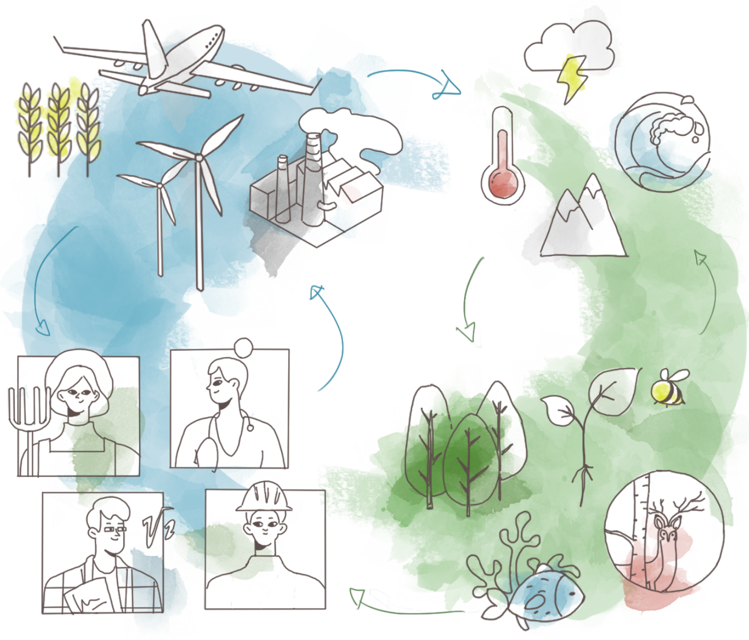
Acta Universitatis Agriculturae Sueciae

Doctoral Thesis No. 2022:70
Faculty of Landscape Architecture, Horticulture and Crop Production Science

Managing coupled human and natural systems (CHANS)

The case of water

Andreas Nicolaidis Lindqvist



Managing coupled human and natural systems (CHANS)

The case of water

Andreas Nicolaidis Lindqvist

Faculty of Landscape Architecture, Horticulture and Crop Production Science

Department of Biosystems and Technology

Alnarp

SLU

SWEDISH UNIVERSITY OF AGRICULTURAL SCIENCES

DOCTORAL THESIS

Alnarp 2022

Acta Universitatis Agriculturae Sueciae
2022:70

Cover: Coupled Human and Natural Systems
(Illustrator: Sandra Lindkvist)

ISSN 1652-6880
ISBN (print version) 978-91-8046-016-3
ISBN (electronic version) 978-91-8046-017-0

Swedish University of Agricultural Sciences, Department of biosystems and technology, Alnarp, Sweden

Print: SLU Service/Repro, Alnarp 2022

Managing coupled human and natural systems (CHANS): The case of water

Abstract

Many sustainability challenges of the 21st century are the result of poor management of coupled human and natural systems (CHANS). Limited understanding of the mechanisms that give rise to complex dynamics in CHANS has contributed to overexploitation and degradation of water and other natural resources around the globe, leading to unintended consequences of well-intentioned policies. This raises the question of whether the tools and methods currently used in environmental management and policy design meet the requirements of complex dynamic systems. In this thesis, qualitative and quantitative research approaches from the fields of systems thinking and simulation modelling were combined with the aim of improving understanding of the dynamics of CHANS, and human-water systems in particular, and developing better methods and tools to support more effective policy and management strategies in the future. The work included a systematic review, qualitative and quantitative system dynamics modelling case studies, method development, and agent-based modelling and simulation.

The results showed that changes in CHANS are driven by observable and unobservable exchanges of energy, matter and information across space and time that give rise to constantly changing, nonlinear dynamics. Many contemporary tools and methods used in management and policy design are not suited to this dynamic complexity and, instead of embracing complexity, seek to reduce it by excluding structural drivers of endogenous behaviour. This can contribute to unsustainable water use and amplify impacts of climate change in coupled human and water systems. This thesis showed that system dynamics-based approaches can effectively complement conventional static management tools, to better account for dynamic complexity. By tapping into the collective intelligence of actors engaged in the system, the approaches can support more realistic models and more effective and sustainable management, leading to establishment of middle-range theories for management of CHANS.

Keywords: coupled human and natural systems, system dynamics, modelling, simulation, water, agent-based models, sustainability, natural resource management.

Förvaltning av sammankopplade sociala, ekologiska och fysikaliska system (SEFS): Fallet vatten

Sammanfattning

Många av de hållbarhetsutmaningar vi står inför under 2000-talet har sina rötter i ineffektiv förvaltning av kopplade sociala, ekologiska och fysikaliska system (SEFS). Begränsad förståelse för hur SEFS fungerar, och för dess komplexa dynamik, gör att välmenande politiska beslut ofta får oavsiktliga negativa systemeffekter. Detta bidrar till överexploatering och omfattande skador på hydrologiska och ekologiska system runt om i världen. Det väcker även frågan om de verktyg och metoder som används inom miljöledning och policyarbete idag är lämpade för dessa komplexa system. I denna avhandling kombinerades kvalitativa och kvantitativa metoder från systemforskning, modellering, och simulering för att: (i) öka förståelsen för hur SEFS, och i synnerhet socio-hydrologiska system, fungerar; och (ii) för att utveckla bättre metoder och analysverktyg, och därmed bättre underlag till policy och förvaltning. Arbetet omfattade en systematisk litteraturstudie, fallstudier baserade på kvalitativ och kvantitativ systemdynamiskmodellering, metodutveckling och agentbaserad modellering och simulering. Resultaten visade att i SEFS sker konstant utbyte av energi, material och information mellan systemens olika delar. Detta ger upphov till en ständigt föränderlig, och ickelinjär, dynamik som många konventionella verktyg inom policy och förvaltning inte är anpassade för. I stället för att omfamna systemens dynamiska komplexitet försöker de minimera denna genom att utesluta drivande strukturer från sina underliggande modeller. Resultaten i avhandlingen visa hur detta bland annat kan bidra till ohållbar vattenanvändning och förstärka effekterna av klimatförändringar. Det visar även att systemdynamisk modellering kan komplettera konventionella statiska beslutsunderlag för att bättre ta hänsyn till dynamisk komplexitet. Genom att utnyttja den kollektiva intelligensen som finns hos de aktörer som lever och verkar i SEFS när vi bygger dessa modeller kan mer realistiska och användbara beslutsunderlag skapas, och teorier för hållbar förvaltning av SEFS utvecklas.

Nyckelord: SEFS, systemdynamik, modellering, simulering, vatten, agentbaserad modellering, hållbarhet, naturresursförvaltning

Dedication

To Julia, my beloved wife, and to everyone who inspired and supported me along the way.

Contents

List of publications

This thesis is based on the work contained in the following papers, referred to by Roman numerals in the text:

I. **Nicolaidis Lindqvist, A.**, Broberg, S., Tufvesson, L., Khalil, S., Prade, T. (2019) Bio-based production systems: Why environmental assessment needs to include supporting systems. *Sustainability* (11), 4678.

II. **Nicolaidis Lindqvist, A.**, Fornell, R., Prade, T., Tufvesson, S., Khalil, S., Kopainsky, B. (2021). Human-water dynamics and their role for seasonal water scarcity – a case study. *Water Resources Management* (35), 3043-3061.

III. **Nicolaidis Lindqvist, A.**, Fornell, R., Prade, T., Tufvesson, L., Khalil, S., Kopainsky, B. (2022). Impacts of future climate on local water supply and demand – a socio-hydrological case study in the Nordic region. *Journal of Hydrology: Regional Studies* (41), 101066.

IV. **Nicolaidis Lindqvist, A.**, Carnohan, S., Fornell, R., Tufvesson, L., Prade, T., Lindhe, A., Sjöstrand, K. Dynamic marginal cost curves: A first iteration in water resources management. (Submitted).

V. **Nicolaidis Lindqvist, A.**, Svensson, P., Carnohan, S., Karlsson, B. Accuracy or alignment: A conflict in the participatory modelling process? (Submitted).

Papers I-III are open access.

The contribution of Andreas Nicolaidis Lindqvist to the papers included in this thesis was as follows:

I. Conceptualisation and method development. Investigation and analysis, together with the co-authors. Writing the original draft.

II. Conceptualisation, data collection and data synthesis. Model development and analysis, together with BK. Writing the original draft with inputs from the co-authors.

III. Conceptualisation, data collection, model construction and simulation experiments, together with BK and RF. Analysis and writing the original draft, with inputs from the co-authors.

IV. Conceptualisation, data collection, model construction and simulation together with RF and SC. Analysis and writing the original draft together with RF and SC, with inputs from the other co-authors.

V. Conceptualisation, method development and writing the original draft, together with the other co-authors. Statistical analysis and inputs to model development.

Publications not appended

In addition to the work presented in this thesis Andreas Nicolaidis Lindqvist has published or contributed significantly to the following publications which are not appended to the thesis:

- Balkan, B.A., **Nicolaidis Lindqvist, A.**, Odoemana, K., Lamb, R., Tiongco, M.A., Gupta, S., Peteru, A., Menendez III, H.M. (2021). Understanding the impact of COVID-19 on agriculture and food supply chains: System dynamics modelling for the resilience of smallholder farmers. *International Journal of Food System Dynamics* (12), 255-270.

- Alsanius, B., **Nicolaidis Lindqvist, A.**, Vågsholm, I. (2022). Green wheel: dilemma regarding heavy electrified road transport of leafy vegetables and impacts on microbial hazards and shelf-life. *SLU Report 2022:6*, [In Swedish: Green wheel – om dilemman rörande tunga batteridrivna vägtransporter och mikrobiella faror samt hållbarhet av bladgrönsaker].

List of tables

List of figures

Abbreviations

ABM	Agent-based model
AI	Artificial intelligence
ALCA	Attributional life cycle assessment
CAS	Complex adaptive systems
CFs	Characterisation factors
CGE	Computed general equilibrium
CHANS	Coupled human and natural systems
CLCA	Consequential life cycle assessment
CLD	Causal loop diagram
DEM	Discrete event model
DSM	Dynamic systems model
GMB	Group model building
LCA	Life cycle assessment
LCI	Life cycle inventory
LCIA	Life cycle impact assessment
MCC	Marginal Cost Curve
ML	Machine learning
MM	Markov model
PE	Partial equilibrium

RQ	Research question
SD	System dynamics
SDG	Sustainable Development Goals
SES	Social-ecological system
SETS	Social, ecological and technical systems
SFD	Stock and flow diagram
SGU	Swedish Geological Survey
ST	Systems thinking
STM	State and transition model
UN	United Nations

1. The CHANS framework and implications for policy, management and planning

Coupled human and natural systems (CHANS) can be defined as complex and adaptive systems (CAS) in which human and natural components interact at different spatial, temporal and organisational scales (Liu *et al.* 2007b) (*Figure 1*). Although the definition is deceptively simple, and despite interactions between man and nature having been acknowledged for well over a century (Marsh 1864; Turner *et al.* 1993; Berkes *et al.* 2000; Hibbard *et al.* 2006; Ostrom 2009), understanding of the underlying mechanisms and processes shaping these interactions, and how the interactions integrate to govern the behaviour of these systems, remains limited (Liu *et al.* 2007a; Kramer *et al.* 2017). Improving the current understanding of the dynamics of CHANS and identifying how to manage CHANS sustainably are central challenges of the 21st century (Kramer *et al.* 2017). Several of the major sustainability challenges facing humanity at the local-to-global scale are the result of ineffective management and governance of these complex webs of human-nature interactions. Examples include climate change, depletion of natural resources, energy and food security, ecosystem degradation, pollution and pandemics (Liu *et al.* 2015). Furthermore, studies suggest we have entered a new geological epoch, known as the Anthropocene (Crutzen 2002; Crutzen 2006), where the scale of human activities is making mankind the primary driver of changes in global ecological and biogeochemical processes (Verburg *et al.* 2016). Effects of escalating environmental change propagate through coupled human-nature systems, triggering feedback loops and tipping points in ways that often cannot be intuitively predicted, increasing the potential for stresses, shocks and unpredictable crises for ecosystems and human societies (Reyers *et al.* 2018).

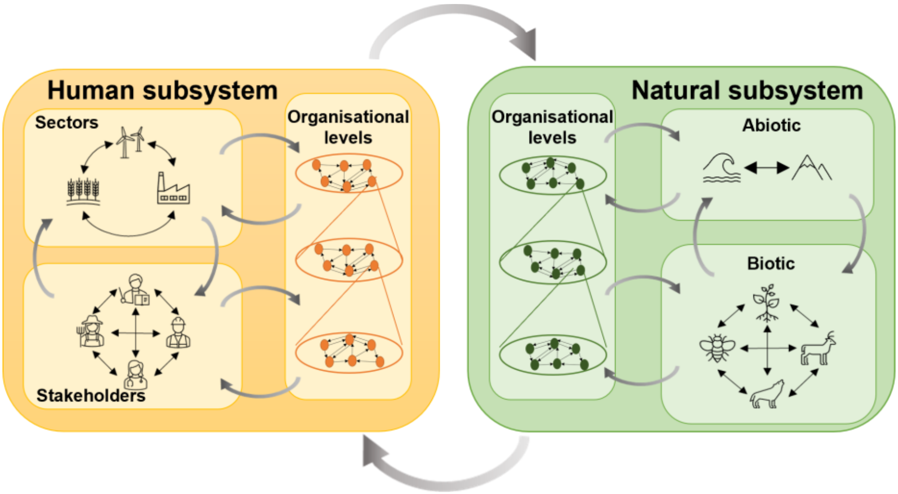


Figure 1. Schematic diagram of coupled human and natural systems. Arrows show interactions within and between subsystems at different organisational levels. Adapted from Liu *et al.* (2021).

Against this background, improving understanding of the interconnected structure and dynamics of CHANS and using this information to guide policy and management design are prerequisites for sustainable development (Liu *et al.* 2015; Kramer *et al.* 2017). This has led to the emergence of CHANS science and closely related, and partly overlapping, concepts such as social-ecological systems (SES) research (Berkes *et al.* 2000), research on integrated social, ecological and technical (SET) systems (Cosens *et al.* 2021), and socio-hydrology research (Sivapalan *et al.* 2012). All these concepts question the utility of conventional reductionist approaches for managing complex human-nature systems (Liu *et al.* 2007b; Levin *et al.* 2012). Instead of actively seeking to reduce complexity, *e.g.* by focusing on specific domains of the system or by designing studies to control for interference and feedback effects, CHANS science and related research fields claim that a holistic, and highly integrated, perspective is needed to design policies and management strategies that can effectively address the sustainability challenges of our time (Michener *et al.* 2001; Levin *et al.* 2012; Liu *et al.* 2021).

This is a remarkable claim that questions the way in which policy and management of human-nature systems have been conducted in the past. It also raises some additional questions, *e.g.* on why CHANS research for policy and management design is particularly needed right now and why traditional/conventional methods and approaches are less suited for policy and management of CHANS.

1.1 The need for research on CHANS

The need for further research on policy and management of CHANS can be attributed to three general trends, each well-supported by the scientific literature. These are: the growing scale of human activity relative to the environment; the increasing degree of coupling between human-environmental subsystems; and the accelerating rate of environmental and social change.

1.1.1 The scale of human activity: Transitioning from an empty world to a full world

In the past century, a dramatic change in the scale of human activity relative to the natural environment has occurred according to perspectives from ecological economics, a field pioneered by Professor Herman Daly and others in the 1980s and 1990s (Daly 1991; Costanza *et al.* 1997). In traditional neoclassical economics, the human economy is envisioned to develop largely unbounded by the natural environment (Daly 2014). Resources are extracted from the vast natural environment to produce goods and services, and waste and pollutants are released back to the environment, which acts as a limitless sink. In this view, the economy is small relative to the natural environment – a notion described as an "empty world" (Daly 2015, p.1). Even if resource constraints and pollution overload were to occur, these are not seen as limitations to growth in neoclassical economics due to an underlying assumption of perfect substitutability of resources, capital and labour. For instance, if a necessary resource is depleted, traditional economic theory assumes that it can be replaced by an alternative resource, more capital or more labour. This leads to the conclusion that physical limits to growth and welfare creation do not exist (Daly 2014). In contrast, in ecological economics the human economy, and all activities therein, is perceived as a subsystem of a finite, non-growing and materially closed, environmental system (Daly 2015). As the economic subsystem expands, it does so by incorporating material and energy from the surrounding natural system, reallocating it from environmental service production to economic service production. In a finite environment, this means that there in fact is a limit to material growth, and that the primary role of environmental policy and management is to manage the conflict between growth of the human subsystem and preservation of the environment and its services.

For most of human history, mankind has indeed lived in a largely "empty world". Until the past few centuries, the human population was small relative to surrounding ecosystems and the extraction rate of natural resources such as fish, water, timber, minerals *etc*. was small in relation to the size of the resource stocks and their reproduction rates. In this period, complex human-nature interactions did exist, but the intensity and scale of the interactions were typically limited in both time and space (Liu *et al.* 2007b; Reyers *et al.* 2018). However, in the past century the largely empty world has rapidly filled up to what is now a "full world" (Daly 2015, p. 4). Since the mid-1900s, the global population has more than tripled. It grew from about two billion in the 1950s to over seven billion today (United Nations 2019). With increasing demand for goods and services, and due to impressive technological innovation, the size of the economy has grown even faster than the global population (Steffen *et al.* 2015a). This is clear on looking at the rapid growth in populations of cattle, pigs, chickens, rice plants, corn stalks *etc*. that make up the living parts of the economy. The non-living part (stocks of capital, buildings, roads, dams, energy infrastructure, cars, cell phones and so on) have grown even faster than the living part (Daly 2015; Steffen *et al.* 2015a). Both the living and non-living parts of the economy rely on metabolic throughput, constant inflows of low-entropy resources from environmental sources, and constant outflows of high-entropy waste to environmental sinks (Daly 2015). As the size of the human subsystem grows, so does the scale and intensity of human-nature interactions and resource exchanges (Daly 2014; Verburg *et al.* 2016). Extraction of resources is already overwhelming the regenerative capacity of many natural resource systems, and accumulation of pollutants is already exceeding the assimilative capacity of many environmental sinks (Daly 2015). Several researchers now warn that the scale of human influence on the Earth System has become so large that it threatens many of the life-supporting Earth System processes on which we all rely (Crutzen 2006; Rockstrom *et al.* 2009). This can trigger nonlinear dynamic effects, with impacts that are difficult to predict (Reyers *et al.* 2018). Thus, to navigate a "full world", the need for further research on policy and management of CHANS is greater than ever before.

1.1.2 Tighter coupling

As the scale of human activity has grown, the number of CHANS and also their degree of coupling have increased (Verburg *et al.* 2016). When mankind was still living in a largely empty world, human-nature interactions were typically local and had an insignificant impact on large-scale natural systems and processes. Since the Industrial Revolution, expansion into new environments, exploitation of new natural resources, increased globalisation, trade *etc*. have resulted in the level of spatial and temporal coupling within and between CHANS now being greater than ever before (Liu 2017). Examples include regional coupling through virtual water trade (Fang *et al.* 2014) and emissions and global distribution of particulate ($PM_{2.5}$) pollutants (Liu *et al.* 2020). Both are coupling effects where resource depletion and environmental pollution in one place are affected by consumption in another place through global production and supply chains.

In short, the human-nature interrelationships and feedbacks influencing the dynamic behaviour of CHANS are vastly more numerous than when mankind was operating in an empty world. The increasing interdependency between social, ecological and technological subsystems of CHANS is causing an exponential increase in complexity, nonlinearity and uncertainty (Cosens *et al.* 2021). This makes system behaviour and response to change more difficult to predict. It increases the risk of unanticipated shocks, tipping points and irreversible change, and poses a significant challenge for effective policy, planning and management (Verburg *et al.* 2016; Kramer *et al.* 2017; Cosens *et al.* 2021).

1.1.3 Accelerating change

The period from 1950 to present is often referred to as 'the Great Acceleration' (Hibbard *et al.* 2006; Steffen *et al.* 2015a). During this period, the world has experienced accelerating socioeconomic development (*e.g.* population, primary energy use, water use, transportation, real GDP *etc*.), and changes in global and regional Earth System structures and processes (atmospheric carbon dioxide, tropical forest loss, marine fish captures, surface temperature *etc*.), at rates never experienced previously (Steffen *et al.* 2015a) (*Figure 2 & Figure 3*). Some of these trends began to stabilise in the past decade, *e.g.* marine fish captures, domestication of land and large dam constructions, but most are still on a trajectory of rapid growth (Steffen *et al.* 2015a).

The acceleration of social and environmental change has important implications for policy and management of increasingly integrated CHANS. First, in a full and interconnected world, rapid changes and related disturbances propagate through the network of coupled human-nature systems faster and more unpredictably than before. As a result, managing uncertainty and systemic risk becomes increasingly problematic (Cosens *et al.* 2021). Second, there is convincing evidence that the Great Acceleration has pushed several key Earth System indicators well outside the natural range of variability within which they have been maintained for the past 12,000 years (Steffen *et al.* 2015b). For instance, concentrations of greenhouse gases in the atmosphere have reached levels well above those observed previously (IPCC 2021), global surface temperatures are at record levels from a human time perspective (Steffen *et al.* 2015a), ocean carbonate chemistry is changing faster than at any time in human history (Hönisch *et al.* 2012), tropical forest loss is at record rates (Steffen *et al.* 2015a), and the rate of biodiversity loss is now on a par with historical mass extinctions (Barnosky *et al.* 2012). The implications of these new extremes are becoming increasingly evident, especially their effects on weather patterns and extreme climate events. For example, natural disturbances such as droughts, extreme rainfall and storms have increased in magnitude, frequency and duration compared with pre-industrial times (Reyers *et al.* 2018). Moreover, alteration and simplification of natural ecosystems (*e.g.* through domestication of land, deforestation and dam construction) and production ecosystems (*e.g.* by replacing diverse cropping systems with monocultures) have reduced their resilience to these disturbances and increased the risk of nonlinear and unpredictable systemic shifts in associated CHANS (Reyers *et al.* 2018).

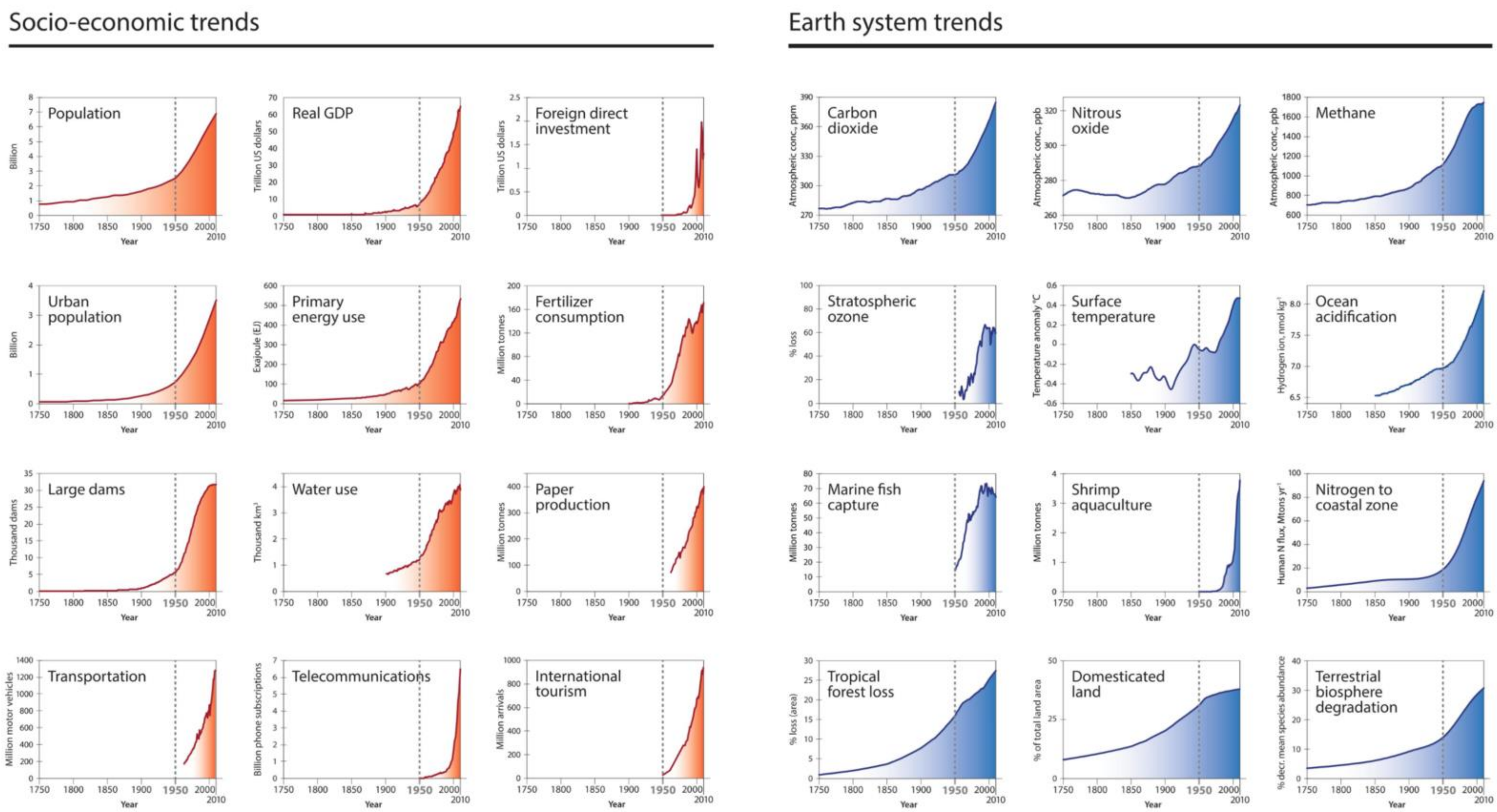


Figure 2 & Figure 3. Charts showing the rapid growth in resource use, socio-economic growth and changes in Earth system structures and processes in the past 70 years. Reproduced with permission from Steffen *et al.* (2015a).

To summarise, in the past century we have transitioned from living in a relatively empty world to a full world (Daly 2015), as illustrated in *Figure 4*. As the scale of human activity has grown relative to the environment, human-nature systems have become increasingly interconnected in both space and time (Verburg *et al.* 2016). Growing interconnectedness increases the potential for nonlinear and counter-intuitive system behaviour, while accelerating environmental and social change have increased the frequency, magnitude and duration of different forms of disturbances and shocks that can trigger these hard-to-predict systemic effects. Together, these trends have enhanced the level of complexity, uncertainty and risk that policymakers and managers of CHANS need to account for (Rockstrom *et al.* 2009; Reyers *et al.* 2018). This creates a need for new tools and methods to support sustainable management and planning (Fischer *et al.* 2015; Mancilla García *et al.* 2020).

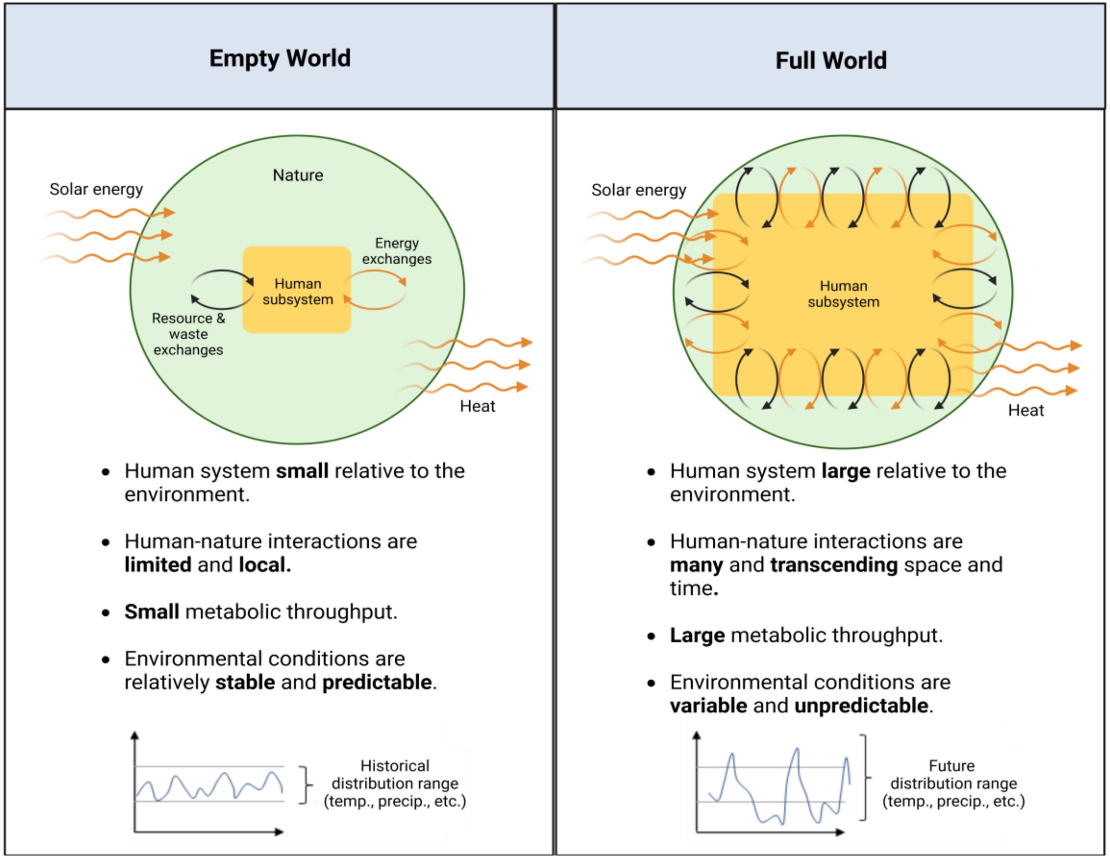


Figure 4. Graphical illustration of the transition from a largely “empty world” to a “full world” that has occurred over the past century. Adapted from Daly (2015). Created with BioRender.com.

1.2 Limitations of conventional methods and approaches for efficient policy and management of CHANS

The science and practice of policy design and management is inherently about understanding and solving real-world problems (Ruiz Estrada 2011; Walker & Daalen 2013; Repenning *et al.* 2017). This entails determining where we are now (our current state) and clearly defining where we want to be (our desired state or goal). It also requires us to understand the processes that affect our current state and drove the change that brought us here, and to apply this understanding in designing and deciding on a course of action to close the gap between our current state and our goal (Repenning *et al.* 2017).

The changing nature and increasing complexity of CHANS have spurred a debate about whether conventional approaches for planning, management and policy design in these systems are up to this task (Holling & Meffe 1996; Levin *et al.* 2012; Kramer *et al.* 2017; Mancilla García *et al.* 2020). Critics argue that most prevailing approaches for studying and supporting management of CHANS are too reductionist (Levin *et al.* 2012) and overlook key features of the underlying systems and the interrelationships between their social, environmental and technical dimensions (Levin *et al.* 2012; Mancilla García *et al.* 2020). Conventional approaches are said to rely too heavily on simplifying assumptions about linear cause-effect relationships (Reyers *et al.* 2018), and to produce results that are only representative for a snapshot in time and therefore do not generate a deep understanding of dynamic system behaviour through time (Cosens *et al.* 2021). Together, these limitations give a misleading representation of how CHANS work, which can lead to adoption of policy and management strategies with unintended outcomes (Levin *et al.* 2012; Gallón 2019). In short, as CHANS have evolved towards increasing complexity, the methods and approaches used for understanding and managing them must evolve too to account for this complexity (Kramer *et al.* 2017; Gallón 2019; Cosens *et al.* 2021).

In efforts to support policy and management of CHANS in the 21st century, several key tasks have been identified in recent studies (Fischer *et al.* 2015; Turner Ii *et al.* 2016; Kramer *et al.* 2017; Guerrero *et al.* 2018; Mancilla García *et al.* 2020; Cosens *et al.* 2021). These tasks include:

i) Enhancing integration of social, ecological and technical dimensions in CHANS research methods.

ii) Improving understanding of the environmental and social drivers of change in CHANS.

iii) Increasing understanding of the complex interactions and resulting dynamics of CHANS at different spatial and temporal scales.

iv) More effectively combining and integrating information from different knowledge systems.

This list of tasks acted as an important source of inspiration and motivation for the studies on which this thesis is based, and for the methods used. However, CHANS is too broad and multifaceted a concept to be covered in a single thesis. Therefore the scope of the research reported in this thesis was purposely limited to focus primarily on management and planning of coupled human-water systems.

2. Water in CHANS

Water is a critical link between nature and society in CHANS (Xiaoming *et al.* 2018) and access to freshwater in sufficient quantities and of sufficient quality is a determinant for the life and prosperity of human societies and natural ecosystems around the globe (Gleick & Cooley 2021). From one perspective the Earth is full of water, since approximately 70% of its surface is covered by water, but only about 2.5-3% of this is freshwater, and only a tiny fraction is easily accessible for human use (Gleick & Cooley 2021). Furthermore, this tiny fraction is unevenly distributed, both geographically and temporally, around the globe. Precipitation and evapotranspiration rates vary significantly across space and throughout the year, causing large temporal and spatial inequalities in water availability (Jaeger *et al.* 2017; Gleick & Cooley 2021). Water is now being withdrawn from groundwater aquifers, rivers and lakes to be used for many competing purposes, including agriculture, industry and domestic use, and lack of water is a driver of both conflict and human suffering (Sultana 2018).

The prominent role that water plays for human prosperity gained official recognition in 2010, when the United Nations (UN) proclaimed access to sufficient, safe-to-drink and affordable water a basic human right (United Nations 2010). Water's role as a prerequisite for sustainable development was further acknowledged by the UN in 2015 with the adoption of 'The 2030 Agenda for Sustainable Development' and the 17 Sustainable Development Goals (SDG) (United Nations 2015). SDG6, on water and sanitation, explicitly targets sustainable management of water resources and the water cycle, with the overarching goal to ensure safe drinking water and sanitation services to all people whilst safeguarding sustainable management of water resources, wastewater flows and freshwater dependent ecosystems. At the time of writing, substantial work is still needed to achieve this goal.

In 2020, one in four people (approximately 2 billion) did not have access to safely managed drinking water within 30 minutes distance from their home (UN-Water 2021). In 2018, the UN estimated that about 10% of the global population lived in countries with high or critical levels of water stress. Most of these countries are in Central and Southern Asia, and in Northern and Western Africa, but moderate, local, or seasonal water stress is also experienced in parts of the US, Europe and Eastern Asia (FAO & UN-Water 2021). Other studies suggest that water stress is significantly more widespread than indicated in UN statistics, *e.g.* Mekonnen and Hoekstra (2016) estimated that up to two-thirds of the global population are experiencing severe water stress for at least one month per year. Between 2008 and 2018, water stress increased by 4-15% across most of the world's regions (Eastern and South-Eastern Asia, Latin America and the Caribbean, Northern and Western Africa, Oceania, sub-Saharan Africa) (FAO & UN-Water 2021). Even in regions where water stress has been on the decline, *e.g.* Europe, North America and Central and Southern Asia (FAO & UN-Water 2021), local assessments show that the frequency and distribution of water stress has been increasing in recent years (European Environment Agency 2021).

