\ land\ drained\ riparian\ buffers = 1$. Therefore, the riparian buffer removal capacity will be equal to the removal efficiency of riparian buffers.

As no accumulation is considered in Eq 12., the nitrogen mass flow to surface water can be estimated as follows :

$nitrogen\ leaching\ to\ surface\ water = nitrogen\ reaching\ NBS - removed\ nitrogen$ (Eq. 17)

Finally, nitrogen pollution is often reported in terms of concentration (c_{NO3}). This variable can be estimated as the ratio of Eq 17 and 8 in the simulation model.

$c_{NO3} = \frac{nitrate\ leaking\ to\ surface\ water}{surface\ water\ flow\ (volumetric)}$ (Eq. 18)

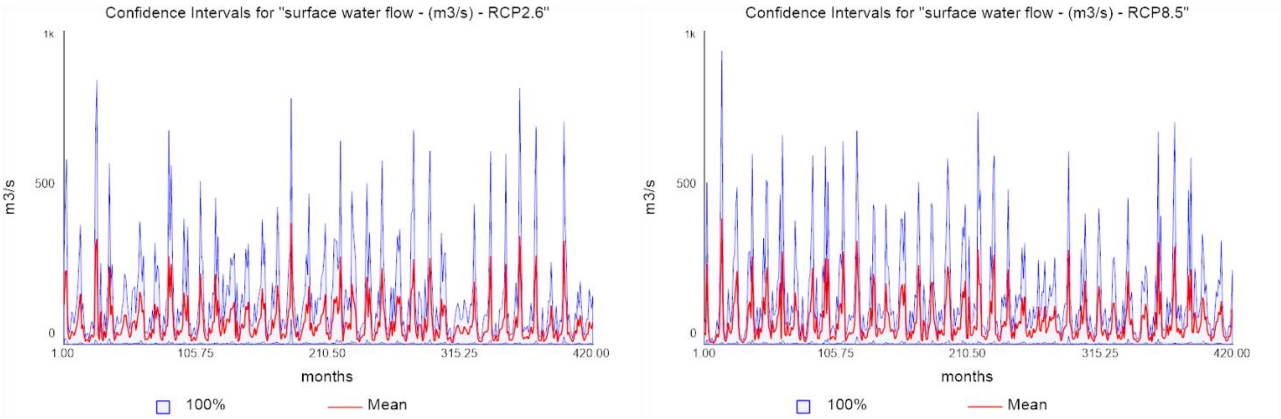

Figure A3. River flow under two different climate change scenarios

1.3.1 Behavioural validation

Table A3. Monthly water quality data at theLielupe's Kalnciems Station, Latvia (2015-2024).

Date	NO_3^- concentration (mg/L)	River flow (m^3/s)	NO_3^- flux (g/s)	NO_3^- flux (tonn/month)
14/01/2015	8.2	255.5	2107.5	5462.7
19/02/2015	6.4	78.6	502.4	1302.1
16/03/2015	6.3	124.7	782.1	2027.3
13/04/2015	6.0	114.4	689.8	1787.9
11/05/2015	2.3	53.3	124.7	323.2
08/06/2015	1.0	21.4	22.1	57.2
21/07/2015	0.0	25.7	1.0	2.7
18/08/2015	0.3	19.6	6.3	16.2
22/09/2015	0.4	15.3	5.5	14.3
19/10/2015	0.2	21.7	3.5	9.2
23/11/2015	0.5	16.3	7.5	19.5
14/12/2015	1.3	60.0	78.1	202.3
12/01/2016	3.1	12.8	39.9	103.5
18/04/2016	7.2	120.3	866.5	2246.0
05/05/2016	3.8	60.7	229.9	595.8
21/06/2016	0.5	7.4	3.7	9.6
12/07/2016	0.0	13.6	0.7	1.7
09/08/2016	0.1	12.0	1.0	2.6
27/09/2016	1.4	13.0	17.9	46.5
25/10/2016	0.9	23.1	19.8	51.4
28/11/2016	7.2	197.1	1419.3	3678.8
12/12/2016	10.3	253.2	2608.1	6760.1
05/01/2017	8.4	116.1	975.5	2528.6
08/02/2017	3.4	43.4	145.3	376.7
21/03/2017	6.2	219.0	1357.5	3518.8
04/04/2017	4.5	115.4	514.9	1334.6
17/05/2017	1.3	37.9	47.4	122.7
07/06/2017	0.3	51.1	17.4	45.0
25/07/2017	0.3	15.5	4.7	12.1

Date	NO_3^- concentration (mg/L)	River flow (m^3/s)	NO_3^- flux (g/s)	NO_3^- flux (tonn/month)
17/08/2017	0.1	8.1	0.4	1.1
12/09/2017	0.5	32.0	15.4	39.9
11/10/2017	1.3	87.7	114.1	295.6
06/11/2017	3.1	202.8	626.6	1624.2
04/12/2017	1.4	256.5	366.7	950.5
18/01/2018	1.8	53.0	93.2	241.6
19/02/2018	3.8	66.4	251.5	652.0
08/03/2018	1.8	15.6	27.5	71.4
11/04/2018	5.1	274.7	1403.6	3638.2
29/05/2018	0.2	38.9	7.0	18.2
07/06/2018	1.0	34.5	34.9	90.3
18/07/2018	0.2	14.9	2.8	7.1
07/08/2018	0.2	41.2	6.3	16.2
11/09/2018	0.0	18.6	0.1	0.3
22/10/2018	0.2	42.3	7.5	19.4
20/11/2018	0.3	32.1	9.6	24.9
06/12/2018	0.5	52.5	25.7	66.7
08/01/2019	1.3	70.8	92.0	238.5
26/02/2019	12.1	262.1	3171.9	8221.5
05/03/2019	8.4	124.0	1041.4	2699.4
01/04/2019	5.6	161.5	898.0	2327.6
02/05/2019	2.4	39.2	92.6	240.0
04/06/2019	0.1	12.2	1.2	3.1
02/07/2019	0.0	16.3	0.7	1.7
13/08/2019	0.1	18.8	2.5	6.5
14/10/2019	1.1	79.5	87.4	226.6
04/11/2019	0.6	28.4	18.1	47.0
02/12/2019	4.0	43.7	175.0	453.5
15/01/2020	10.3	139.3	1434.7	3718.7
25/02/2020	11.1	233.3	2589.5	6712.0
05/03/2020	9.4	222.5	2091.7	5421.6
07/04/2020	8.5	68.4	581.4	1506.9

Date	NO_3^- concentration (mg/L)	River flow (m^3/s)	NO_3^- flux (g/s)	NO_3^- flux (tonn/month)
14/05/2020	1.6	63.8	102.7	266.2
09/06/2020	0.4	18.8	7.7	20.0
07/07/2020	0.1	6.1	0.6	1.6
17/08/2020	0.0	16.9	0.1	0.3
17/09/2020	0.1	13.4	1.6	4.2
14/10/2020	0.1	39.0	5.7	14.8
11/11/2020	0.3	28.2	8.5	22.0
10/12/2020	4.0	49.5	199.3	516.7
12/01/2021	6.9	16.2	111.7	289.6
11/02/2021	11.2	67.8	759.8	1969.4
09/03/2021	14.9	234.9	3500.6	9073.6
14/04/2021	5.9	99.1	587.4	1522.6
18/05/2021	6.8	58.3	396.7	1028.2
09/06/2021	4.8	32.4	156.4	405.4
13/07/2021	0.3	33.0	8.2	21.4
18/08/2021	0.1	10.8	0.9	2.2
02/09/2021	0.2	42.7	6.5	16.9
14/10/2021	0.8	51.4	42.6	110.5
09/11/2021	2.9	189.6	555.4	1439.6
09/12/2021	9.1	83.4	758.9	1967.1
11/01/2022	13.8	445.5	6147.3	15933.9
23/02/2022	7.3	988.9	7218.7	18711.0
09/03/2022	6.9	248.3	1713.0	4440.2
13/04/2022	2.8	141.3	388.7	1007.4
18/05/2022	1.5	67.9	100.5	260.5
14/06/2022	3.4	39.3	135.3	350.7
27/07/2022	1.7	13.7	23.9	61.8
16/08/2022	2.2	47.4	103.3	267.7
13/09/2022	0.9	27.5	25.3	65.5
18/10/2022	1.1	32.1	35.6	92.2
09/11/2022	0.7	21.8	14.6	37.8
13/12/2022	1.7	17.0	29.3	76.0

Date	NO_3^- concentration (mg/L)	River flow (m^3/s)	NO_3^- flux (g/s)	NO_3^- flux (tonn/month)
18/01/2023	8.9	526.8	4688.2	12151.9
14/02/2023	8.0	192.6	1540.7	3993.5
16/03/2023	6.7	427.9	2866.9	7430.9
19/04/2023	5.0	105.6	528.1	1368.9
18/05/2023	1.1	15.5	16.6	43.0
27/06/2023	0.1	7.1	1.0	2.7
27/07/2023	0.1	9.0	0.6	1.6
15/08/2023	0.1	32.6	4.8	12.4
20/09/2023	1.8	34.9	62.1	161.1
05/10/2023	1.4	47.2	64.7	167.7
14/12/2023	4.0	84.1	336.4	872.1
16/01/2024	7.0	71.1	497.4	1289.4
06/02/2024	4.9	432.1	2108.5	5465.3
12/03/2024	5.0	192.2	961.1	2491.2
11/04/2024	2.4	230.9	551.9	1430.5
08/05/2024	2.8	83.8	231.3	599.4
06/06/2024	0.8	7.7	6.1	15.9
10/07/2024	0.1	14.2	0.7	1.9
08/08/2024	0.4	37.5	15.0	38.9
05/09/2024	0.5	16.8	7.9	20.5
07/10/2024	1.3	28.3	37.6	97.5
06/11/2024	0.5	32.7	17.3	44.9
10/12/2024	2.0	19.2	38.1	98.8