The state of water resources, and challenges associated with achieving SDG 6, can be linked to the three trends described above (growing scale of human activity, tighter coupling, accelerating environmental and social change), as they combine to intensify the pressures and competition for water resources in CHANS around the globe.

2.1 Scale, coupling and accelerating social and environmental change in coupled human-water systems

Based on the trends and scale of water use, we are now clearly operating in the full world paradigm. Between 1900 and 2010, global freshwater withdrawal grew by approximately 700% (Gleick 2000), driven by a combination of a fast-growing global population (the global population in 2000 was 3.5 times the size of that in 1900), agricultural and industrial expansion (land under irrigation increased more than five-fold in the past century), and improvements in living standards leading to growing per-capita water use (Gleick 2000; Gleick 2003b). The most rapid growth occurred between the 1950s and 1980s, when withdrawal increased by on average 30% per decade (Gleick & Cooley 2021), and society is now exploiting more of available freshwater resources than ever before (Xiaoming *et al.* 2018). It is only in the past few decades that there have been indications of a slowdown, and even reductions, in withdrawal rates in some Western economies. For instance, in the US, per-capita water withdrawal declined by 20% between 1980 and 2000, and global water withdrawals only grew by 2.7% between 2000 and 2010, despite continued population and economic growth (Gleick 2003b; Gleick & Cooley 2021). This levelling off is most probably due to many regions approaching limits in terms of available rivers for large dam construction and many of the available groundwater aquifers already being exploited (Steffen *et al.* 2015a).

In many other regions, however, population and water use both continue to grow (Cosgrove & Loucks 2015) and some estimate that global water use will increase by a further 20-30% by 2050 (Boretti & Rosa 2019). Population growth and continued urbanisation will result in dramatic increases in domestic water use and contribute to increasing demand for agricultural water to meet rising demand for food and other agricultural products. Industrial water use is also projected to increase dramatically, especially in Africa and parts of Asia as these regions industrialise. As demand accumulates, this will increase the pressure on water resources, many of which are already over-exploited (Cosgrove & Loucks 2015; Gleeson *et al.* 2020). For instance, human abstraction is already causing seasonal depletion of groundwater aquifers in large parts of the Indian peninsula and rapid urban growth has led to localised groundwater depletion in many Asian, American, European and Middle Eastern cities. In California and the Midwestern United States, unsustainable pumping has led to widespread depletion of large groundwater aquifers (Lall *et al.* 2020), and remote sensing studies have shown that more than 30% of the largest groundwater systems in the world are experiencing rapid depletion (Richey *et al.* 2015). Surface water sources, such as rivers and lakes, are also undergoing severe overexploitation and quality degradation. Abstraction, damming and alteration of flow regimes have led to dramatic reductions in flow in several of the world's major rivers, to the extent that many are no longer reaching the sea (Gleick 2003a). In up to 65% of global river outlets, upstream abstraction and pollution are already affecting downstream freshwater ecosystems and water security (Vörösmarty *et al.* 2010).

If population growth and economic and agricultural activities are the key drivers of water demand, then weather and climate are among the key determinants of water supply. Historically, global and regional climate and weather patterns have been relatively stable. Despite natural variability, precipitation and evapotranspiration, and thus local hydrological cycles, have been comparatively predictable (Rockström *et al.* 2014). This predictability has served humanity well, as water availability has typically not fluctuated beyond a well-defined envelope of variability, allowing water resource management and planning to be conducted under the assumption of dynamic stationarity (Milly *et al.* 2008). However, it is now clear that human-induced climate change is challenging assumptions about a stable and stationary water supply (Milly *et al.* 2008; Famiglietti 2014; Rockström *et al.* 2014; Wu *et al.* 2020). Even slight changes in climate can alter the entire probability distribution of extreme weather events, increasing both the frequency and intensity of droughts and floods (Cosgrove & Loucks 2015). Studies suggest that even in a scenario where global warming is limited to 2 °C, this could lead to a 40% increase in the number of people experiencing absolute water scarcity. Already today, climate change is having observable impacts on precipitation patterns and groundwater recharge (Taylor *et al.* 2013; Falkenmark *et al.* 2019). For instance, in recent years extreme weather events have caused local-to-regional seasonal water scarcity in typically water-abundant regions (Ahopelto *et al.* 2019; Stensen *et al.* 2019), and simulation studies predict large changes in water availability in the coming century (Jaeger *et al.* 2017; Wunsch *et al.* 2022). While our capacity to forecast future climate impacts is constantly improving, our understanding of the social, hydrological and ecological effects on CHANS remains limited due to the complexity of these systems (Jaeger *et al.* 2017; Xiaoming *et al.* 2018).

2.2 A new water management paradigm and new research questions

To achieve the UN target of universal access to safe, sustainably managed and affordable drinking water by 2030, current efforts in water resources management need to quadruple in the coming years (World Health Organization 2021). There is an emerging consensus that to meet the challenges of increasing demand and accelerating environmental and social change in a sustainable way, water management and planning must account for the complexity of CHANS (Gleick 2000; Gleick 2003a; Pahl-Wostl 2007; Schoeman *et al.* 2014; Xiaoming *et al.* 2018; Pahl-Wostl 2020). Holistic, integrated approaches to water resources management are needed, where interdependencies between the hydrological system, the environment, and socioeconomic development are accounted for in order to achieve efficient, just and sustainable management of water resources. This is a major shift from the conventional command-and-control paradigm of water resources management rooted in an "empty world", where human and water systems were modelled independently, human impact on water resources was considered fairly limited, and natural and social disturbances were perceived as relatively predictable (Milly *et al.* 2008; Xiaoming *et al.* 2018). With this change in paradigm, the challenges of water resources management should be addressed as a subset of the challenges of CHANS research described above.

3. Thesis aim and objectives

As described in Chapters 1 and 2, research on CHANS is an expanding scientific domain. It is motivated by growing awareness about the importance of understanding human-nature interactions to ensure sustainable management and planning in these systems. The CHANS paradigm has gained considerable traction in water resources management research, but key research gaps still exist. To this end, the overall aim of this thesis was:

to contribute to sustainable management of CHANS in general, and human-water systems in particular, by increasing understanding of their drivers of change and by improving upon, and developing new, methods and tools to support policy design, planning and management.

Thus the target was knowledge creation with regard to the dynamics of CHANS (why does the system behave as it does and how is it likely to behave in the future?) and operationalisation of this knowledge into the methods and tools used to make policy and management decisions (how should we act to make the system behave as we want it to?).

3.1 Research questions

With the above aim in mind, the work in this thesis addressed the following overarching research questions (RQ):

RQ 1. How are CHANS represented in contemporary methods for assessment, management and planning of human activities, and what is required from methods to support sustainable management of these systems? [Papers I & IV]

RQ 1 contributes to the overall aim of the thesis by identifying concrete gaps and problems in contemporary management tools and methods when applied to CHANS.

RQ 2. What processes govern the dynamics of drinking water supply and demand in coupled human-water systems? [Paper II & III]

RQ 2 facilitates systemic understanding of the structural drivers of change in CHANS, exemplified in the domain of human-water systems.

RQ 3. How will climate change influence drinking water supply, and what dynamic effects may this have on socioeconomic development, and subsequent water demand, in coupled human-water systems? [Paper III]

RQ 3 supports sustainable management of coupled human-water systems by exploring how exogenous and endogenous drivers of change influence future system behaviour.

RQ 4. How can analytical methods be improved to better support policy, planning and management of CHANS? [Papers II-V]

RQ 4 facilitates advancement of decision-support tools adapted to the requirements of CHANS.

3.2 Papers included in this thesis

The research questions presented above are addressed in the five papers on which this thesis is based, as schematically illustrated in *Figure 5*.

Paper I Explores the extent to which defining features of CHANS are accounted for in mainstream life cycle assessment (LCA) methods – an approach frequently used to support planning, management and policy assessment. Bio-based production systems are used as an example of CHANS, and key limitations and avenues for improvements are suggested.

Paper II Applies participatory systems mapping to explore how human-water interactions influence water supply and demand at the local-to-regional scale in a Swedish case study. Socio-hydrological feedbacks give rise to water supply-demand cycles and system lock-in effects, and contribute to seasonal water scarcity and policy resistance.

Paper III Develops a system dynamics model of a coupled human-water system on the island of Gotland, Sweden. Simulation experiments explore how future climate scenarios may affect local drinking water supplies, and subsequent dynamic effects on housing development, the tourist sector and municipal water supply services, between 2020 and 2050.

Paper IV Develops a system dynamics-based approach to marginal cost assessment of water scarcity mitigation strategies. It shows how accounting for temporal dynamics and interaction effects between physical and behavioural processes in CHANS can provide added policy insights not accounted for by conventional marginal cost curve approaches.

Paper V Revisits the participatory systems mapping approaches applied in Papers II and III and uses an agent-based modelling (ABM) approach to investigate how social influence during the mapping process affects the accuracy of the model produced and the degree of group alignment around that model. The paper ultimately explores whether, and under what conditions, there is a conflict between accuracy and alignment, and how the intervention can be designed to mitigate this.

As illustrated in *Figure 5*, Paper I served as a starting point and inspiration for the other four papers. Limitations of contemporary LCA methods identified in Paper I inspired the use of qualitative and quantitative system dynamics in the case studies conducted in Papers II and III. The study in Paper IV was in turn inspired by the simulation results from Paper III and the findings from Paper I. The former indicated a need for significant investments in water resources management to reduce the risk of future water scarcity, while the latter indicated a need for dynamic, systems-oriented methods to assess and evaluate such interventions. Lastly, Paper V built on conclusions from Paper I, and inspiration and questions that arose through the work conducted in Papers II, III and IV. Paper I identified a need for context-specific models to assess the impact of change in CHANS and in Paper II a participatory modelling approach was applied, drawing from the mental models of a diverse group of stakeholders, to produce such models. In Paper V, the effects of group dynamics during such participatory modelling approaches was studied, drawing on theories on social behaviour to explore what makes system mapping in small groups successful or unsuccessful.

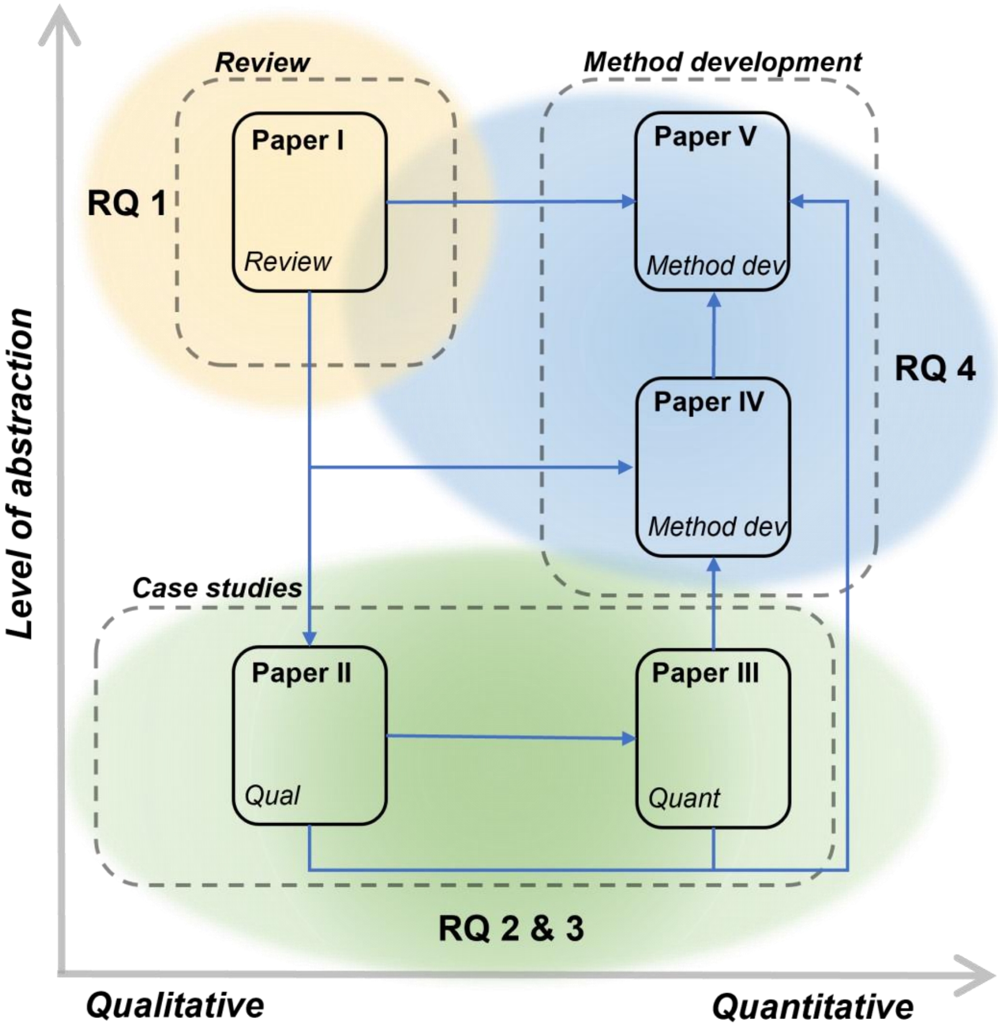


Figure 5. Schematic description of progress in the research project (blue arrows), the research questions (RQs) addressed in Papers I-V (blue, yellow and green fields) and the type of study represented by each paper (dashed boxes).

4. Methodology and research approach

In the field of CHANS research, there are few strict rules or protocols to guide research design. As a result, difficult decisions need to be made by the researcher about the methodological approach to adopt and about theories and frameworks to draw from (de Vos *et al.* 2021). The literature recommends that researchers embrace a diversity of models and methods when studying complex phenomena in human and nature systems, as this will enrich understanding and improve the chances of managing the system successfully (Hong & Page 2004; Alberti *et al.* 2011; Page 2018). Furthermore, researchers need to acknowledge that a true or optimal method for studying CHANS rarely exists and that there are often several equally viable strategies and methods for studying a given phenomenon (de Vos *et al.* 2021).

The research approach taken in this thesis draws strongly on the framework of complex adaptive systems (CAS) and employs methods from systems thinking (ST), system dynamics (SD), and simulation modelling to study, understand and manage CHANS. The following sections provide an overview of the CAS framework, and justify the methods used in Papers I-V.

4.1 Understanding CHANS as complex adaptive systems

In the literature, CHANS and its sibling concepts (SES, SET, socio-hydrological systems *etc*.) are referred to as complex adaptive systems (CAS) (Liu *et al.* 2007a; Levin *et al.* 2012; Preiser *et al.* 2018; Preiser *et al.* 2021). These can be defined as systems consisting of multiple interdependent and interacting components that produce emergent systemic properties. CAS exhibit adaptive capacities that allow the system to change and evolve over time in response to feedbacks, shocks and changes in the internal or surrounding context (Holland 2006; Preiser *et al.* 2018). The CAS framework has proven to be a powerful and unifying way to study and describe systems across many different contexts and disciplines (Carmichael & Hadžikadić 2019). In 2018, Preiser *et al.* (2018) defined six general organising principles of CAS that can be used to understand the behaviour of these systems, and to inform approaches and methods to study and manage them:

i. CAS are *constituted relationally*, meaning that the relationships between system components are more important for understanding their behaviour and system-level properties than the detailed properties of their individual components.

ii. CAS have *adaptive capacities*, where the interrelations within and between subsystems of CAS create feedbacks that allow the system to adapt and adjust continuously to externally or internally changing conditions. This capacity enables originally similar systems to take on unique developmental trajectories, but also provides systems with "memory" that can constrain and shape future development.

iii. CAS are characterised by *non-linear dynamic relations***,** meaning that relationships between components are rarely uniform or proportional. Non-linearity arises due to feedback effects, time lags, path dependencies and constantly changing, non-equilibrium, interactions between elements within the focal system, and between the focal system and the surroundings. These non-linearities make the system difficult to predict and control.

iv. CAS are *radically open and lack clear system boundaries***.** CAS are constantly interacting with their broader environment, and the processes of interaction constitute parts of the system itself. This makes defining system boundaries and determining which components belong inside or outside the system very difficult.

v. CAS are *context-dependent*. As the context of the system changes, the components, relationships and functions of the system may also change and it may take on new functions.

vi. *Complex causality and emergence* are defining characteristics of CAS. Cause and effect in these systems are typically bidirectional and characterised by complex dynamic causal chains. These systems exhibit emergent properties and a high level of dynamic complexity that arise through interactions of system components. This means that the system-level properties are fundamentally different from those expected from combining the properties of the underlying individual elements. Together, complex causality and emergent properties mean that the essential dynamics and qualities of CAS cannot be observed or understood through isolated studies of the individual system components.

Understanding CHANS as CAS has important implications in terms of the types of knowledge that can be expected to be acquired about these systems (ontological implications), how that knowledge can be acquired (epistemological implications), and what methods and approaches that are suitable for studying them (practical implications).

4.2 Ontological and epistemological implications

From a CAS-based world view, unobservable, relational and organisational interactions in CHANS give rise to non-material causes and dynamics with real, system-level ontological and observable effects. This corresponds to what Mancilla García *et al.* (2020) describe as a process-relational view of the world where processes and relations between elements are considered the primary constituents of reality. This is fundamentally different from the traditional Newtonian world view, where objects are the primary elements of reality, and where properties that cannot be observed, isolated and measured lack ontological status (Preiser *et al.* 2021). In the Newtonian perspective, the only way of achieving scientific knowledge is through a process of observation, experimentation and measurement of phenomena under conditions of independent verifiability and reproducibility (Preiser *et al.* 2021). To put it differently, a Newtonian world view assumes that all real-world phenomena can be fully understood and predicted in the same way as an engineer can predict, with high accuracy, the behaviour of a rocket moving through space, *i.e*. by studying the isolated properties of its smallest components and extrapolating these to the larger system. In this view, the natural world is in equilibrium and deterministic, based solely on the well-defined parts of matter that are inherently inert and passive (Arthur 2014). Through a Newtonian lens, ‘the system is equal to the sum of its parts’.

In the CAS-based/process-relational world view, on the other hand, systems are not simply parts connected by mechanical interactions that can be fully understood by isolating and studying them one by one. Instead, relationships and organisational processes are the nuts and bolts that define the system construct. CAS come about and behave in ways that are the result of the underlying relational structure between its components. Thus, from a process-relational lens, CAS do not exist independently from the relations and interactions that constitute them (Mingers 2000; Cilliers 2002) and system understanding can only be achieved by studying the system as a whole (Preiser *et al.* 2021). The immune system serves as an illustrative example: it cannot be extracted and studied as an isolated single entity, nor does it sit in any particular organ, but exists only as observable, emergent, system-level properties driven by the interactions between various body functions, processes and organs – ‘the system is different from the sum of its parts’ (Preiser *et al.* 2021).

Several studies suggest that to understand, manage and explore CHANS and other complex adaptive living and non-living systems, a shift from the traditional Newtonian world view towards a more process-relational view is needed (Arthur 1999; Meadows 2009; Ulanowicz 2009; Levin *et al.* 2012; Hertz *et al.* 2020; Mancilla García *et al.* 2020). This shift has practical implications for how CHANS research should be designed to develop empirically valid and meaningful data, and to build actionable knowledge and theories to support effective management and planning.

4.3 Practical implications

The complexity and emerging nature of CHANS mean that they can never be fully understood, and that any knowledge acquired about them will always be partial and dependent on the spatial, temporal, and historical context. Any insights derived will also be influenced by the conscious or unconscious choices and assumptions made by the researcher (de Vos *et al.* 2021). Because of this, universal theories of CHANS can rarely be derived (Preiser *et al.* 2018; Schlüter *et al.* 2019b). Instead, research should focus on reflexive interpretation and evaluation of these systems (Preiser & Cilliers 2010), and on understanding contextual and dynamic contexts that may give rise to observed problematic or puzzling system-level phenomena. This requires a systemic perspective to be adopted (Meadows 2009; Levin *et al.* 2012; de Vos *et al.* 2021; Preiser *et al.* 2021), where human and natural subsystems are studied in an integrated approach, using methods that allow information and knowledge from a diversity of sources, theories and disciplines to be combined (Kelly *et al.* 2013; Guerrero *et al.* 2018).

For research aiming to guide policy design, management and planning, the objective is to provide information that supports decision-makers to make knowledgeable choices among alternative options (Walker & Daalen 2013). Given that knowledge of CHANS will always be partial and context-dependent, this requires specific emphasis on transparency throughout the entire research endeavour. Assumptions, biases, the disciplinary background of the researchers, the contextual framing of the project *etc*. will colour method choices, analysis and interpretation of results. Thus, it is important to strive for a transparent and open process, to allow critical choices and assumptions made to be scrutinised by other researchers and non-researchers, and to ensure that they are accounted for in the decision-making process (Maeda *et al.* 2021; Schlüter *et al.* 2021a). Maintaining a high-level of transparency is also particularly important to build trust and alignment around the results obtained, especially when addressing problems of considerable complexity (Voinov *et al.* 2018).

This can be boiled down to a set of requirements that guide design and method choice in CHANS research:

Methods for studying and supporting policy and management in CHANS should strive for efficient treatment of system structure, the feedback effects created by the structure, and how these play out dynamically over time, embedded in a transparent and reflexive process.

Transparency

CHANS research should aspire to a high level of transparency and reflexivity in terms of the underlying logic, assumptions and choices made to arrive at conclusions. This is particularly important in the sphere of policy and management and when working with external stakeholders (Voinov *et al.* 2018; Maeda *et al.* 2021; Schlüter *et al.* 2021a).

Treatment of structure

CHANS research should focus on understanding the structural drivers and causal mechanisms that give rise to system-level phenomena, rather than the isolated properties of individual system components (Costanza & Ruth 1998; de Vos *et al.* 2021). This entails accounting for both natural forces (*e.g.* first principles and physical flows and accumulations *etc.*) and social and behavioural mechanisms (*e.g.* economic forces, flow and accumulation of information, and how people behave and act under different conditions) that give rise to an observed effect (Sterman 2000; Gerring 2008; Biesbroek *et al.* 2017; Wang *et al.* 2018).

Treatment of time

Since CHANS are inherently dynamic and constantly evolving, they need to be studied over time and not as isolated individual events (Levin *et al.* 2012).

Treatment of feedback

Methods for studying CHANS need to be able to handle nonlinear dynamics and feedback effects between system components (Levin *et al.* 2012; Preiser *et al.* 2018).

4.4 Use of modelling and simulation for studying and managing CHANS – selecting the right tool for the task

Conceptual and formal models are well-suited, and commonly used, for studying and managing CHANS (Sterman 2001; Levin *et al.* 2012; Schlüter *et al.* 2019c; Biggs *et al.* 2021b). However, given the diversity of different model types and modelling paradigms available, selecting one (or several) that fits the requirements of a study is not always a straightforward task (Schlüter *et al.* 2019c). Therefore the following section provides a short introduction to some of the reasons why models are used, and lists a set of criteria that can be used to evaluate whether a model is fit for the intended purpose. A (far from exhaustive) overview of different types of modelling available to the CHANS researcher is provided, with the types categorised based on how well they can handle the requirements of CHANS research presented above. This characterisation is used to justify the modelling choices made in the presented papers.

4.4.1 The purpose of modelling

The reason for modelling is to better understand the world and the problems around us. Pearl (2000, p. 202) defines a model as “an idealized representation of reality that highlights some aspects and ignores others”. A good model is one that represents the aspects of reality necessary to understand the problem at hand by mimicking the relevant features of the system under study. By studying features of the model, inferences can be made about the real world. Colloquially, however, this is often misunderstood as modelling being all about *prediction*, when it is often truly more about *understanding*. This leads to the mistaken perception that a model is a crystal ball that is fed with known inputs and produces predictions of the future (Epstein 2008). From this viewpoint, the value and validity of a model are judged solely on the accuracy of those predictions. While prediction can indeed be the main reason for building models, it is far from the only reason (Epstein 2008; Page 2018). Page (2018, p 15) illustrates the richness of model uses by suggesting seven broad categories of modelling purposes (reason, explain, design, communicate, act, predict, explore), easily remembered by the acronym REDCAPE (Table 1).

Table 1. *Seven reasons for building and using models (adapted from Page (2018, p 15)*

Purpose	***Description***
Reason	To identify conditions and deduce logical implications.
Explain	To provide (testable) explanations for observed phenomena.
Design	To design institutions, policies and rules.
Communicate	To relate knowledge and understanding.
Act	To guide choices, management and actions.
Predict	To make predictions about future unknown phenomena.
Explore	To investigate possibilities and hypotheticals.

A model can be built to accommodate a few of these purposes at once, but catering for all seven would require the use of multiple models and model types. The reason is that different model types rest on different approaches and founding assumptions. This has resulted in a diversity of models being developed, each with its own strengths and weaknesses that need to be matched with the purpose and requirements of the study. To this end, Levins (1966), supported by several subsequent studies (Costanza & Ruth 1998; Evans 2012; Dickey-Collas *et al.* 2014), proposed a minimum of three evaluation criteria to be used when matching model type with study requirements. These are: model realism, precision and generality (*Figure 6*). In all modelling activities, trade-offs need to be made between these qualities, since according to Levins (1966), any model can at best perform well in two of the three.

It should also be acknowledged that these model desirables are not perfectly fixed for a given model type. Models built using the same modelling approach tend to occupy the same domain of the triangle in *Figure 6*, but the exact location depends on the context of application (Dickey-Collas *et al.* 2014). For instance, Ip *et al.* (2013) showed that even within the same modelling family, the degree of realism, generality and precision can be fine-tuned to match the requirements of the application.

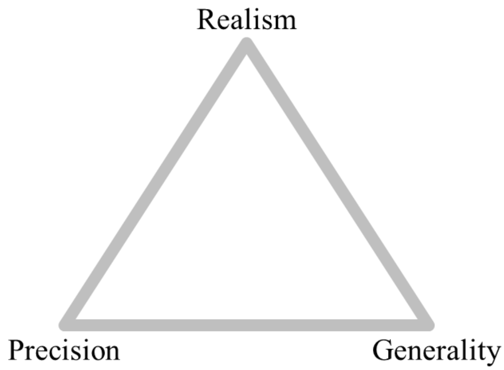


Figure 6. All models occupy a space somewhere between the extremes of high realism, high precision and high generality. According to Levins (1966), no model can perform well in all three of these qualities, as one always will be compromised when increasing the other two. Diagram based on Levins (1966) and adapted from Dickey-Collas *et al.* (2014).

Realism refers to the number of underlying elements and processes giving rise to observed patterns and behaviours represented in the model (detail realism), and the capacity of the model to represent system structure and behaviour in a qualitatively realistic way (structure-behavioural realism).

Precision refers to replicating system behaviour with a high level of quantitative accuracy.

Generality refers to the ability to represent a broad range of system behaviour with the same model, and the extent to which a model can be applied in new contexts, systems and domains.

In addition to the three model criteria proposed by Levins (1966), model interpretability (or tractability as proposed by Silverman (2018)) can be added as a fourth criterion. Interpretability has become a characteristic of increasing relevance with the rapid growth in computer power, which has increased the capacity to build and use models so complex that they become impenetrable for the human mind (Silverman 2018; Rudin 2019).

Interpretability is associated with transparency and refers to whether the assumptions of the model, and the internal logic that it applies to derive outputs from inputs, are clear and can be analysed and scrutinised. In other words, it must be possible to analyse and understand why the model produces the results that it does.

It should be noted that the first three criteria (realism, precision, generality) are fixed characteristics of a model, whereas interpretability is determined both by the specifics of the model and by the characteristics and skills of the user. For instance, the logic of a model might be fully understandable for an experienced modeller, but a black box for a decision maker without modelling experience. Poor interpretability can hide critical flaws in model logic (Voosen 2017) and, if the model is intended to support understanding and guide decision making, can erode credibility and trust in the results produced (Voosen 2017; Maeda *et al.* 2021).

Since no single model performs well on all four criteria (Costanza & Ruth 1998; Silverman 2018), it is important to find a balance between realism, precision, generality and interpretability that fits the requirements and purpose of the study, and the skill and demand of the intended users. In the modelling studies presented in this thesis, the primary purposes were:

- To build system understanding and to provide explanations of system-level phenomena (Paper II)
- To explore potential future scenarios (Paper III)
- To reason about the implications of higher-level theories and explore their implications under different hypotheticals (Paper V).
- To develop tools to support and guide management and policy actions (Papers IV and V).

This required interpretable models with an emphasis on structural realism that were behaviourally accurate only to the degree needed to support decisions. A moderate level of generality was aspired for, to allow for a certain degree of theory development and for extension of insights to new contexts. The emphasis on predictive power was relatively lower.

The practical requirements for studying CHANS presented in section 4.3. (treatment of structure, time and feedback), and the model criteria described above, were used as a framework for evaluating and selecting among different modelling approaches for the studies.

4.4.2 Choosing a modelling approach

Figure 7 provides a rough overview, or a taxonomy, of different model types available to the CHANS researcher.

Physical models

The first level of distinction is made between physical, informal and formal models. A typical example of a physical model is a small-scale copy of an object (*e.g*. a ship or a bridge) that is studied to derive conclusions about the properties of its full-size counterpart. Physical models can also be model organisms (*e.g*. fruit fly, *Drosophila melanogaster*), studied to draw inferences about the biological processes in other organisms, or analogue machines constructed to represent some (set of) processes. Naturally, these are rarely applicable to studying CHANS.

Informal models

Informal models are conceptual representations of a system, its components and their relationships. These can take the form of narratives, diagrams, drawings *etc*. In CHANS research, causal loop diagrams (CLD) are frequently used as conceptual representations of the causal structure of the system under study. These are often created using participatory system mapping approaches, such as group model building (GMB) (Vennix 1999; Hovmand *et al.* 2011), where qualitative information about the system is extracted and synthesised in close collaboration with experts and stakeholders living and interacting in the system (Voinov *et al.* 2018). These models are naturally qualitative and static in nature but, as shown in Paper II, they can create a rich understanding of the processes, drivers and feedbacks responsible for system-level behaviour (Banitz *et al.* 2022). They can be used to explore and synthesise different perspectives of the system under study (Aminpour *et al.* 2021), and guide further data collection and formal modelling (Luna-Reyes & Andersen 2003).

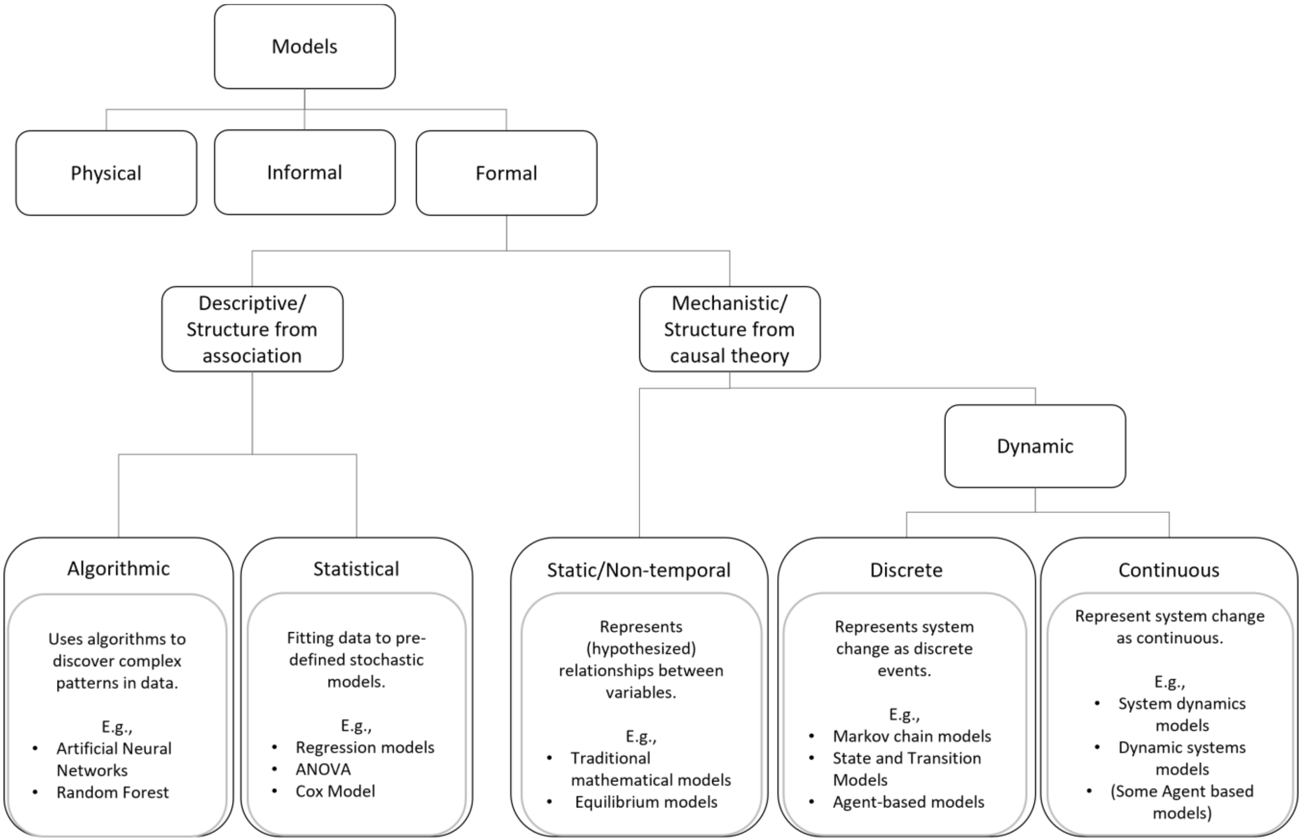


Figure 7. Taxonomy of model types.

Formal models

Formal models are quantitative, precise statements describing the components of the system under study and their relationships, typically formulated using mathematical equations. Formal models can be further subdivided into descriptive (or phenomenological) and mechanistic models, which differ fundamentally in their approach and the type of data and information they provide about the system.

Formal descriptive models

Descriptive models seek to discover relationships among a set of measured variables by identifying patterns in empirical data. They can be further subdivided into statistical models and algorithmic models, best known as artificial intelligence (AI) or machine learning (ML) models (Breiman 2001). Both modelling cultures start with known data inputs and response outputs, and both have the primary goal of predicting accurately the response outputs for future data inputs. However, the approaches they use to do this are fundamentally different.

Statistical models

In statistical modelling, a stochastic model class (*e.g*. Gaussian distribution or Cox model) is selected to represent the statistical relationships (which are often not the causal relationship) between the data inputs and the response outputs. The model is then fitted to the data by calibrating its parameters and validated using different goodness-of-fit tests (Breiman 2001). These models tend to occupy the left side of the triangle in *Figure 6*, with typically a high level of precision and potentially high level of detail realism (Dickey-Collas *et al.* 2014). However, since the structure of these models is based on historical associations, rather than the causal mechanisms of the real-world system, the structure-behavioural realism of the model is typically low. Any conclusions drawn will be about the mechanisms of the model, not about the mechanisms of the system (Breiman 2001). It follows that model generality tends to be relatively low (Dickey-Collas *et al.* 2014), and the utility of statistical models for studying novel conditions, policy changes and system change outside the range of historical data is limited (Evans 2012).

Algorithmic models

In contrast to statistical models, in algorithmic models no direct causal assumptions are made by the modeller about the real-life process that produces observed relationships between inputs and response. Instead, the modeller uses ML algorithms (*e.g*. artificial neural network, random forest *etc*.) to find patterns in the data, which allows the model generated to operate on the inputs and predict the outputs (Breiman 2001; Schoenberg & Swartz 2021). These models occupy the leftmost corner of the triangle in *Figure 6*, as they typically have a very high level of predictive power (Breiman 2001).

However, because these models lack an underlying causal structure grounded in real-world mechanisms, they say nothing about the causal connections between input and output variables (Ellner & Guckenheimer 2011). Furthermore, because of the lack of realistic representation of the underlying causality mechanisms, once trained/calibrated, generality tends to be low and even though much attention is paid to "over-fitting" (a measure of the utility of the generated model's structure to datasets beyond the one used to initially construct the model), their predictive power is limited to the conditions and data range to which they were originally calibrated/trained (Baker *et al.* 2018). Thus ML models struggle to predict outcomes that were not in the original training dataset (Kim *et al.* 2017). In dynamic and constantly evolving systems like CHANS, where conditions and context can make abrupt shifts and system behaviour is expected to stray outside previously observed ranges, this condition is frequently violated (Schlüter *et al.* 2019a; Quinn & Quinn 2020). However, the greatest weakness of most advanced algorithmic models is lack of interpretability, and lack of ability to answer the question 'why?' (Voosen 2017). The mathematical structure of the underlying machine algorithms typically cannot be interpreted in any meaningful way, and the logic of their learned associations is not understood even by the experts who build them (Rudin 2019; Schoenberg & Swartz 2021). These models are like black boxes and therefore have limited use for building system understanding or exploring different policy and management scenarios (Maeda *et al.* 2021).

Formal mechanistic models

In contrast to descriptive models, mechanistic models seek to establish mechanistic relationships between variables and mimic the real-world processes that give rise to observed patterns and phenomena (Baker *et al.*

2018). The mechanisms by which different elements in the system influence one another are typically derived from theory, first- and second-order principles, and qualitative and quantitative empirical data.

Several sub-categories of mechanistic models exist. The categorisation below is based on how they treat time.

Static models

Static, or non-temporal, models make no reference to time. Their focus is on understanding the logical or quantitative relationships between system variables in a given instance (Hunt *et al.* 2008). Non-temporal models can be used to find the equilibrium (steady) state of a system variable given knowledge about other system variables, and the models can then be used to predict what the state of the system *would be* in a counterfactual situation where the inputs or relationships are manipulated (Law *et al.* 2007).

Examples include many traditional analytical models, *e.g.* the relationship $d = rt$, where the rate of travel, r, multiplied by the time spent travelling, t, equals distance travelled, d. This model accurately describes the relationship between r, t and d, but not how the system might evolve over time (Hunt *et al.* 2008). This type of analytical model is fully interpretable and found at the base of the triangle in *Figure 6*, *i.e.* it has a low level of realism, but high generality and high precision (Dickey-Collas *et al.* 2014).

Conventional partial equilibrium (PE) models and computed general equilibrium (CGE) models also belong to the category of non-temporal models (Mitra-Kahn 2008). These models use sector- or economy-wide input-output tables to identify quantitative relationships between all monetary flows of resource inputs and production outputs in a system. The parameters of the functional relationships between inputs and outputs are calibrated to find an equilibrium between supply and demand that corresponds to some historical baseline (usually a year for which there are good data). Policy scenarios are then explored by shocking the model by introducing a step change in some exogenous variable and observing how the change propagates through the system as it adjusts to a new equilibrium state (Mitra-Kahn 2008).

An important note on the static/non-temporal family of models is that these models describe the system as having no "memory". When the inputs change, the system finds a new equilibrium based solely on these new input values, regardless of its previous state. In other words, these models operate under the assumption that history (other than the historical data used for calibration) does not matter to the future state of the system (Anderson & Cavendish 2001). Another aspect stemming from the lack of reference to time in these models is that the transition path the system takes to move from one equilibrium state to another, and the time required to make this transition, is unknown (Böhringer & Löschel 2006). In other words, one cannot be certain when, or if, the system will reach the anticipated new equilibrium.

Dynamic models

In contrast to static models, dynamic models are used explicitly to study how a system changes over time as a result of the causal interactions between system components. The evolution of the modelled system is represented either as occurring in discrete steps or as continuous with respect to time, and the models are typically resolved using computer simulation techniques (Page 1999; Law *et al.* 2007). These models occupy the right side of the triangle in *Figure 6*, with greater emphasis on realism and generality than on point-by-point precision (Dickey-Collas *et al.* 2014). They try to mimic real-world processes by using mathematics to represent the system as consisting of elements that can take on different states (*e.g.* the number of people in a city or the balance in a bank account). The state of an element at any point in time is dependent on its value in the previous time steps, and it can influence the state of other elements in the system. Feedback effects and endogenous dynamics can be modelled by allowing the current state of an element to directly or indirectly influence its future state (Schoenberg & Swartz 2021).

Discrete dynamic models

Discrete dynamic models, or discrete event models (DEM) (Law *et al.* 2007), include *e.g.* Markov models (MM) (Page 2018), state and transition models (STM) (Bestelmeyer *et al.* 2017), and most agent-based models (ABM) (Bonabeau 2002). All these apply the assumption that the elements in the system can take on and move between different states, and that this transition occurs instantaneously at a countable number of discrete time points. These time points represent events happening in the system (*e.g.* an arrival or departure, or an ecosystem transitioning from forest to savannah) at synchronous or asynchronous time increments (Law *et al.* 2007). In other words, in DEM time moves forward by regular or irregular “leaps” at which the system states are updated to take on new values.

In MM, which are frequently used in social systems modelling (Page 2018), and in STM, which are commonly used for studying landscape and ecosystem dynamics (Daniel *et al.* 2016; Bestelmeyer *et al.* 2017), the elements move between the different states according to a set of transition probabilities. These represent how likely it is that an element will move from one state to any other state and can be fixed (Page 2018), or influenced by other state variables in the system (Daniel *et al.* 2016).

In contrast to MM and STM, in ABM the system is represented as consisting of autonomous agents that interact with one another and the environment according to predefined rules (Bonabeau 2002). Each agent has internal state variables representing different attributes, and the agent can transition between these states when triggered by its internal dynamics or by its interactions with the surroundings. It is the internal state of each individual agent that, through the behavioural rules it is defined to follow, determines its actions. A defining feature of ABM is that feedbacks are not explicitly coded into the model structure as is the case in *e.g.* STM and SD models (see below). Instead, feedback can occur as an emergent property when an action taken by an agent to modify its own state later comes back to affect the same agent and the same state in a future time period (Schlüter *et al.* 2021b; Schoenberg & Swartz 2021).

Overall, ABM are very flexible and can realistically simulate emergent patterns from human behaviour in ways most other modelling methods cannot (Bonabeau 2002). The approach is particularly suitable when modelling individual entities with discontinuous behaviour and which cannot be well described at the aggregate level, and/or when the complexity of behavioural rules makes equation-based representations intractable (Bonabeau 2002). ABM provide the additional advantage that they allow for the creation of models in situations where knowledge about interdependencies at the aggregate level is limited. For instance, one may know very little about the global feedback structure of a system but still have some information about the behaviour rules of the individual elements. In this case, the behavioural rules can be used to create an ABM that can be used to obtain information about the global feedback structure (Borshchev & Filippov 2004).

The flexibility of ABM and their focus on emergence have made this modelling approach very popular for studying CHANS and other complex social systems (Hammond 2015; Schlüter *et al.* 2019b). However, the modelling approach is less suitable for modelling physical systems (Herrera *et al.* 2018), and the approach in ABM to mimic reality from the micro level up means that, even for small models, the level of detail complexity can make it very challenging to understand the processes driving simulation results (*why* the model behaves as it does) (León-Medina 2017). This can cause difficulties in analysing simulation results and in effectively communicating results to external users (Schlüter *et al.* 2021b).

Continuous dynamic models

The last group of models presented here are continuous dynamic models. These include SD models, but also dynamic systems (DS) models (Borshchev & Filippov 2004) and some variations of continuous ABM. Since ABM are covered above and DS modelling is typically restricted to non-human systems (Borshchev & Filippov 2004), the focus in this sub-section is on SD modelling, which is frequently used in CHANS research (Schlüter *et al.* 2019c; de Vos *et al.* 2021).

In contrast to ABM, SD models are typically constructed at a higher level of aggregation, focusing on dynamic complexity as opposed to detailed complexity. The world is represented as consisting of material and information stocks (state variables where material and information are stored and accumulated) that are regulated by flows representing the rates of change between stocks. Mathematically, SD models consist of simulated differential or difference equations (Richardson 2009), but they are typically constructed using graphical notation (see section 4.6.4), making the model structure explicit and more easily interpretable also for non-modellers (Black 2013).

All SD modelling is built around the founding principle that system structure is a key determinant of dynamic behaviour (Richardson 2009). To this end, the method has an explicit focus on the role of feedbacks and time delays in understanding real-world problems. Feedbacks occur when the level of a stock at one point in time directly or indirectly influences its own future flow(s) and can be either positive (reinforcing past changes in the stock) or negative (dampening/balancing past changes in the stock). Delays between cause and effect, created by the fact that it takes time for information and material stocks to adjust to change, are common in complex systems and, together with feedback loops, they are a major source of complex dynamic behaviour (Sterman 2000). In dynamically complex systems, multiple feedback loops operate simultaneously and, as the system evolves, the relative strength of the different loops changes and gives rise to complex nonlinear dynamics (Sterman 2000).

The group of models in *Figure 7* that satisfactorily fulfils all three requirements of treatment of structure, time and feedback are those belonging to the mechanistic dynamic group. Within this family of models, SD was selected as the primary modelling approach in studies of CHANS in this thesis, due to its potential to produce models of considerable realism and generality (Ip *et al.* 2013) with maintained interpretability. SD is also well-documented for its utility in environmental management (Kelly *et al.* 2013) and for exploring and evaluating effects of alternative policy strategies and social and environmental disturbances (Schlüter *et al.* 2019c). In studies of micro-level social interactions, on the other hand, ABM was used due to its suitability for realistic modelling of human interactions and its capacity to elicit global-level emergence from local-level behavioural rules (Schlüter *et al.* 2021b).

4.5 Research methods used in Papers I-V

The studies conducted in this thesis ranged from theoretical literature-based studies (Paper I) to highly contextualised, case study-based research (Papers II and III) and simulation-based method development research (Papers IV and V). Despite this difference, the work in all papers was rooted in a systems perspective of the world. Therefore, the research approach chosen drew heavily from the fields of systems thinking, system dynamics modelling and agent-based modelling.

Sub-sections 4.5.1-4.5.4 elaborate on the research approach and methods used in Papers I-V. For interested readers, a more in-depth description of key methods can be found in section 4.6 and in the attached papers. Readers already familiar with the methods, or who only want an overview of the overall design of each study, can skip section 4.6 and move straight to Chapter 5.

4.5.1 Paper I – a qualitative systematic review of the literature

Paper I in brief

Type of study: Qualitative systematic review

Purpose: Explore how CHANS are represented in the LCA methodology.
Synthesise methodological limitations.
Suggest avenues for improvements.

Knowledge type: Descriptive & prescriptive

In Paper I**,** a qualitative systematic review (Grant & Booth 2009) was conducted to explore the extent to which defining features of CHANS, exemplified by bio-based production systems, are accounted for in mainstream LCA methods used for environmental policy assessment, planning and management. Other objectives were to synthesise limitations of contemporary methods used for environmental sustainability assessment of these systems, and to suggest avenues for methodological improvement. The review was conducted according to the well-established PRISMA (Preferred Reporting Items for Systematic Reviews and Meta-Analyses) guidelines for systematic literature reviews (Liberati *et al.* 2009) (*Figure 8*). A total of 780 published scientific papers and 104 book sections, technical reports and policy briefs were identified in bibliometric searches conducted in Web of Science and Google Scholar. Following a systematic screening process, titles and abstracts were scanned and papers were selected based on three inclusion criteria:

(1) Studies identifying and/or addressing limitations and weaknesses of LCA and other life cycle-based methods for environmental sustainability assessment.

(2) Studies addressing environmental sustainability assessment of bioeconomy or bio-based production systems.

(3) Studies focused on improving or developing approaches and tools for environmental assessment and planning of bio-based production systems.

The final dataset consisted of 107 scientific articles and 28 book sections and non-academic texts, which were analysed and synthesised using a matrix-based method (Goldman & Schmalz 2004).

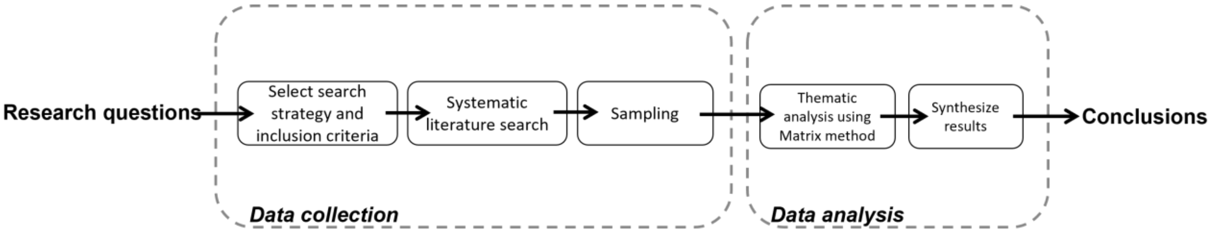


Figure 8. Graphical methods description for Paper I.

4.5.2 Papers II & III – qualitative and quantitative systems modelling to understand the past, present and future of a coupled social-hydrological system

Paper II in brief

Type of study: Qualitative modelling case study.
Purpose: Identify the socio-hydrological drivers of water supply and demand.
Explain endogenous mechanisms to policy resistance.
Identify system leverage points.
Main methods: Literature review
Key informant interviews
Participatory system mapping
Feedback loop analysis
Knowledge type: Descriptive, exploratory & explanatory

In Paper II and Paper III, system modelling case studies were conducted on Fårö island (57.9°N, 19.1°E), part of Gotland municipality, Sweden (*Figure 9*). This is one of the most water-stressed regions in Sweden, with significant seasonal variations in water supply and demand due to relatively limited groundwater availability and a large tourist sector that competes for water with other users. In Paper II, literature studies, empirical data, key informant interviews (Shackleton *et al.* 2021) and participatory system mapping methods (Lopes & Videira 2017) were used to collect and generate data about key causal drivers of water supply and demand on Fårö. Through an iterative process of data triangulation, theory development, participatory validation and theory adjustment, the data were synthesised into a causal loop diagram (CLD), a conceptual model of the socio-hydrological feedback processes governing the dynamics of water supply and demand on Fårö. Qualitative feedback loop analysis was conducted to examine the causal drivers of seasonal water scarcity, and to develop a structure-based theory of the systemic causes of policy resistance to water scarcity mitigation measures experienced in the past 20 years. The results were also used to provide directions for future research and to inform future water management strategies.

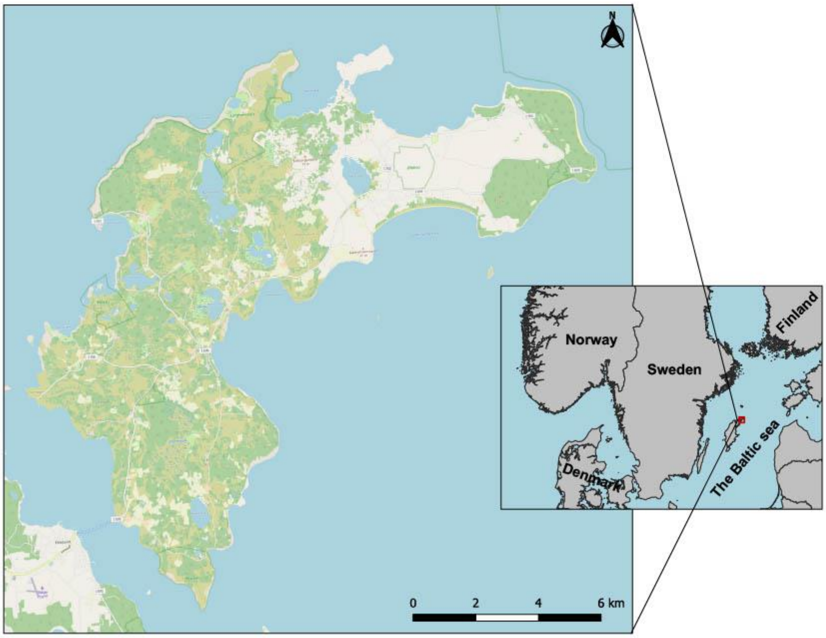


Figure 9. Map of Fårö. Location in the Baltic Sea indicated by red box in the small map of the Nordic region. Reproduced with permission from Nicolaidis Lindqvist *et al.* (2022).

Paper III in brief

Type of study: Quantitative modelling case study.

Purpose: Explore effects of future climate scenarios on drinking water supply, socioeconomic development, and water demand.

Main methods: Key informant interviews
System dynamics modelling
Budyko-based hydrological modelling
Monte Carlo simulations

Knowledge type: Descriptive, exploratory & prescriptive

Following a mixed-methods approach (*Figure 10*), insights from Paper II informed the research direction for Paper III, where an integrated social and hydrological dynamic simulation model was developed to explore how future climate scenarios (2020-2050) on Fårö may affect drinking water supply, and subsequent dynamic effects on housing, tourism and municipal water services. Data collected in Paper II were complemented by further qualitative and quantitative data collection through additional interviews and dialogue with key informants from the municipal water utilities company, hydrogeologists from the Swedish Geological Survey (SGU) and representatives from the tourist sector on Fårö. Municipal and public databases were used for collecting empirical data on historical water use, tourism development, climate and groundwater levels. The model was developed using the SD modelling method, applying a Budyko-based approach for water balance modelling with limited data (Zhang *et al.* 2008). It was calibrated to available historical data and validated using both statistical procedures (calculation of Theil inequality statistics, testing for bias, unequal variance and unequal covariance between simulated results and historical data (Sterman 1984)) and non-statistical procedures (including direct and indirect structure tests, parameter sensitivity tests, extreme condition tests *etc*. (Barlas 1996; Schwaninger & Groesser 2016)). To explore a wide outcome space of plausible futures, multivariate Monte Carlo simulations were conducted in a series of 1000 simulation experiments where model parameters governing future climate, housing supply and demand, tourism growth, groundwater quality and quantity, and per capita water use were randomly varied. The mean and extremes of the simulated ensemble were compared against those in two historical reference periods (1961-1990 and 2000-2020) to evaluate effects on future water availability. Socioeconomic impacts were evaluated by comparing the ensemble results to a hypothetical scenario with unconstrained water supply.

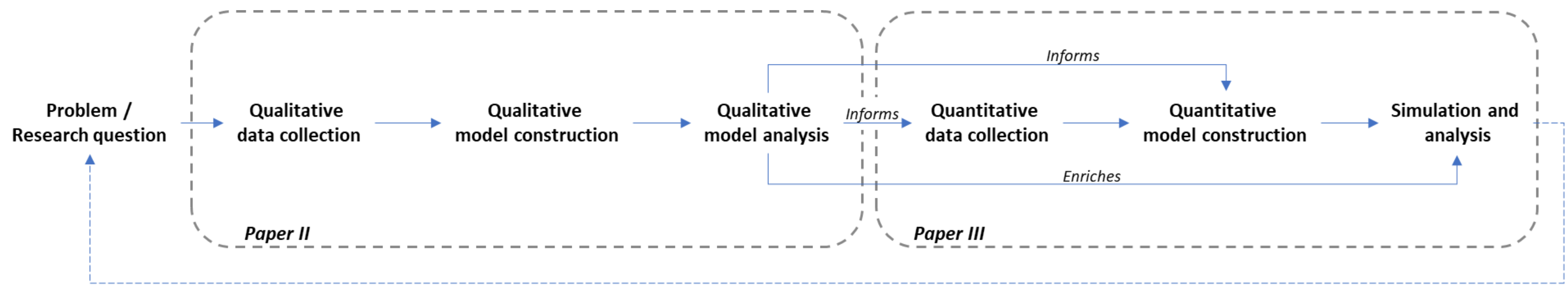


Figure 10. Conceptual diagram of the mixed methods approach applied in Papers II and III. Adapted from (Creswell 2009).

4.5.3 Paper IV – Dynamic marginal cost curves for assessing water scarcity mitigation measures

Paper IV in brief

Type of study: Method development study.

Purpose: Address limitations of conventional MCC methods.

Main methods: Marginal cost curve calculations
System dynamics modelling

Knowledge type: Prescriptive

A marginal cost curve (MCC) is a decision support tool frequently used for assessing and ranking the marginal cost effectiveness of alternative measures, investments and strategies in environmental management and policy design (Kesicki & Strachan 2011; Jiang *et al.* 2020). The method has been criticised for not accounting for important systemic features in the assessment: ancillary benefits and costs, systemic complexity, temporal dynamics, and interaction effects between measures (Kesicki & Ekins 2012; Huang *et al.* 2016; Jiang *et al.* 2020). In Paper IV, a method development study applying SD modelling was conducted with the aim of addressing these limitations and exploring new policy insights that a dynamic MCC can provide to decision makers compared with conventional MCC methods.

Using published data (Sjöstrand *et al.* 2019), complemented by literature studies, a SD model for marginal cost assessment of four water scarcity mitigation measures was developed and applied in a semi-hypothetical case study of a Swedish city. Simulation experiments were conducted for a total of 15 mitigation mixes. The marginal and average cost for each mitigation mix were analysed, and MCCs for each mix were derived. Effects on four ancillary outcome indicators (groundwater withdrawal, service capacity, net water supply, municipal water price) were also analysed, and the mitigation mixes were ranked based on their performance for each indicator.

4.5.4 Paper V – Accuracy or alignment: A conflict in the participatory modelling process?

Paper V in brief

Type of study: Simulation experiment and method development study.

Purpose: Explore the effects of social dynamics on participatory system mapping effectiveness and guide design of future mapping interventions.

Main methods: ABM

Knowledge type: Exploratory & prescriptive

Paper V revisited the participatory system mapping approach applied in Papers II and III, and in numerous other studies in sustainability transition research (Stave 2010; Stave *et al.* 2017; Sterling *et al.* 2019; Carnohan *et al.* 2021; Coletta *et al.* 2021). A desirable result of a participatory modelling intervention is to reach alignment around an accurate shared model of the causal structure of the problem at hand, or the system one is trying to understand (Vennix 1999; Smetschka & Gaube 2020). The method is grounded in the belief that complex problems can best be understood and addressed through integration of information and perspectives from a diversity of mental models (Costanza & Ruth 1998; Videira *et al.* 2009). Research has shown that by combining information from a cognitively heterogeneous group of people, the "collective intelligence" of the group often outperforms even expert individuals exposed to the same task (Page 2007; Woolley *et al.* 2010). If this is the case, it can be used to leverage the decision making process (Aminpour *et al.* 2021). However, there is an equally rich base of literature showing how social dynamics in small groups can amplify pre-existing mental biases and impact individual perceptions about the world (Bang & Frith 2017; Becker *et al.* 2017), leading to herding behaviour (or "group think") (Bikhchandani *et al.* 1992; McCauley 1998; Muchnik *et al.* 2013). This often reduces the accuracy, but increases alignment, in the group (Lorenz *et al.* 2011; Bang & Frith 2017; Kao *et al.* 2018).

Based on the above dichotomy, the starting hypothesis in Paper V was there can be a conflict between accuracy and alignment in participatory modelling settings. An ABM was developed, drawing from the group model building (GMB) literature, and well-established theories on social behaviour in small groups. The system mapping process of a GMB intervention was simulated and the participants were represented as autonomous and interacting agents. Each agent had its own mental model of a hypothetical real-world system, represented as a unidirectional network with 18 nodes and 23 edges. The mental models of the agents were updated and changed as a consequence of their social interactions with one another. Just as in a GMB intervention, the objective of the agents was to combine information from their different mental models to create an accurate consensus model of the real-world system. The hypothesis was tested by simulation experiments that explored the interplay between accuracy and alignment under different group compositions. The accuracy and alignment of the group were measured as the Jaccard distance (Jaccard 1912) between the consensus model and the real-world system, and between the mental models of the agents. The student's t-test was used to analyse whether the workshop process improved the accuracy and alignment of the group, and principal component analysis (PCA) was used to evaluate the determining factors of these improvements.

4.6 Methods and tools discussed

4.6.1 Qualitative systematic review [Paper I]

To push the research frontier and advance the state of current knowledge, one must first know where the current frontier lies. This entails understanding the breadth and depth of current knowledge, as well as its weaknesses, inconsistencies, gaps or contradictions. To this end, reviewing existing prior work is an essential part of the research process in academia (Xiao & Watson 2019). Through the process of reviewing, summarising, analysing and synthesising the available literature on a topic, it is possible to test hypotheses, develop new theories and evaluate the validity and quality of these against predefined criteria (Paré *et al.* 2015).

In scientific research, a distinction is made between a "scoping review" and a "systematic review" (Jesson *et al.* 2011, p. 105). The former is often conducted in a typical literature review and serves as part of the introduction and empirical background of empirical research studies. It provides the theoretical foundation, justifies the study, substantiates the research problem and/or frame the research question/s, hypotheses and research approach (Paré *et al.* 2015). The aim of the scoping review is often to gain a broad understanding and 'paint a big picture' of a field of research. It is conducted without a clearly framed research question, and without the requirements of a systematic, documented, and repeatable process for identifying, selecting, assessing, synthesising, and analysing the relevant literature.

In contrast to the scoping review, the systematic review is conducted with tightly defined aims, objectives and research questions (Jesson *et al.* 2011). It is often used as an approach to assess whether contemporary knowledge and practices in a field are based on relevant (sometimes conflicting) evidence, in order to identify gaps or deficiencies in the current evidence or practices and thereby guide future research in the area (Munn *et al.* 2018). A fundamental feature of the systematic review lies in the requirement for validity, transparency, and repeatability. Hence, a systematic review should have a rigorous, transparent, documented, and repeatable process for all steps of the review process (*Figure 11*). The process includes formulating the problem and research question, developing a review protocol (documenting the search strategy, inclusion and exclusion criteria, a strategy for quality assurance, methods for analysing results, *etc*. (Jesson *et al.* 2011)), a literature search, screening identified papers for inclusion, quality assessment, data extraction, analysis and interpretation of results, and reporting findings. The systematic review must provide a methodological report, including the review protocol, where the process conducted is thoroughly documented to ensure transparency and repeatability of results (Xiao & Watson 2019).

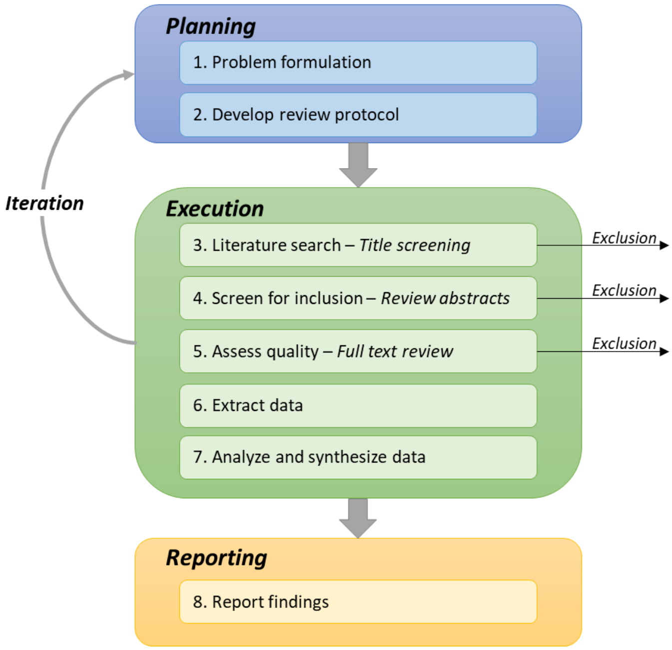


Figure 11. Conceptual representation of the systematic review process. Adapted from Xiao and Watson (2019).

Over the years, several subcategories of reviews have been developed in different fields and for different purposes. For instance, Grant and Booth (2009) describe 14 review types that apply different degrees of systematic approaches. In Paper I, a qualitative systematic review (Grant & Booth 2009; Paré *et al.* 2015) was conducted. This is an approach, adhering to a systematic review process, used to integrate and compare qualitative research findings by means of narrative synthesis and thematic analysis (Grant & Booth 2009).

4.6.2 Life cycle assessment (LCA) [Paper I]

LCA is an environmental assessment and planning tool that compiles and evaluates the environmental impact of a product or production system throughout its life cycle (ISO 2006; Sala *et al.* 2016). Two broad types of LCA exist: attributional LCA (ALCA) and consequential LCA (CLCA). ALCA is typically used retrospectively for product-level comparative studies (*e.g.* to compare the environmental performance of two functionally equivalent products) and it only accounts for the immediate physical flows and direct environmental impacts associated with a product's life cycle (Zamagni *et al*. 2012; Sala *et al.* 2016). In contrast, CLCA is forward-looking and is used to assess how the global environmental burden of a system will change in response to the introduction of a new policy or other disturbances (*e.g*. "what would be the environmental impact of policy X?"). This requires the system boundary of the CLCA to be expanded to include both direct consequences to the primary system under study, and indirect consequences to surrounding systems affected by the change (Yang & Heijungs 2018; Ekvall 2019). The prospective nature and consequential focus of CLCA make it the preferred type for policy analysis (Sala *et al.* 2016; Reale *et al.* 2017; Yang & Heijungs 2018) and it is the approach focused on in this thesis and in Paper I.

In practice, LCA consists of four steps. First, the goals, objectives, and the functional unit of the assessment are defined. The second step is life cycle inventory (LCI), where the basic principle is to define a boundary between the product system under study (the 'technosphere') and the surrounding natural and human environment. The flows of material and energy inputs, and emissions and waste outputs (collectively referred to as elementary flows) between the product system and the environment are then tracked through all stages of the studied life cycle. These flows are aggregated and presented as a "total emissions of substance X and total use of resource Y" (Hauschild 2005). The third step is the life cycle impact assessment (LCIA), where the inventory of aggregated elementary flows is translated into information about the potential environmental impact of the product system. This is done by mapping the elementary flows in the LCI to relevant environmental impact categories and then multiplying the flows by substance specific characterisation factors (CFs) to derive potential environmental midpoint and endpoint impacts. Results are normalised to common impact specific units, and the total damage to each impact category is calculated as the sum of impacts from all elementary flows mapped to that category. This yields a profile of environmental impact and resource use of the product system, which is interpreted and evaluated in relation to a reference scenario in the fourth step of the assessment (Hauschild 2005; Hauschild *et al.* 2013).

Models are used in both the LCI and LCIA steps in the LCA. In more basic assessments, the LCI typically makes use of simple process or input-output (IO) models to predict how the elementary flows change in response to a policy change (Yang & Heijungs 2018). These approaches have been criticised for being too simplistic and for relying on unrealistic assumptions about linearity. For instance, IO models implicitly assume that an X% increase in demand for a product will lead to a proportional change in elementary flows for all levels of demand. This assumption ignores effects of resource supply constraints and assumes an unsaturable market for all products and by-products (Yang & Heijungs 2018). Due to these limitations, more advanced equilibrium-based models are increasingly being used for LCI modelling, as these allow market mechanisms and nonlinear cause-effect relationships to be incorporated into the LCI assessment (Yang & Heijungs 2018). In the LCIA step, model use is embedded in the CFs. These CFs are derived from substance-specific impact assessment models of the environmental mechanisms (the impact pathway) linking physical flows of resources and pollutants with their potential environmental impacts (Hauschild *et al.* 2013). In practice, the impact models are typically not developed as part of the LCA, but regional or global CFs for each elementary flow are obtained using pre-existing model databases such as ReCiPe (Goedkoop *et al.* 2008) or IMPACT2002+ (Jolliet *et al.* 2003).

In modern LCA software, inventory databases and impact assessment methods are both built-in features. This significantly simplifies and speeds up the LCA process, but it also limits the flexibility of the approach and the possibility to conduct context-specific assessments (Deutsch & Troell 2021).

4.6.3 Case study research [Papers II & III]

Case study research involves close, in-depth and detailed scientific investigation of a real-life phenomenon within its specific context (Ridder 2017). It is a particularly useful method in descriptive, explanatory and exploratory studies (*e.g.* inquiries about how or why an observed phenomena occurs), when the studied phenomenon cannot be isolated from its real-world context and/or when the investigator has limited control over events (Yin 2009). Case studies are frequently used in CHANS research because the method allows the researcher to closely examine the relationships and interactions between system components in their natural settings (Herrera *et al.* 2018), and because of its usefulness for developing new theory by collective and comparative case study analysis (Magliocca *et al.* 2018; Pahl-Wostl *et al.* 2021).

A challenge in case study research lies in the richness and extensiveness of the real-life context compared with that of carefully controlled experimental design approaches. In the laboratory it is possible to control for confounding variables, but when conducting case study research, especially when studying CHANS, the confounding variables and their interactions are part of the research interest. Furthermore, there will often be more variables of interest than data points (Yin 2009). This often calls for use of multiple sources of evidence, converged through a process of careful and iterative triangulation (Yin 2009; Crowe *et al.* 2011; Biggs *et al.* 2021b), as done in the case study conducted in Papers II and III.

Crowe *et al.* (2011) describe the case study approach as having four phases:

(i) Defining the phenomenon of interest

Carefully scoping the phenomenon of interest and formulating relevant research question(s), informed by existing literature, theory and/or local or theoretical puzzling issues, are both part of defining the case. This step also entails defining the spatial, temporal and contextual boundaries of the study (Crowe *et al.* 2011). In Papers II and III**,** the phenomenon of interest was social-hydrological mechanisms of water supply and demand, and the endogenous mechanisms of policy resistance, studied from a contemporary Nordic perspective.

(ii) Selecting what case(s) to study

Selecting what case(s) to use depends on the objectives and research questions of the inquiry. In more intrinsic studies, the objective is to learn about a unique phenomenon that is distinguished from the norm, or from what could be expected from existing theory. In these studies, the case(s) is selected because of its uniqueness, not because it is representative of other similar cases. Case studies can also be of a more instrumental type where a "typical" case that is considered representative of the phenomenon is studied to gain a broader understanding of the issue and complement previous studies (Crowe *et al.* 2011). The Fårö case in Papers II and III had both instrumental and intrinsic features. Social-hydrological case studies have been conducted in other parts of the world and theories of human-water interactions have been developed for similar cases (see *e.g.* Penny and Goddard (2018)). However, the geographical and institutional setting of the study in Papers II and III (within the Nordic region) makes the case unique and novel. Lastly, case study research can also be of a collective and comparative type where multiple cases are studied and compared to build even broader knowledge and understanding about the phenomenon, the conditions in which it occurs, and to test existing, or develop new, theories (Crowe *et al.* 2011; Pahl-Wostl *et al.* 2021).