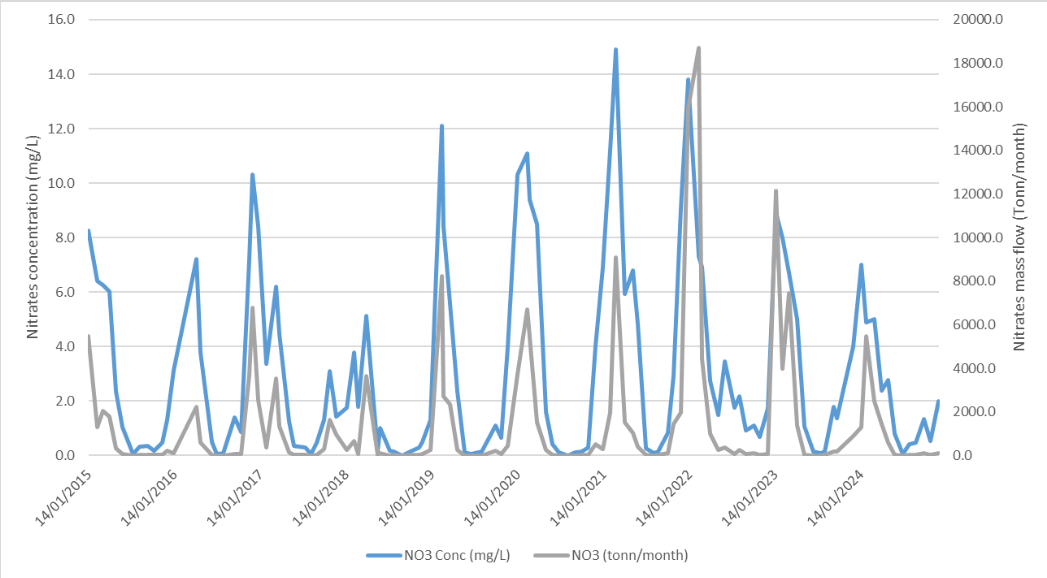

Figure A4. Observed water quality parameters 2015-2024.

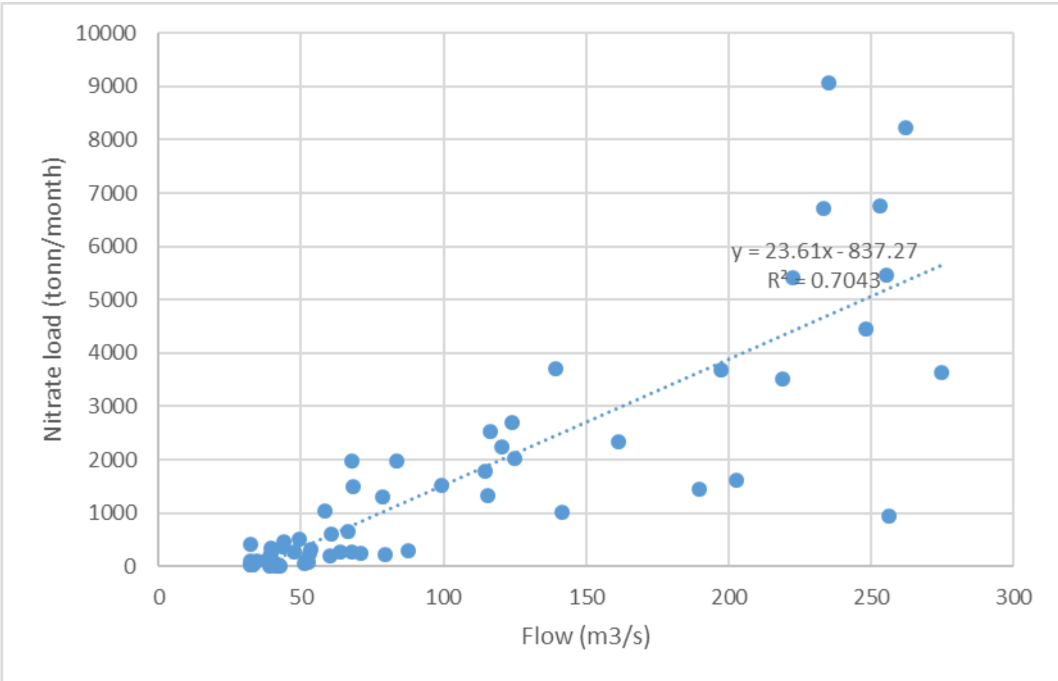

Figure A5. Scatter plot Nitrate load vs. Flow – Q>30m3/s

Table A4. Regression analysis - Q>30m3/s

SUMMARY OUTPUT

Regression Statistics	
Multiple R	0.839209038
R Square	0.70427181
Adjusted R Square	0.699083596
Standard Error	1188.003223
Observations	59

ANOVA

	df	*SS*	*MS*	*F*	*Significance F*
Regression	1	1.92E+08	1.92E+08	135.7446	1.04E-16
Residual	57	80447044	1411352		
Total	58	2.72E+08			

	Coefficients	*Standard Error*	*t Stat*	*P-value*	*Lower 95%*	*Upper 95%*	*Lower 95.0%*	*Upper 95.0%*
Intercept	-837.2718887	262.6533	-3.18775	0.002329	-1363.23	-311.318	-1363.23	-311.318
Flow (m3/s)	23.6101995	2.026462	11.65095	1.04E-16	19.55228	27.66812	19.55228	27.66812

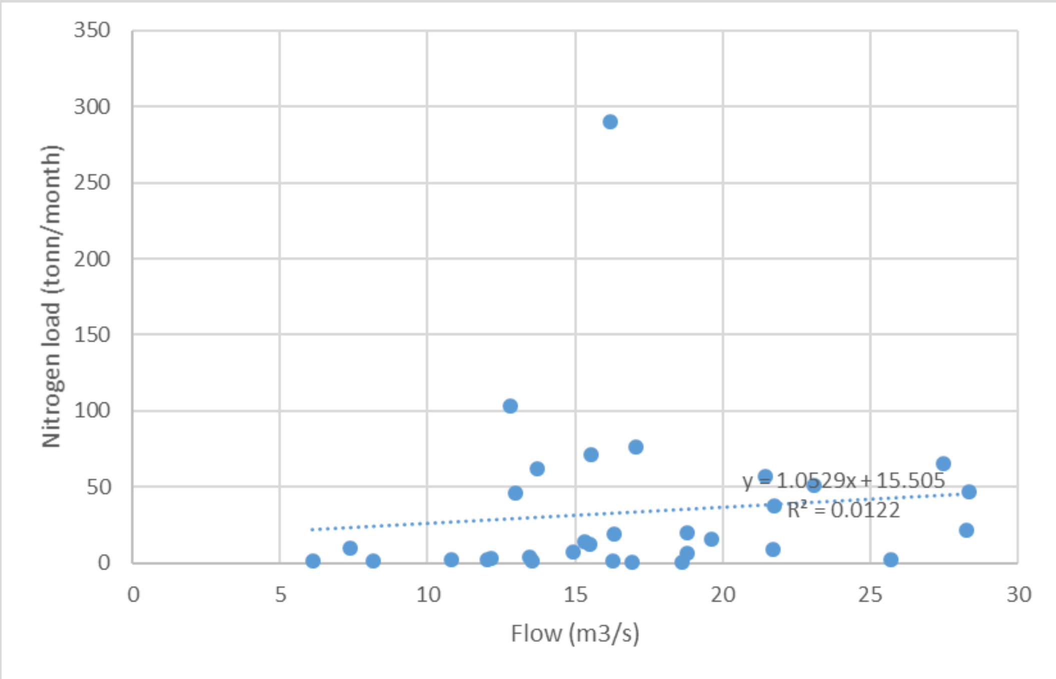

Figure A6. Scatter plot Nitrate load vs. Flow – Q<30m3/s

Table A5. Regression analysis – Q<30m3/s

SUMMARY OUTPUT

Regression Statistics	
Multiple R	0.545726
R Square	0.297817
Adjusted R Square	0.276539
Standard Error	365.6553
Observations	35

ANOVA

	df	SS	MS	F	Significance F
Regression	1	1871361	1871361	13.99632	0.000697
Residual	33	4412226	133703.8		
Total	34	6283587			

	Coefficients	Standard Error	t Stat	P-value	Lower 95%	Upper 95%	Lower 95.0%	Upper 95.0%
Intercept	30.9843	102.3047	0.302863	0.763895	-177.156	239.1248	-177.156	239.1248
Flow (m3/s)	4.786908	1.279523	3.741165	0.000697	2.183699	7.390118	2.183699	7.390118

Table A6. Regression analysis - Observed measurements vs. Model output – nitrates concentration (mg/L)

SUMMARY OUTPUT

Regression Statistics	
Multiple R	0.748056

R Square	0.5595 87
Adjusted R Square	0.5386 15
Standard Error	1.8939 75
Observations	23

ANOVA

	df	*SS*	*MS*	*F*	*Significa nce F*
Regression	1	95.71407	95.71 407	26.68 254	4.06E-05
Residual	21	75.32999	3.587 142		
Total	22	171.0441			

	Coeffic ients	*Standard Error*	*t Stat*	*P-value*	*Lower 95%*	*Upper 95%*	*Lower 95.0%*	*Upper 95.0%*
Intercept	0.1726 89	0.646492	0.267 117	0.791 985	-1.17177	1.5171 43	- 1.17177	1.51714 3
median	0.9253 02	0.179131	5.165 514	4.06E -05	0.552779	1.2978 24	0.55277 9	1.29782 4