(iii) Collecting the data

Case study research typically involves collecting data from multiple sources of evidence, often combining qualitative (*e.g.* interviews, focus groups, observations *etc*.) and quantitative techniques (*e.g.* questionnaires, statistical data *etc*.) as done in Paper II and III. These mixed-method approaches are common as they allow for a deeper understanding of the phenomenon under study than any single method could achieve on its own (Crowe *et al.* 2011). As all methods have their limitations and biases, combining different data collection methods that complement one another is an efficient way to increase the internal validity of the study (Schlüter *et al.* 2021a).

(iv) Analysing, interpreting and reporting results

A process of careful and iterative triangulation is often used to analyse the combination of qualitative and quantitative data obtained in case study research (*e.g.* combining quantitative statistical methods with qualitative methods such as grounded theory, systems mapping or thematic analysis) (Yin 2009). This process allows the researcher, as exemplified in Papers II and III, to converge towards a common interpretation of the phenomenon under study. Findings from case studies can be used for testing, challenging, and strengthening existing theory, and for providing explanations for the phenomenon in certain circumstances. It can also be used to develop new theory, and to generalise beyond the studied case (Flyvbjerg 2006; Crowe *et al.* 2011).

4.6.4 System dynamics modelling [Papers II, III & IV]

System dynamics modelling was briefly introduced together with a set of other modelling approaches in section 4.4.2. This section provides more historical and philosophical background, and practical details of the method.

The field of SD originated in the early work of Professor W.J. Forrester at Massachusetts Institute of Technology (MIT) in the 1950s, with the cornerstones of the underlying philosophy and methodology presented in Forrester's *Industrial Dynamics* (Forrester 1961), and in the later *Principles of Systems* (Forrester 1968) and *Urban Dynamics* (Forrester 1969). In his 1988 Killian Faculty Award lecture, Forrester described SD as a field of research dealing with "high order, nonlinear, systems involving the interactions of people, nature, and technology, based on a feedback structure viewpoint" (Forrester 1988). SD is extensively drawing data and information from "the mental models in the world around us, converting those into computer models so that we come to understand better what those models imply" (Forrester 1988). Richardson (2009) extended this definition by describing SD as an approach to build theories, analyse policies, and provide strategic decision support from an endogenous point of view with the help of computer simulations. The approach is applicable to dynamic problems arising in complex systems characterised by interdependencies, delays, nonlinear behaviour, and material and information feedback (Richardson 2009).

At the heart of the SD approach is the understanding that the dynamic behaviour of systems arises endogenously from their internal structure (Sterman 2000). The process of constructing SD models thus involves identifying and linking the relevant pieces of structure, and using simulation to study the behaviour it generates (Radzicki 2010). From the SD perspective, all systems are made up of the same basic set of structural elements: stocks, flows and chains of causal connections (information links) forming feedback loops (Richardson 2011). In addition to these key structural features, SD models typically also include converter variables. These represent constants, limit factors, or auxiliary operations that detail the cause-effect relationships between variables in the system.

Stocks are state variables. They represent the accumulation of flows of material or information over time. This characteristic also makes stocks responsible for decoupling of flows, for creating delays, and for preserving system memory. Examples of stocks include the volume of water in a lake, CO_2 levels in the atmosphere, the population of a city, the level of national debt, or an individual's perceived level of stress.

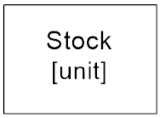
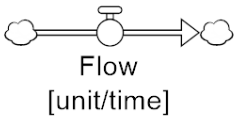
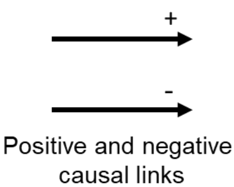
Flows represent the fluxes of material, resources and information between stocks in a system. The amount of material or information in a stock can only be regulated by changing the rate of its inflows and outflows. Therefore, the net inflow or outflow of a stock represents its rate of change.

Feedback loops are circular chains of information about the level of information or material in a system's stock that operate by affecting its own flows. Feedback occurs when information about a change in the level of a stock travels, directly or indirectly, from the stock back to its flow(s), causing a change in the rate of the flow(s) in a future time step. Feedback loops can act to either reinforce the original change (termed positive feedback), resulting in self-reinforcing behaviour, or counteract/dampen it (termed negative or balancing feedback), giving rise to stabilising or goal-seeking behaviour.

Mathematically, SD models consist of series of simulated differential or difference equations (Richardson 2009) that are implemented using a visual, object-oriented approach in the form of stock-and-flow diagrams (SFD). The continuous evolution of time in real-world systems is approximated in the simulation by breaking up time into small incremental time steps (ΔT). For each such interval, the net flow of material and information into and out of the system's stocks is calculated and added to the level from the previous time step. The process is repeated throughout the simulation, to approximate the continuous evolution of the real-world system (Radzicki 2010).

A summary of the key elements in SD models is provided in Table 2 and *Figure 12*.

Table 2. *Description of the key elements in system dynamics models*

Element	Description
	Stock variables are the integration of all inflows and outflows. $Stock(t) = \int_{t0}^{t} \big(inflow(s) - outflow(s)\big)dt + Stock(t0)$ (Eq. 1) Thus, the change in the level of a stock is equal to its net flows over the simulated time step. $Stock_t = Stock_{t-1} + DT * (inflow\ - outflow)_{t-1}$ (Eq. 2)
	Flows represent the flux of material and information in a system, and they thus govern the change over time in their associated stocks. One stock can have multiple inflows and outflows, and flows can connect multiple stocks to represent supply chains, ageing chains or information cascades. The clouds represent the system boundaries of the model.
	Converters can represent system constants, limiting factors or auxiliary calculations defined by the modeller to represent the causal relationships between variables in the system. Converters can also be used to represent empirical nonlinear relationships between system variables using graphical functions (lookup tables).
	Causal relationships between variables are represented by arrows in the stock-and-flow diagram (SFD). A + or – sign at the arrowhead indicates the direction of causality. A positive (+) causal link means that a change in the tail variable causes the head variable to change in the *same* direction. A negative (-) causal link means that a change in the tail variable causes the head variable to change in the *opposite* direction.

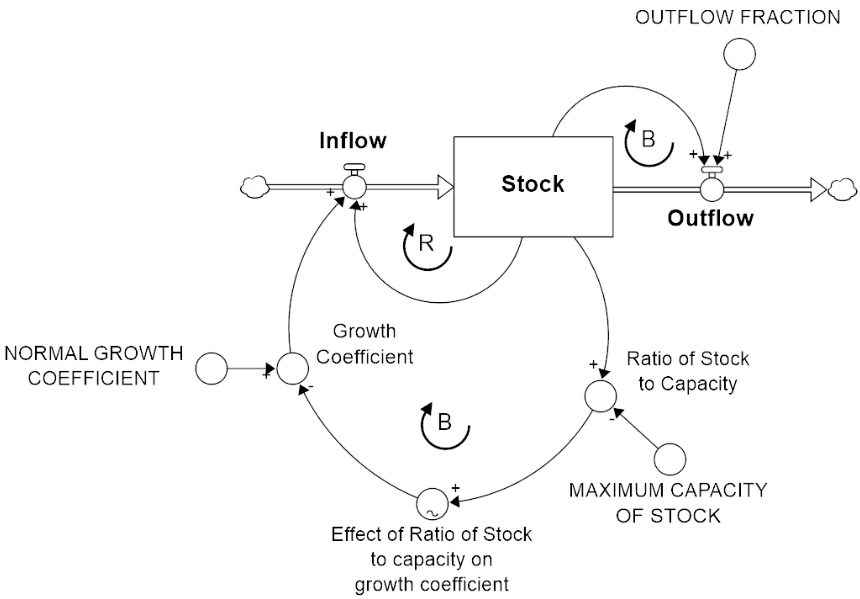


Figure 12. Example of a simple system dynamics (SD) model with one stock, two flows and three feedback loops indicated by curved arrows, with a capital R (reinforcing) or B (balancing) to indicate their effect. Constants and limit factors are indicated by capital letters, and graphical functions are denoted by a converter with a "~" sign. Adapted from Radzicki (2010).

Constructing, testing and evaluating an SD model is a highly iterative process that has been described in detail by several authors (Forrester 1994; Sterman 2000; Martinez-Moyano & Richardson 2013). The main steps involved are:

1. *Problem identification and definition*. This entails clearly defining the problem to be addressed, articulating why this is a problem and to whom, describing the problem in terms of its behaviour over time, identifying key variables and concepts, and setting the system boundaries.

2. *System conceptualisation***.** This step involves formulating a dynamic hypothesis about the system structure responsible for endogenously generating the problematic behaviour. The dynamic hypothesis is typically presented in the form of a stock-and-flow diagram, causal loop diagram, or some other system mapping tool.

3. *Model formulation***.** In the model formulation step, the conceptual model from step 2 is translated into a mathematical simulation model by defining mathematical relationships between variables, defining decision rules, estimating parameter values, and setting initial conditions.

4. *Model testing and evaluation***.** Model testing and evaluation is conducted to evaluate the level of realism and precision of the model, and to build confidence in the simulation results. This involves an exhaustive set of model tests, including structure behaviour tests, behaviour replication tests, extreme condition testing, sensitivity tests *etc*. For a detailed description of model testing, see Sterman (2000).

5. *Policy design, test and evaluation.* Once it has been developed and tested, the model can be used to inform design of new policies to address the problem, and to test and evaluate these through simulation. This step involves "what if" scenarios where new scenarios and policies are tested by adding necessary decision rules, strategies and physical structures to the model structure, and conducting simulation experiments and running sensitivity tests to evaluate the results.

The whole process is iterative and frequently involves close collaboration between the modeller and different stakeholders, problem owners, and domain experts from the different parts of the system under study (Martinez-Moyano & Richardson 2013).

4.6.5 The Budyko framework for hydrological modelling [Paper III]

A hydrological model simulates changes in water storage and fluxes above and below ground through the application of water balance equations (Horton *et al.* 2022). These equations are derived from the water balance law, which is essentially based on the law of conservation of mass and states that the total inflow of water to any system is equal to the outflows plus the change in storage during a time interval (Sutcliffe 2004). The general water balance equation can be applied at different scales and takes the form $P = Q + ET + \Delta S$, where P is precipitation, Q is runoff, ET is evapotranspiration and ΔS is change in storage. In hydrological studies, it is often represented by compartment-type models at different levels of aggregation depending on the purpose of the study (Simonović 2012) (*Figure 13*).

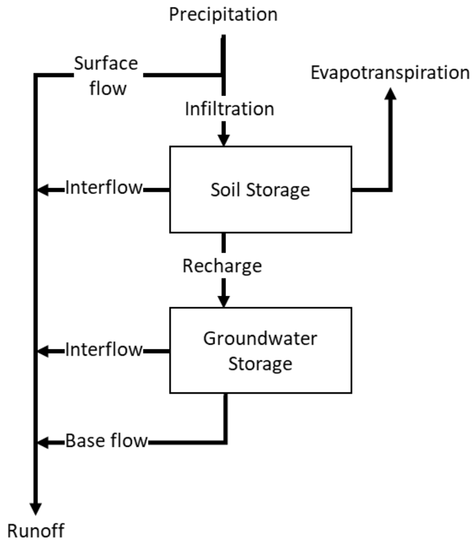


Figure 13. Schematic illustration of a simple hydrological compartment-type model.

The Budyko framework (Budyko 1961) is a top-down approach (Sivapalan *et al.* 2003) to hydrological modelling based on the concept that catchment water balance is ultimately controlled by the relationship between water availability and atmospheric demand. The framework has been extended to include additional explanatory factors, such as vegetation and land cover type (Zhang *et al.* 2001), and it has been applied to study hydrological dynamics at different temporal and spatial scales (Zhang *et al.* 2008).

In Budyko-type models, available water (W) (precipitation minus runoff) is partitioned between evapotranspiration (ET), groundwater recharge (GWR) and soil storage (SS) based on the notion of supply-demand competition (Gan *et al.* 2021). When potential evapotranspiration (PET) is very large relative to W (very dry conditions), more water is partitioned to ET and water availability is the limiting factor. In contrast, during very wet conditions ET will approach PET. This means that all available energy will be used for evapotranspiration, energy becomes the limiting factor, and GWR and SS increases (*Figure 14* left panel). The shape of the supply-demand curve is determined by the physical properties of the catchment, reflected by a shape parameter, *e.g.* parameter w as suggested by Fu (1981) (*Figure 14* right panel).

In Paper III, a Budyko-based approach as described by Zhang et al. (2008) was adapted to model groundwater dynamics on Fårö island. The same general supply-demand approach as described above was applied to model the partitioning of precipitation into direct runoff and infiltration, and partitioning of soil water into groundwater recharge, evapotranspiration, and soil storage. See Paper III for details.

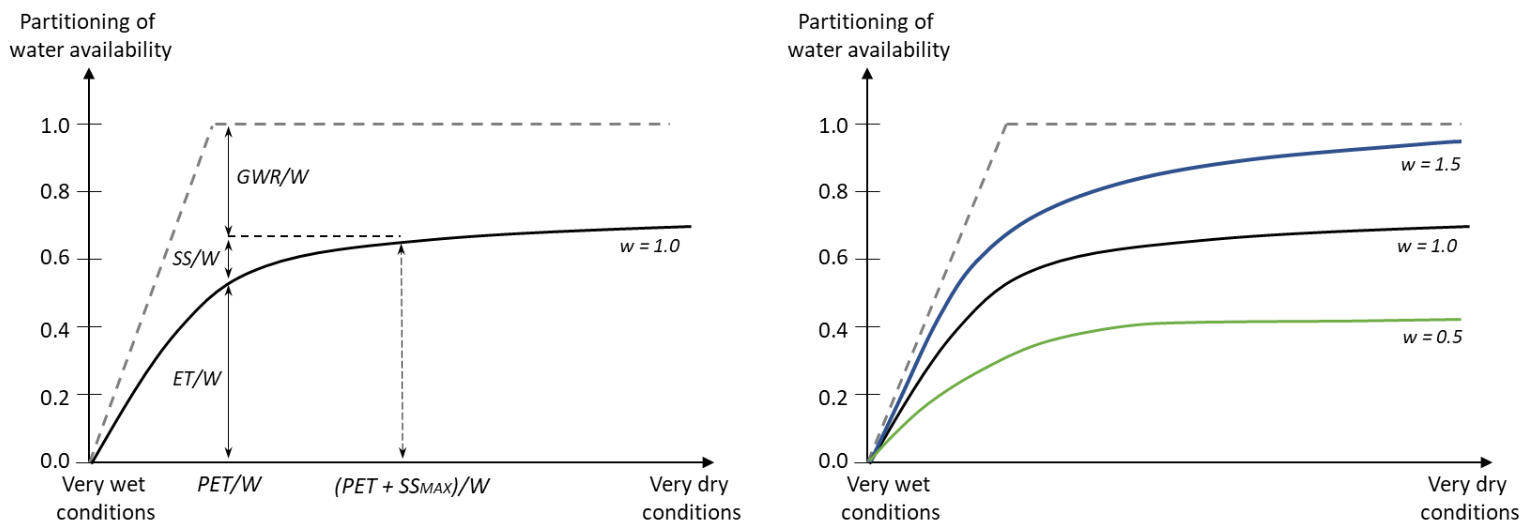


Figure 14. Schematic representation of Budyko-type curves for partitioning available water (W) between evapotranspiration (ET), groundwater recharge (GWR) and soil storage (SS) (left panel) for different hypothetical values of a shape factor *w* (right panel). The grey dashed lines represent the demand and supply limits and SS_{MAX} is the maximum soil storage capacity. Reproduced with modifications from Zhang *et al.* (2008).

4.6.6 Monte Carlo simulations [Paper III]

Monte Carlo simulation is a method commonly used to estimate the outcome of a given uncertain process or event by a procedure of repeated random sampling. The basic concept is to have a process or, as was the case in Paper III, a mathematical model representing the causal structure of a system under study. The model uses a set of input parameters which are processed through the mathematical structure of the model to generate a set of outputs (*Figure 15*). Very often the true values of the input parameters are not known with complete certainty, or the input parameters may vary stochastically according to some estimated probability distribution. This causes problems in analysis of model outputs, as these will depend on the unique combination of input parameters chosen.

In Paper III, this problem was solved using Monte Carlo simulations, by running *n* repeated simulation experiments where new values were randomly sampled for all input parameters in each run. The parameter values were sampled from predefined probability distributions derived from historical data or expert estimates. Each new set of input parameters generates a set of outputs that represents one unique outcome scenario. As *n* increases, the output sampling distribution converges towards the normal distribution and this represents an outcome space which can be used in further statistical analysis (Raychaudhuri 2008).

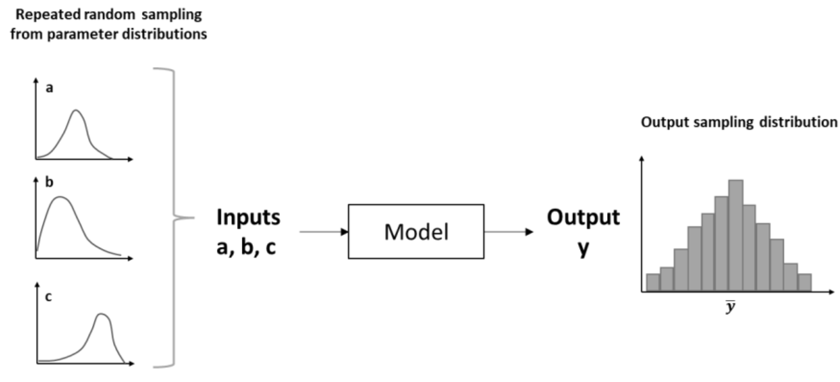


Figure 15. Illustration of the Monte Carlo simulation process. Parameter values for inputs *a*, *b* and *c* are repeatedly sampled from their respective probability distributions *n* times and simulated through the mathematical model to generate output *y*. Results from the simulations are summarised by a frequency plot that converges towards a normal distribution as *n* increases, yielding an estimate of *y*.

The model developed in Paper III contained several parameters where the true value was highly uncertain, including parameters related to future climate, housing development, tourism growth *etc*. Therefore, the Monte Carlo approach was used to explore an ensemble of plausible futures, and to inform decision makers about the bounds within which future policies should be designed to operate (Bankes 1993).

4.6.7 Marginal cost curve (MCC) [Paper IV]

A marginal cost curve (MCC) is a decision support tool used for assessing and ranking the cost-effectiveness of alternative investments, options or strategies in environmental management and policy design (Kesicki & Strachan 2011; Jiang *et al.* 2020). The MCC is presented as a graph that specifies the potential of a measure (or combination of measures) on the horizontal axis, and the marginal costs associated with the measure(s) on the vertical axis (Kesicki 2011) (see *Figure 16*). Generally speaking, there are two types of MCC: expert-based and model-based (Kesicki & Ekins 2012; Levihn *et al.* 2014).

Expert-based MCC are derived from expert estimates of the potential and costs of discrete measures, which are ranked from lowest to highest marginal cost. The total cost of a measure, or a combination of measures, required to reach the predefined reduction target (*e.g.* for greenhouse emissions abatement), savings target (*e.g.* for energy efficiency improvements) or mitigation target (*e.g.* for water scarcity mitigation) is calculated as the integral of the area under the curve (Kesicki & Ekins 2012). Model-based MCC, on the other hand, use formal models to derive the relationship between potential and marginal cost of a measure. Broadly speaking, either top-down (economy-oriented) or bottom-up (engineering-oriented) models are used (Kesicki 2011). The most common top-down models are computed general equilibrium (CGE) with a market-oriented focus, and linear programming models with a microeconomics focus (Huang *et al.* 2016). In bottom-up modelling, marginal costs are derived from engineering-based simulation or optimisation models (Kesicki 2011). In both top-down and bottom-up approaches, the MCC is derived by running the model using varying strict limit functions and calculating the corresponding costs. For instance, the marginal costs for emissions reduction can be derived from repeated simulations with increasingly strict emissions limits (alternatively, the model can be run with different emissions prices to derive the resulting emissions levels). The emissions-cost pairs are then plotted to form the MCC curve (Kesicki 2011; Du *et al.* 2015).

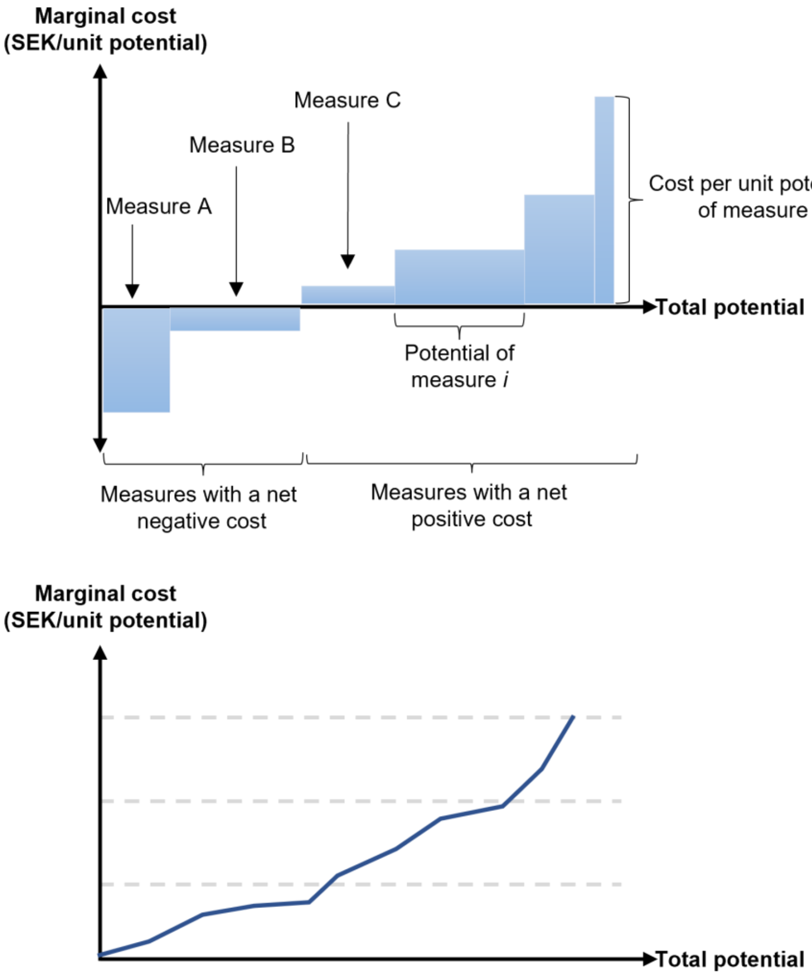


Figure 16. Simple example of (upper panel) an expert-based marginal cost curve (MCC) and (lower panel) a model-based MCC. Diagrams adapted from Sjöstrand (2020) and Kesicki (2011).

Expert-based and model-based MCCs both have their unique strengths and weaknesses. For instance, the expert-based approach is conceptually simple to execute and interpret, and by tapping into the knowledge of local experts one can incorporate considerable level context- and technology-specific detail into the assessment (Kesicki 2011). A disadvantage of using expert estimates is that the underlying mechanisms and assumptions behind the estimates may be unclear for external users and non-experts. Further, the approach is unable to capture interaction effects between measures and it does not provide any information to the decision maker about how the intended effects of the measures are distributed over the assessment time horizon (Kesicki 2011; Jiang *et al.* 2020).

Among the model-based approaches, top-down models derive the MCC from a whole-economy, highly aggregated perspective, whereas bottom-up models are typically sector-specific with a high level of technical detail (Kesicki 2011; Du *et al.* 2015). This makes top-down models better for studying the macroeconomic effects and feedbacks of measures or policies, whereas bottom-up models are better for assessing sector-specific effects of interventions, but they cannot capture economic interactions with surrounding systems (Kesicki 2011; Du *et al.* 2015).

Paper IV describes four overarching limitations of conventional MCC approaches that limit the utility of the method as a tool for policy, planning and management. It also presents a SD-based approach for MCC calculations and assesses the capacity of that approach to address the main limitations of MCC.

4.6.8 Agent-based modelling (ABM) [Paper V]

Agent-based modelling is a relatively young modelling approach with its roots in the theory of CAS (Arthur *et al.* 2015), emerging in the 1970s in the fields of complexity science, economics, social sciences and computer science (Hare & Deadman 2004). An ABM is a computer program that consists of a group of autonomous[1] agents (often individuals, *e.g.* humans or organisms, but also organisations, countries or even physical objects) that interact with one another and the environment over time. In each time step of the simulation, the agents are programmed to take actions based on (i) their internal state(s), (ii) interactions with other agents, and (iii) changes in their environment. The decisions and actions an agent decides to take are determined by a set of pre-programmed behaviour rules (Schlüter *et al.* 2021b). These rules are often in the form of if-then-else functions or decision

[1]Autonomous means that the behaviour of the agents is not centrally controlled.

trees (Schlüter *et al.* 2019c). For example, if the state variable "energy level" drops below "desired energy level", this will trigger the action "eat". The actions taken by the agents can modify their internal states and thereby change their behaviour in subsequent time steps. For example, if the action "eat" is taken at time 1, this will increase the energy level of the agent to match its "desired energy level" and change the behaviour from eating to not eating at time 2. This feature allows the agents to adapt to changes in context, which is a defining feature, and often a feature of great interest, when studying CAS (Schlüter *et al.* 2021b). A schematic illustration of the structure of ABM models is provided in *Figure 17*.

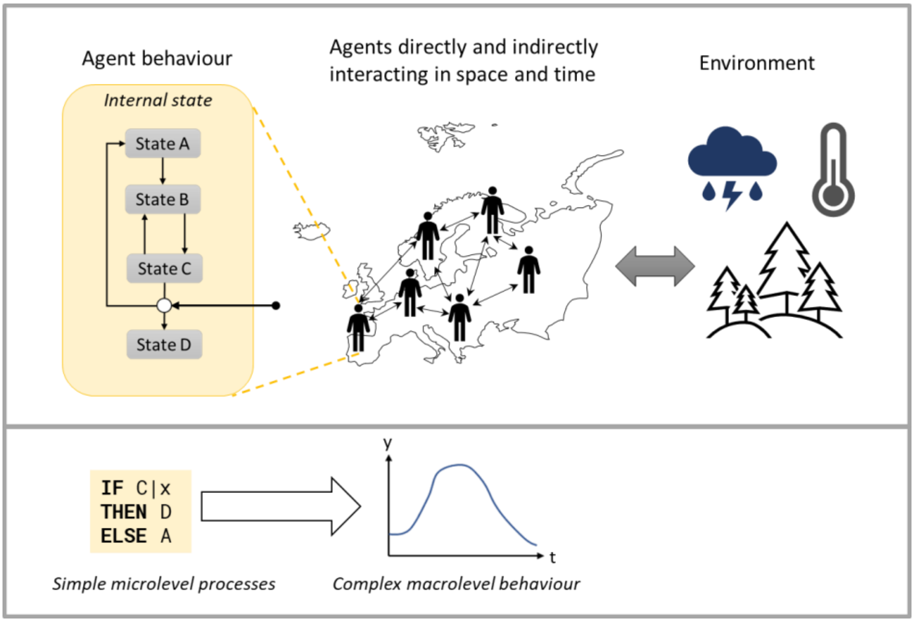


Figure 17. Agents interact directly and indirectly with each other and with the environment. Transitions between internal states are governed by simple rules at the micro level, but give rise to complex patterns and behaviours at the macro level. Diagram adapted from (Borshchev & Filippov 2004).

Agent-based modelling allows for exploration and explanation of how micro-level interactions of multiple agents can give rise to counter-intuitive, and highly complex, macro-level dynamics. Studies have shown that even with a surprisingly small number of simple behavioural rules, ABM can produce macro-level phenomena with a high level of realism (see *e.g.* Reynolds (1987) for a simulation of the flocking behaviour of birds and

Schelling (1971) for the endogenous drivers of segregation). Paper V leveraged this potential to explore how micro-level social interactions between agents in a GMB setting influence the outcomes of accuracy and alignment at the macro-level.

5. Summary of results and discussion

In this chapter, the main results of the thesis are presented and discussed in relation to the overall aim and research questions addressed.

5.1 Defining the problem: Limitations of contemporary tools for managing CHANS

RQ 1: How are CHANS represented in contemporary methods for assessment, management, and planning of human activities, and what is required from methods to support sustainable management of these systems?

Tools aiming to support management, policy design and planning in CHANS must account for their complex causality and constantly evolving nature. The work in Papers I and IV explored the extent to which contemporary decision support tools (LCA and MCC) can cope with this challenge. However, both of these are tools under constant development and, since the publication of Paper I, research on life cycle-based methods has evolved (as indeed have my personal experience and understanding of CHANS). Thus, in the following section the main findings from Paper I and IV are presented and expanded upon, with the discussion grounded in the contemporary literature and the requirements of CHANS methods described in Chapter 4.

Treatment of structure

System structure refers to the causal relationships and mechanisms linking system components and giving rise to observable phenomena and dynamic behaviour (Sterman 2000; Gerring 2008). Because CHANS consist of integrated human and natural components, the structure of these systems entails both physical relationships and first principles, as well as social, economic and human-behavioural mechanisms and, crucially, the interactions between these. Both LCA and MCC are biased in terms of which of these structures they include when assessing the consequences of an introduced change. First, the LCI step of a standard LCA has a purely technological focus. It isolates the technological subsystem from its wider social and ecological context, and it assumes that the elementary flows in and out of the technological subsystem will respond linearly to the hypothetical change (Yang & Heijungs 2018; Pizzol *et al.* 2021). This approach ignores the economic, behavioural and environmental mechanisms, driven by interactions with surrounding subsystems, that may influence elementary flows. These could be physical, economic or behavioural demand- and supply-side constraints (Gutowski 2018; Yang & Heijungs 2018; Hicks 2022), economies or diseconomies of scale (Yang & Heijungs 2018), or learning curves and tipping points (Pizzol *et al.* 2021; Rovelli *et al.* 2021).

In more advanced LCAs, partial or general equilibrium models are used in the LCI step to account for how market mechanisms influence supply and demand for products and by-products (Yang & Heijungs 2018). Even though this adds more economic theory to the assessment, and it enables certain non-linear features to be accounted for, these models rest on profoundly unrealistic neoclassical assumptions about human behaviour, and they lack important structural features necessary to assess the consequences of system change (Gutowski 2018; Yang & Heijungs 2018; Hicks 2022). To start with, neoclassical equilibrium models assume that individuals are completely rational, companies always maximise their utility, markets are perfectly efficient and all actors have access to perfect information at all times. These are all assumptions that have been repeatedly proven not to be representative of how the real world operates (Thaler & Ganser 2015; van der Werf *et al.* 2020). In reality, people do not behave like the Econs of neoclassical economics. They interact with their social, ecological and technical environment, and make boundedly rational decisions based on partial information and their goals, attitudes and beliefs (Simon 1957; Thaler & Ganser 2015; Hicks 2022). Human behaviour is potentially even more relevant to evaluate the consequences of a policy than the technologies the policy entails, and ignoring these features of the system has repeatedly been shown to lead to perverse outcomes (Dahmus 2014). Therefore, Gutowski (2018) argues that people, and not the products or the technology, should be at the centre of policy assessments. Unfortunately, realistic representations of the mechanisms driving human behaviour remain largely lacking from the LCA and MCC literature.

In Paper I, the focus was primarily on the impact assessment step of LCA, *i.e*. the step that translates elementary flows into potential environmental impacts. From this analysis, the following additional structural limitations of the conventional LCA approach were identified:

- The impact models used to translate elementary flows into environmental impacts are often highly simplified and not contextualised to the place and context of the study.

- Many important local and regional impact categories are under-represented in environmental assessments because they lack reliable impact models.

- Many biophysical and ecological functions and services are the result of interactions between environmental and ecological processes belonging to different impact categories in the LCA framework. However, because interaction effects between impact categories are not accounted for in LCIA, synergistic effects between environmental impacts are not included in the assessment.

Treatment of time

Both LCA and MCC are static, flow-and-accounting tools relying on the non-temporal type of models presented in section 4.4.2. Both tools assess the consequences of a shock introduced to a system by comparing alternative stable states that the system may take with or without the introduced change. In LCA, the changes in elementary flows caused by the shock are aggregated and translated to environmental impacts for a single point in time representative of one such static state. In MCC, the corresponding aggregation is made for all costs, benefits and utility produced, over the time horizon of the assessment. The consequences of the introduced shock are then assessed by comparing the original stable state (no shock) with the alternative stable state (with shock).

- Because time is not accounted for, neither LCA nor MCC provides any information about the transition path that the system takes to go from one stable state to the other, how long the transition would take, or even if whether is at all possible for the system to transition freely from one state to the other.

- In environmental management and planning, ignoring the transition pathway limits the usefulness of the assessment, as the timing and temporal distribution of environmental stressors can strongly influence the impacts on the receiving system. For instance, temporal variations in pollution load may influence the damage caused by polluting activities.

- Similarly, in economic assessments, knowing how the costs and benefits of a policy are distributed over time is as important as the predicted endpoint (Anderson & Cavendish 2001). For instance, if the objective of a policy is water scarcity mitigation, then selecting a mitigation strategy that adds new water to the system early in time can be of greater value than choosing the most cost-effective strategy.

Treatment of feedback

Feedback can occur at the local level (*e.g.* within one subsystem) and, as is significant for CHANS, at the global level (*e.g.* between human and natural subsystems). As discussed above under "Treatment of structure", both LCA and MCC are typically biased towards only representing the technological subsystem of CHANS in some detail. The human subsystem is often completely ignored in the representation and the natural environment is treated as an exogenous source and sink from which resources are extracted and pollutants are expelled. Thus, the path of cause and effect is unidirectional: from the introduced change, through the technical system, to the environment.

- As concluded in Paper I and discussed above, interaction effects between environmental impact categories in LCA are not accounted for, and thus many environmental dynamics are ignored.

- Because there is no feedback from the damage imposed on the environmental subsystem back to the human subsystem (*i.e.* from the LCIA back to the LCI), it is questionable whether the true consequences of a studied change can ever be assessed.

- As illustrated in Paper IV, accounting for feedback effects can have a significant impact on the assessment. Thus, ignoring local and global feedback can severely limit the reliability of both LCA and MCC results, and thus reduce the usefulness of these tools for guiding policy and management in CHANS.

5.2 Engage with the system: Dynamics of water supply and demand in coupled human-water systems – past, present and future.

In-depth case studies exploring the systemic drivers of water supply and demand on Fårö island, Sweden, were conducted (Paper II) and the results were used to assess how future climate may impact water availability and socioeconomic development in the region (Paper III).

5.2.1 Understanding past and present social-hydrological drivers of water supply and demand

RQ 2: What processes govern the dynamics of drinking water supply and demand in coupled human-water systems?

In Paper II**,** a qualitative SD model was developed to identify the socio-hydrological drivers of water supply and demand and assess why historical policies to mitigate water scarcity had been ineffective. Through triangulation and integration of multidisciplinary local and expert knowledge, empirical data and secondary data, a CLD illustrating how water is an interconnecting link between the housing, tourist and municipal sectors was constructed. Through close coupling, decisions in one of these sectors have cascading effects on other parts of the system. The full CLD is presented in *Figure 18*. For detailed variable definitions and feedback loop descriptions, see Paper II.

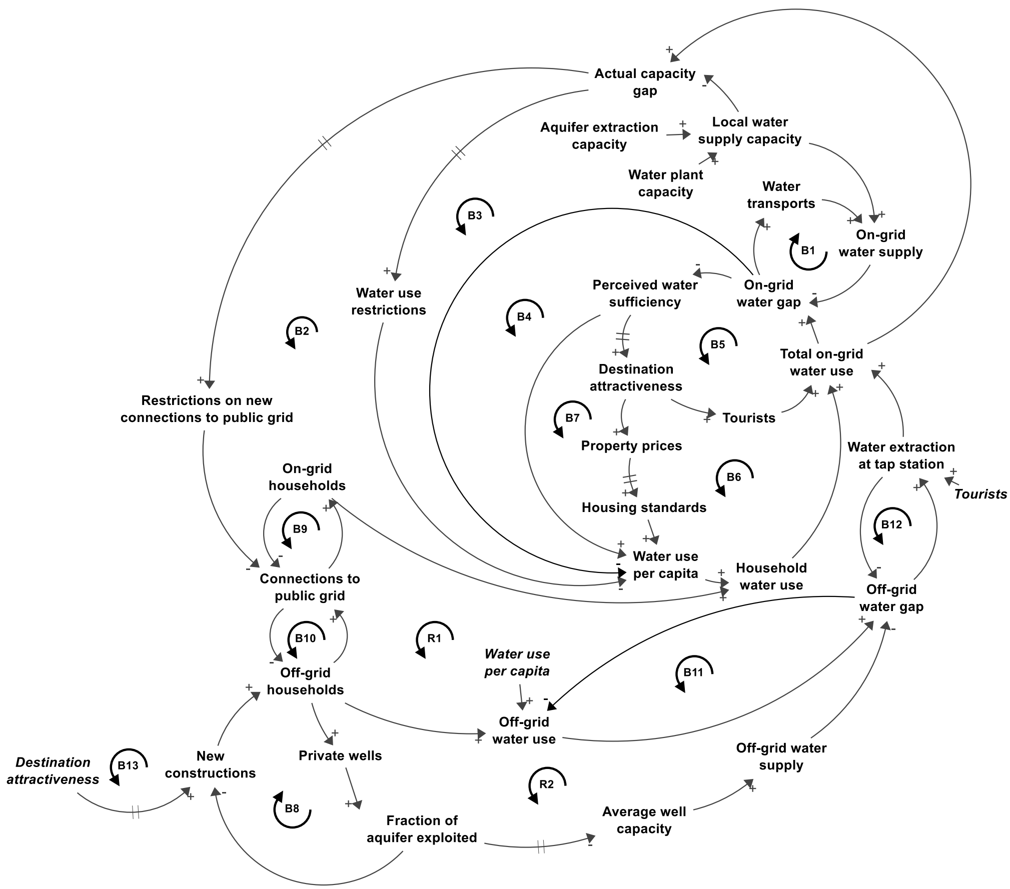


Figure 18. Causal loop diagram developed in Paper II. Causal links with double dashed bars indicate a time delay between cause and effect. Curved arrows with a capital B/R represent balancing and reinforcing feedback loops, respectively. Reproduced with permission from Nicolaidis Lindqvist *et al.* (2021).

The main insights from Paper II were:

- Historical policies to mitigate water scarcity on Fårö have primarily been oriented towards increasing supply by inter-basin water transport. This reduces water scarcity in the short term, but creates a gap between consumer-perceived state of water resources and the actual state.

- When water is perceived as more plentiful, incentives for water use efficiency erode and water-demanding capital investments continue (*e.g.* more hotels are built and the housing stock is improved). This is known as a supply-demand cycle, a form of rebound effect (Alcott 2005), and has also been documented in previous studies (Kallis 2010; Scarrow 2014).

- Water-consuming capital investments have a long lifetime, and new investments are often made with the expectation that the level of water availability at the time of the investment will remain stable in the future. It is therefore very challenging to phase out unsustainable supply-oriented policies once capital investments have been made. Thus, short-term solutions to the water scarcity problem contribute to systemic lock-in effects and unsustainable consumption (Unruh 2000; Truong *et al.* 2022).

The above insights were summarised into a condensed and generalised CLD consisting of two balancing and one reinforcing feedback loops (*Figure 19*). A supply-demand gap occurs when water demand exceeds local supply. The gap can be closed by either reducing demand (the lower balancing loop) or increasing supply (the upper balancing loop). In a well-functioning system, the lower loop dominates and demand self-adjusts to the local carrying capacity of the system. However, if supply-targeting policies are emphasised, the upper balancing loop may dominate and the gap is closed by increased exogenous supply. This erodes efficiency incentives and attracts further investments in long-lived water-demanding capital (the lock-in effect), creating a reinforcing feedback loop that drives escalating water demand (the supply-demand cycle).

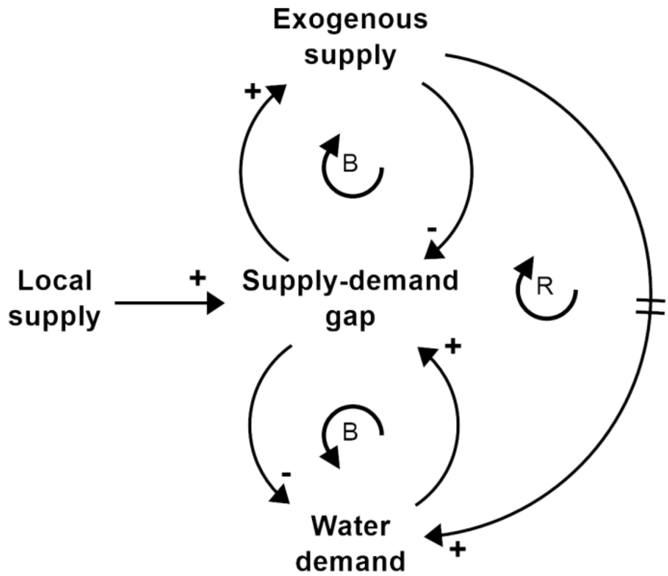


Figure 19. The three high-level feedback loops governing drinking water supply and demand in the human-water system studied in Paper II.

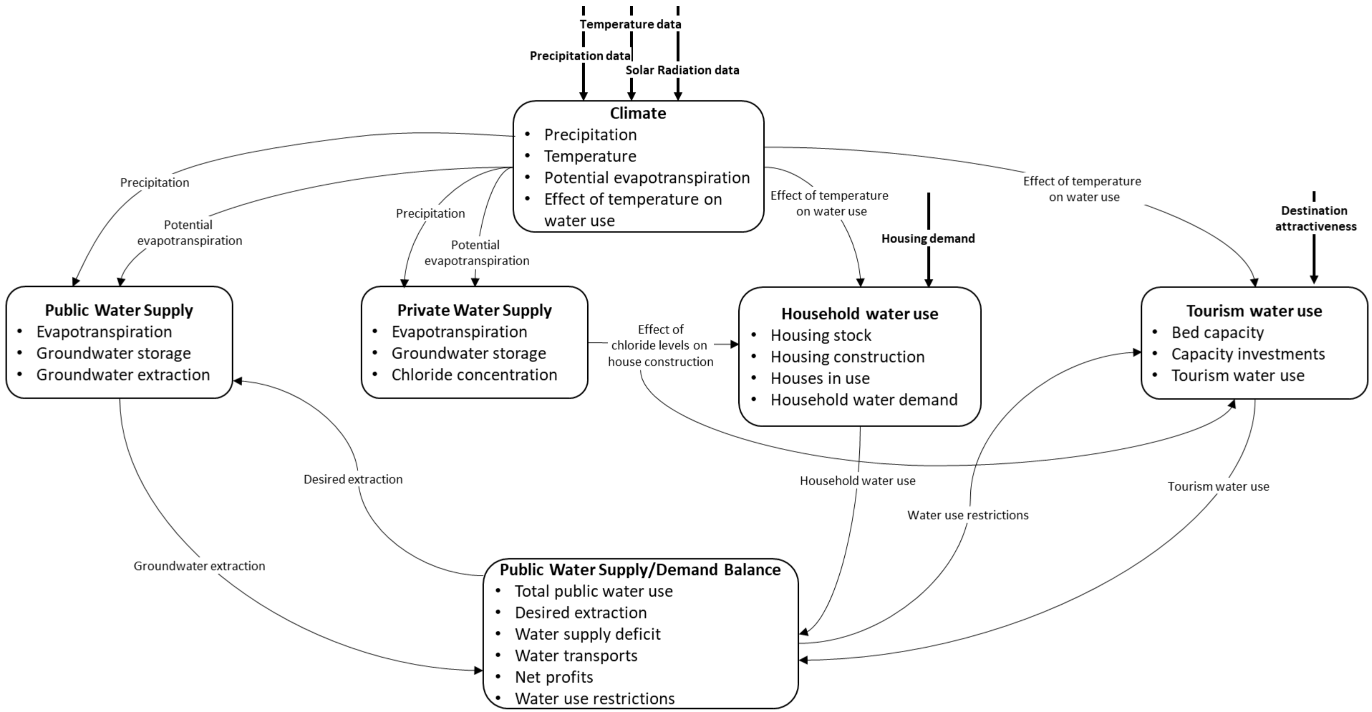
The problem structure illustrated in *Figure 19* is not unique to the Fårö case, or even to the water resources management domain. Moallemi *et al.* (2022) refer to this generic causal structure as the "band-aid solution" archetype, where relatively easy interventions that lead to immediate but temporary improvements (*e.g*. meeting scarcity by increasing exogenous water supplies) have the unintended side-effect of diminishing the perceived need, and undermining the incentives, for more fundamental changes (*e.g*. reducing total water demand).

This example illustrates the important role that information flows play in determining the consequences of management actions and overall system performance. Increasing exogenous supply of water effectively weakens the information feedback between the true and perceived state of the system, causing unintended side-effects of otherwise well-intentioned policies. Understanding how information flows through the system, and the feedback processes it may trigger, is key for effective management and policy design. However, ignoring the flow of information is one of the most common causes of system malfunction (Meadows 2009). Therefore, any attempt to model (conceptually or formally) the effects of policy and management actions must not aim to represent the system as it “should” work, *i.e*. in an idealised clockwork fashion where agents act with perfect and immediate access to information. Rather, the system should be modelled as it actually works, where information spreads slowly and actors make boundedly (sometimes) rational decisions based on the information they have at the moment (Sterman 2000).

5.2.2 Exploring future social-hydrological impacts of climate change

RQ 3: How will climate change influence drinking water supply, and what dynamic effects may this have on socioeconomic development, and subsequent water demand, in coupled human-water systems?

The results from Paper II were primarily of a descriptive and explanatory nature, *i.e*. the study revealed the structure of the coupled human-water system on Fårö and provided a theory on the drivers of the historical trajectory of the system (Biggs *et al.* 2021b). Paper III was more forward-looking and exploratory. It drew on the results in Paper II to assess how future climate may influence water supply, and what dynamic effects this may have on the hydrology and socioeconomic development on the island. A SD simulation model was constructed, consisting of six interconnected submodules to simulate future climate, groundwater levels, groundwater quality, municipal and private water supply, the housing and tourist sectors, municipal water transport, and water use restrictions. A schematic representation of the different submodules, information exchanges and the main computations performed in each module is provided in Fi*gure 20*. A detailed description of the model can be found in appendix A in Paper III. The model was calibrated to the period 2000-2020 and Monte Carlo simulations, fed with data from regional climate scenarios (RCP2.5 and RCP8.5) provided by SMHI (Asp *et al.* 2015), were conducted to explore the likely outcome space for the system in the period 2020-2050.



F*igure 20*. Graphical representation of the model constructed in Paper III. Boxes represent the six submodules, with their key processes and stock variables indicated. Arrows represent exogenous data inputs (bold) and information exchange between modules. Reproduced with permission from Nicolaidis Lindqvist *et al*. (2022).

The main findings from Paper III were:

- Groundwater levels on Fårö have been historically low in the past 20 years. In the simulated future climate scenarios groundwater storage remained critically low, and in 60-70% of the simulations the groundwater head fell to levels beyond the most extreme year experienced since the 1960s (*Figure 21*). This will cause seasonal water scarcity to become more frequent and widespread and it is likely to increase the risk of saltwater intrusion into groundwater wells.

- Limited access to water of sufficient quality and quantity is expected to constrain housing construction on Fårö by up to 11%, and expansion of the tourist sector by up to 30%, compared with an unconstrained scenario.

- If no changes are made to the current municipal water management strategy, Monte Carlo simulations suggested that the need for supplementary inter-basin water transport will increase by on average about 25% compared with current levels by 2050.

- Worryingly, available local municipal water supplies were insufficient to meet demand across all simulated scenarios. In other words, even in the most optimistic of future scenarios the island will still require supplementary water transport to meet demand in the summer months. To become water self-sufficient, fundamental improvements in water-use efficiency and diversification of water supply solutions are needed.

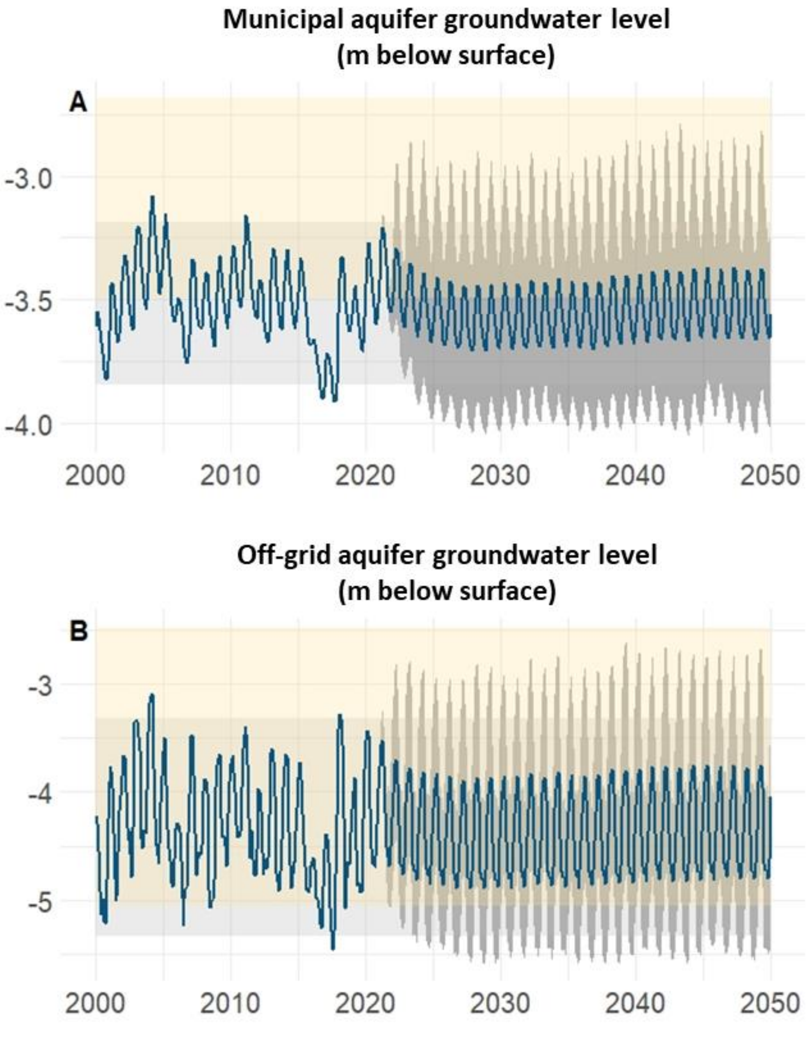


Figure 21. Simulated groundwater level in (A) municipal and (B) private aquifers on Fårö. Blue lines are mean groundwater levels of the simulated ensemble, shaded areas represent the 95% confidence intervals, and the yellow and grey bands indicate the normal groundwater range (mean level +/- two standard deviations) for reference period P1 (1961-1990) and P2 (2000-2020), respectively. Reproduced with permission from Nicolaidis Lindqvist *et al.* (2022).

To my knowledge, Paper III is the first study to explore local impacts of future climate using an integrated social and hydrological model in Sweden, and possibly in Scandinavia. The lack of local assessments for this region is understandable for two reasons. First, water has hitherto been a plentiful resource in Sweden, a country with greater freshwater availability per capita than many other countries in Europe (Eurostat 2022). Thus, water scarcity has not been a prioritised issue. Second, downscaling global climate model projections to the subregional scale remains a substantial challenge, and the uncertainties associated with such predictions are substantial (Oreskes *et al.* 2010). Nevertheless, with changing weather patterns and growing abstraction rates, shifts in both global and local hydrological cycles are becoming increasingly evident (Schewe *et al.* 2014; Falkenmark *et al.* 2019; Wu *et al.* 2020). These changes are expected to cause seasonal water shortages to become more common also in formerly water-abundant regions (Asp *et al.* 2015; Ahopelto *et al.* 2019). Thus, navigating the local social and hydrological drivers and impacts of water scarcity is becoming increasingly crucial for effective water management (United Nations 2018). Because of the uncertainty regarding future climate (Deser 2020), and the frequent lack of detailed hydrological and water use data (Tegegne *et al.* 2017), predictive projections of future scenarios are rarely possible (or even appropriate) (Bankes 1993). Instead, the approach presented in Paper II and Paper III, *i.e.* using participatory approaches to engage with the system, collaborating with local academic and non-academic experts, and exploring an *ensemble* of possible futures, is preferable. This approach can facilitate the design of policies, strategies and solutions that are rooted in the local context and perform satisfactorily under a wide range of circumstances (Bankes 1993; Malekpour *et al.* 2016; de Vos *et al.* 2021).

5.3 Designing solutions: decision-support adapted to CHANS

RQ 4: How can analytical methods be improved to better support policy, planning and management of CHANS?

As mentioned earlier in this thesis, policy and management is inherently about understanding and solving problems. In the field of CHANS these are often system-level problems, emerging from constant dynamic interactions between social, economic and technical processes in the human subsystem, and physical and ecological processes in the natural subsystem (Biggs *et al.*

2021a). Successful management of these problems cannot be achieved by studying the constituent subsystems and processes in static isolation. Instead it requires holistic methods focused on how system components behave and interact dynamically over time (Liu *et al.* 2008). In the decision-making process, this requires new and adapted tools for assessment and evaluation. Paper IV showed the value of adapting conventional static type analyses to incorporate more of the causal structure of the system under study and using dynamic simulations to evaluate policy interventions (*Figure 22*). Key findings were:

- Compared with using conventional approaches, applying a system dynamics-based approach to derive the MCC can bring new policy insights, reveal unintended consequences of decisions, and more effectively exhibit the ancillary benefits and costs of different measures.

- Using system dynamics-based simulation models to derive the MCC gives valuable insights about when in time the costs and benefits of different actions occur.

- Complementing the formal simulation model with a CLD, or other visualisation tool, can make the underlying logic and structural assumptions of the model more accessible to non-modellers, making the decision-making process more transparent.

- Overall, this can support more informed decisions, as the underlying model accounts for more defining features of CHANS than conventional approaches typically do.

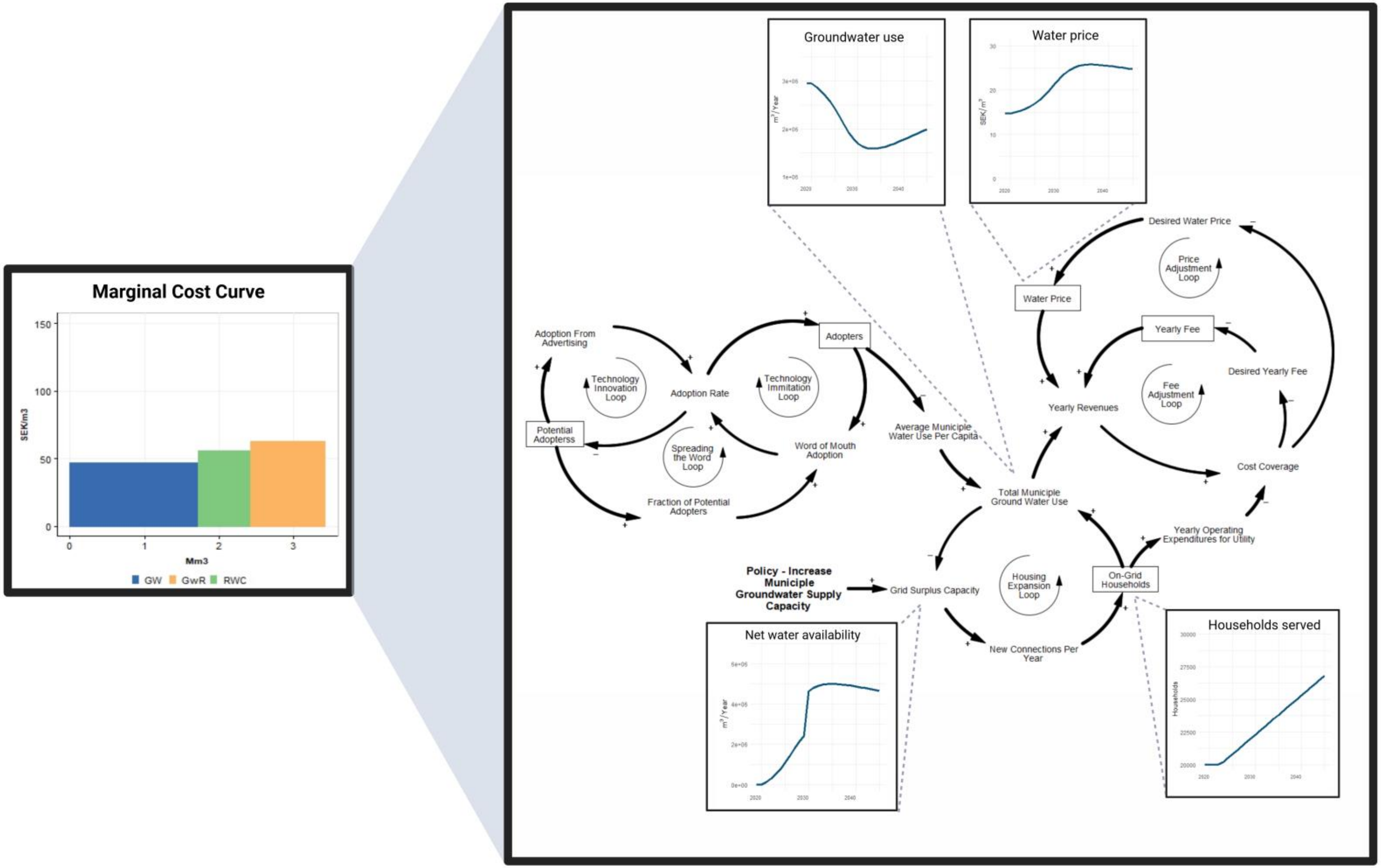


Figure 22. Extracts from Paper IV, illustrating how the system dynamics-based approach generates both a marginal cost curve (bar graph) as well as the underlying simulated dynamic effects (timeseries graphs) on water availability, service capacity, groundwater use, and consumer water price. The results are both derived and interpreted using the underlying model structure (central causal loop diagram). Figure created with BioRender.com.

Another insight from the work conducted in this thesis is that, given that knowledge and understanding of CHANS will always be partial and context-dependent (Preiser *et al.* 2021), finding an optimised solution to these problems is rarely possible.

The necessity of accounting for system structure to effectively manage CHANS, and the fact that much of this structure is context-dependent and at least partly unobservable, creates a dilemma. In section 5.2, it was shown that tapping into a diversity of knowledge sources and perspectives to understand system structure, *e.g.* by involving stakeholders in participatory modelling approaches, is a promising way to address this dilemma. Participatory modelling approaches to create formal representations and to guide decision making of a complex reality have been frequently used in environmental and social system management (Stave 2010; Voinov *et al.* 2018; Aminpour *et al.* 2021). However, how information is collected from the participants, the composition of the group, their personal attributes and their social interactions during the process, can influence both the accuracy of the model produced (Woolley *et al.* 2010), and the extent to which the group members align around a shared system understanding (Bang & Frith 2017; Becker *et al.* 2017). These aspects are still largely unexplored in the participatory modelling literature, even though system conceptualisation is a key step in any modelling activity (Jakeman *et al.* 2006; Martinez-Moyano & Richardson 2013) and has major implications for the modelling outcomes (Luna-Reyes 2003).

Paper V examined these effects using simulation experiments, replicating the system mapping step of a GMB intervention. The aim was to explore the determinants of model accuracy and group alignment in a participatory system mapping setting, and then derive guiding "rules of thumb" to support the design of future interventions. The main conclusions from Paper V were:

- The composition of the group can have a strong impact on GMB outcomes. The system map produced through the simulated GMB process in Paper V was more accurate than the mental model of the average group member in ~67% of cases. However, in the remaining ~33% of cases, there was either no significant difference in accuracy or the model produced by the group was *less* accurate than that of the average member (see *Figure 23*).

- Alignment increased significantly in all simulation experiments (*Figure 23*). The increase was greatest in groups where the members had a high level of social status, as this facilitated convergence towards a shared mental model of the system under study.

- Social status (a proxy for perceived personal credibility and persuading power) was a strong determinant of group alignment, and initial mental model accuracy was a strong determinant of group accuracy.

- To improve the chances of a desired outcome (high accuracy and high alignment), selecting a group with intelligent individuals and a moderate level of social status when designing GMB workshops is recommended (but hard to control for).

- To reduce the risk of an outcome with high alignment around an inaccurate model, controlling, or at least moderating, the influence of socially dominant individuals in the GMB process is advisable.

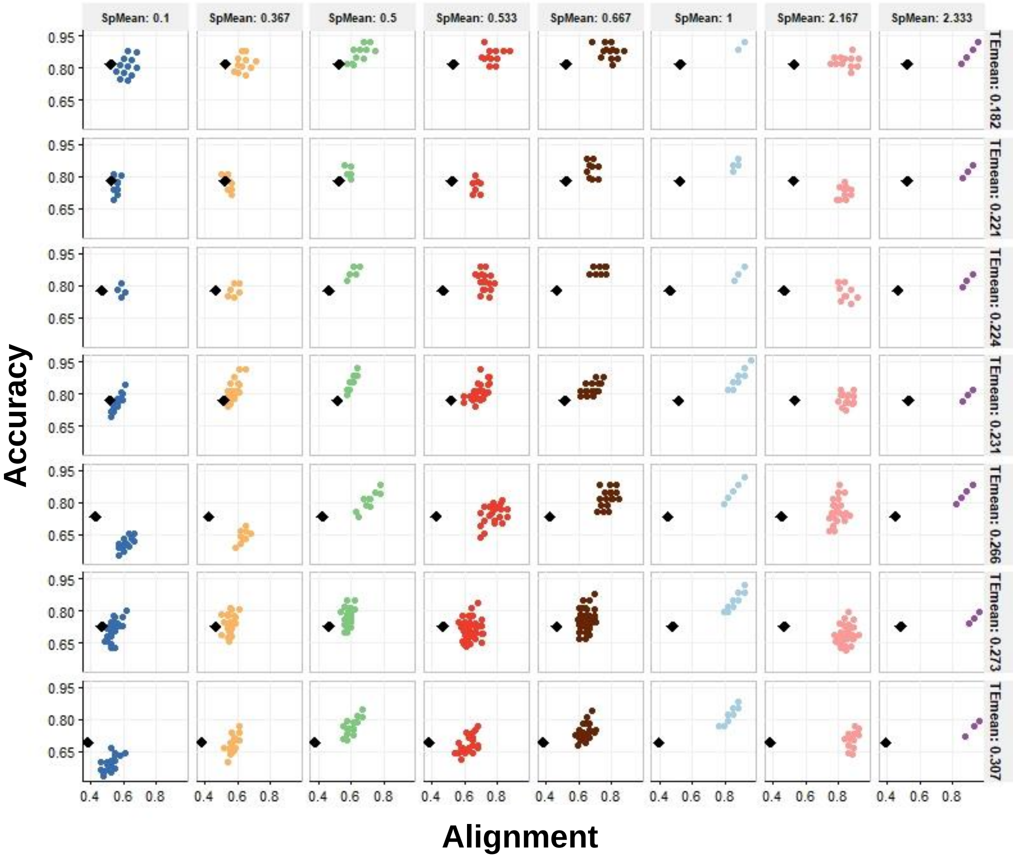


Figure 23. Plots for simulation experiments showing the level of accuracy and alignment at the end of the simulation. Each plot represents a batch of 30 repeated simulations with a given level of initial accuracy (TEmean) and mean social status (SpMean) of the group. Initial accuracy decreases from the top down, and social status increases from left to right. Black diamonds indicate the initial accuracy and alignment of the group at the start of the workshop. Reproduced from Paper V.

6. Concluding discussion and contributions to society

Through the work described in this thesis, the aims were to: (i) contribute to knowledge on the patterns and processes that govern the dynamics of CHANS, focusing in particular on coupled human-water systems, in order to support more effective policy and management; (ii) improve understanding of how climate and social change interact to influence future water supply and demand; and (iii) assess and develop analytical tools and methods to support future policy, planning and management of human activities in these systems.

6.1 Rethinking CHANS and how to model CHANS

No universal theory of CHANS, or blueprint for how to successfully and sustainably manage them, is presented in this thesis. However, the results provide some insights into the anatomy of these systems and indicate that one reason why previous policies and management strategies for CHANS have often failed is because they have been based on inaccurate models.

The dynamics in CHANS are driven by both observable and unobservable exchanges of energy, matter and information between and within the social, economic, technical, environmental and ecological processes in the human and natural subsystems. These interactions form feedback loops that give rise to complex, and often counter-intuitive, patterns and non-equilibrium behaviours, making it difficult to distinguish cause from effect. These behaviours are emergent properties of CHANS and thus they cannot be isolated and studied by reductionist approaches.

Any policy or management intervention imposed on CHANS will influence not only the observable features of the system (*e.g.* the physical infrastructure or material flows), but also the unobservable features (*e.g.* human perceptions, ambitions and information exchanges). This will trigger the endogenous dynamics embedded in the causal structure of the system. Thus, any tool or method intended for understanding and managing CHANS should aim to incorporate both observable and unobservable system structure, study the system over time, and acknowledge the important role that human-nature feedbacks play in shaping the effects of policy interventions.

Unfortunately, most contemporary tools and methods used to support policy and management in CHANS (*e.g.* LCA and MCC as assessed in this work) are not adapted to these requirements. First, the tools are based on predominantly linear and static models but are applied to nonlinear and highly dynamic systems. They provide snapshots of the precise quantitative relations between system variables at a given point in time, but they ignore how these relations will evolve over time. Second, they are biased towards accounting primarily for the observable technical and physical parts of system structure, while they tend to ignore unobservable features such as information flows, temporal delays and feedbacks between humans and the natural environment. Third, they are rooted in a neo-classical world view where systems exhibit equilibrium properties – an implicit assumption that does not hold in real-world, constantly evolving, complex systems (Costanza *et al.* 1993). In order to fit CHANS into an equilibrium frame, feedback loops are cut, or simply ignored, in the models used in conventional assessment and planning tools. This reduces the dynamic complexity of the model and facilitates identification of closed-end "solutions", but it also reduces model realism, and hence model utility for assessing and designing new management or policy interventions is substantially reduced (Costanza *et al.* 1993; Anderson & Cavendish 2001).

Identification of simple explanations to complex problems is often promoted in both research and practice (Edmonds & Moss 2004; Edmonds 2007). However, oversimplified representations of complex systems tend to favour quick fixes, simple and universal solutions, or panaceas that unfortunately often fail (Ostrom 2007; Axelrod & Cohen 2008). The reliance on inaccurate models is suggested as a major reason why previous environmental and social policies have not delivered as expected (Meadows 2009; Levin *et al.* 2012; Laitos & Okulski 2017). The work in this thesis was an attempt to support a shift from the use of linear, reductionist, and static models of CHANS towards more realistic representations that encapsulate more of the true complexity. The need for such a transition has been highlighted in recent research (Kramer *et al.* 2017; Laitos & Okulski 2017; Preiser *et al.* 2021; Reyers *et al.* 2022). This does not mean that every possible detail of the system must be incorporated for a model to be useful for policy and management purposes. Abstractions and simplifications must still be made, but one may need to be more selective in how the famous Occam's razor is applied. After all, if the real-world structures that drive system change are omitted from models, the simulation results cannot be expected to realistically represent the effects of policy or management changes.

6.2 A canary in the mine

In management of human-water systems, the limitations of conventional approaches are evidenced in the failure of well-founded policies to achieve their intended goals. Historically, measures to mitigate water scarcity have been dominated by supply-side interventions. To meet growing demand, local extraction rates are often pushed to their limits, followed by increasing reliance on engineering-type solutions that redistribute water in time and space (Allan 2005). Large-scale water reservoirs for surface water storage and inter-basin water transfer projects for water relocation are examples of supply-side policies commonly used to cope with drought and water shortages. However, when implemented in isolation, these interventions often have the unintended side-effects of weakening the incentives for more fundamental changes (*e.g.* reducing total water demand) and increasing long-term water consumption (Mirchi *et al.* 2012; Di Baldassarre *et al.* 2018). When occurring in tandem with systemic lock-in effects (Markolf *et al.* 2018), the water scarcity trap becomes pervasive and makes adaptation to changing hydrological conditions increasingly difficult and costly due to historical long-lived capital investments.

The "band-aid solution" and "lock-in" problems are just two examples of recurring causal structure system archetypes (Senge *et al.* 1990), causing unanticipated and problematic behaviours in human-water systems (Mirchi *et al.* 2012; Bano *et al.* 2022) and CHANS in general (Moallemi *et al.* 2022). They highlight the co-evolutionary nature of the social and hydrological sides of coupled human and water systems (Kallis 2010) and show how ignoring the feedback effects that drive this evolution in management and policy design can have severe unintended consequences.

From a climate adaptation perspective, the findings from Fårö are highly relevant for other Swedish regions, and for regions elsewhere. With the large-scale changes in global and regional weather patterns already occurring as an effect of climate change (Taylor *et al.* 2013; Falkenmark *et al.* 2019), water availability can no longer be taken for granted (Milly *et al.* 2008). The temporal and spatial distribution of water is already changing in Sweden and other countries (Sjökvist *et al.* 2015; Wu *et al.* 2020) and this will most likely force changes in the management and allocation of water resources. Hence Fårö could well be the canary in the mine warning that fundamental change is needed. Management decisions in regions facing unfamiliar water shortages could be guided by the structural insights from Fårö and similar studies to mitigate escalating water scarcity. Introducing demand-side measures prior to expanding supplies, improving alignment of public perceptions with the actual state of water resources, and designing infrastructure investments so that the risk of lock-in effects are minimised, could be the way to proceed. However, more local-to-regional social-hydrological case studies will most likely also be needed to guide these actions and to avoid false panacea-type solutions.