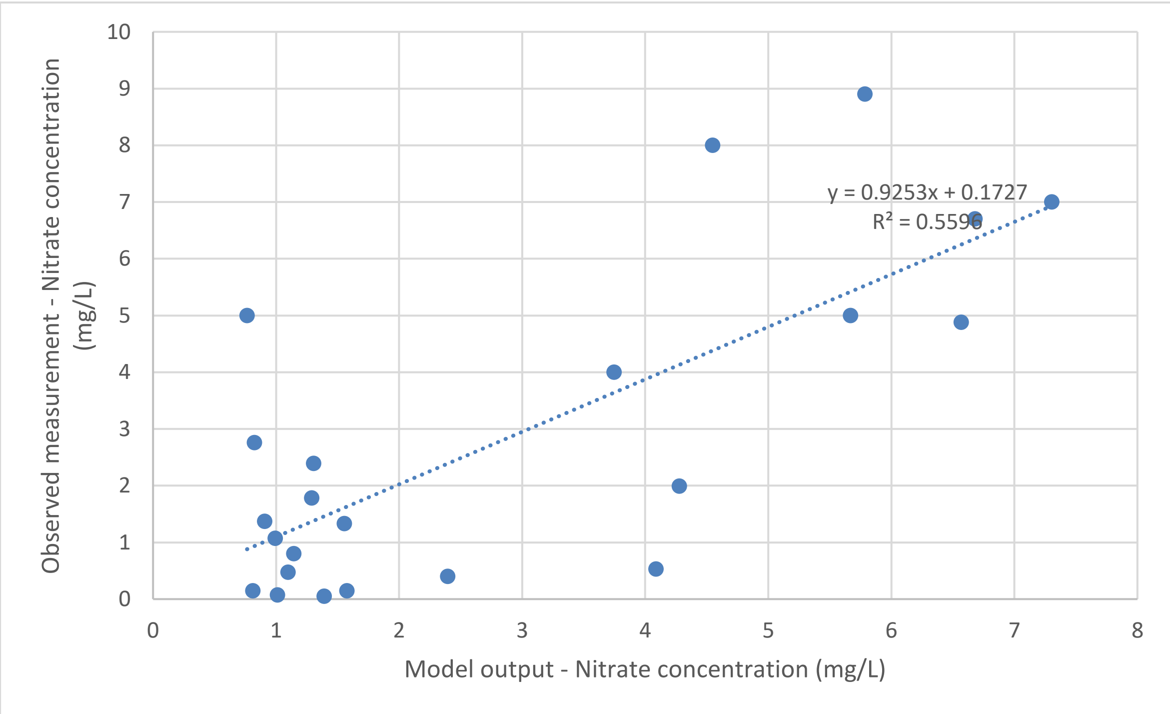

Figure A7. Observed measurements vs. Model output – nitrates concentration (mg/L)

Table A7. Percentiles - Q<30m3/s

Point	**NO_3^- *flux* *(tonn/month)***	***Rank***	***Percent***
16	289.6285553	1	100.00%
7	103.4601984	2	96.70%
20	75.9684096	3	93.50%
15	71.36484856	4	90.30%
30	65.537856	5	87.00%
11	61.82568	6	83.80%
25	57.2025744	7	80.60%
28	51.44214528	8	77.40%
32	47.0403072	9	74.10%
8	46.47748526	10	70.90%
27	37.7950752	11	67.70%
31	21.96558	12	64.50%
22	19.9532322	13	61.20%
18	19.464624	14	58.00%
24	16.21073088	15	54.80%
13	14.276736	16	51.60%
14	12.05465143	17	48.30%
2	9.56016	18	45.10%
26	9.177913108	19	41.90%
12	7.14996288	20	38.70%
23	6.526560295	21	35.40%
9	4.1798592	22	32.20%
6	3.05983008	23	29.00%
29	2.6635392	24	25.80%
5	2.616892625	25	22.50%

Point	**NO_3^-** ***flux (tonn/month)***	***Rank***	***Percent***
4	2.21012928	26	19.30%
17	1.728146215	27	16.10%
10	1.68684768	28	12.90%
1	1.567480098	29	9.60%
3	1.097458899	30	6.40%
21	0.28969488	31	3.20%
19	0.262960992	32	0.00%

1.4 Ecosystems sector

Our approach to the *Ecosystems* as a sector is based on land use. Forests and grasslands are the two main 'natural' land-use types in the basin, and in the wider Baltic region (Eriksson & Cousins, 2014). A widely used indicator for comparing different types of biomes in terms of vegetation density is the *carbon mass in vegetation* ($cveg$). Long-term estimates of the variable were done using six impact-models from ISIMIP2b, as described by Trabucco et al. (2024). This variable was estimated for both land use types, as well as for arable land at the river basin level. Figures 7 and 8 summarise the time series confidence interval for *carbon mass in vegetation* in two climate change scenarios (i.e. RCP 2.6 and RCP 8.5). By multiplying the *carbon mass in vegetation* and its corresponding land use area, it is possible to obtain an estimate of the *total carbon mass in vegetation* ($tcveg$) by land use type, as presented in Eq. 19 (see Figures 9 and 10).

$$tcveg_{land\ use\ type} = cveg_{land\ use\ type} * area_{land\ use\ type} \quad \text{(Eq. 19)}$$

In turn, the sum of $tcveg_{land\ use\ type}$ for every land use type is a proxy for the total carbon mass in vegetation for the basin (see Eq. 20).

$$tcveg_{river\ basin} = \sum_{land\ use\ types} cveg_{land\ use\ type} * area_{land\ use\ type} \quad \text{(Eq. 20)}$$

Another ecosystems-related variable that was considered is the species richness (sr) for certain animal classes. Species richness for mammals, birds, and amphibians time-series were obtained from the Globio 4 biodiversity model (Schipper et al., 2020) as further described by Trabucco et al. (2024) (see Table 8). Respective species richness time series are available for RCP 2.6 and 8.5 climate change scenarios (see Figures 11 and 12).

Regression analyses show that $tcveg_{river\ basin}$ and sr are significantly correlated for mammals and birds in the long-term future in two RCP scenarios (see Table 9). Building on this finding, the proposed model explicitly accounts for biodiversity variations as a function of carbon mass in vegetation as presented in Eq. 21. The estimation of carbon mass in vegetation, in turn, is dynamic and a function of the areas of different land uses (see Eq. 20).

$$sr_{animal\ class,RCP} = \beta_{cveg-animal\ class,RCP} * tcveg_{river\ basin,RCP} + c, \forall\ animal\ class\ , RCPs \quad \text{(Eq. 21)}$$

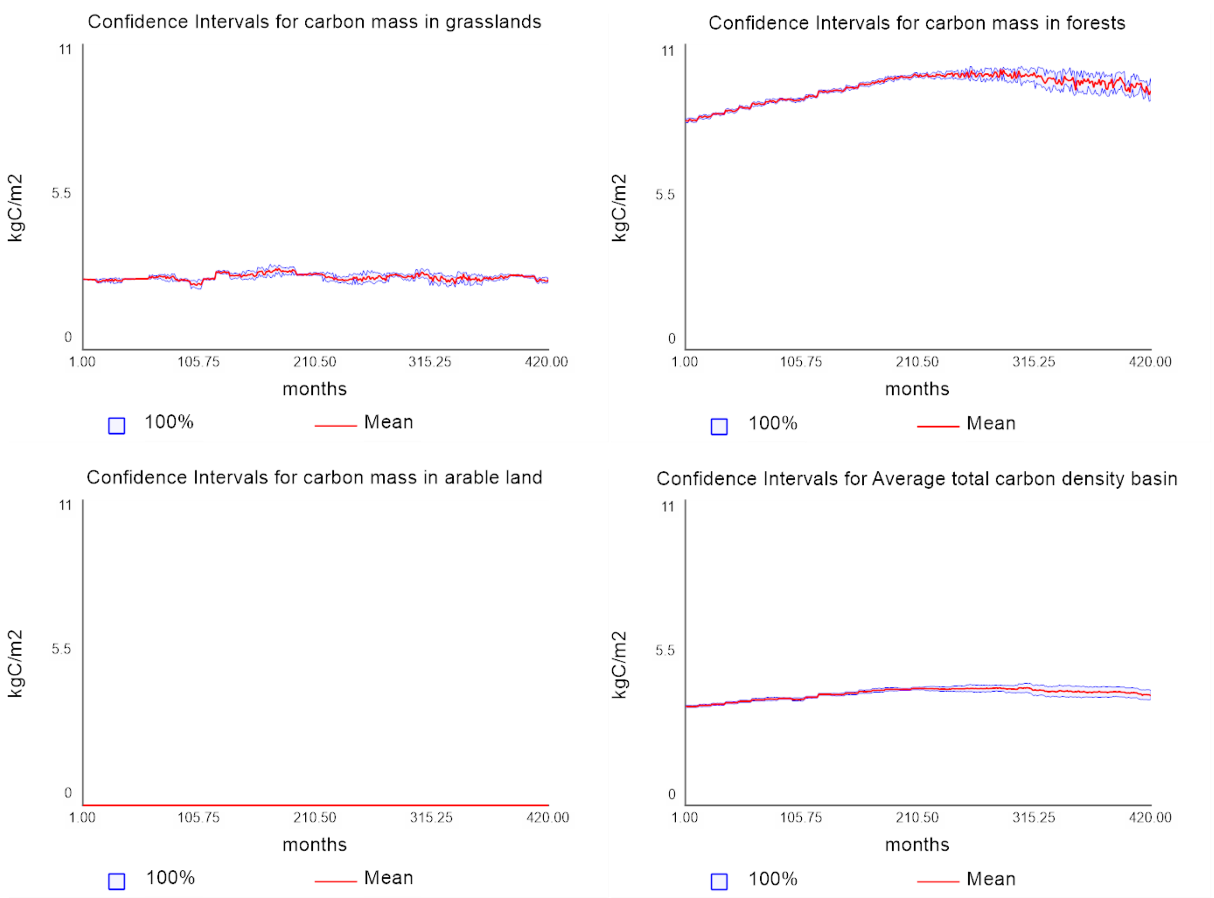

Figure A8. Cveg by land use type – RCP 2.6

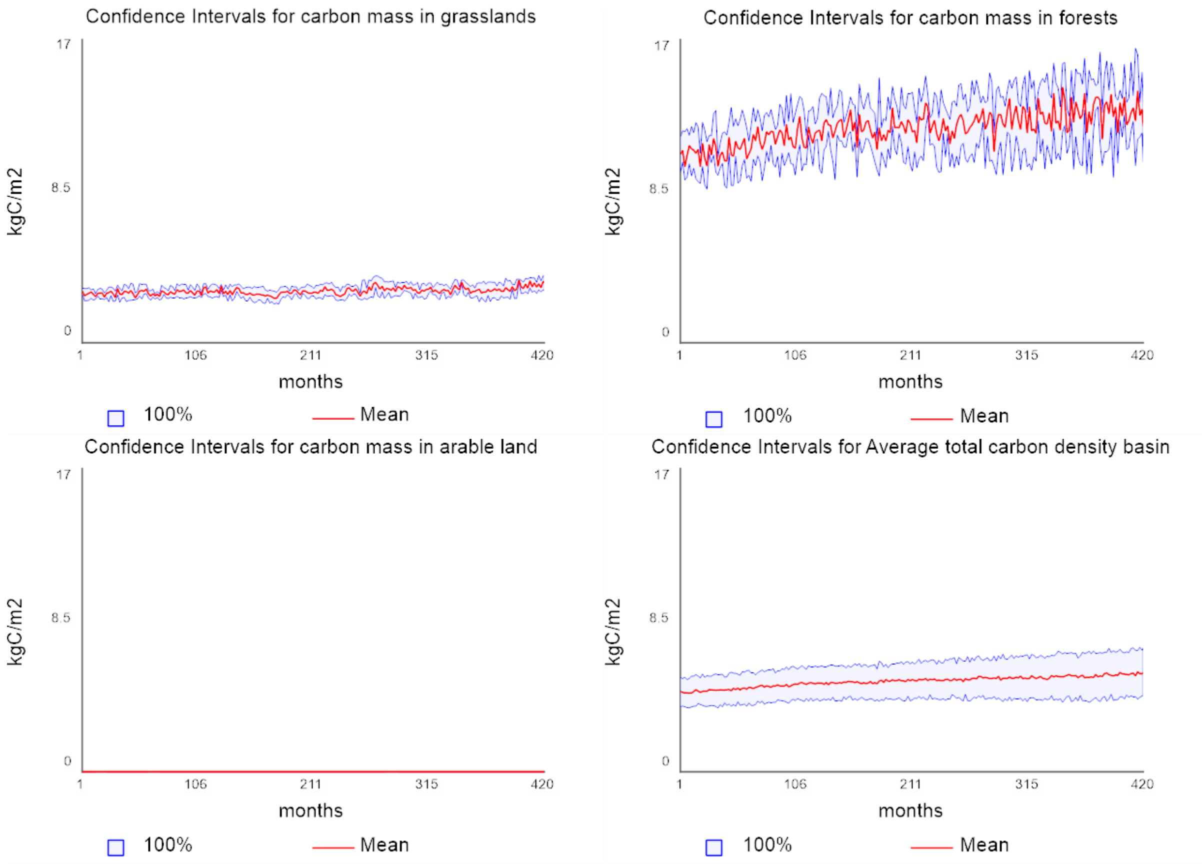

Figure A9.Cveg by land use type – RCP 8.5

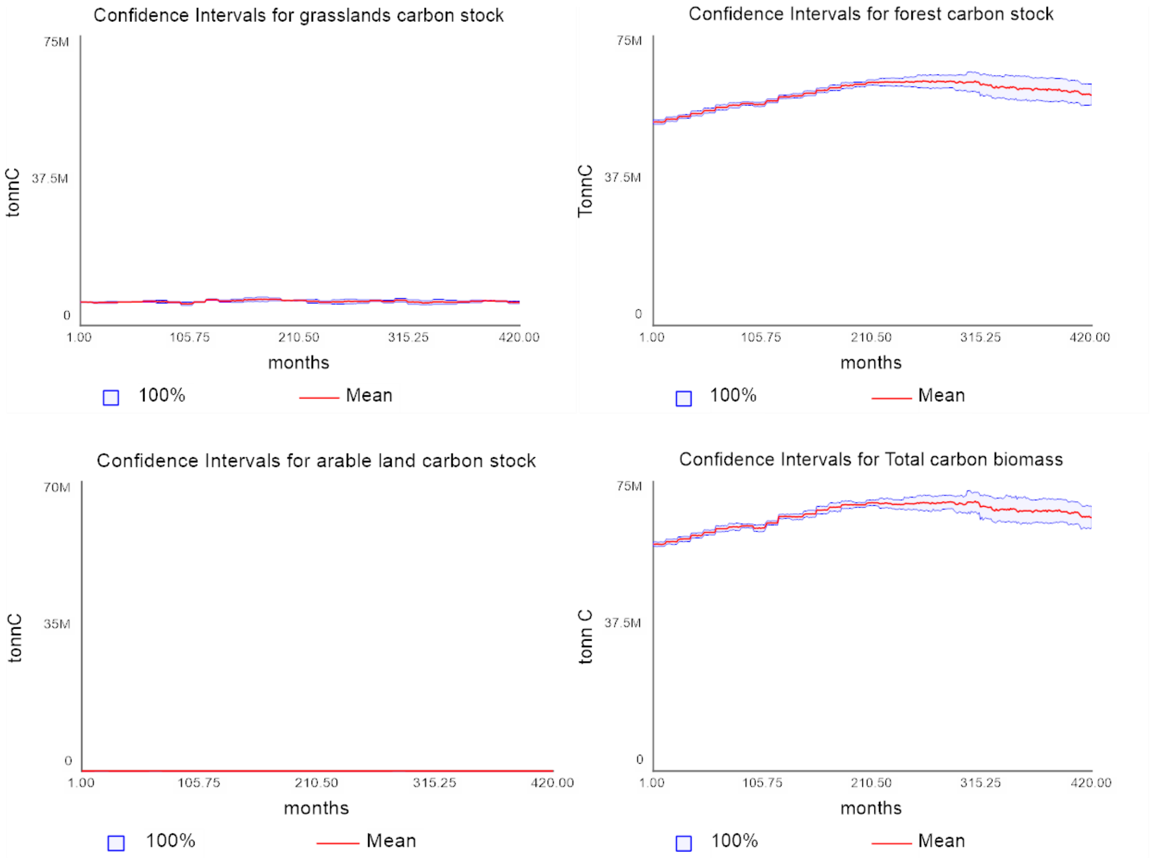

Figure A10. Tcveg by land use type – RCP 2.6

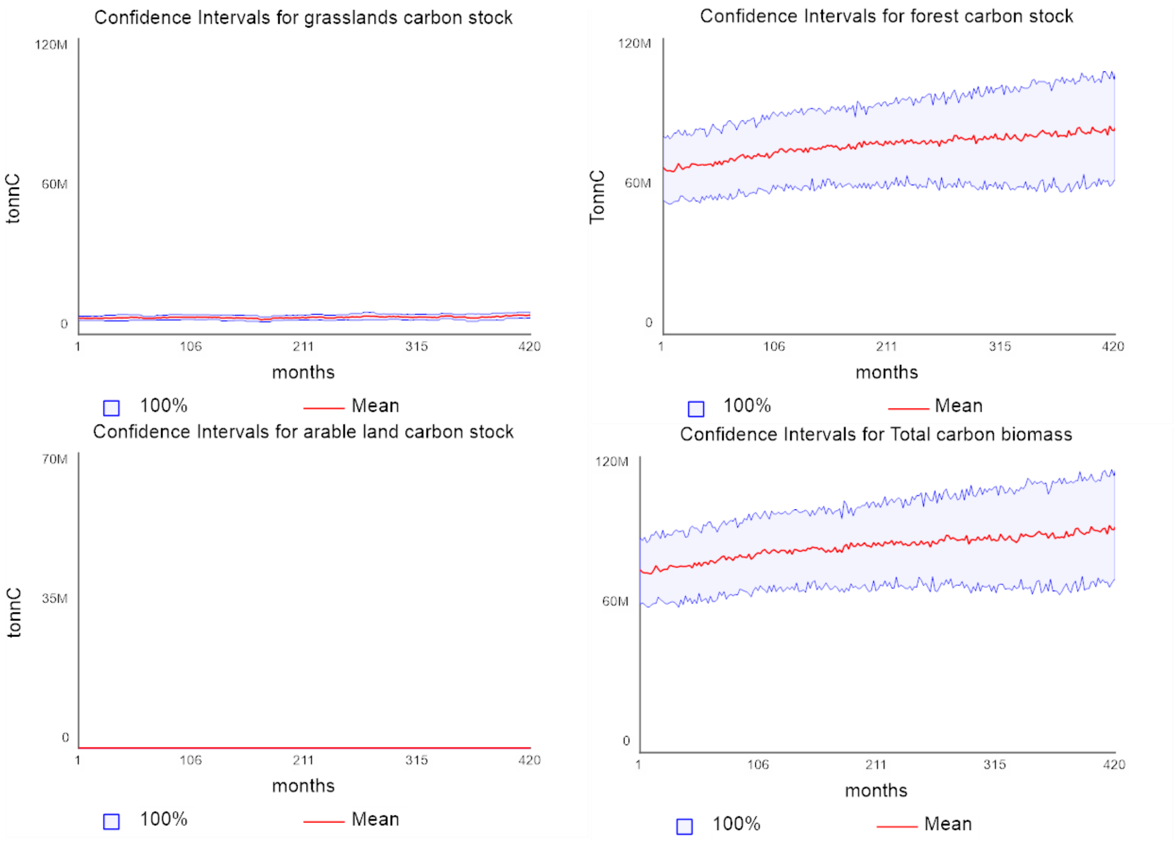

Figure A11. Tcveg by land use type – RCP 8.5

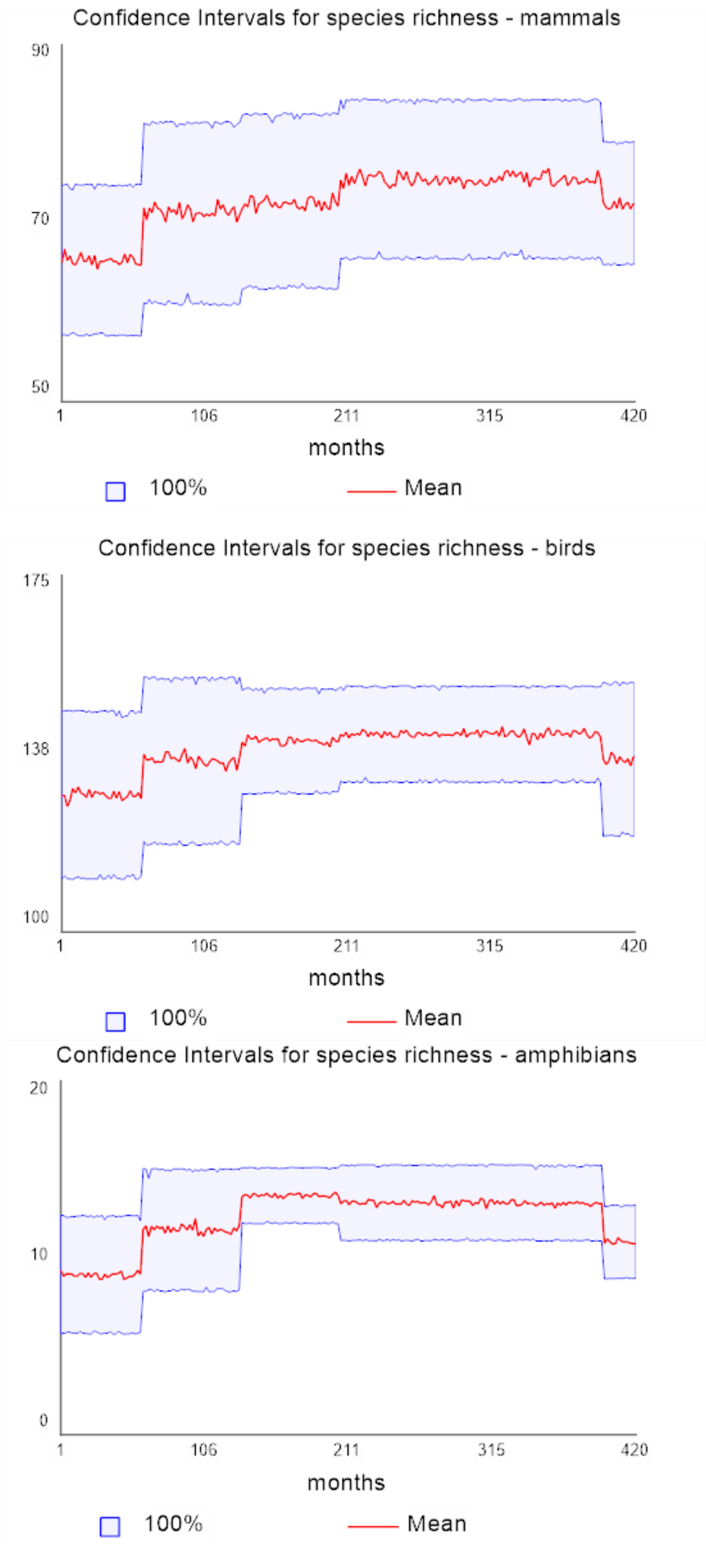

Figure A12. SR - RCP 2.6

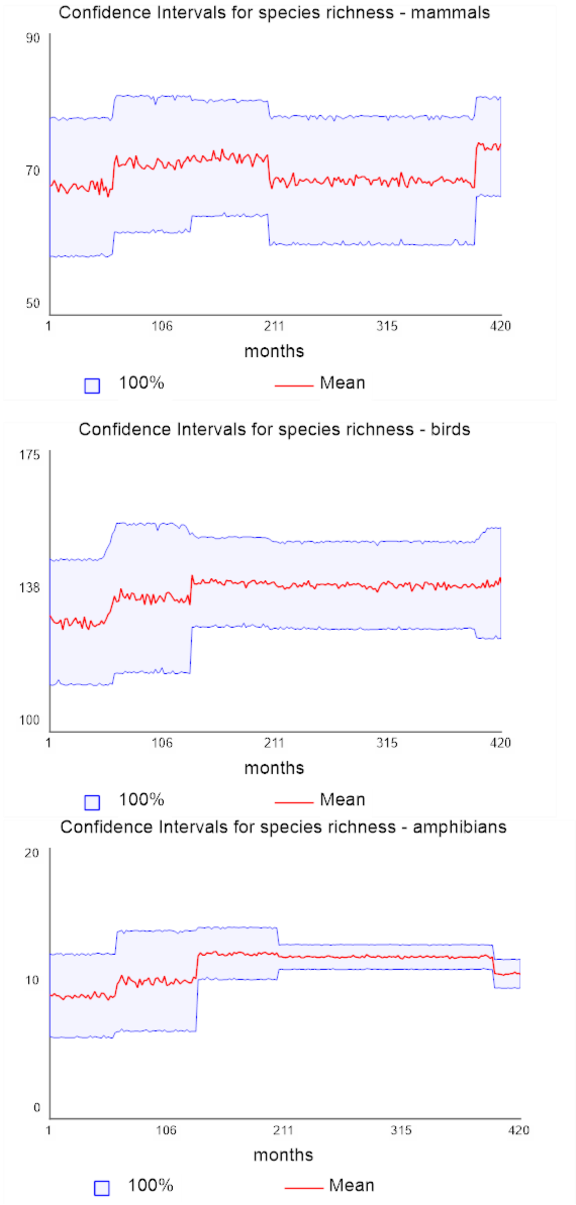

Figure A13. SR - RCP 8.5

Table A8. Species richness and carbon mass in vegetation forecasts (RCP 2.6 and 8.5)

Year	Species richness - amphibians (# of species) - RCP2.6	Species richness - amphibians (# of species) - RCP8.5	Species richness - birds (# of species) - RCP2.6	Species richness - birds (# of species) - RCP8.5	Species richness - mammals (# of species) - RCP2.6	Species richness - mammals (# of species) - RCP8.5	Total carbon mass in vegetation (kg C/m2) - RCP2.6	Total carbon mass in vegetation (kg C/m2) - RCP8.5
2010	9.15	8.67	127.24	126.99	62.95	66.45	3.78	4.72
2020	11.13	10.54	139.06	139.58	71.23	71.06	4.02	4.92
2026	13.25	12.59	139.35	140.61	73.29	73.24	4.20	5.21
2032	13.67	11.99	141.90	140.64	75.56	70.49	4.44	5.38
2048	10.81	10.54	138.93	145.61	72.11	73.61	4.30	5.60
2050	12.38	12.13	141.10	140.50	72.68	72.68	4.26	5.64
2052	10.19	10.45	145.66	149.87	72.58	73.32	4.32	5.72
2056	10.38	13.09	146.68	140.00	71.07	75.41	4.39	5.84

Table A9. Summary of regression models - Species richness vs. Carbon mass in vegetation

Output variable	Predictor	Coefficient	P-value (coef)	Intercept	P-value (intercept)	R2
Species richness - amphibians (# of species) - RCP2.6	Total carbon mass in vegetation (kg C/m2) - RCP2.6	3.67	0.20	-4.09	0.72	0.25
Species richness - amphibians (# of species) - RCP8.5	Total carbon mass in vegetation (kg C/m2) - RCP8.5	2.19	0.11	-0.55	0.93	0.37
Species richness - birds (# of species) - RCP2.6	Total carbon mass in vegetation (kg C/m2) - RCP2.6	24.08	0.00	38.53	0.14	0.77
Species richness - birds (# of species) - RCP8.5	Total carbon mass in vegetation (kg C/m2) - RCP8.5	11.96	0.04	76.15	0.02	0.54
Species richness - mammals (# of species) - RCP2.6	Total carbon mass in vegetation (kg C/m2) - RCP2.6	14.66	0.01	9.64	0.55	0.74
Species richness - mammals (# of species) - RCP8.5	Total carbon mass in vegetation (kg C/m2) - RCP8.5	5.76	0.01	41.05	0.00	0.72

Table A10. Summary of stochastic parameters with a time series function

Sector	Parameters	Probability distribution	Unit
Ecosystems	Species richness amphibians - lower bound (min_amph_sr)	Deterministic (Time series)	Number of species
Ecosystems	Species richness amphibians - upper bound (max_amph_sr)	Deterministic (Time series)	Number of species
Ecosystems	Species richness amphibians	UNIFORM (min_amph_sr, max_amph_sr)	Number of species
Ecosystems	Species richness mammals - lower bound (min_mammals_sr)	Deterministic (Time series)	Number of species
Ecosystems	Species richness mammals - upper bound (max_mammals_sr)	Deterministic (Time series)	Number of species