6.3 Aligning at the intersection of independent lies

To support effective policy and management in CHANS, the tools and methods used to inform decisions must account for the structural drivers of system behaviour and be adapted to context, and users must acknowledge that observable and unobservable feedbacks mean that optimised solutions are rarely attainable. The underlying models of these tools (formal or informal) must aim to represent these systems not as they “should” work in an idealised reality, but as they really work. If only the observable physical and technical parts of the system are modelled, while ignoring unobservable features of system structure, the drivers of system change can never be understood.

As exemplified in this thesis, transdisciplinary qualitative and quantitative system dynamics-based approaches can be a valuable complement to conventional static and equilibrium-based approaches in this regard. These approaches require an exploratory mindset, incorporating information and knowledge from a wide variety of sources, perspectives and mental models. This can facilitate the necessary transition from treating complex CHANS as consisting of independent elements with stable steady-states to viewing them as integrated and dynamic structures where the outcomes of interventions are fluid, shaped by the context and the constant interplay between co-evolving subsystems. Triangulating between different mental models, perspectives and information sources can be very effective in building representations of the dynamic relationships between the social, environmental and technological dimensions of CHANS. Participatory modelling is one such approach that can add richness to the understanding of reality and help tap into the collective intelligence of cognitively diverse actors who are all part of the system to be managed (Page 2007; Aminpour *et al.* 2021). However, participatory approaches also have their challenges, *e.g.* social dynamics, psychological biases and power relations may (consciously or unconsciously) influence the information expressed in group discussions and, crucially, what remains left unsaid (Lorenz *et al.* 2011; Bang & Frith 2017). This will influence the accuracy of the model, the mental models of the participants and ultimately the management actions chosen. In the best case, participatory modelling can support accurate analysis and lead to alignment around high-leverage, transformative policies. In the worst case, participatory model building can lead to unintended alignment around a representation of reality that is in fact inaccurate, resulting in ineffective and counterproductive policy decisions.

Large gaps still exist in our understanding of the dichotomy between accuracy and alignment in participatory modelling settings. The work in this thesis merely scratched the surface, but the findings indicate that the social dynamics at play during these interventions can strongly influence the model produced. Therefore, the design choices made when building models with others need to be carefully evaluated. On the one hand, one must accept that no-one can fully understand the structure and behaviour of complex systems but that the truth sits somewhere “at the intersection of independent lies” (Levins 1966, p. 423), so all perspectives need to be acknowledged. On the other hand, the participatory modelling process itself is dynamic and participants consciously or unconsciously influence one another through their interactions. The challenge lies in mitigating social influence effects that lead to inaccurate models without losing the leverage achieved by mental model alignment around a shared understanding of the problem. Tuning down social status of the participants is one possible solution (Paper V), but there are many more determinants and factors of social influence (Dechêne *et al.* 2010; Lorenz *et al.* 2011; Becker *et al.* 2017). These need to be studied to further improve the design of participatory approaches to modelling and research more broadly.

6.4 From panaceas to middle-range theories

If we succeed in embracing the complexity of CHANS in our attempts to model and manage them, and if we engage repeatedly with CHANS in different contexts, and study them from different perspectives and through different disciplinary lenses, this may bring us closer to establishing middle-range theories (Meyfroidt *et al.* 2018; Reyers *et al.* 2022) instead of false panaceas. Middle-range theories are context-specific generalisations that describe the causal mechanisms driving well-defined dynamic phenomena, and the conditions that enable or prevent these causal chains (Meyfroidt *et al.* 2018). They sit between single-case descriptions and universal explanatory theories (Schlüter *et al.* 2019d; de Vos *et al.* 2021) and they can help guide Occam's razor when building models for policy and management. Developing middle-range theories of CHANS is still at the forefront of current research (de Vos *et al.* 2021) and supporting this quest, although in small and insufficient steps, is arguably the greatest contribution of this thesis to society.

7. Limitations and future research

In research, there is always more that *could* have been done. More data could have been collected, more interviews or workshops held, more experiments performed, more treatments tested, more aspects explored or more details added to the model. With limited resources, all these '*coulds*' can never realistically be catered for, as the time and money available set practical limitations.

One such limitation in this thesis was the number of case studies, as Fårö island was used as the sole case in Papers II and III. Close engagement with this one geographical case, and extensive collaboration with local stakeholders, facilitated deep understanding of the drivers of system behaviour, but the generalisability of the results is admittedly limited (Pahl-Wostl *et al.* 2021). Although this limitation was partially addressed through the process of triangulating between theoretical synthesis, expert knowledge and other case study literature to support the findings, additional empirical case studies, combined with structured comparative case study analysis methods (Pahl-Wostl *et al.* 2021), would support the development of middle-range theories and yield a better understanding of the conditions under which the insights gained apply (Schlüter *et al.* 2019d). Employing recent approaches and methods for comparative case study analysis, as exemplified by *e.g.* Schlüter *et al.* (2019d) and Pahl-Wostl *et al.* (2021), would be a natural next step in this direction.

Another limitation was the heavy reliance on simulation experiments in Papers IV and V. Simulation models are versatile and powerful tools for understanding complex phenomena (Sterman 2001; Page 2018), but they are also persuasive (Edmonds 2000). Because of the relative ease of simulation model construction with modern software and computational power, simulation models (if carelessly used) can give the illusion of increasing understanding of the mechanisms behind a phenomenon, even though no such progress is made in reality (Edmonds 2000). This risk can be mitigated by complementing any modelling exercise with careful and through use of empirical data, observations, experiments and field studies whenever possible. For instance, the system dynamics-based MCC approach developed in Paper IV should be tested and further evaluated in real-life settings. Similarly, the simulation model of the group model building process applied in Paper V should be complemented, and further developed, by experimental studies exploring how accuracy and alignment evolve in controlled system mapping interventions.

Besides appropriate use of data, embedding the model building process and application in a larger, carefully designed, transparent cooperative process is necessary (Norström *et al.* 2020; Maeda *et al.* 2021). As discussed in Chapter 6, both accuracy and alignment around a model are needed for it to eventually support sustainable transitions. However, much more research is needed to understand the mechanisms that drive these outcomes. Unresolved questions include how the modelling approach can be designed so that it encourages reflexivity, mitigates inequality and power dynamics, leverages collective intelligence, and supports collective action; how the process and context influence the model co-production process; and how to develop instruments to measure model accuracy and group alignment in practical settings. These questions open up exciting opportunities for closer collaboration with other disciplines, such as behavioural science, psychology, metrology, *etc*.

Lastly, there are dimensions of CHANS which were not covered in depth in this thesis. For instance, the main focus was on human-water interactions, while not fully addressing the interplay with the ecological and biological parts of the natural subsystem. Ecological and biological systems are integral parts of the CHANS framework (Liu *et al.* 2021). Changes in water availability and use will impact several biological and ecosystem functions and services (Brauman *et al.* 2007) and most likely give rise additional human-natural interactions that should be examined in future studies.

References

Ahopelto, L., Veijalainen, N., Guillaume, J.H.A., Keskinen, M., Marttunen, M. & Varis, O. (2019). Can There be Water Scarcity with Abundance of Water? Analyzing Water Stress during a Severe Drought in Finland. *Sustainability*, 11(6), 1548. https://www.mdpi.com/2071-1050/11/6/1548

Alberti, M., Asbjornsen, H., Baker, L., Brozovic, N., Drinkwater, L., Drzyzga, S., Jantz, C., Fragoso, J., Holland, D., Kohler, T., Liu, J., McConnell, W., Maschner, H., Millington, J., Monticino, M., Podesta, G., Pontius, R., Redman, C., Reo, N. & Urquhart, G. (2011). Research on Coupled Human and Natural Systems (CHANS): Approach, Challenges, and Strategies. *Bulletin of the Ecological Society of America*, 92, 218-228. https://doi.org/10.1890/0012-9623-92.2.218

Alcott, B. (2005). Jevons' paradox. *Ecological Economics*, 54(1), 9-21. https://doi.org/https://doi.org/10.1016/j.ecolecon.2005.03.020

Allan, J.A. (2005). Water in the Environment/Socio-Economic Development Discourse: Sustainability, Changing Management Paradigms and Policy Responses in a Global System. *Government and Opposition*, 40(2), 181-199. https://doi.org/https://doi.org/10.1111/j.1477-7053.2005.00149.x

Aminpour, P., Gray, S.A., Singer, A., Scyphers, S.B., Jetter, A.J., Jordan, R., Murphy, R. & Grabowski, J.H. (2021). The diversity bonus in pooling local knowledge about complex problems. *Proceedings of the National Academy of Sciences*, 118(5), e2016887118. https://doi.org/10.1073/pnas.2016887118

Anderson, D. & Cavendish, W. (2001). Dynamic Simulation and Environmental Policy Analysis: Beyond Comparative Statics and the Environmental Kuznets Curve. *Oxford Economic Papers*, 53(4), 721-746. http://www.jstor.org/stable/3488610 [2022/08/13/]

Arthur, W.B. (1999). Complexity and the Economy. *Science*, 284(5411), 107-109. https://doi.org/doi:10.1126/science.284.5411.107

Arthur, W.B. (2014). *Complexity and the Economy*. Oxford University Press.

Arthur, W.B., Durlauf, S.N. & Lane, D.A. (2015). Process and Emergence in the Economy. *Complexity and the economy*, 89-102.

Asp, M., Berggreen-Clausen, S., Berglöv, G., Björck, E., Mårtensson, J.A., Lylén, L., Ohlsson, A., Persson, H., Sjökvist, E. & Persson, G. (2015). *Framtidsklimat i Gotlands län - enligt RCP-scenarier [Future climate in Gotland county - according to the RCP scenarios]*. (Klimatologi). SMHI. https://www.smhi.se/publikationer/publikationer/framtidsklimat-i-gotlands-lan-enligt-rcp-scenarier-1.96171

Axelrod, R. & Cohen, M.D. (2008). *Harnessing complexity*. Basic books.

Baker, R.E., Peña, J.-M., Jayamohan, J. & Jérusalem, A. (2018). Mechanistic models versus machine learning, a fight worth fighting for the biological community? *Biology Letters*, 14(5), 20170660. https://doi.org/doi:10.1098/rsbl.2017.0660

Bang, D. & Frith, C.D. (2017). Making better decisions in groups. *Royal Society Open Science*, 4(8), 170193. https://doi.org/doi:10.1098/rsos.170193

Banitz, T., Hertz, T., Johansson, L.-G., Lindkvist, E., Martí-nez-Peña, R., Radosavljevic, S., Schlüter, M., Wennberg, K., Ylikoski, P. & Grimm, V. (2022). Visualization of causation in social-ecological systems. *Ecology and Society*, 27(1). https://doi.org/10.5751/ES-13030-270131

Bankes, S. (1993). Exploratory Modeling for Policy Analysis. *Operations Research*, 41, 435-449. https://doi.org/10.1287/opre.41.3.435

Bano, R., Khiadani, M. & Nyam, Y.S. (2022). System archetypes underlying formal-informal urban water supply dynamics. *Water Resources Management*, 1-16.

Barlas, Y. (1996). Formal aspects of model validity and validation in system dynamics. *System Dynamics Review*, 12(3), 183-210. https://doi.org/https://doi.org/10.1002/(SICI)1099-1727(199623)12:3<183::AID-SDR103>3.0.CO;2-4

Barnosky, A.D., Hadly, E.A., Bascompte, J., Berlow, E.L., Brown, J.H., Fortelius, M., Getz, W.M., Harte, J., Hastings, A., Marquet, P.A., Martinez, N.D., Mooers, A., Roopnarine, P., Vermeij, G., Williams, J.W., Gillespie, R., Kitzes, J., Marshall, C., Matzke, N., Mindell, D.P., Revilla, E. & Smith, A.B. (2012). Approaching a state shift in Earth's biosphere. *Nature*, 486(7401), 52-58. https://doi.org/10.1038/nature11018

Becker, J., Brackbill, D. & Centola, D. (2017). Network dynamics of social influence in the wisdom of crowds. *Proceedings of the National Academy of Sciences*, 114(26), E5070-E5076. https://doi.org/doi:10.1073/pnas.1615978114

Berkes, F., Folke, C. & Colding, J. (2000). *Linking Social and Ecological Systems: Management Practices and Social Mechanisms for Building Resilience*. Cambridge University Press. https://books.google.se/books?id=XixuNvX2zLwC

Bestelmeyer, B.T., Ash, A., Brown, J.R., Densambuu, B., Fernández-Giménez, M., Johanson, J., Levi, M., Lopez, D., Peinetti, R., Rumpff, L. & Shaver, P. (2017). State and Transition Models: Theory, Applications, and Challenges. In: Briske, D.D. (ed.) *Rangeland Systems: Processes, Management and Challenges*. Cham: Springer International Publishing. 303-345. https://doi.org/10.1007/978-3-319-46709-2_9

Biesbroek, R., Dupuis, J. & Wellstead, A. (2017). Explaining through causal mechanisms: resilience and governance of social–ecological systems. *Current Opinion in Environmental Sustainability*, 28, 64-70. https://doi.org/https://doi.org/10.1016/j.cosust.2017.08.007

Biggs, R., Clements, H., De Vos, A., Folke, C., Manyani, A., Maciejewski, K., Martín-López, B., Preiser, R., Selomane, O. & Schlüter, M. (2021a). What

are social-ecological systems and social-ecological systems research? In. 3-26. https://doi.org/10.4324/9781003021339-1
Biggs, R., De Vos, A., Preiser, R., Clements, H., Maciejewski, K. & Schlüter, M. (2021b). *The Routledge handbook of research methods for social-ecological systems*. Taylor & Francis.
Bikhchandani, S., Hirshleifer, D. & Welch, I. (1992). A theory of fads, fashion, custom, and cultural change as informational cascades. *Journal of political Economy,* 100(5), 992-1026.
Black, L.J. (2013). When visuals are boundary objects in system dynamics work. *System Dynamics Review*, 29(2), 70-86. https://doi.org/https://doi.org/10.1002/sdr.1496
Bonabeau, E. (2002). Agent-based modeling: Methods and techniques for simulating human systems. *Proceedings of the National Academy of Sciences*, 99(suppl_3), 7280-7287. https://doi.org/doi:10.1073/pnas.082080899
Boretti, A. & Rosa, L. (2019). Reassessing the projections of the World Water Development Report. *npj Clean Water,* 2(1), 15. https://doi.org/10.1038/s41545-019-0039-9
Borshchev, A. & Filippov, A. (2004). From system dynamics and discrete event to practical agent based modeling: reasons, techniques, tools. In: *Proceedings of the 22nd international conference of the system dynamics society*: Citeseer. 25-29.
Brauman, K.A., Daily, G.C., Duarte, T.K.e. & Mooney, H.A. (2007). The Nature and Value of Ecosystem Services: An Overview Highlighting Hydrologic Services. *Annual Review of Environment and Resources*, 32(1), 67-98. https://doi.org/10.1146/annurev.energy.32.031306.102758
Breiman, L. (2001). Statistical Modeling: The Two Cultures (with comments and a rejoinder by the author). *Statistical Science*, 16(3), 199-231, 33. https://doi.org/10.1214/ss/1009213726
Budyko, M.I. (1961). The heat balance of the earth's surface. *Soviet Geography,* 2(4), 3-13.
Böhringer, C. & Löschel, A. (2006). Computable general equilibrium models for sustainability impact assessment: Status quo and prospects. *Ecological Economics,* 60(1), 49-64. https://doi.org/https://doi.org/10.1016/j.ecolecon.2006.03.006
Carmichael, T. & Hadžikadić, M. (2019). The Fundamentals of Complex Adaptive Systems. In: Carmichael, T., Collins, A.J. & Hadžikadić, M. (eds) *Complex Adaptive Systems: Views from the Physical, Natural, and Social Sciences*. Cham: Springer International Publishing. 1-16. https://doi.org/10.1007/978-3-030-20309-2_1
Carnohan, S.A., Clifford-Holmes, J.K., Retief, H., McKnight, U.S. & Pollard, S. (2021). Climate change adaptation in rural South Africa: Using stakeholder narratives to build system dynamics models in data-scarce environments. *Journal of Simulation,* 15(1-2), 5-22. https://doi.org/10.1080/17477778.2020.1762516

Cilliers, P. (2002). *Complexity and postmodernism: Understanding complex systems*. routledge.
Coletta, V.R., Pagano, A., Pluchinotta, I., Fratino, U., Scrieciu, A., Nanu, F. & Giordano, R. (2021). Causal Loop Diagrams for supporting Nature Based Solutions participatory design and performance assessment. *Journal of Environmental Management*, 280, 111668. https://doi.org/https://doi.org/10.1016/j.jenvman.2020.111668
Cosens, B., Ruhl, J.B., Soininen, N., Gunderson, L., Belinskij, A., Blenckner, T., Camacho, A.E., Chaffin, B.C., Craig, R.K., Doremus, H., Glicksman, R., Heiskanen, A.-S., Larson, R. & Similä, J. (2021). Governing complexity: Integrating science, governance, and law to manage accelerating change in the globalized commons. *Proceedings of the National Academy of Sciences*, 118(36), e2102798118. https://doi.org/doi:10.1073/pnas.2102798118
Cosgrove, W.J. & Loucks, D.P. (2015). Water management: Current and future challenges and research directions. *Water Resources Research*, 51(6), 4823-4839. https://doi.org/10.1002/2014wr016869
Costanza, R., Cumberland, J.H., Daly, H., Goodland, R. & Norgaard, R.B. (1997). *An introduction to ecological economics*. CRC Press.
Costanza, R. & Ruth, M. (1998). Using dynamic modeling to scope environmental problems and build consensus. *Environmental Management*, 22(2), 183-195. https://link.springer.com/content/pdf/10.1007/s002679900095.pdf
Costanza, R., Wainger, L., Folke, C., xe, ler, K.-G., xf & ran (1993). Modeling Complex Ecological Economic Systems. *Bioscience*, 43(8), 545-555. https://doi.org/10.2307/1311949
Creswell, J.W. (2009). *Research Design: Qualitative, Quantitative, and Mixed Methods Approaches*. SAGE Publications. https://books.google.se/books?id=bttwENORfhgC
Crowe, S., Cresswell, K., Robertson, A., Huby, G., Avery, A. & Sheikh, A. (2011). The case study approach. *BMC Med Res Methodol*, 11, 100. https://doi.org/10.1186/1471-2288-11-100
Crutzen, P.J. (2002). Geology of mankind. *Nature*, 415(6867), 23-23. https://doi.org/10.1038/415023a
Crutzen, P.J. (2006). The “anthropocene”. In: *Earth system science in the anthropocene*. Springer. 13-18.
Dahmus, J.B. (2014). Can Efficiency Improvements Reduce Resource Consumption? *Journal of Industrial Ecology*, 18(6), 883-897. https://doi.org/https://doi.org/10.1111/jiec.12110
Daly, H. (2015). Economics for a Full World. Great Transition Initiative.
Daly, H.E. (1991). *Steady-state economics: with new essays*. Island press.
Daly, H.E. (2014). *Beyond growth: the economics of sustainable development*. Beacon Press.
Daniel, C.J., Frid, L., Sleeter, B.M. & Fortin, M.J. (2016). State-and-transition simulation models: a framework for forecasting landscape change. *Methods in Ecology and Evolution*, 7(11), 1413-1423.

de Vos, A., Maciejewski, K., Bodin, Ö., Norström, A., Schlüter, M. & Tengö, M. (2021). The practice and design of social-ecological systems research. In: *The Routledge Handbook of Research Methods for Social-Ecological Systems*. Routledge. 47-63.

Dechêne, A., Stahl, C., Hansen, J. & Wänke, M. (2010). The truth about the truth: A meta-analytic review of the truth effect. *Personality and Social Psychology Review*, 14(2), 238-257.

Deser, C. (2020). "Certain Uncertainty: The Role of Internal Climate Variability in Projections of Regional Climate Change and Risk Management". *Earth's Future*, 8(12), e2020EF001854. https://doi.org/https://doi.org/10.1029/2020EF001854

Deutsch, L. & Troell, M. (2021). Flow and impact analysis. In: *The Routledge Handbook of Research Methods for Social-Ecological Systems*. Routledge. 412-425.

Di Baldassarre, G., Wanders, N., AghaKouchak, A., Kuil, L., Rangecroft, S., Veldkamp, T.I.E., Garcia, M., van Oel, P., Breinl, K. & Van Loon, A. (2018). Water shortages worsened by reservoir effects. *Nature Sustainability*, 1. https://doi.org/10.1038/s41893-018-0159-0

Dickey-Collas, M., Payne, M.R., Trenkel, V.M. & Nash, R.D.M. (2014). Hazard warning: model misuse ahead. *ICES Journal of Marine Science*, 71(8), 2300-2306. https://doi.org/10.1093/icesjms/fst215

Du, L., Hanley, A. & Wei, C. (2015). Estimating the Marginal Abatement Cost Curve of CO2 Emissions in China: Provincial Panel Data Analysis. *Energy Economics*, 48, 217-229. https://doi.org/https://doi.org/10.1016/j.eneco.2015.01.007

Edmonds, B. (2000). Complexity and scientific modelling. *Foundations of Science*, 5(3), 379-390.

Edmonds, B. (2007). Simplicity is not truth-indicative. *Gershenson, C. et al.(2007), Philosophy and Complexity. World Scientific*, 65-80.

Edmonds, B. & Moss, S. (2004). From KISS to KIDS–an 'anti-simplistic'modelling approach. In: *International workshop on multi-agent systems and agent-based simulation*: Springer. 130-144.

Ekvall, T. (2019). Attributional and consequential life cycle assessment. In: *Sustainability Assessment at the 21st century*. IntechOpen.

Ellner, S.P. & Guckenheimer, J. (2011). 1. What Are Dynamic Models? In: *Dynamic Models in Biology*. Princeton University Press. 1-30. https://doi.org/doi:10.1515/9781400840960-004

Epstein, J.M. (2008). Why model? *Journal of Artificial Societies and Social Simulation*, 11(4), 12.

European Environment Agency (2021). *Water resources across Europe - confronting water stress: an updated assessment*. https://doi.org/doi:10.2800/320975

Eurostat (2022). *Water Statistics*. https://ec.europa.eu/eurostat/statistics-explained/index.php/Water_statistics

Evans, M.R. (2012). Modelling ecological systems in a changing world. *Philosophical transactions of the Royal Society of London. Series B, Biological sciences,* 367(1586), 181-190. https://doi.org/10.1098/rstb.2011.0172

Falkenmark, M., Wang-Erlandsson, L. & Rockström, J. (2019). Understanding of water resilience in the Anthropocene. *Journal of Hydrology X,* 2, 100009. https://doi.org/https://doi.org/10.1016/j.hydroa.2018.100009

Famiglietti, J.S. (2014). The global groundwater crisis. *Nature Climate Change,* 4(11), 945-948. https://doi.org/10.1038/nclimate2425

Fang, K., Heijungs, R. & de Snoo, G.R. (2014). Theoretical exploration for the combination of the ecological, energy, carbon, and water footprints: Overview of a footprint family. *Ecological Indicators*, 36, 508-518. https://doi.org/https://doi.org/10.1016/j.ecolind.2013.08.017

FAO & UN-Water (2021). *Progress on the level of water stress. Global status and acceleration needs for SDG indicator 6.4.2*. Rome. https://doi.org/10.4060/cb6241en

Fischer, J., Gardner, T.A., Bennett, E.M., Balvanera, P., Biggs, R., Carpenter, S., Daw, T., Folke, C., Hill, R., Hughes, T.P., Luthe, T., Maass, M., Meacham, M., Norström, A.V., Peterson, G., Queiroz, C., Seppelt, R., Spierenburg, M. & Tenhunen, J. (2015). Advancing sustainability through mainstreaming a social–ecological systems perspective. *Current Opinion in Environmental Sustainability,* 14, 144-149. https://doi.org/https://doi.org/10.1016/j.cosust.2015.06.002

Flyvbjerg, B. (2006). Five Misunderstandings About Case-Study Research. *Qualitative Inquiry,* 12(2), 219-245. https://doi.org/10.1177/1077800405284363

Forrester, J.W. (1961). *Industrial dynamics*. Cambridge, MA, USA: MIT Press.

Forrester, J.W. (1968). *Principles of Systems*. Waltham, MA, USA: Pegasus Communications.

Forrester, J.W. (1969). *Urban Dynamics*. Waltham, MA, USA: Pegasus Communications.

Forrester, J.W. (1988). The Common Foundation Underlying Physical and Social Systems. *The 16th Annual Killian faculty award lecture* Cambridge, MA, USA.

Forrester, J.W. (1994). System dynamics, systems thinking, and soft OR. *System Dynamics Review,* 10(2-3), 245-256. https://doi.org/10.1002/sdr.4260100211

Fu, B. (1981). On the calculation of the evaporation from land surface. *Sci. Atmos. Sin,* 5(1), 23-31.

Gallón, L. (2019). Systemic Thinking. In: Leal Filho, W., Azul, A.M., Brandli, L., Özuyar, P.G. & Wall, T. (eds) *Quality Education*. Cham: Springer International Publishing. 1-11. https://doi.org/10.1007/978-3-319-69902-8_58-1

Gan, G., Liu, Y. & Sun, G. (2021). Understanding interactions among climate, water, and vegetation with the Budyko framework. *Earth-Science Reviews,* 212, 103451. https://doi.org/https://doi.org/10.1016/j.earscirev.2020.103451

Gerring, J. (2008). The Mechanismic Worldview: Thinking inside the Box. *British Journal of Political Science,* 38(1), 161-179. http://www.jstor.org/stable/27568337 [2022/08/23/]

Gleeson, T., Cuthbert, M., Ferguson, G. & Perrone, D. (2020). Global Groundwater Sustainability, Resources, and Systems in the Anthropocene. *Annual Review of Earth and Planetary Sciences,* 48(1), 431-463. https://doi.org/10.1146/annurev-earth-071719-055251

Gleick, P. (2003a). Global Freshwater Resources: Soft-Path Solutions for the 21st Century. *Science (New York, N.Y.),* 302, 1524-8. https://doi.org/10.1126/science.1089967

Gleick, P.H. (2000). A Look at Twenty-first Century Water Resources Development. *Water International,* 25(1), 127-138. https://doi.org/10.1080/02508060008686804

Gleick, P.H. (2003b). Water Use. *Annual Review of Environment and Resources,* 28(1), 275-314. https://doi.org/10.1146/annurev.energy.28.040202.122849

Gleick, P.H. & Cooley, H. (2021). Freshwater Scarcity. *Annual Review of Environment and Resources,* 46(1), 319-348. https://doi.org/10.1146/annurev-environ-012220-101319

Goedkoop, M., Heijungs, R., Huijbregts, M., Schryver, A., Struijs, J. & Zelm, R. (2008). ReCiPE 2008: A life cycle impact assessment method which comprises harmonised category indicators at the midpoint and the endpoint level.

Goldman, K.D. & Schmalz, K.J. (2004). The matrix method of literature reviews. *Health Promotion Practice,* 5(1), 5-7.

Grant, M.J. & Booth, A. (2009). A typology of reviews: an analysis of 14 review types and associated methodologies. *Health Information & Libraries Journal,* 26(2), 91-108. https://doi.org/https://doi.org/10.1111/j.1471-1842.2009.00848.x

Guerrero, A.M., Bennett, N.J., Wilson, K.A., Carter, N., Gill, D., Mills, M., Ives, C.D., Selinske, M.J., Larrosa, C., Bekessy, S., Januchowski-Hartley, F.A., Travers, H., Wyborn, C.A. & Nuno, A. (2018). Achieving the promise of integration in social-ecological research: a review and prospectus. *Ecology and Society,* 23(3). https://doi.org/10.5751/ES-10232-230338

Gutowski, T.G. (2018). A Critique of Life Cycle Assessment; Where Are the People? *Procedia CIRP,* 69, 11-15. https://doi.org/https://doi.org/10.1016/j.procir.2018.01.002

Hammond, R.A. (2015). Considerations and best practices in agent-based modeling to inform policy. In: *Assessing the use of agent-based models for tobacco regulation*. National Academies Press (US).

Hare, M. & Deadman, P. (2004). Further towards a taxonomy of agent-based simulation models in environmental management. *Mathematics and computers in simulation,* 64(1), 25-40.
Hauschild, M.Z. (2005). Assessing environmental impacts in a life-cycle perspective. ACS Publications.
Hauschild, M.Z., Goedkoop, M., Guinée, J., Heijungs, R., Huijbregts, M., Jolliet, O., Margni, M., De Schryver, A., Humbert, S., Laurent, A., Sala, S. & Pant, R. (2013). Identifying best existing practice for characterization modeling in life cycle impact assessment. *The International Journal of Life Cycle Assessment,* 18(3), 683-697. https://doi.org/10.1007/s11367-012-0489-5
Herrera, H., Bianchi, C. & Kopainsky, B. (2018). *Public policy planning to enhance the resilience of socio-ecological systems to climate change: operationalising resilience concepts from a dynamic perspective.* Doctoral dissertation, University of Bergen.
Hertz, T., Mancilla Garcia, M. & Schlüter, M. (2020). From nouns to verbs: How process ontologies enhance our understanding of social-ecological systems understood as complex adaptive systems. *People and Nature,* 2(2), 328-338. https://doi.org/https://doi.org/10.1002/pan3.10079
Hibbard, K., Crutzen, P., Lambin, E., Liverman, D., Mantua, N., McNeill, J., Messerli, B. & Steffen, W. (2006). Decadal-scale interactions of humans and the environment. *Sustainability or Collapse? An Integrated History and Future of People on Earth*.
Hicks, A. (2022). Seeing the people in LCA: Agent based models as one possibility. *Resources, Conservation & Recycling Advances,* 15, 200091. https://doi.org/https://doi.org/10.1016/j.rcradv.2022.200091
Holland, J.H. (2006). Studying Complex Adaptive Systems. *Journal of Systems Science and Complexity,* 19(1), 1-8. https://doi.org/10.1007/s11424-006-0001-z
Holling, C.S. & Meffe, G.K. (1996). Command and Control and the Pathology of Natural Resource Management. *Conservation Biology,* 10(2), 328-337. http://www.jstor.org/stable/2386849 [2022/04/21/]
Hong, L. & Page, S.E. (2004). Groups of diverse problem solvers can outperform groups of high-ability problem solvers. *Proceedings of the National Academy of Sciences,* 101(46), 16385-16389.
Horton, P., Schaefli, B. & Kauzlaric, M. (2022). Why do we have so many different hydrological models? A review based on the case of Switzerland. *WIREs Water,* 9(1), e1574. https://doi.org/https://doi.org/10.1002/wat2.1574
Hovmand, P., Rouwette, E., Andersen, D., Richardson, G., Calhoun, A., Rux, K. & Hower, T. (2011). Scriptapedia: A Handbook of Scripts for Developing Structured Group Model Building Sessions. *Social Science & Medicine - SOC SCI MED*.
Huang, S.K., Kuo, L. & Chou, K.-L. (2016). The applicability of marginal abatement cost approach: A comprehensive review. *Journal of Cleaner Production,* 127, 59-71. https://doi.org/https://doi.org/10.1016/j.jclepro.2016.04.013

Hunt, C.A., Ropella, G.E.P., Park, S. & Engelberg, J. (2008). Dichotomies between computational and mathematical models. *Nature Biotechnology*, 26(7), 737-738. https://doi.org/10.1038/nbt0708-737

Hönisch, B., Ridgwell, A., Schmidt, D.N., Thomas, E., Gibbs, S.J., Sluijs, A., Zeebe, R., Kump, L., Martindale, R.C. & Greene, S.E. (2012). The geological record of ocean acidification. *Science*, 335(6072), 1058-1063.

Ip, E.H., Rahmandad, H., Shoham, D.A., Hammond, R., Huang, T.T.-K., Wang, Y. & Mabry, P.L. (2013). Reconciling Statistical and Systems Science Approaches to Public Health. *Health Education & Behavior*, 40(1_suppl), 123S-131S. https://doi.org/10.1177/1090198113493911

IPCC (2021). *Climate Change 2021: The Physical Science Basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge, United Kingdom and New York, NY, USA: Press, C.U. https://doi.org/10.1017/9781009157896.

ISO (2006) *ISO 14044:2006 Environmental management — Life cycle assessment — Requirements and guidelines*: International Organization for Standardization.

Jaccard, P. (1912). THE DISTRIBUTION OF THE FLORA IN THE ALPINE ZONE.1. *New Phytologist*, 11(2), 37-50. https://doi.org/https://doi.org/10.1111/j.1469-8137.1912.tb05611.x

Jaeger, W.K., Amos, A., Bigelow, D.P., Chang, H., Conklin, D.R., Haggerty, R., Langpap, C., Moore, K., Mote, P.W., Nolin, A.W., Plantinga, A.J., Schwartz, C.L., Tullos, D. & Turner, D.P. (2017). Finding water scarcity amid abundance using human-natural system models. *Proceedings of the National Academy of Sciences*, 114(45), 11884-11889. https://doi.org/doi:10.1073/pnas.1706847114

Jakeman, A.J., Letcher, R.A. & Norton, J.P. (2006). Ten iterative steps in development and evaluation of environmental models. *Environmental Modelling & Software*, 21(5), 602-614. https://doi.org/https://doi.org/10.1016/j.envsoft.2006.01.004

Jesson, J., Matheson, L. & Lacey, F.M. (2011). Doing your literature review: Traditional and systematic techniques.

Jiang, H.-D., Dong, K.-Y., Zhang, K. & Liang, Q.-M. (2020). The hotspots, reference routes, and research trends of marginal abatement costs: A systematic review. *Journal of Cleaner Production*, 252. https://doi.org/10.1016/j.jclepro.2019.119809

Jolliet, O., Margni, M., Charles, R., Humbert, S., Payet, J., Rebitzer, G. & Rosenbaum, R. (2003). IMPACT 2002+: A new life cycle impact assessment methodology. *The International Journal of Life Cycle Assessment*, 8(6), 324-330. https://doi.org/10.1007/BF02978505

Kallis, G. (2010). Coevolution in water resource development: The vicious cycle of water supply and demand in Athens, Greece. *Ecological Economics*, 69(4), 796-809. https://doi.org/https://doi.org/10.1016/j.ecolecon.2008.07.025

Kao, A.B., Berdahl, A.M., Hartnett, A.T., Lutz, M.J., Bak-Coleman, J.B., Ioannou, C.C., Giam, X. & Couzin, I.D. (2018). Counteracting estimation bias and social influence to improve the wisdom of crowds. *Journal of The Royal Society Interface,* 15(141), 20180130. https://doi.org/doi:10.1098/rsif.2018.0130

Kelly, R.A., Jakeman, A.J., Barreteau, O., Borsuk, M.E., ElSawah, S., Hamilton, S.H., Henriksen, H.J., Kuikka, S., Maier, H.R., Rizzoli, A.E., van Delden, H. & Voinov, A.A. (2013). Selecting among five common modelling approaches for integrated environmental assessment and management. *Environmental Modelling & Software,* 47, 159-181. https://doi.org/https://doi.org/10.1016/j.envsoft.2013.05.005

Kesicki, F. (2011). Marginal abatement cost curves for policy making – expert-based vs . model-derived curves. In.