Ecosystems	Species richness mammals	UNIFORM (min_mammals_sr, max_mammals_sr)	Number of species
Ecosystems	Species richness birds - lower bound (min_birds_sr)	Deterministic (Time series)	Number of species
Ecosystems	Species richness birds - upper bound (max_birds_sr)	Deterministic (Time series)	Number of species
Ecosystems	Species richness birds	UNIFORM (min_birds_sr, max_birds_sr)	Number of species
Ecosystems	Carbon mass in vegetation - decidious forest - lower bound (min_cveg_d_forest)	Deterministic (Time series)	kgC/m2
Ecosystems	Carbon mass in vegetation - decidious forest - upper bound (max_cveg_d_forest)	Deterministic (Time series)	kgC/m2
Ecosystems	Carbon mass in vegetation - decidious forest	UNIFORM (min_cveg_d_forest, max_cveg_d_forest)	kgC/m2
Ecosystems	Carbon mass in vegetation - evergreen forest - lower bound (min_cveg_e_forest)	Deterministic (Time series)	kgC/m2
Ecosystems	Carbon mass in vegetation - evergreen forest - upper bound (max_cveg_e_forest)	Deterministic (Time series)	kgC/m2
Ecosystems	Carbon mass in vegetation - evergreen forest	UNIFORM (min_cveg_e_forest, max_cveg_e_forest)	kgC/m2
Ecosystems	Carbon mass in vegetation - grasslands - lower bound (min_cveg_grasslands)	Deterministic (Time series)	kgC/m2

Sector	Parameters	Probability distribution	Unit
Ecosystems	Carbon mass in vegetation - grasslands - upper bound (max_cveg_grasslands)	Deterministic (Time series)	kgC/m2
Ecosystems	Carbon mass in vegetation - grasslands	UNIFORM (min_cveg_grasslands, max_cveg_grasslands)	kgC/m2
Food	Crop yield - field peas - lower bound (field_peas_min)	Deterministic (Time series)	Tonnes/ha*month
Food	Crop yield - field peas - upeer bound (field_peas_max)	Deterministic (Time series)	Tonnes/ha*month
Food	Crop yield - field peas	UNIFORM (field_peas_min, field_peas_max)	Tonnes/ha*month
Food	Crop yield - maize - lower bound (maize_min)	Deterministic (Time series)	Tonnes/ha*month
Food	Crop yield - maize - upeer bound (maize_max)	Deterministic (Time series)	Tonnes/ha*month
Food	Crop yield - maize	UNIFORM (maize_min, maize_max)	Tonnes/ha*month
Food	Crop yield - rapeseed - lower bound (rapeseed_min)	Deterministic (Time series)	Tonnes/ha*month
Food	Crop yield - rapeseed - upeer bound (rapeseed_max)	Deterministic (Time series)	Tonnes/ha*month
Food	Crop yield - rapeseed	UNIFORM (rapeseed_min, rapeseed_max)	Tonnes/ha*month
Food	Crop yield - summer wheat - lower bound (summer_wheat_min)	Deterministic (Time series)	Tonnes/ha*month
Food	Crop yield - summer wheat - upeer bound (summer_wheat_max)	Deterministic (Time series)	Tonnes/ha*month
Food	Crop yield - summer wheat	UNIFORM (summer_wheat_min, summer_wheat_max)	Tonnes/ha*month
Food	Crop yield - winter wheat - lower bound (winter_wheat_min)	Deterministic (Time series)	Tonnes/ha*month
Food	Crop yield - winter wheat - upeer bound (winter_wheat_max)	Deterministic (Time series)	Tonnes/ha*month
Food	Crop yield - winter wheat	UNIFORM (winter_wheat_min, winter_wheat_max)	Tonnes/ha*month
Water	Minimum surface water flow (flow_min)	Deterministic (Time series)	mm/s

Sector	Parameters	Probability distribution	Unit
Water	Median surface water flow (flow_median)	Deterministic (Time series)	mm/s
Water	Maximum surface water flow (flow_maximum)	Deterministic (Time series)	mm/s
Water	Surface water flow	TRIANGULAR (flow_min, flow_med, flow_max))	mm/s

Table A11. Summary of stochastic parameters with a probability distribution function

Sector	Stochastic parameters	Median value	Probability distribution and parameters	Units	Sources	Additional Comments
Food	Nitrogen content in manure	6.25	UNIFORM (1.67, 8.12)	kg/(head *month)	Aplocina et al. (2016)	
Food	Relative yield ratio field peas (Organic/Conventional farming practice)	0.72	TRIANGULAR(0.53, 0.72, 0.92)	-	Tuomisto et al. (2012)	field peas not available in the meta-analysis, we assumed their ranges as the average of the rest of the crops
Food	Relative yield ratio maize (Organic/Conventional farming practice)	0.67	TRIANGULAR(0.48, 0.67, 0.83)	-	Tuomisto et al. (2012)	
Food	Relative yield ratio rapeseed (Organic/Conventional farming practice)	0.82	TRIANGULAR(0.53, 0.82, 1.11)	-	Tuomisto et al. (2012)	
Food	Relative yield ratio summer wheat (Organic/Conventional farming practice)	0.78	TRIANGULAR(0.7, 0.78, 0.87)	-	Tuomisto et al. (2012)	
Food	Relative yield ratio winter wheat (Organic/Conventional farming practice)	0.62	TRIANGULAR(0.41, 0.62, 0.86)	-	Tuomisto et al. (2012)	
Nature-based solutions	Nitrate leaching rate	0.00275	TRIANGULAR (0.00035, 0.00275, 0.00525)	(TonnN)/(ha*month)	Zhou (2014)	Ranges for wheat and maize (the dominant crops in the basin)
Nature-based solutions	Bioreactor nitrogen removal efficiency	0.56	UNIFORM (0.12, 1)	-	Christianson et al. (2012)	

Sector	Stochastic parameters	Median value	Probability distribution and parameters	Units	Sources	Additional Comments
Nature-based solutions	Constructed wetland nitrogen removal efficiency	0.51	UNIFORM(0.25, 0.78)	-	Cassman et al. (2002)	
Nature-based solutions	Riparian buffers nitrogen removal efficiency	0.48	TRIANGULAR(0.34, 0.48, 0.81)	-	Zhu et al. (2020)	
Nature-based solutions	Organic farming nitrogen removal efficiency	0.31	TRIANGULAR(0, 0.31, 0.45)		Tuomisto et al. (2012)	
Population	Domestic per capita nitrogen generation rate	0.21	UNIFORM (0.09, 0.32)	kg/(cap* month)	Mesdaghinia et al. (2015)	

2. RESULTS' SUPPLEMENTARY TABLES.

Table 0.12. Key variables' output for each combination of transboundary cooperation and policy ambition.

Level of transboundary cooperation	Variables	Perc.	Level of policy ambition: Low 2020	Low 2030	Low 2040	Low 2050	Medium 2020	Medium 2030	Medium 2040	Medium 2050	High 2020	High 2030	High 2040	High 2050
Bilateral	Total cereal production	5%	117203	102165	105261	103639	110455	102739	98354	91179	110126	94867	90440	91564
Bilateral	Total cereal production	50%	189732	168269	167192	176365	187817	163328	155962	164432	179463	152029	144799	148196
Bilateral	Total cereal production	95%	260483	236131	227114	253546	259741	223890	214464	236809	250088	214950	196551	210956
Bilateral	Total carbon mass in vegetation	5%	70831229	79190132	79025834	74378517	70889293	79401726	79223606	74470819	70947215	79481893	79391157	74688053
Bilateral	Total carbon mass in vegetation	50%	71450116	80238979	81147000	77253325	71429752	80407780	81340917	77321120	71512845	80513911	81446461	77592502
Bilateral	Total carbon mass in vegetation	95%	71990555	81349632	83128016	80260945	72038127	81469844	83360560	80487838	72087657	81600643	83572630	80594834
Bilateral	Nitrogen concentration	5%	0.5	0.7	0.8	0.3	0.5	0.6	0.6	0.2	0.4	0.6	0.4	0.2
Bilateral	Nitrogen concentration	50%	5.3	5.5	4.2	3.0	5.1	4.8	3.3	2.5	4.8	4.2	2.3	1.7
Bilateral	Nitrogen concentration	95%	8.0	7.7	6.5	5.7	7.7	7.0	5.4	4.6	7.2	6.0	4.2	3.5
Unilateral	Total cereal production	5%	113647	107926	111262	107738	116303	110963	106952	102112	121430	109685	103979	103157
Unilateral	Total cereal production	50%	189283	179255	174668	188296	190780	176304	169956	181939	190407	171805	167383	176535
Unilateral	Total cereal production	95%	265547	248705	239957	266283	263551	244627	237046	263611	266690	238126	229738	252731
Unilateral		5%	70483464	78388215	78243941	73457261	70511273	78456308	78284785	73584442	70533538	78485967	78289125	73664921

	Variables	Perc.	Low				Medium				High			
			2020	2030	2040	2050	2020	2030	2040	2050	2020	2030	2040	2050
Unilateral (Latvia)	Total carbon mass in vegetation	50%	71068033	79356198	80275498	76414965	71095658	79444125	80319259	76511691	71118243	79467091	80462270	76814349
		95%	71645801	80365356	82314020	79419221	71656052	80427446	82309314	79413313	71669537	80503802	82492312	79588495
	Nitrogen concentration	5%	0.6	0.7	0.8	0.3	0.6	0.7	0.7	0.3	0.5	0.6	0.7	0.3
		50%	5.4	5.8	4.6	3.4	5.2	5.6	4.2	3.1	5.1	5.1	3.7	2.7
		95%	8.0	8.1	7.1	6.3	7.9	7.8	6.5	5.6	7.7	7.3	5.9	5.1
	Total cereal production	5%	116151	111754	113062	106816	120604	110523	112162	108944	119642	111194	108192	103342
		50%	191431	178462	179244	189430	191259	180459	176789	189281	193092	177206	176572	183443
		95%	268538	247020	242019	265960	263195	243418	238113	266969	261119	241312	238304	263008
	Total carbon mass in vegetation	5%	70402778	78059318	77868457	73208855	70422567	78083020	77964042	73222997	70419168	78176867	78099962	73422128
		50%	70987118	79049984	79901587	76070733	70983598	79120705	80011463	76055377	70986492	79110677	80077273	76339188
		95%	71559002	80064104	82072188	79120930	71555929	80068685	82085722	79146806	71545459	80130839	82153115	79377411
	Nitrogen concentration	5%	0.5	0.7	0.9	0.3	0.4	0.6	0.6	0.3	0.4	0.7	0.6	0.3
		50%	5.3	5.8	4.8	3.6	5.2	5.6	4.4	3.3	5.3	5.5	4.0	2.8
		95%	8.0	8.2	7.2	6.5	8.0	7.9	6.9	5.9	7.7	7.6	6.4	5.6

Level of policy ambition: Low, Medium, High.

Table A13. Open policy exploration overall results

Variables	Perc.	2020	2030	2040	2050
Total cereal production	5%	118702	108973	108285	102622
	50%	189492	179069	172194	184072
	95%	263280	245332	236448	258040
Total carbon mass in vegetation	5%	70242384	77457946	77570021	73007056
	50%	71059329	79254432	80331163	76345749
	95%	71928033	81062159	82833372	79849808
Nitrogen concentration	5%	0.5	0.6	0.7	0.2
	50%	5.2	5.2	3.8	2.7
	95%	7.8	7.5	6.3	5.5

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APPENDIX B

Supplementary information: 'Opening the black box: tracking model evolution and learning in environmental participatory modelling'

Table B1. Summary of the workshops in the participatory modelling process in the Lielupe case study (NEXOGENESIS project)

Workshop	**1**	**2**	**3**	**4**	**5**	**6**
Date	10/02/2022	02/11/2022	15/06/2023	06/02/2024	02/10/2024	27/05/2025
Location	Online	Riga, LV	Vilnius, LT	Riga, LV	Riga, LV	Riga, LV
Phase of the participatory modelling cycle	Modelling foundations	Modelling foundations	Modelling foundations	Model formulation and confidence building	Model use and policy evaluation	Model use and policy evaluation
Phase of the stakeholder engagement approach	Co-exploration	Co-exploration	Co-design	Co-design	Co-development	Co-development
Number of participants	10	10	10	18	11	17
Workshop format	Online	In person	In person	In person	In person	In person
Workshop global purpose	Identification of main nexus issues in the basin	Discussion about the current state of the basin and the required policies to improve it	Nexus policies prioritisation for the basin	Presentation of preliminary results of a simulation model with policies for the basin	Stakeholder interaction with a web-based decision support system	Discussing the practical implications of policy alternatives in the context of the local/transboundary governance roadmap

Workshop	1	2	3	4	5	6
Facilitation approach	Small group discussions	Plenary discussion Collective brainstorm on policy alternatives for Nexus sectors	Plenary discussion World Café (WEFE sectors) on policy instruments Dot voting on policies prioritisation per WEFE sector	Plenary discussion about preliminary results of the model Q&A session - modelling features, assumption and limitations Feedback (MentiMeter) on tool functionalities	Aided testing of the tool in groups (+task) Feedback (plenary) to improve the model and tool functionalities	Plenary discussion on policy package validation Dot voting on activities reflection for priorities and challenges to reach the policy goals
Driving questions	Which are the main Nexus issues in the basin? How do current policies affect Nexus interlinkages?	Which policies are needed in the basin?	How to improve the current river basin situation?	Is this model useful to understand the Nexus issues in the basin? How do you prefer to interact with a simulation model of the river basin?	Is the tool operational to identify the “must have” policies for achieving the policy goals?	How achieving the policy goals are supported by set of activities and policy instruments to be implemented?
Inputs (modelling cycle tools)	Early conceptual map	Early conceptual map	Conceptual model Draft policies by sector	Simulation model results	Decision support system (with the model in the background)	A set of policy tools and their expected (modelled) performance