Kesicki, F. & Ekins, P. (2012). Marginal abatement cost curves: a call for caution. *Climate Policy,* 12(2), 219-236. https://doi.org/10.1080/14693062.2011.582347

Kesicki, F. & Strachan, N. (2011). Marginal abatement cost (MAC) curves: confronting theory and practice. *Environmental Science & Policy,* 14(8), 1195-1204. https://doi.org/10.1016/j.envsci.2011.08.004

Kim, B.S., Kang, B.G., Choi, S.H. & Kim, T.G. (2017). Data modeling versus simulation modeling in the big data era: case study of a greenhouse control system. *SIMULATION,* 93(7), 579-594. https://doi.org/10.1177/0037549717692866

Kramer, D.B., Hartter, J., Boag, A.E., Jain, M., Stevens, K., Nicholas, K.A., McConnell, W.J. & Liu, J. (2017). Top 40 questions in coupled human and natural systems (CHANS) research. *Ecology and Society,* 22(2). https://doi.org/10.5751/ES-09429-220244

Laitos, J. & Okulski, J. (2017). *Why environmental policies fail*. Cambridge University Press.

Lall, U., Josset, L. & Russo, T. (2020). A Snapshot of the World's Groundwater Challenges. *Annual Review of Environment and Resources,* 45(1), 171-194. https://doi.org/10.1146/annurev-environ-102017-025800

Law, A.M., Kelton, W.D. & Kelton, W.D. (2007). *Simulation modeling and analysis*. (3). Mcgraw-hill New York.

León-Medina, F.J. (2017). Analytical sociology and its 'syntax' for explanation. *Social Science Information,* 56(4), 502-525. https://doi.org/10.1177/0539018417733820

Levihn, F., Nuur, C. & Laestadius, S. (2014). Marginal abatement cost curves and abatement strategies: Taking option interdependency and investments unrelated to climate change into account. *Energy,* 76, 336-344. https://doi.org/10.1016/j.energy.2014.08.025

Levin, S.A., Xepapadeas, T., Crépin, A.-S., Norberg, J., de Zeeuw, A., Folke, C., Hughes, T.P., Arrow, K.J., Barrett, S., Daily, G.C., Ehrlich, P.R., Kautsky, N., Mäler, K.-G., Polasky, S., Troell, M., Vincent, J.R. & Walker, B.

(2012). Social-ecological systems as complex adaptive systems: modeling and policy implications. *Environment and Development Economics,* 18, 111 - 132.

Levins, R. (1966). THE STRATEGY OF MODEL BUILDING IN POPULATION BIOLOGY. *American Scientist,* 54(4), 421-431. http://www.jstor.org/stable/27836590 [2022/07/27/]

Liberati, A., Altman, D.G., Tetzlaff, J., Mulrow, C., Gøtzsche, P.C., Ioannidis, J.P.A., Clarke, M., Devereaux, P.J., Kleijnen, J. & Moher, D. (2009). The PRISMA Statement for Reporting Systematic Reviews and Meta-Analyses of Studies That Evaluate Health Care Interventions: Explanation and Elaboration. *PLOS Medicine,* 6(7), e1000100. https://doi.org/10.1371/journal.pmed.1000100

Liu, H., Fang, C. & Fang, K. (2020). Coupled Human and Natural Cube: A novel framework for analyzing the multiple interactions between humans and nature. *Journal of Geographical Sciences,* 30(3), 355-377. https://doi.org/10.1007/s11442-020-1732-9

Liu, J. (2017). Integration across a metacoupled world. *Ecology and Society,* 22(4). https://doi.org/10.5751/ES-09830-220429

Liu, J., Dietz, T., Carpenter, S., Folke, C., Alberti, M., Redman, C., Schneider, S., Ostrom, E., Pell, A., Lubchenco, J., Taylor, W., Ouyang, Z., Deadman, P., Kratz, T. & Provencher, W. (2008). Coupled human and natural systems. *Ambio,* 36, 639-49.

Liu, J., Dietz, T., Carpenter, S.R., Alberti, M., Folke, C., Moran, E., Pell, A.N., Deadman, P., Kratz, T., Lubchenco, J., Ostrom, E., Ouyang, Z., Provencher, W., Redman, C.L., Schneider, S.H. & Taylor, W.W. (2007a). Complexity of Coupled Human and Natural Systems. *Science,* 317(5844), 1513-1516. https://doi.org/doi:10.1126/science.1144004

Liu, J., Dietz, T., Carpenter, S.R., Folke, C., Alberti, M., Redman, C.L., Schneider, S.H., Ostrom, E., Pell, A.N. & Lubchenco, J. (2007b). Coupled human and natural systems. *AMBIO: a journal of the human environment,* 36(8), 639-649.

Liu, J.G., Dietz, T., Carpenter, S.R., Taylor, W.W., Alberti, M., Deadman, P., Redman, C., Pell, A., Folke, C., Ouyang, Z.Y. & Lubchenco, J. (2021). Coupled human and natural systems: The evolution and applications of an integrated framework. *Ambio,* 50(10), 1778-1783. https://doi.org/10.1007/s13280-020-01488-5

Liu, J.G., Mooney, H., Hull, V., Davis, S.J., Gaskell, J., Hertel, T., Lubchenco, J., Seto, K.C., Gleick, P., Kremen, C. & Li, S.X. (2015). Systems integration for global sustainability. *Science,* 347(6225). https://doi.org/10.1126/science.1258832

Lopes, R. & Videira, N. (2017). Modelling feedback processes underpinning management of ecosystem services: The role of participatory systems mapping. *Ecosystem Services,* 28, 28-42. https://doi.org/https://doi.org/10.1016/j.ecoser.2017.09.012

Lorenz, J., Rauhut, H., Schweitzer, F. & Helbing, D. (2011). How social influence can undermine the wisdom of crowd effect. *Proceedings of the National Academy of Sciences,* 108(22), 9020-9025. https://doi.org/10.1073/pnas.1008636108

Luna-Reyes, L. (2003). Model Conceptualization: a Critical Review.

Luna-Reyes, L.F. & Andersen, D.L. (2003). Collecting and analyzing qualitative data for system dynamics: methods and models. *System Dynamics Review,* 19(4), 271-296. https://doi.org/10.1002/sdr.280

Maeda, E.E., Haapasaari, P., Helle, I., Lehikoinen, A., Voinov, A. & Kuikka, S. (2021). Black Boxes and the Role of Modeling in Environmental Policy Making. *Frontiers in Environmental Science,* 9. https://doi.org/10.3389/fenvs.2021.629336

Magliocca, N.R., Ellis, E.C., Allington, G.R.H., de Bremond, A., Dell'Angelo, J., Mertz, O., Messerli, P., Meyfroidt, P., Seppelt, R. & Verburg, P.H. (2018). Closing global knowledge gaps: Producing generalized knowledge from case studies of social-ecological systems. *Global Environmental Change,* 50, 1-14. https://doi.org/https://doi.org/10.1016/j.gloenvcha.2018.03.003

Malekpour, S., de Haan, F.J. & Brown, R.R. (2016). A methodology to enable exploratory thinking in strategic planning. *Technological Forecasting and Social Change,* 105, 192-202.

Mancilla García, M., Hertz, T., Schlüter, M., Preiser, R. & Woermann, M. (2020). Adopting process-relational perspectives to tackle the challenges of social-ecological systems research. *Ecology and Society,* 25(1). https://doi.org/10.5751/ES-11425-250129

Markolf, S.A., Chester, M.V., Eisenberg, D.A., Iwaniec, D.M., Davidson, C.I., Zimmerman, R., Miller, T.R., Ruddell, B.L. & Chang, H. (2018). Interdependent Infrastructure as Linked Social, Ecological, and Technological Systems (SETSs) to Address Lock-in and Enhance Resilience. *Earth's Future,* 6(12), 1638-1659. https://doi.org/10.1029/2018ef000926

Marsh, G.P. (1864). *Man and Nature: Or, Physical Geography as Modified by Human Action*. Sampson Low, Son and Marston. https://books.google.se/books?id=4tKNdhQYypgC

Martinez-Moyano, I. & Richardson, G. (2013). Best practices in system dynamics modeling. *System Dynamics Review,* 29. https://doi.org/10.1002/sdr.1495

McCauley, C. (1998). Group Dynamics in Janis's Theory of Groupthink: Backward and Forward. *Organizational Behavior and Human Decision Processes,* 73(2), 142-162. https://doi.org/https://doi.org/10.1006/obhd.1998.2759

Meadows, D.H. (2009). *Thinking in systems : a primer*. London ; Sterling, VA : Earthscan, 2009. ©2008. https://search.library.wisc.edu/catalog/9910100084402121

Mekonnen, M.M. & Hoekstra, A.Y. (2016). Four billion people facing severe water scarcity. *Science Advances,* 2(2), e1500323. https://doi.org/doi:10.1126/sciadv.1500323

Meyfroidt, P., Roy Chowdhury, R., de Bremond, A., Ellis, E.C., Erb, K.H., Filatova, T., Garrett, R.D., Grove, J.M., Heinimann, A., Kuemmerle, T., Kull, C.A., Lambin, E.F., Landon, Y., le Polain de Waroux, Y., Messerli, P., Müller, D., Nielsen, J.Ø., Peterson, G.D., Rodriguez García, V., Schlüter, M., Turner, B.L. & Verburg, P.H. (2018). Middle-range theories of land system change. *Global Environmental Change*, 53, 52-67. https://doi.org/https://doi.org/10.1016/j.gloenvcha.2018.08.006

Michener, W.K., Baerwald, T.J., Firth, P., Palmer, M.A., Rosenberger, J.L., Sandlin, E.A. & Zimmerman, H. (2001). Defining and Unraveling Biocomplexity. *Bioscience*, 51(12), 1018-1023. https://doi.org/10.1641/0006-3568(2001)051[1018:Daub]2.0.Co;2

Milly, P.C.D., Betancourt, J., Falkenmark, M., Hirsch, R.M., Kundzewicz, Z.W., Lettenmaier, D.P. & Stouffer, R.J. (2008). Stationarity Is Dead: Whither Water Management? *Science*, 319(5863), 573-574. https://doi.org/doi:10.1126/science.1151915

Mingers, J. (2000). The contribution of critical realism as an underpinning philosophy for OR/MS and systems. *Journal of the Operational Research Society*, 51(11), 1256-1270.

Mirchi, A., Madani, K., Watkins, D. & Ahmad, S. (2012). Synthesis of System Dynamics Tools for Holistic Conceptualization of Water Resources Problems. *Water Resources Management*, 26(9), 2421-2442. https://doi.org/10.1007/s11269-012-0024-2

Mitra-Kahn, B. (2008). WP 2008-1 Debunking the Myths of Computable General Equilibrium Models.

Moallemi, E.A., Hosseini, S.H., Eker, S., Gao, L., Bertone, E., Szetey, K. & Bryan, B.A. (2022). Eight Archetypes of Sustainable Development Goal (SDG) Synergies and Trade-Offs. *Earth's Future*, 10(9), e2022EF002873. https://doi.org/https://doi.org/10.1029/2022EF002873

Muchnik, L., Aral, S. & Taylor, S.J. (2013). Social Influence Bias: A Randomized Experiment. *Science*, 341(6146), 647-651. https://doi.org/doi:10.1126/science.1240466

Munn, Z., Peters, M.D.J., Stern, C., Tufanaru, C., McArthur, A. & Aromataris, E. (2018). Systematic review or scoping review? Guidance for authors when choosing between a systematic or scoping review approach. *BMC Medical Research Methodology*, 18(1), 143. https://doi.org/10.1186/s12874-018-0611-x

Nicolaidis Lindqvist, A., Fornell, R., Prade, T., Khalil, S., Tufvesson, L. & Kopainsky, B. (2022). Impacts of future climate on local water supply and demand – A socio-hydrological case study in the Nordic region. *Journal of Hydrology: Regional Studies*, 41, 101066. https://doi.org/https://doi.org/10.1016/j.ejrh.2022.101066

Nicolaidis Lindqvist, A., Fornell, R., Prade, T., Tufvesson, L., Khalil, S. & Kopainsky, B. (2021). Human-Water Dynamics and their Role for Seasonal

Water Scarcity – a Case Study. *Water Resources Management*, 35(10), 3043-3061. https://doi.org/10.1007/s11269-021-02819-1

Norström, A.V., Cvitanovic, C., Löf, M.F., West, S., Wyborn, C., Balvanera, P., Bednarek, A.T., Bennett, E.M., Biggs, R., de Bremond, A., Campbell, B.M., Canadell, J.G., Carpenter, S.R., Folke, C., Fulton, E.A., Gaffney, O., Gelcich, S., Jouffray, J.-B., Leach, M., Le Tissier, M., Martín-López, B., Louder, E., Loutre, M.-F., Meadow, A.M., Nagendra, H., Payne, D., Peterson, G.D., Reyers, B., Scholes, R., Speranza, C.I., Spierenburg, M., Stafford-Smith, M., Tengö, M., van der Hel, S., van Putten, I. & Österblom, H. (2020). Principles for knowledge co-production in sustainability research. *Nature Sustainability*, 3(3), 182-190. https://doi.org/10.1038/s41893-019-0448-2

Oreskes, N., Stainforth, D.A. & Smith, L.A. (2010). Adaptation to Global Warming: Do Climate Models Tell Us What We Need to Know? *Philosophy of Science,* 77(5), 1012-1028. https://doi.org/10.1086/657428

Ostrom, E. (2007). A diagnostic approach for going beyond panaceas. *Proceedings of the National Academy of Sciences,* 104(39), 15181-15187. https://doi.org/doi:10.1073/pnas.0702288104

Ostrom, E. (2009). A General Framework for Analyzing Sustainability of Social-Ecological Systems. *Science*, 325(5939), 419-422. https://doi.org/10.1126/science.1172133

Page, S.E. (1999). Computational models from A to Z. *Complexity,* 5(1), 35-41. https://doi.org/https://doi.org/10.1002/(SICI)1099-0526(199909/10)5:1<35::AID-CPLX5>3.0.CO;2-B

Page, S.E. (2007). *The Difference: How the Power of Diversity Creates Better Groups, Firms, Schools, and Societies*. Princeton University Press. https://books.google.se/books?id=FAFVHnJ7uK0C

Page, S.E. (2018). *The model thinker: What you need to know to make data work for you*. Basic Books.

Pahl-Wostl, C. (2007). Transitions Towards Adaptive Management of Water Facing Climate and Global Change. *Water Resources Management,* 21, 49-62. https://doi.org/10.1007/s11269-006-9040-4

Pahl-Wostl, C. (2020). Adaptive and sustainable water management: from improved conceptual foundations to transformative change. *International Journal of Water Resources Development,* 36(2-3), 397-415. https://doi.org/10.1080/07900627.2020.1721268

Pahl-Wostl, C., Basurto, X. & Villamayor-Tomas, S. (2021). Comparative case study analysis. In: *The Routledge Handbook of Research Methods for Social-Ecological Systems*. Routledge. 282-294.

Paré, G., Trudel, M.-C., Jaana, M. & Kitsiou, S. (2015). Synthesizing information systems knowledge: A typology of literature reviews. *Information & Management*, 52(2), 183-199. https://doi.org/https://doi.org/10.1016/j.im.2014.08.008

Pearl, J. (2000). *Causality: Models, reasoning, and inference*. (Causality: Models, reasoning, and inference.). New York, NY, US: Cambridge University Press.

Penny, G. & Goddard, J.J. (2018). Resilience principles in socio-hydrology: A case-study review. *Water Security,* 4-5, 37-43. https://doi.org/https://doi.org/10.1016/j.wasec.2018.11.003

Pizzol, M., Sacchi, R., Köhler, S. & Anderson Erjavec, A. (2021). Non-linearity in the Life Cycle Assessment of Scalable and Emerging Technologies. *Frontiers in Sustainability,* 1. https://doi.org/10.3389/frsus.2020.611593

Preiser, R., Biggs, R., De Vos, A. & Folke, C. (2018). Social-ecological systems as complex adaptive systems: organizing principles for advancing research methods and approaches. *Ecology and Society,* 23(4). https://doi.org/10.5751/ES-10558-230446

Preiser, R. & Cilliers, P. (2010). Unpacking the ethics of complexity: concluding reflections. In: *Complexity, difference and identity*. Springer. 265-287.

Preiser, R., Schlüter, M., Biggs, R., García, M.M., Haider, J., Hertz, T. & Klein, L. (2021). Complexity-based social-ecological systems research: philosophical foundations and practical implications. In: *The Routledge Handbook of Research Methods for Social-Ecological Systems*. Routledge. 27-46.

Quinn, J. & Quinn, C. (2020). Coupled Human and Natural Systems: A Review and Anthrome Case Study. In. https://doi.org/10.1016/B978-0-12-409548-9.12435-2

Radzicki, M. (2010). System Dynamics and Its Contribution to Economics and Economic Modeling. In. 727-737. https://doi.org/10.1007/978-1-4419-7701-4_39

Raychaudhuri, S. (2008). Introduction to monte carlo simulation. In: *2008 Winter simulation conference*: IEEE. 91-100.

Reale, F., Cinelli, M. & Sala, S. (2017). Towards a research agenda for the use of LCA in the impact assessment of policies. *International Journal of Life Cycle Assessment,* 22(9), 1477-1481. https://doi.org/10.1007/s11367-017-1320-0

Repenning, N.P., Kieffer, D. & Astor, T. (2017). The most underrated skill in management.

Reyers, B., Folke, C., Moore, M.-L., Biggs, R. & Galaz, V. (2018). Social-Ecological Systems Insights for Navigating the Dynamics of the Anthropocene. *Annual Review of Environment and Resources,* 43(1), 267-289. https://doi.org/10.1146/annurev-environ-110615-085349

Reyers, B., Moore, M.-L., Haider, L.J. & Schlüter, M. (2022). The contributions of resilience to reshaping sustainable development. *Nature Sustainability,* 5(8), 657-664. https://doi.org/10.1038/s41893-022-00889-6

Reynolds, C.W. (1987). Flocks, herds and schools: A distributed behavioral model. In: *Proceedings of the 14th annual conference on Computer graphics and interactive techniques*. 25-34.

Richardson, G. (2009). System Dynamics, The Basic Elements of. In. 856-862. https://doi.org/10.1007/978-1-4419-7701-4_48
Richardson, G.P. (2011). Reflections on the foundations of system dynamics. *System Dynamics Review*, 27(3), 219-243. https://doi.org/10.1002/sdr.462
Richey, S., Thomas, B., Lo, M.-H., Reager, J.T., Famiglietti, J., Voss, K., Swenson, S. & Rodell, M. (2015). Quantifying Renewable Groundwater Stress with GRACE. *Water Resources Research,* 51. https://doi.org/10.1002/2015WR017349
Ridder, H.-G. (2017). The theory contribution of case study research designs. *Business Research,* 10(2), 281-305. https://doi.org/10.1007/s40685-017-0045-z
Rockstrom, J., Steffen, W., Noone, K., Persson, A., Chapin, F.S., Lambin, E.F., Lenton, T.M., Scheffer, M., Folke, C., Schellnhuber, H.J., Nykvist, B., de Wit, C.A., Hughes, T., van der Leeuw, S., Rodhe, H., Sorlin, S., Snyder, P.K., Costanza, R., Svedin, U., Falkenmark, M., Karlberg, L., Corell, R.W., Fabry, V.J., Hansen, J., Walker, B., Liverman, D., Richardson, K., Crutzen, P. & Foley, J.A. (2009). A safe operating space for humanity. *Nature,* 461(7263), 472-475. https://doi.org/10.1038/461472a
Rockström, J., Falkenmark, M., Allan, T., Folke, C., Gordon, L., Jägerskog, A., Kummu, M., Lannerstad, M., Meybeck, M., Molden, D., Postel, S., Savenije, H.H.G., Svedin, U., Turton, A. & Varis, O. (2014). The unfolding water drama in the Anthropocene: towards a resilience-based perspective on water for global sustainability. *Ecohydrology,* 7(5), 1249-1261. https://doi.org/10.1002/eco.1562
Rovelli, D., Cornago, S., Scaglia, P., Brondi, C., Low, J.S.C., Ramakrishna, S. & Dotelli, G. (2021). Quantification of Non-linearities in the Consequential Life Cycle Assessment of the Use Phase of Battery Electric Vehicles. *Frontiers in Sustainability,* 2. https://doi.org/10.3389/frsus.2021.631268
Rudin, C. (2019). Stop explaining black box machine learning models for high stakes decisions and use interpretable models instead. *Nature Machine Intelligence,* 1(5), 206-215. https://doi.org/10.1038/s42256-019-0048-x
Ruiz Estrada, M.A. (2011). Policy modeling: Definition, classification and evaluation. *Journal of Policy Modeling,* 33(4), 523-536. https://doi.org/https://doi.org/10.1016/j.jpolmod.2011.02.003
Sala, S., Reale, F., Cristobal Garcia, J. & Pant, R. (2016). *Life cycle assessment for the impact assessment of policies*. https://doi.org/10.2788/318544
Scarrow, R.M. (2014). Sustainable Migration to the Urban West. *International Journal of Sociology,* 44(4), 34-53. https://doi.org/10.1080/00207659.2014.979706
Schelling, T.C. (1971). Dynamic models of segregation. *The Journal of Mathematical Sociology,* 1(2), 143-186. https://doi.org/10.1080/0022250X.1971.9989794
Schewe, J., Heinke, J., Gerten, D., Haddeland, I., Arnell, N.W., Clark, D.B., Dankers, R., Eisner, S., Fekete, B.M., Colón-González, F.J., Gosling, S.N.,

Kim, H., Liu, X., Masaki, Y., Portmann, F.T., Satoh, Y., Stacke, T., Tang, Q., Wada, Y., Wisser, D., Albrecht, T., Frieler, K., Piontek, F., Warszawski, L. & Kabat, P. (2014). Multimodel assessment of water scarcity under climate change. *Proceedings of the National Academy of Sciences,* 111(9), 3245-3250. https://doi.org/doi:10.1073/pnas.1222460110

Schlüter, M., Biggs, R., Clements, H., de Vos, A., Maciejewski, K. & Preiser, R. (2021a). Synthesis and emerging frontiers in social-ecological systems research methods. In: *The Routledge Handbook of Research Methods for Social-Ecological Systems*. Routledge. 453-479.

Schlüter, M., Haider, L.J., Lade, S.J., Lindkvist, E., Martin, R., Orach, K., Wijermans, N. & Folke, C. (2019a). Capturing emergent phenomena in social-ecological systems an analytical framework. *Ecology and Society,* 24(3). https://www.jstor.org/stable/26796977 [2022/09/26/]

Schlüter, M., Lindkvist, E., Wijermans, N. & Polhill, G. (2021b). Agent-based modelling. *The Routledge Handbook of Research Methods for Social-Ecological Systems*, 383.

Schlüter, M., Müller, B. & Frank, K. (2019b). The potential of models and modeling for social-ecological systems research the reference frame ModSES. *Ecology and Society,* 24(1). https://doi.org/10.2307/26796919

Schlüter, M., Müller, B. & Frank, K. (2019c). The potential of models and modeling for social-ecological systems research: the reference frame ModSES. *Ecology and Society,* 24(1). https://doi.org/10.5751/ES-10716-240131

Schlüter, M., Orach, K., Lindkvist, E., Martin, R., Wijermans, N., Bodin, Ö. & Boonstra, W.J. (2019d). Toward a methodology for explaining and theorizing about social-ecological phenomena. *Current Opinion in Environmental Sustainability,* 39, 44-53. https://doi.org/https://doi.org/10.1016/j.cosust.2019.06.011

Schoeman, J., Allan, C. & Finlayson, C.M. (2014). A new paradigm for water? A comparative review of integrated, adaptive and ecosystem-based water management in the Anthropocene. *International Journal of Water Resources Development,* 30(3), 377-390. https://doi.org/10.1080/07900627.2014.907087

Schoenberg, W. & Swartz, J. (2021). Building More Robust System Dynamics Models Through Validation. *The 2021 System Dynamics Conference - Virtually Chicago*, Chicago, Illinois, USA.

Schwaninger, M. & Groesser, S. (2016). System Dynamics Modeling: Validation for Quality Assurance. In. 1-20. https://doi.org/10.1007/978-3-642-27737-5_540-3

Senge, P.M., Senge, P.M., Doubleday & Company (1990). *The Fifth Discipline: The Art and Practice of the Learning Organization*. Doubleday/Currency. https://books.google.se/books?id=bVZqAAAAMAAJ

Shackleton, S., Bezerra, J.C., Cockburn, J., Reed, M.G. & Abu, R. (2021). Interviews and surveys. In: *The Routledge Handbook of Research Methods for Social-Ecological Systems*. Routledge. 107-118.

Silverman, E. (2018). Modelling in Population Biology. In: Silverman, E. (ed.) *Methodological Investigations in Agent-Based Modelling: With Applications for the Social Sciences*. Cham: Springer International Publishing. 61-81. https://doi.org/10.1007/978-3-319-72408-9_4

Simon, H.A. (1957). Models of man; social and rational.

Simonović, S.P. (2012). *Managing water resources: methods and tools for a systems approach*. Routledge.

Sivapalan, M., Blöschl, G., Zhang, L. & Vertessy, R. (2003). Downward approach to hydrological prediction. *Hydrological Processes*, 17(11), 2101-2111. https://doi.org/https://doi.org/10.1002/hyp.1425

Sivapalan, M., Savenije, H.H.G. & Blöschl, G. (2012). Socio-hydrology: A new science of people and water. *Hydrological Processes*, 26(8), 1270-1276. https://doi.org/https://doi.org/10.1002/hyp.8426

Sjökvist, E., Mårtensson, J.A., Dahné, J., Köplin, N., Björck, E., Nylén, L., Berglöv, G., Brunell, J.T., Nordborg, D., Hallberg, K., Södling, J. & Clausen, S.B. (2015). *Klimatscenarier för Sverige - Bearbetning av RCP-scenarier för meteorologiska och hydrologiska effektstudier [Climate scenarios for Sweden - Processing of the RCP scenarios for metrological and hydrological effect studies]*. (Klimatologi). SMHI.

Sjöstrand, K. (2020). *Decision support for sustainable water security*. Chalmers Tekniska Hogskola (Sweden).

Sjöstrand, K., Lindhe, A., Söderqvist, T., Dahlqvist, P. & Rosén, L. (2019). Marginal Abatement Cost Curves for Water Scarcity Mitigation under Uncertainty. *Water Resources Management*, 33(12), 4335-4349. https://doi.org/10.1007/s11269-019-02376-8

Smetschka, B. & Gaube, V. (2020). Co-creating formalized models: Participatory modelling as method and process in transdisciplinary research and its impact potentials. *Environmental Science & Policy*, 103, 41-49. https://doi.org/10.1016/j.envsci.2019.10.005

Stave, K. (2010). Participatory System Dynamics Modeling for Sustainable Environmental Management: Observations from Four Cases. *Sustainability*, 2. https://doi.org/10.3390/su2092762

Stave, K., Kopainsky, B., Wubie, M., Mengistu, A., Yibeltal, M., Aynalem, S., Adugna, B.T., Wassie, A., Aragaw, C., Getenet, B., Beyene, B., Abebe, A., Goshu, G., Tadesse, A. & Ayenew, M. (2017). Participatory System Dynamics Mapping for Collaboration and Socioecological Integration in the Lake Tana Region. In. 615-630. https://doi.org/10.1007/978-3-319-45755-0_34

Steffen, W., Broadgate, W., Deutsch, L., Gaffney, O. & Ludwig, C. (2015a). The trajectory of the Anthropocene: The Great Acceleration. *The Anthropocene Review*, 2(1), 81-98. https://doi.org/10.1177/2053019614564785

Steffen, W., Richardson, K., Rockström, J., Cornell, S.E., Fetzer, I., Bennett, E.M., Biggs, R., Carpenter, S.R., Vries, W.d., Wit, C.A.d., Folke, C., Gerten, D., Heinke, J., Mace, G.M., Persson, L.M., Ramanathan, V., Reyers, B. &

Sörlin, S. (2015b). Planetary boundaries: Guiding human development on a changing planet. *Science,* 347(6223), 1259855. https://doi.org/doi:10.1126/science.1259855
Stensen, K., Krunegård, A., Rasmusson, K., Matti, B. & Hjerdt, N. (2019). *Sveriges vattentillgång utifrån perspektivet vattenbrist och torka : – Delrapport 1 i regeringsuppdrag om åtgärder för att motverka vattenbrist i ytvattentäkter [Swedish water resources from a water scarcity and drought perspective: - report 1]*. (Hydrologi 02837722 (ISSN)). http://urn.kb.se/resolve?urn=urn:nbn:se:smhi:diva-5434 [2019-10-02t10:41:00.116+02:00]
Sterling, E.J., Zellner, M., Jenni, K.E., Leong, K., Glynn, P.D., BenDor, T.K., Bommel, P., Hubacek, K., Jetter, A.J., Jordan, R., Olabisi, L.S., Paolisso, M. & Gray, S. (2019). Try, try again: Lessons learned from success and failure in participatory modeling. *Elementa: Science of the Anthropocene,* 7. https://doi.org/10.1525/elementa.347
Sterman, J. (2000). *Business Dynamics: Systems Thinking and Modeling for a Complex World*. McGraw-Hill. https://books.google.se/books?id=hxxlPwAACAAJ
Sterman, J.D. (1984). Appropriate Summary Statistics for Evaluating the Historical Fit of System Dynamics Models. *Dynamica,* 10(2), 51-66.
Sterman, J.D. (2001). System dynamics modeling: Tools for learning in a complex world. *California Management Review,* 43(4), 8-+. https://doi.org/10.2307/41166098
Sultana, F. (2018). Water justice: why it matters and how to achieve it. *Water International,* 43(4), 483-493. https://doi.org/10.1080/02508060.2018.1458272
Sutcliffe, J.V. (2004). *Hydrology: a question of balance*. IAHS Press.
Taylor, R.G., Scanlon, B., Döll, P., Rodell, M., van Beek, R., Wada, Y., Longuevergne, L., Leblanc, M., Famiglietti, J.S., Edmunds, M., Konikow, L., Green, T.R., Chen, J., Taniguchi, M., Bierkens, M.F.P., MacDonald, A., Fan, Y., Maxwell, R.M., Yechieli, Y., Gurdak, J.J., Allen, D.M., Shamsudduha, M., Hiscock, K., Yeh, P.J.F., Holman, I. & Treidel, H. (2013). Ground water and climate change. *Nature Climate Change,* 3(4), 322-329. https://doi.org/10.1038/nclimate1744
Tegegne, G., Park, D.K. & Kim, Y.-O. (2017). Comparison of hydrological models for the assessment of water resources in a data-scarce region, the Upper Blue Nile River Basin. *Journal of Hydrology: Regional Studies,* 14, 49-66. https://doi.org/https://doi.org/10.1016/j.ejrh.2017.10.002
Thaler, R.H. & Ganser, L. (2015). Misbehaving: The making of behavioral economics.
Truong, N., Trencher, G. & Matsubae, K. (2022). How Does Socio-Technical Lock-In Cause Unsustainable Consumption in Cities? A Framework and Case Study on Mobility in Bangkok. *Frontiers in Sustainable Cities,* 4. https://doi.org/10.3389/frsc.2022.770984

Turner, B.L., Clark, W.C., Kates, R.W., Richards, J.F., Mathews, J.T. & Meyer, W.B. (1993). *The Earth as Transformed by Human Action: Global and Regional Changes in the Biosphere over the Past 300 Years*. Cambridge University Press. https://books.google.se/books?id=7Gl0AAAAIAAJ

Turner Ii, B.L., Esler, K.J., Bridgewater, P., Tewksbury, J., Sitas, N., Abrahams, B., Chapin, F.S., Chowdhury, R.R., Christie, P., Diaz, S., Firth, P., Knapp, C.N., Kramer, J., Leemans, R., Palmer, M., Pietri, D., Pittman, J., Sarukhán, J., Shackleton, R., Seidler, R., van Wilgen, B. & Mooney, H. (2016). Socio-Environmental Systems (SES) Research: what have we learned and how can we use this information in future research programs. *Current Opinion in Environmental Sustainability,* 19, 160-168. https://doi.org/https://doi.org/10.1016/j.cosust.2016.04.001

Ulanowicz, R. (2009). A third window. In: *Natural Life beyond Newton and Darwin*. Templeton Press Conshohocken (Pensylvania).

UN-Water (2021). Summary Progress Update 2021: SDG 6—water and sanitation for all. *Geneva, Switzerland*.

United Nations (2010). The human right to water and sanitation :resolution. New York: UN.

United Nations (2015). Transforming our world: the 2030 Agenda for Sustainable Development. *New York, NY, USA.*

United Nations (2018). *Sustainable Development Goal 6: Synthesis Report 2018 on Water and Sanitation*. New York, NY, USA: United Nations. https://doi.org/https://doi.org/10.18356/e8fc060b-en

United Nations (2019). *World Population Prospect 2019*. https://population.un.org/wpp/

Unruh, G.C. (2000). Understanding carbon lock-in. *Energy Policy,* 28(12), 817-830. https://doi.org/https://doi.org/10.1016/S0301-4215(00)00070-7

van der Werf, H.M.G., Knudsen, M.T. & Cederberg, C. (2020). Towards better representation of organic agriculture in life cycle assessment. *Nature Sustainability,* 3(6), 419-425. https://doi.org/10.1038/s41893-020-0489-6

Vennix, J.A.M. (1999). Group model-building: tackling messy problems. *System Dynamics Review,* 15(4), 379-401. https://doi.org/https://doi.org/10.1002/(SICI)1099-1727(199924)15:4<379::AID-SDR179>3.0.CO;2-E

Verburg, P.H., Dearing, J.A., Dyke, J.G., Leeuw, S.v.d., Seitzinger, S., Steffen, W. & Syvitski, J. (2016). Methods and approaches to modelling the Anthropocene. *Global Environmental Change,* 39, 328-340. https://doi.org/https://doi.org/10.1016/j.gloenvcha.2015.08.007

Videira, N., Antunes, P. & Santos, R. (2009). Scoping river basin management issues with participatory modelling: The Baixo Guadiana experience. *Ecological Economics,* 68(4), 965-978. https://doi.org/https://doi.org/10.1016/j.ecolecon.2008.11.008

Voinov, A., Jenni, K., Gray, S., Kolagani, N., Glynn, P.D., Bommel, P., Prell, C., Zellner, M., Paolisso, M., Jordan, R., Sterling, E., Schmitt Olabisi, L., Giabbanelli, P.J., Sun, Z., Le Page, C., Elsawah, S., BenDor, T.K., Hubacek, K., Laursen, B.K., Jetter, A., Basco-Carrera, L., Singer, A., Young, L., Brunacini, J. & Smajgl, A. (2018). Tools and methods in participatory modeling: Selecting the right tool for the job. *Environmental Modelling & Software*, 109, 232-255. https://doi.org/https://doi.org/10.1016/j.envsoft.2018.08.028

Voosen, P. (2017). The AI detectives. *Science,* 357(6346), 22-27. https://doi.org/doi:10.1126/science.357.6346.22

Vörösmarty, C., McIntyre, P., Gessner, M., Dudgeon, D., Proussevitch, A., Green, P., Glidden, S., Bunn, S., Sullivan, C., Reidy Liermann, C. & Davies, P. (2010). Global Threats to Human Water Security and River Biodiversity. *Nature*, 468, 334. https://doi.org/10.1038/nature09549

Walker, W. & Daalen, C. (2013). System Models for Policy Analysis. In. 157-184. https://doi.org/10.1007/978-1-4614-4602-6_7

Wang, S., Fu, B., Zhao, W., Liu, Y. & Wei, F. (2018). Structure, function, and dynamic mechanisms of coupled human–natural systems. *Current Opinion in Environmental Sustainability*, 33, 87-91. https://doi.org/https://doi.org/10.1016/j.cosust.2018.05.002

Woolley, A.W., Chabris, C.F., Pentland, A., Hashmi, N. & Malone, T.W. (2010). Evidence for a collective intelligence factor in the performance of human groups. *Science,* 330(6004), 686-688.