Workshop	1	2	3	4	5	6
Outputs (modelling cycle tools)	Identification of the key Nexus issues in the basin	Draft CLD	Prioritised Nexus policies (sectors) A context-specific definition of sustainability	Stakeholder feedback and requests for: updating the model and designing a DSS to use it	Stakeholder feedback about: the DSS functionalities and improving the precision of some key modelling outputs	Stakeholder feedback about: Opportunities and limitations of taking the proposed policy packages into action.
respondents learning surveys (N, %)	NA	NA	10, 100%	NA	9, 73%.	15, 88%
% female (and male) participants	40% (60%)	70% (30%)	60% (40%)	72% (28%)	64% (36%)	71% (29%)
% WEFE Sectoral participants (Water, Energy, Food, Ecosystems)*	60%, 30%, 0, 50%	30%, 10%, 10%, 30%	50%, 30%, 0%, 40%	33%, 6%, 6%, 44%	36%, 18%, 0%. 45%	41%, 6%, 18%, 82%
% Latvian (and Lithuanian) participants	80% (20%)	100% (0%)	60% (40%)	56% (44%)	82% (18%)	59% (41%)
Online summary	Link	Link	Link	Link	Link	Link

*Percentages do not necessarily add up to 100% because stakeholders can represent more than one sector

Table B2. Summary of the organisations represented across the different stakeholder workshops (WS) and focus groups (FG) in Latvia (LV) and Lithuania (LT)

SH/Institutions in Latvia	Country	WS 1	WS 2	WS 3	WS 4	WS 5	WS 6	FG (TV)	FG (LT)
Latvia University of Life Sciences and Technologies	Latvia	X	X		X	X	X	X	
Latvian Environment, Geology and Meteorology Centre	Latvia	X		X	X	X	X		
Bauska County municipality	Latvia		X	X	X	X	X	X	
Ministry of Agriculture	Latvia	X	X		X		X	X	
University of Latvia	Latvia	X	X						
NGO Green Liberty	Latvia	X	X						
NGO Association “Farmers’ Parliament”	Latvia		X		X		X		
Jelgava municipality oper ative information center (Jelgava Digital Center)	Latvia		X		X	X			
Zemgale Planning Region	Latvia	X		X		X			
NGO Latvian Fund for Nature	Latvia		X						
NGO Business Development Group	Latvia	X							
Latvian water and wastewater works association	Latvia		X						
Laflora Ltd.	Latvia			X					
World Wide Fund for Nature (WWF), Latvia	Latvia			X					
Latvian State Forest Research Institute Silava	Latvia				X				
NGO Salgale rural support association	Latvia				X	X	X	X	
Ministry of Environmental Protection and Regional development	Latvia				X				
LPKS "LATRAPS"	Latvia				X	X			
Center for Environmental Policy (AAPC)	Lithuania			X	X	X	X		X

SH/Institutions in Latvia	Country	WS 1	WS 2	WS 3	WS 4	WS 5	WS 6	FG (TV)	FG (LT)
Kaunas University of Technology	Lithuania	X							
Vytautas Magnus University, Lithuania	Lithuania	X							
Ministry of Environment	Lithuania			X					X
Lithuanian Energy Agency	Lithuania			X					
Lithuanian Hydrometeorological Service	Lithuania				X		X		X
Panevėžys City Municipality	Lithuania				X	X	X		
Biržai District Municipality	Lithuania				X		X		
Environmental Protection Agency	Lithuania								X
Ministry of Energy	Lithuania						X		

Table B3. Summary of the focus groups in the participatory modelling process in the Lielupe case study (NEXOGENESIS project)

Date	**27/03/2025**	**29/04/2025**
Location	Riga, LV	Vilnius, LT
Phase of the participatory modelling cycle	Model use and policy evaluation	Model use and policy evaluation
Phase of the stakeholder engagement approach	Co-development	Co-development
Number of participants	5	5
Focus group format	In person	In person
Focus group purpose	Stakeholder interaction with a web-based decision support system and select policy packages for reaching certain policy goals in the Lielupe RB	Stakeholder interaction with a web-based decision support system and select policy packages for reaching certain policy goals in the Lielupe RB

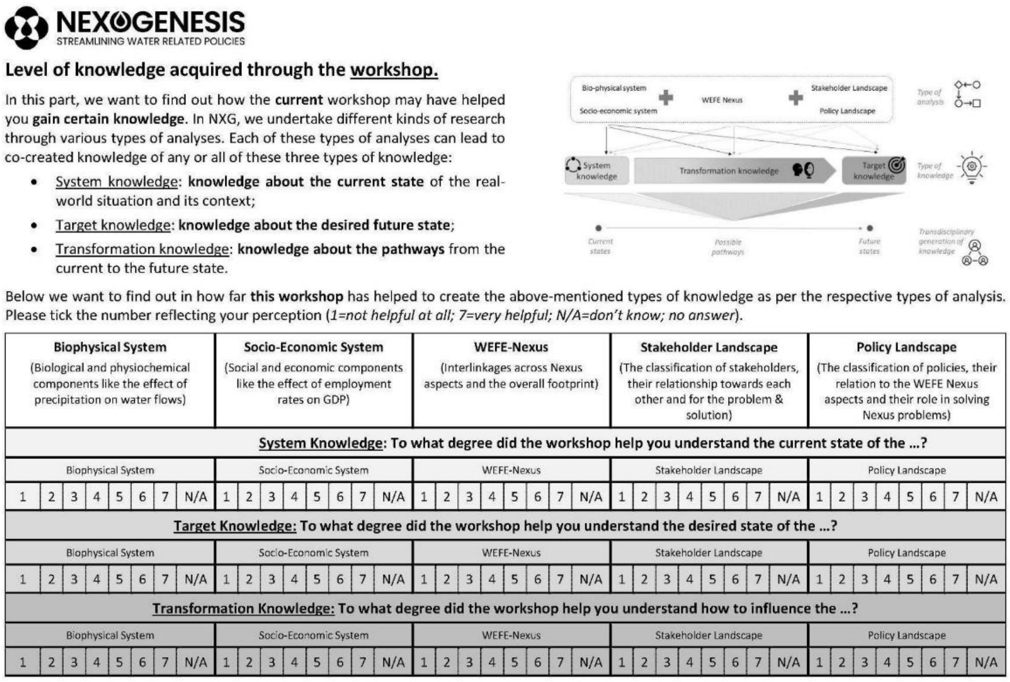
Facilitation approach	Aided testing of the tool in groups (+task) Feedback (plenary) on the model and tool functionalities	Aided testing of the tool in groups (+task) Feedback (plenary) on the model and tool functionalities
Driving questions	Is the tool practically applicable to recommend policy packages for reaching the policy goals?	Is the tool practically applicable to recommend policy packages for reaching the policy goals?
Key insights of the session	The task was given to participants to test the practical application of the tool and explore recommendations of the decision support system to achieve policy goals (i) nitrogen pollution reduction in Latvia; (ii) equitable contribution for implementation of measures in Latvia and Lithuania; (iii) reaching the biodiversity target in the Lielupe RB and (iv) simultaneously achieving all goals in the Lielupe RB. Outcomes indicate that packages of multiple (4-9) policy instruments are required. It was reflected that policy packages selected seem to be reasonable.	The task was given to participants to test the practical application of the tool and explore recommendations of the decision support system to achieve policy goals (i) nitrogen pollution reduction in Lithuania; and (ii) simultaneously achieving all goals in the Lielupe RB. Outcomes indicate that packages of multiple (2-11) policy instruments are required. It was reflected that water pollution with Nitrogen is the key issue and policy packages should be focussed to address it.

Table B4. Stakeholder workshop activities classified by type of analysis, timing across the PM cycle, and the intended knowledge type to be generated

Type of Analysis	Activity or Method applied	Timing of presentation to SHs or activity of co-creation	Type of intended knowledge type to be generated
Biophysical	Conceptual map development	I - exercise during WS2	system
	Model development	-	System and transformation
	Data gathering	-	system
	Scenario development	-	target
	Model results shown to SHs	Demonstration during WS4	transformation
	Model integrated into NEPAT	I - during WS3	transformation

Type of Analysis	Activity or Method applied	Timing of presentation to SHs or activity of co-creation	Type of intended knowledge type to be generated
Socio-economic	Scenario development	-	target
	Data gathering	-	system
	Inclusion into model	-	system
WEFE Nexus	Interlinkages in the conceptual map	I - exercise during WS2	system
	WEFE Nexus Footprint development	-	target
	Connection of Footprint to model	-	system
	Integration of Footprint into NEPAT	III - showcasing of WEFE footprint results at WS5	transformation
SH landscape	SH register, identification and classification	-	
	1st SHE plan	-	
	2nd SHE plan	I - Presentation of CS specific SH landscape to SHs at WS3	system
		I - Discussion on SH sustainment at WS3	target
	3rd SHE plan	III - Presentation of CS specific SH landscape to SHs at WS5	transformation
		III - Decision on SH sustainment at WS5	target
		III - Presentation of project-wide SH landscapes to SHs at WS6	transformation
Policy landscape		I - interviews in the 2nd project year with key SH	System and transformation

Type of Analysis	Activity or Method applied	Timing of presentation to SHs or activity of co-creation	Type of intended knowledge type to be generated
	NXG governance and policy coherence assessment	I - Presentation of results at WS3	System and transformation
	Selection of policies to be implemented in the NEPAT	I - Exercise with SHs at WS3	Target
		III - Presentation of initial policy implementation in the NEPAT at WS5	Transformation
	Selection of user-validated policy packages (UVPP)	III - Exercise at WS5	Target
	Development of governance roadmaps for UVPPs	III - Conversations throughout the last year of the project and at WS6	Transformation


NEXOGENESIS
STREAMLINING WATER RELATED POLICIES

Level of knowledge acquired through the workshop.

In this part, we want to find out how the **current** workshop may have helped you **gain certain knowledge**. In NXG, we undertake different kinds of research through various types of analyses. Each of these types of analyses can lead to co-created knowledge of any or all of these three types of knowledge:

- System knowledge: **knowledge about the current state** of the real-world situation and its context;
- Target knowledge: **knowledge about the desired future state**;
- Transformation knowledge: **knowledge about the pathways** from the current to the future state.

Below we want to find out in how far **this workshop** has helped to create the above-mentioned types of knowledge as per the respective types of analysis. Please tick the number reflecting your perception (*1=not helpful at all; 7=very helpful; N/A=don't know; no answer*).

Biophysical System (Biological and physiochemical components like the effect of precipitation on water flows)	**Socio-Economic System** (Social and economic components like the effect of employment rates on GDP)	**WEFE-Nexus** (Interlinkages across Nexus aspects and the overall footprint)	**Stakeholder Landscape** (The classification of stakeholders, their relationship towards each other and for the problem & solution)	**Policy Landscape** (The classification of policies, their relation to the WEFE Nexus aspects and their role in solving Nexus problems)
System Knowledge: To what degree did the workshop help you understand the current state of the …?				
Biophysical System	Socio-Economic System	WEFE-Nexus	Stakeholder Landscape	Policy Landscape
1 2 3 4 5 6 7 N/A	1 2 3 4 5 6 7 N/A	1 2 3 4 5 6 7 N/A	1 2 3 4 5 6 7 N/A	1 2 3 4 5 6 7 N/A
Target Knowledge: To what degree did the workshop help you understand the desired state of the …?				
Biophysical System	Socio-Economic System	WEFE-Nexus	Stakeholder Landscape	Policy Landscape
1 2 3 4 5 6 7 N/A	1 2 3 4 5 6 7 N/A	1 2 3 4 5 6 7 N/A	1 2 3 4 5 6 7 N/A	1 2 3 4 5 6 7 N/A
Transformation Knowledge: To what degree did the workshop help you understand how to influence the …?				
Biophysical System	Socio-Economic System	WEFE-Nexus	Stakeholder Landscape	Policy Landscape
1 2 3 4 5 6 7 N/A	1 2 3 4 5 6 7 N/A	1 2 3 4 5 6 7 N/A	1 2 3 4 5 6 7 N/A	1 2 3 4 5 6 7 N/A

Figure B1. Knowledge generation survey, as distributed to workshop participants

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About the Author

Henry Daniel Amorocho-Daza completed his PhD at the IHE Delft Institute for Water Education and Delft University of Technology. His doctoral research was conducted within NEXOGENESIS, a European Commission-funded project, where he led the development of simulation models to facilitate stakeholder dialogue in the Lielupe River Basin (Latvia and Lithuania).

An interdisciplinary researcher and systems thinker, he specialises in quantitative methods—including systems dynamics, multi-criteria decision analysis, and statistics—to support decision-making in complex environmental contexts. Throughout his career, Henry has built experience across international collaborations spanning academia, NGOs, and the public sector.

He holds a BSc in Industrial and Environmental Engineering from Universidad de los Andes (Colombia) and an MSc with Honours in Water Management and Governance from IHE Delft (The Netherlands).

Journal articles

Amorocho-Daza, H., Avellán, T., Sušnik, J., Brēmere, I., Indriksone, D., Ryfisch, S., van der Zaag., P. & Slinger, J. (under review). Opening the black box: tracking model evolution and learning in environmental participatory modelling. Under review in *Systems Research and Behavioural Science*

Amorocho-Daza, H., Sušnik, J., Slinger, J. H., & van der Zaag, P. (2026). A participatory system dynamics approach to assess transboundary nutrient pollution: modelling the water-energy-food-ecosystems nexus in the Lielupe River Basin, Lithuania and Latvia. *Ecological Modelling*, *513*, 111417. https://doi.org/10.1016/j.ecolmodel.2025.111417

Amorocho-Daza, H., Sušnik, J., van der Zaag, P., & Slinger, J. H. (2025). A model-based policy analysis framework for social-ecological systems: Integrating uncertainty and participation in system dynamics modelling. *Ecological Modelling, 499,* 110943. https://doi.org/10.1016/j.ecolmodel.2024.110943

Amorocho-Daza, H., van der Zaag, P., & Sušnik, J. (2024). Ethical considerations of using system dynamics in participatory settings: a social-ecological-systems perspective. *System Dynamics Review, 40*(2), e1755. https://doi.org/10.1002/sdr.1755

Lucca, E., Kofinas, D., Avellan, T., Kleemann, J., Mooren, C. E., Blicharska, M., …, **Amorocho-Daza, H.**, … Laspidou, C. (2025). Integrating "nature" in the water-energy-food Nexus: Current perspectives and future directions. *Science of The Total Environment, 966*, 178600. https://doi.org/10.1016/j.scitotenv.2025.178600

Conference presentations

Amorocho-Daza, H., Brēmere, I., Indriksone, D., van der Zaag, P., Slinger, J. & Sušnik, J. (2024). Using System Dynamics participatory modelling to support international river basin policy discussions: The case of the Lielupe River Basin (Lithuania and Latvia) Water-Energy-Food-Ecosystems Nexus. Oral presentation at International System Dynamics Conference 2024, 4-8 August 2024, Bergen, Norway.

Amorocho-Daza, H., Sušnik, J, van der Zaag, P.& Slinger, J. (2025). A Policy Analysis Approach to Transboundary Nutrient Pollution: Insights from a Co-Created System Dynamics Model of the Lielupe River Basin (Lithuania and Latvia). Oral presentation at European System Dynamics Workshop 2025, 22-23 May 2025, Delft, The Netherlands.

Amorocho-Daza, H., van der Zaag, P. & Sušnik, J. (2023). Ethical considerations of participatory modelling in the context of sustainable development. Oral presentation at 18th International Conference on Environmental Science and Technology 2023, 30 August – 2 September 2023, Athens, Greece.

Amorocho-Daza, H., van der Zaag, P. & Sušnik, J. (2023). A statistical model to quantify the water-human development relation. Oral presentation at International Conference on Water, Energy, Food and Sustainability, 10-12 May 2023, Leiria, Portugal.

Amorocho-Daza, H., van der Zaag, P. & Sušnik, J. (2022). A System Dynamics approach to evaluate complex Water-Energy-Food-Ecosystems (WEFE) Nexus policy interventions under uncertainty at river basin scale. Oral presentation at 23rd WaterNet/WARFSA/GWP-SA Symposium, 19-21 October 2022. Sun City, South Africa.

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educational PhD programme of SENSE.

Delft, the 4th of June 2026

SENSE coordinator PhD education

Dr Ir Peter Vermeulen

The SENSE Director

Dr Jampel Dell'Angelo

The SENSE Research School declares that **Henry Daniel Amorocho Daza** has successfully fulfilled all requirements of the educational PhD programme of SENSE with a work load of 44.9 EC, including the following activities:

SENSE PhD Courses

- Environmental research in context (2022)
- Research in context activity: 'Organising a panel discussion on Challenges and opportunities of interdisciplinarity in river studies (2023)
- Grasping sustainability (2023)

Other PhD and Advanced MSc Courses

- Ethics in a Global Context, University of Lucerne, Switzerland (2022)
- Managing myself, leading others, TU Delft (2022)
- Research data management, TU Delft (2022)
- Project Management for PhD Candidates, TU Delft (2022)
- The Art of Presenting Science, TU Delft (2023)
- Popular Scientific Writing, TU Delft (2023)
- Cross Cultural Communication skills in Academia, TU Delft (2022)
- Creative Tools for Scientific Writing, TU Delft (2022)
- Sharing your Research and Work as Simple as a TEDx Talk, TU Delft (2022)
- Exploring a research career outside academia, TU Delft (2023)
- Personal branding, presenting yourself effectively, TU Delft (2023)
- Managing the Academic Publication Review Process, TU Delft (2023)
- Basic Problem Solving & Decision-making for researchers, TU Delft (2023)
- Facilitating Group Model Building to support the Sustainable Development Goals, Radboud University (2023)
- Tidy data transformation and visualization with , PE&RC and WIMEK (2023)

Management and Didactic Skills Training

- PhD Association Board representative – IHE Delft (2022-2023)

Oral Presentations

- *A policy evaluation framework for the Water-Energy-Food-Ecosystems Nexus.* IHE PhD symposium 13-14 October 2022, Delft, The Netherlands
- *Ethical considerations of participatory modelling in the context of sustainable development*. IHE PhD Symposium, 19-20 October 2022, Delft, The Netherlands
- *A statistical model to quantify the water-human development relation. International* Conference on Water, Energy, Food and Sustainability, 10-12 May 2023, Leiria, Portugal
- *Ethical considerations of participatory modelling in the context of sustainable development.* 18th International Conference on Environmental Science and Technology 2023, 30 August – 2 September 2023, Athens Greece

IHE DELFT

Institute for Water Education under the auspices of UNESCO

TU Delft Delft University of Technology

This dissertation engages with the wicked problem of agriculture-driven nutrient pollution in transboundary basins. The Lielupe River Basin, shared between Latvia and Lithuania, provides a practical example of this issue. This thesis proposes ways forward to address this problem by crafting and using environmental models in participatory and systematic ways.

Modelling the environment with stakeholders adopts a systems perspective in connecting theory with practice. Theoretically, we investigate how participation, uncertainty and ethics can be included in model development for socio-environmental problems. Practically, we explore the added value of implementing participatory modelling in transboundary river management using the Lielupe basin as a case study.

Researchers and practitioners can find two theoretical contributions: (1) a flexible 3-phase modelling framework for addressing socio-environmental issues; (2) a structured set of ethical questions when engaging in participatory modelling.

Empirical contributions are specific to the Lielupe River Basin yet provide insights that are potentially transferable to other cases. Our results indicate that moderate cooperative land-use change scenarios outperform ambitious unilateral actions, helping to achieve basin-wide water quality objectives a decade faster — a critical finding for the Baltic region and the achievement of EU water quality objectives. We also show that model evolution and participants' learning are socially driven across and beyond a modelling cycle.

This thesis contributes towards a more reflective and transparent modelling practice. It provides environmental modellers with practical tools for jointly developing and using models with stakeholders. The dissertation concludes with a call to modellers to consider how modelling can support broader participatory engagement processes rather than how participatory engagement can support modelling.

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