World Health Organization (2021). *Progress on household drinking water, sanitation and hygiene 2000-2020: five years into the SDGs*. (https://creativecommons.org/licenses/by-nc-sa/3.0/igo). Geneva: World Health Organization. https://apps.who.int/iris/handle/10665/345081

Wu, W.-Y., Lo, M.-H., Wada, Y., Famiglietti, J.S., Reager, J.T., Yeh, P.J.F., Ducharne, A. & Yang, Z.-L. (2020). Divergent effects of climate change on future groundwater availability in key mid-latitude aquifers. *Nature Communications,* 11(1), 3710. https://doi.org/10.1038/s41467-020-17581-y

Wunsch, A., Liesch, T. & Broda, S. (2022). Deep learning shows declining groundwater levels in Germany until 2100 due to climate change. *Nature Communications,* 13(1), 1221. https://doi.org/10.1038/s41467-022-28770-2

Xiao, Y. & Watson, M. (2019). Guidance on Conducting a Systematic Literature Review. *Journal of Planning Education and Research,* 39(1), 93-112. https://doi.org/10.1177/0739456x17723971

Xiaoming, F., Qinglong, L., Lichang, Y., Fu, B. & Yongzhe, C. (2018). Linking water research with the sustainability of the human–natural system. *Current Opinion in Environmental Sustainability,* 33, 99-103. https://doi.org/https://doi.org/10.1016/j.cosust.2018.05.012

Yang, Y. & Heijungs, R. (2018). On the use of different models for consequential life cycle assessment. *The International Journal of Life Cycle Assessment,* 23(4), 751-758. https://doi.org/10.1007/s11367-017-1337-4

Yin, R.K. (2009). *Case study research: Design and methods*. (5). Sage Publications, Inc.

Zamagni, A., Guinee, J., Heijungs, R., Masoni, P. & Raggi, A. (2012). Lights and shadows in consequential LCA. *International Journal of Life Cycle Assessment*, 17(7), 904-918. https://doi.org/10.1007/s11367-012-0423-x

Zhang, L., Dawes, W.R. & Walker, G.R. (2001). Response of mean annual evapotranspiration to vegetation changes at catchment scale. *Water Resources Research*, 37(3), 701-708. https://doi.org/10.1029/2000WR900325

Zhang, L., Potter, N., Hickel, K., Zhang, Y. & Shao, Q. (2008). Water balance modeling over variable time scales based on the Budyko framework – Model development and testing. *Journal of Hydrology*, 360(1), 117-131. https://doi.org/https://doi.org/10.1016/j.jhydrol.2008.07.021

Popular science summary

In coupled human and natural systems (CHANS), man and nature are constantly interacting through exchanges of material, energy and information. Thus, changes on one side of these systems are affected by changes on the other side. Processes and conditions in the natural system are shaped by human decisions, while conditions in the human system are influenced by natural processes. Understanding these interactions is fundamental to achieving human well-being and environmental sustainability, but with the rapid expansion and globalisation of human activities in the past century these webs of interactions have grown increasingly complex. Today, human actions in one place or time can have unintended impacts on people or ecosystems miles or years apart. For instance, expanding water use in one region can affect food security in another, investments in polluting technologies today may hamper advances towards cleaner technologies in the future, and policies producing benefits in the short term may have unintended long-term consequences. These are systemic impacts that need to be accounted for when designing strategies and policies for managing CHANS in order to avoid unanticipated consequences. However, widespread overexploitation of the environment, despite well-intentioned management actions, shows that our current understanding of CHANS at the system level is still limited, and that current management practices are not fit for purpose.

In this thesis, the literature on CHANS was reviewed to evaluate how thoroughly common tools used for making management and policy decisions account for the interactions between man and the environment. In a case study on Fårö, northern Gotland, Sweden, the role and management of drinking water in CHANS was studied in particular. For this case area, conceptual and mathematical models were developed and used to map how

society and water interact and co-evolve. These models (digital representations of the real world) were used to run computer simulation experiments, exploring effects of changes in climate and alternative social-economic scenarios on water security. In a follow-up project, a model-based tool was developed to help decision makers compare the marginal cost-effectiveness of alternative water management options and how these interact with one another over time. These models can help managers and decision makers to understand how the interactions between man and the environment shape the effects of management choices. However, building these models is no easy task. Since producing an accurate representation of reality requires knowledge, information and perspectives from different people and different viewpoints, how these different "mental models" are synthesised into one model of reality is critical. This is often done in workshops together with stakeholders and experts, but this thesis showed that workshop design can be critical for the quality of the model produced and for the likelihood of it being used effectively.

This thesis also showed that many of the tools used to design policies and management strategies for the future are not adapted to complex and dynamic systems like CHANS. They tend to leave out important information flows between different parts of the system and to break up the system into independent parts, instead of seeing it as an interconnected whole. Therefore, these tools fail to see how a policy may produce unintended impacts at a different place, or at a different time. In the case study of water resources on the island of Fårö, this was found to result in well-intentioned policies for increasing water availability in the short term leading to escalating water scarcity in the long term. However, the results also showed that by thinking more systemically about water resources and using dynamic simulation models to guide decisions, some of these pitfalls can be avoided. These insights can be applied to any part of CHANS management. However, many challenges still exist, such as how to ensure that the models used to guide management actions in CHANS are accurate and accepted by the intended users, and how to leverage insights from individual cases so they can be used more broadly to support transformative change.

Populärvetenskaplig sammanfattning

I kopplade sociala, ekologiska och fysikaliska system (SEFS) sker ett ständigt utbyte av material, energi och information mellan det mänskliga samhället och de omgivande naturliga miljöerna (bio-, geo-, hydro- och atmosfären). I stort sett all mänsklig aktivitet är beroende av naturliga resurser i någon form, och i stort sett all aktivitet genererar någon form av avfall som till slut hamnar i naturen. På så vis sätter tillgången på resurser och miljömässiga betingelser ramarna och kursen för hur mänskliga samhällen utvecklas, och samtidigt formar vi människor miljön runtomkring oss och dess ekologiska och fysikaliska processer. Detta skapar en väv av ömsesidigt beroende som är viktig att förstå för att uppnå långsiktig social, ekonomisk och ekologisk hållbarhet. Men, på grund av ökande mänsklig aktivitet, omfattande globalisering, och snabb ekonomisk tillväxt har väven som sammanlänkar människa och miljö blivit alltmer komplex och svår att överblicka. Idag kan beslut som fattas vid en tidpunkt i en världsdel få konsekvenser för ekosystem och samhällen många mil eller år bort. Vattenanvändning i en region kan påverka tillgången på mat i en annan, investeringar i fossil teknologi idag kan minska viljan att investera i alternativa teknologier i framtiden, och kortsiktiga politiska beslut kan ha oförutsedda långsiktiga konsekvenser. För att uppnå en hållbar och effektiv förvaltning av SEFS måste vi kunna navigera den här typen av komplexa systemeffekter.

Tyvärr vittnar omfattande miljöförstörelse, storskalig utarmning av naturresurser, och den pågående klimatkrisen om att vi ännu idag har svårt att förstå hur dessa komplexa system fungerar. Trotts omfattande miljölagstiftning och välmenande förvaltningsåtgärder ser vi fortfarande få tecken som tyder på förändring i en hållbar riktning. Det är således motiverat att ställa sig frågan huruvida de metoder och verktyg som idag används i förvaltning av SEFS är lämpliga för ändamålet.

I denna avhandling har litteraturen om SEFS granskats för att utvärdera hur de verktyg och metoder som används som underlag till förvaltnings- och policybeslut redogör för interaktionerna mellan människa och miljö. I en fallstudie på Fårö, norra Gotland, studerades dricksvattnets roll och förvaltning i SEFS. Systemmodeller utvecklades och användes för att kartlägga samspelet mellan vatten och samhällsutveckling, samt hur valet av vattenförsörjningssystem påverkar risken för att vattenbrist uppstår i framtiden. Simuleringsexperiment användes för att utforska sannolikheten för omfattande vattenbrist under ett stort antal klimat- och socioekonomiska framtidsscenarier, samt vilka konsekvenser detta skulle få för regionens hushåll, turistnäring, och vattenförsörjning.

I ett uppföljningsprojekt utvecklades ett modellbaserat verktyg för att hjälpa beslutsfattare att jämföra vilken eller vilka vattenförsörjningsstrategier som var mest kostnadseffektiva. Verktyget jämför marginalkostnaden per kubikmeter vatten som en strategi tillför till systemet, men även hur olika lösningar interagerar och påverkar varandra om de implementeras parallellt, samt hur deras kostnaderna och nyttor fördelas över tid. Dessa modeller kan hjälpa beslutsfattare att förstå hur effekten av ett beslut eller en investering formas av samspelet mellan människa, teknologi och miljö i det sammanhang där det implementeras.

Att modellera komplexa system är dock ingen lätt uppgift. För att skapa en realistisk kopia av verkligheten i en modell krävs kunskap, data och information från många olika perspektiv och kunskapsområden. Hur dessa olika "mentala modeller" sedan kombineras till en matematisk representation av verkligheten är avgörande för huruvida modellen i slutändan är användbar. Ofta gör man detta tillsammans med intressenter och ämnesexperter genom interaktiva workshops och möten, men denna avhandling visade att hur dessa workshops designas kan vara avgörande både för kvaliteten på den producerade modellen, och för sannolikheten att den faktiskt kommer till användning.

Avhandling visade också att många av de verktyg som används i policy- och förvaltningsarbete inte är anpassade för så komplexa och dynamiska system som SEFS. Ofta förbises flödet av information mellan systemets olika delar, och för att underlätta analysen tenderar konventionella metoder dela upp komplexa system i mindre komponenter i stället för att se dem som en sammankopplad helhet. Resultatet blir att man ofta missbedömer vilka systemeffekter ett beslut i en del av systemet kan få på omkringliggande delar. Fallstudien på Fårö visade hur detta kan göra att välmenande åtgärder för att öka vattentillgången på kort sikt kan bidra till eskalerande vattenbrist på lång sikt. Men resultaten i avhandlingen visade också att genom att tänka mer systemiskt kring vattenresurser, och genom att använda simulering och systemstudier för att vägleda beslut, kan vissa av dessa fallgropar undvikas. Detta gäller inte enbart vid förvaltning av gemensamma vattenresurser utan kan tillämpas även på andra delar av SEFS. Många utmaningar kvarstår dock. Till exempel, hur kan man säkerställa att de beslutstödsmodeller som utvecklas är korrekta och att beslutsfattarna har förtroende för dem, och hur kan man dra nytta av lärdomar från enskilda fallstudier så att de kan användas för att stödja transformativ förändring i andra sammanhang?

Acknowledgements

The last five years have been the most challenging, exciting, rewarding, stressful and humbling years of my life. Throughout this roller-coaster ride, I received support, motivation and help from my family, friends and many colleagues. To all of you, I would like to express my deepest gratitude.

To **Julia**, my lovely wife. Thank you for supporting and encouraging me along this entire journey. Your kind words helped me push through even when things were tough.

To my supervisor group, thank you for helping me grow as a researcher and for guiding me when I went off track. **Sammar**, thank you for always being so kind and helpful, and for encouraging me to explore new research interests. **Thomas**, thank you for always giving constructive and thoughtful feedback, and for your great attention to detail. I would like to thank **Linda**, for always being so positive and for helping me understand LCA. Thanks to **Rickard** for all great discussions about systems thinking, modelling and simulation, for challenging me in my thinking and for pushing me to the next level. **Birgit**, thank you joining the supervisor group and mentoring me in SD when I needed it the most. Without you, this thesis would look completely different and so would my mental models of the world.

A big thank you to my friends and colleagues in the **Microbial Horticulture** group at SLU. I was admittedly an odd bird in this group of horticultural and microbial specialists, but I have learned a lot from you all and had a lot of fun along the way. Special thanks to **Beatrix**, for supporting, mentoring and encouraging me to become the best researcher I can possibly be.

I also would like to say thanks to my friends and colleagues at **RISE**: **Sarah**, **Liisa**, the **Industrial transition unit** and many more. Special thanks to **Shane**, **Pontus** and **Bodil** for working around the clock with me to finish Paper V in time for the deadline.

Thanks **Jake** and **Len** for supporting me in my system dynamics journey. One day I hope to be as great a modeller and mentor as you. Lastly, thanks to **Lars** and **Mikael** at Region Gotland for providing data for the studies, and to **Partnerskap Alnarp** and **SMHI** for providing funding for selected parts of the PhD project.

Andreas Nicolaidis Lindqvist
Alnarp, Sweden, 2022

I

sustainability

MDPI

Review

Bio-Based Production Systems: Why Environmental Assessment Needs to Include Supporting Systems

Andreas Nicolaidis Lindqvist [1,2,*], Sarah Broberg [1], Linda Tufvesson [2], Sammar Khalil [2] and Thomas Prade [2]

1 RISE Research Institutes of Sweden, Ideon Beta5, Scheelevägen 17, 22370 Lund, Sweden
2 Swedish University of Agricultural Sciences, Department of Biosystems and Technology, P.O. Box 103, SE-230 53 Alnarp, Sweden
* Correspondence: andreas.nicolaidis@ri.se

Received: 8 July 2019; Accepted: 22 August 2019; Published: 28 August 2019

Abstract: The transition to a bio-based economy is expected to deliver substantial environmental and economic benefits. However, bio-based production systems still come with significant environmental challenges, and there is a need for assessment methods that are adapted for the specific characteristics of these systems. In this review, we investigated how the environmental aspects of bio-based production systems differ from those of non-renewable systems, what requirements these differences impose when assessing their sustainability, and to what extent mainstream assessment methods fulfil these requirements. One unique characteristic of bio-based production is the need to maintain the regenerative capacity of the system. The necessary conditions for maintaining regenerative capacity are often provided through direct or indirect interactions between the production system and surrounding "supporting" systems. Thus, in the environmental assessment, impact categories affected in both the primary production system and the supporting systems need to be included, and impact models tailored to the specific context of the study should be used. Development in this direction requires efforts to broaden the system boundaries of conventional environmental assessments, to increase the level of spatial and temporal differentiation, and to improve our understanding of how local uniqueness and temporal dynamics affect the performance of the investigated system.

Keywords: bioeconomy; bio-based economy; bio-based production systems; environmental assessment; sustainability assessment; LCA; environmental management; systems analysis

1. Introduction

Transitioning to a bioeconomy (or bio-based economy) is a high political priority on both the national and the European level. According to the strategy and action plan of the European Commission, *Innovating for sustainable growth: a bioeconomy for Europe* [1], and the subsequent *Updated Bioeconomy Strategy* [2], the bioeconomy "*encompasses the production of renewable biological resources and the conversion of these resources and waste streams into value-added products, such as food, feed, bio-based products, and bioenergy*". The objective of the transformation is to achieve sustainable development by tackling several societal challenges simultaneously, e.g., ensuring food security, sustainable management of resources, replacing non-renewable resources with renewables, mitigation of and adaptation to climate change, job creation, and maintaining economic competitiveness [3]. In addition to the EU strategy, several countries are currently developing, or have developed, their own bioeconomy strategies, including Sweden, the Netherlands, the US, Malaysia, South Africa, Germany, Finland, and France [4,5]. Even though there are variations in definitions and wording, and there are differences in aims and objectives, driving forces, sustainability perspectives, spatial focus, and in the role of technology innovations (see for example Bugge, et al. [6]), there are certain key characteristics common to most bioeconomy strategies. First, the transition to a bioeconomy calls for the increased production and extraction of biomass. The biomass should be utilized to provide food and feed, as well as broadly replace non-renewable resources across sectors, including the transportation sector, the energy sector, chemical industries, construction, life sciences, etc. Second, enabling the widespread replacement of non-renewable resources by renewables requires research and the commercialization of "green technologies", such as biomass processing, biotechnology, and biorefinery concepts. With innovative technologies, biomass has the potential to be a substitute to oil, gas, and coal in most of their current applications, and thereby reduce our reliance on fossil resources. Third, resource efficiency should be achieved through a cascading use of resources, the valorization of residuals, and the adoption of circular economy principles. Fourth, the transformation towards a circular bioeconomy, and the application of green technologies, is part of the solution to several of the sustainability challenges facing society today (including climate change, ecosystem degradation, resource depletion, biodiversity loss, food insecurity, etc.). Fifth, by expanding the market for bio-based products, transforming the industrial sector, fostering biotechnological innovation and supporting rural development, the bioeconomy enhances economic growth and creates new jobs across all sectors of society [1,4–7].

Ensuring that the multiple sustainability objectives of the bioeconomy transformation are achieved is a significant and challenging task. In this study, we have therefore narrowed our scope by directing our efforts to the specific challenges associated with assessing the environmental dimension of sustainability (from here on referred to simply as sustainability) in bio-based production. Such an assessment requires methods that are both comprehensive and profound, as well as being adapted to the particular characteristics and inherent complexity of the bio-based production systems [8]. Life Cycle Assessment (LCA) has often been described as the key assessment and planning tool for this purpose [3]. LCA is one of the most commonly used tools for environmental assessment, it has been applied for environmental assessment and planning in multiple sectors globally, and it is widely used as a decision support tool for product development, environmental benchmarking, management, and policy development [3,9]. However, despite its popularity, researchers have raised concerns regarding some of the limitations and weaknesses of the LCA methodology, and worries have been expressed that, if not addressed, these limitations and weaknesses can have significant implications for the applicability and reliability of the assessments [9–16]. For example, Reap et al. [12,17], conducted an extensive literature review on unresolved problems related to LCA methods and application, and identified multiple problem areas, with potentially significant implications in terms of reliability and usefulness as a sustainability assessment tool [12,17]. Furthermore, in light of the current bioeconomy discourse, Cristobal et al. [18] state that current limitations in the LCA methodology severely limit our understanding of the environmental implications of bioeconomy value chains, and this constitutes a significant problem for management and policy development. Thus, improving our capacity to assess the environmental impacts of bioeconomy development is of great importance for ensuring the sustainability of the transition at hand.

This need for robust assessment methods, adapted for the specific needs of bio-based systems, and the expressed concerns regarding the inherent limitations of the LCA methodology, constitute the starting point for this study. The aim was to investigate: (1) What are the key characteristics of bio-based production systems that need to be taken into account when assessing their long-term sustainability? (2) How do these affect the suitability and reliability of LCA as the primary assessment and planning tool for the bioeconomy? A third aim was to (3) provide guidance and direction for future research, in terms of important aspects to consider in future assessment and planning of bio-based production systems. Achieving these aims requires both a comprehensive understanding of how bio-based production systems differs from the current, largely fossil based, economic discourse, and how these differences influence the requirements of the sustainability assessment. To this end, the objectives of this paper are: (a) to outline environmental aspects that are of specific importance to address in sustainability assessments of bio-based production systems, compared to systems based on non-renewable resources; (b) to explore the requirements these aspects impose on the sustainability assessment, and to what extent the current LCA methodology fulfils these requirements, and; (c) to provide recommendations on areas to improve in future assessment and planning efforts of bio-based production systems, based on the identified aspects and requirements.

2. Materials and Methods

To answer the first two questions above, we conducted extensive literature studies on the bioeconomy concept and the debate regarding its environmental and ecological sustainability, environmental aspects of bio-based production systems, natural resource management theory, and the role of LCA in the bioeconomy transition. Additionally, a literature review was conducted focusing on the limitations of the LCA methodology when applied to bio-based production systems. The review included published scientific papers (both reviews and articles), books on LCA methodology, reports, and governmental publications. The primary source used for scientific publications was Web of Science. Google Scholar and Google were also utilized for retrieving "grey literature", such as reports, government documents or publications by non-governmental organizations, and for finding references not covered by the Web of Science databases. The scope of the literature search in Web of Science was limited to publications between 2008 and 2018, and further limited by using topic-based searches combining the terms "Sustainability assessment" AND bioeconomy, "Sustainability assessment" AND "bio-based economy", "Sustainability assessment" AND "bio-based system*", LCA AND Weakness*, LCA AND Limitation*, LCA AND "Research need*". The broad choice of search terms was intentionally used to ensure that relevant studies without an explicitly stated bioeconomy/bio-based focus were not excluded. No geographical restrictions were applied, and only studies published in English or Swedish were considered. With these search criteria, 616 publications were found. Additionally, to broaden the scope and add information from other sustainability assessment methods, a reviews-only search was conducted with the terms "sustainability assessment" AND methods, yielding 88 additional publications, giving a total of 704 scientific papers. Titles and abstracts were scanned, and selection criteria for identifying relevant articles, based on the above stated aims and objectives, were applied: (1) to focus on limitations and problem areas in LCA and other methods for assessing environmental sustainability; (2) to focus on bioeconomy and/or bio-based production systems; and (3) to focus on the need for, and approaches to, improving the environmental assessment methodology as a tool for the assessment and planning of bio-based production systems. Articles considered relevant to one or more of the criteria were selected and read in detail. When appropriate, key references of selected articles were also retrieved and included in the literature review. The selection process was carried out by the review first author, however, to ensure the appropriateness of search criteria, selection criteria, coverage, interpretation of data, etc. The full group of authors was regularly consulted throughout the process. In total, 107 scientific articles and review papers, and 28 books, book sections, reports, and other "grey literature" sources were included in the review.

The selected literature was qualitatively analyzed and the results are presented in the chapters below. First, we analyzed the documented key differences between bio-based production systems and systems based on non-renewable resources and explored the requirements for bio-based production systems to be considered sustainable (Section 3.1). Second, we investigated what the characteristics and requirements of bio-based production systems mean for the sustainability assessment in terms of scope, system boundaries and choice of impact categories. We identified impact categories documented as particularly important for the assessment of bio-based production systems, and we explored to what degree these are covered my mainstream assessment methods (Section 3.2). Third, we evaluated what challenges the identified impact categories impose on the sustainability assessment and to what extent current assessment methodology addresses these challenges (Section 3.3). In chapter four, we discuss three areas in need for targeted efforts to address the methodological challenges associated with sustainability assessment of bio-based production systems and, in chapter five, we provide our own reflections and recommendations for future researchers and practitioners to keep in mind, in order to improve the environmental sustainability assessment of bio-based production systems.

3. Results

3.1. What Is a Sustainable Bioeconomy?

Even though environmental sustainability is at the core of the European bioeconomy strategy [2], a fossil-free economy, built upon bio-based production, the cascading use of resources, and advanced green technologies, is not sustainable by default. Biodiversity loss, ecosystem degradation, land-use-change, freshwater depletion, and greenhouse gas emissions are all examples of possible environmental impacts from unsustainably managed bio-based systems [4,19–21]. Therefore, planning and transitioning towards a sustainable bioeconomy calls for assessment methods that are tailored towards the specific environmental issues of bio-based production [19,22]. This, in turn, requires a comprehensive understanding of the specific characteristics of these systems (mode of operation, critical environmental aspects, etc.), and how their environmental issues and sustainability challenges differ from those of the current fossil-dependent discourse [18,19].

In this study, we define bio-based production systems as open, or semi-open, social–ecological systems that combine human technology and biological processes to utilize the ecosystems, their services and biological resources, for the production of food, fiber, biomass or other bio-based products [23]. The concept encompasses traditional cropping and animal systems for food and feed, forestry for timber and energy purposes, fisheries and aquaculture, as well as more novel systems for the production of biofuel and bio-chemicals (e.g., algae farming or bio-energy cropping systems) [23]. Bio-based systems are unique in their inherent capacity of regeneration, allowing biological resource stocks to replenish after extraction. In theory, biological resources can be continuously exploited for eternity as long as two fundamental conditions are met: (a) the rate of extraction does not exceed the rate of regeneration [24], and (b) the extraction, processing, and utilization of the resource, and other external factors, do not diminish its regenerative capacity of the system. If these two criteria are met, the resource can be considered renewable, which is a prerequisite for the system to be considered sustainable [19,24,25].

In contrast, production systems based on fossil resources, minerals, and metal ores, are non-renewable. This means that there is a finite amount of these resources available in the earth's crust and no regeneration occurs (or the regeneration rate and the processes involved in regeneration are so slow that they are neglectable from a human time perspective). Since fossil/non-renewable resources do not regenerate, these systems are not depending on the maintenance of a regenerative capacity. Therefore, in the sustainability assessment of fossil resource systems, greater emphasis should be on ensuring that waste emissions from extraction and utilization do not lead to the degradation of surrounding systems, and that the rate of extraction should not be faster than the rate of development of renewable substitutes to replace the fossil resource [24]. Assessments of bio-based production systems also need to focus on minimizing emissions and damage to surrounding systems, however it is the need to ensure that the regenerative capacity of the system is maintained that is the key difference that makes sustainable management of these systems fundamentally different from their non-renewable counterparts.

It is important to note that the regenerative capacity of biological resources is not static. On the contrary, it is tightly correlated with both the state of the resource stock itself and the state and availability of other limited resources (e.g., water, land, nutrients, soil or suitable habitats [19,22]). This entails that in order for condition (b) to be met, these critical resources need to be maintained within required limits to support regeneration.

Another important difference is that bio-based production systems are typically tightly connected, and in constant interaction, with their surrounding systems [26–28]. For example, a forest is in constant interaction with the surrounding atmospheric system through the exchange of CO_2 and oxygen [29], agricultural systems are tightly connected to, and affected by, the surrounding hydrological system [30], and many fisheries are influenced by the state of distant river and freshwater systems for spawning [31]. Very often, it is through these system interactions that the critical resources for regeneration are maintained within required limits. For example, the productivity and regenerative capacity of an agricultural field is influenced by the capacity of the soil to replenish, retain water, and provide necessary plant nutrients; the surrounding hydrological system influences crop water availability; and pollination is influenced by the capacity and resilience of neighboring ecosystems. These interdependencies are often bilateral. The soil quality is affected by agricultural practices, such as biomass extraction and fertilizer use, and the hydrological cycle is influenced by irrigation practices, and how this change evapotranspiration and water retention time, etc. Thus, interactions with surrounding systems are constantly affecting the rate of regeneration in bio-based systems, and thereby influencing future resource extraction.

In contrast, fossil resources typically do not interact with surrounding systems (ecosystems, social systems, physical systems). This is either because the resource itself is inert (e.g., many metal ores are chemically unreactive), or, as for many petroleum resources, because of physical boundaries isolating the resource from its surroundings, e.g., oil reservoirs are typically confined by some geological formations (e.g., impermeable rock or salt). This physical confinement plays an important role in the chemical formation of the petroleum resource and, more importantly, it isolates the oil and gas from any interactions with surrounding systems. Even though the process of extracting, processing, and utilizing fossil resources often has significant environmental impacts, in the form of greenhouse gas emissions, land degradation, forest clearance, chemical pollution, etc. [32–34]. The change in the state of the fossil resource (the size or quality of the resource stock) does not profoundly influence the surrounding systems. For example, a deep-sea oil deposit does not interact with the surrounding marine ecosystems, with the marine food web, or with the fishing communities utilizing the surrounding waters. Thus, surrounding systems are not affected by changes in the size or state of the resource stock. The same is true in the other direction. Since the future extraction of fossil resources is not dependent on a maintained regenerative capacity, any changes in state of the surrounding systems have very limited influence on the future extraction, i.e., the eutrophication of surrounding waters does not affect the size or state of the oil deposit because the production/formation of the oil has no connection to the state of the surrounding systems.

In summary, a sustainable bioeconomy requires that bio-based production systems are managed so that the rate of extraction does not exceed the rate of regeneration, and that the regenerative capacity of the resource stock is maintained. For this to be possible, management must also consider the interactions between the biological resource stock and the surrounding "supporting" systems responsible for providing the necessary conditions for regeneration. This makes the management of bio-based production systems much more complex than the management of fossil-based systems. Fossil resources are typically systemically inactive, and management primarily needs to focus on minimizing environmental impact from extraction, processing, and utilization. Draining the fossil resource stock typically has no direct implications for surrounding systems, neither do surrounding systems influence the size of the stock, or change the conditions required for exploitation. Thus, exploitation and management of fossil and biological resources require fundamentally different strategies. The latter requires a comprehensive system perspective, where not only the size and state of the primary resource stock is maintained, but also the size and the state of surrounding systems involved in providing the necessary conditions for regeneration.

3.2. Assessing the Environmental Sustainabilityt of Bio-Based Production

Since the production and regenerative capacity of biological resources depend on both the state of the production system itself, and on the state of, and interactions with, neighboring systems, these must all be included in the sustainability assessment. In LCA, this means that impact categories should be chosen so that effects on both the production system and on supporting systems are included in the assessment. For example, in agricultural production, the soil system is one of the supporting systems providing the necessary conditions for biomass production and regeneration (providing plant nutrients, water retention, and growth substrate). Thus, in order to truly assess the sustainability of the production system, the system boundaries of the study need to be sufficiently wide so that impact categories related to the state of important soil parameters are included in the assessment [35].

On a conceptual level, having broad system boundaries, and including multiple, parallel impact categories in the analysis is not a problem. In fact, one of the well-documented assets of the LCA methodology is its capacity to address multiple environmental issues simultaneously, and that this helps avoid burden shifting between environmental impacts, and across time and space [36,37]. Covering multiple impact categories should ensure that efforts for lowering one environmental impact does not unintentionally cause trade-offs with another one, e.g., reducing greenhouse gas emissions at the expense of increased eutrophication [37–39]. However, this is not often the case, as the number of impact categories considered is often restricted to a selected few [40–42]. Also, scanning the LCA literature shows that not only is the representation of impact categories often incomplete, but it is also often highly skewed. Some impact categories are overrepresented, and others are only rarely represented [36,43]. For example, Global Warming Potential (GWP) was included in 98% of livestock LCA studies reviewed by McClellande et al. [42] (from a total of 173 papers published between 2000 and 2016, 169 studies included climate change as an impact category) and in 97% of LCA studies on biofuels reviewed by Lazarevic and Martin [36]. Biodiversity and ecosystem services (ESs), on the contrary, were only covered in 3% of the studies reviewed by McClellande et al. [42], and in none of the ones investigated by Lazarevic and Martin [36]. Water resource depletion and biotic resource depletion were not included as separate impact categories in any of the papers reviewed by Lazarevic and Martin [36], nor by McClellande et al. [42]. Instead, these were incorporated as part of the impact category "resource depletion" (broadly including both biotic and abiotic resources). The often limited and uneven coverage of impact categories in the LCA literature can be explained by a number of factors. Limited time, budget, and data availability are common issues constraining the choice of categories [18]. Trends in politics, research, and media focus are other influential factors [44]. For some impact categories, the availability of quality data and the lack of well-established impact models are other bottlenecks. As examples, assessments of impacts on biodiversity, ESs, and water use often suffer from a lack of quality data and available impact models [9,12]. Thus, these are less likely to be included in a study, compared to other categories with less complex, or better documented, cause–effect chains [30,40,45–47]. The multifunctional nature of bio-based systems, and the system–system interactions they rely on, make adequate impact category coverage particularly important. For example, Lorilla et al. showed how the state of Mediterranean agricultural production systems affects the functioning of several ESs through complex system interactions [48] and in a study focusing on land-use impacts on soil quality parameters, Vida Legaz et al. presented intricate cause–effect relationships shaping the impact pathway, from changes in soil conditions to impacts on biomass production, freshwater provisioning, climate regulation, biodiversity, etc. [35]. Capturing these types of synergies and feedbacks, and ensuring they are covered in the impact assessment, is a challenge in LCA [13,26].

Given time and funding restrictions, limiting the number of impact categories is often the only option available [49], and it is sometimes legitimized as a way of reducing the complexity of the study and providing a clearer message to the audience [18]. However, the uneven coverage of environmental impacts can be problematic for the credibility and usability of LCA results, as it increases the risk of problem shifting [36,42]. It can also give the impression that some environmental issues are non-existent when, in reality, they have simply not been covered by the analysis. This was demonstrated by Berger, et al. [50] in a study comparing the water and carbon footprints of biofuels with those of fossil fuels. The results showed that, if focusing only on carbon footprint, biofuels perform better than fossil fuels due to their relatively lower net CO^2 emissions. However, when adding water footprint, and impacts on freshwater reserves, the sustainability of biofuel production was in many cases less obvious. Thus, impact category choice needs to be justified at an early stage of the assessment [51], and the choice should be tailored to the system studied. For bio-based systems, this entails including necessary impact categories to assess impacts affecting the stock of the resource and its regenerative capacity. Our analysis suggests that there are four impact categories that are particularly important for this purpose:

biotic resource depletion; freshwater use; biodiversity loss; and the degradation of ESs [20,26,52]. Interestingly, these impact categories are also among the least represented in the LCA literature, and several researchers have expressed the need to develop the assessment methodology to better account for these environmental issues [15,42,43,52,53].

3.3. Implications for LCA

Thus far, we have concluded that the sustainable management of bio-based systems requires different strategies compared to systems based on non-renewables, and that some of the most important impact categories to consider in the assessment of these systems are: biotic resource depletion; freshwater use; biodiversity loss; and impacts on ESs. We have also seen that these impact categories are underrepresented in the LCA literature, and partially this is because of limited data availability, and the lack of reliable models describing the effects bio-based systems may have on these impact categories. Due to these limitations, studies with restrictions in terms of time and budget tend to prioritize other impact categories, where data are more easily accessible and impact pathways more well-documented. Next, we will investigate in more detail why these impact categories are so challenging to assess, what requirements they impose on the sustainability assessment, and to what extent current LCA methodology can fulfil these requirements.

3.3.1. Biotic Resource Depletion

"Biotic resources" is a broad concept, encompassing a wide array of biological products and capital, including fish, wood, soil, etc. [54]. Historically, biotic resources have received limited attention in LCA and, in assessments of production systems based on biotic resources, the impacts from the depletion of the resources themselves are not accounted for in most cases [43,55]. The reasons are, in part, because of the lack of reliable indicators for many biotic resources, limited understanding of their associated elementary flows, and missing impact models that account for impacts on both the resource stock itself and indirect impacts on surrounding systems [37,43]. It is only in the last few years, with the growth of the bioeconomy, that the criticality and need for improvements in the impact assessment of biotic resource use have started to become recognized [1,43,55]. Yet, to date, these resources remain poorly addressed in LCA research (e.g., top-soil, forest biomass, and fish stocks) [18,43,56], and there is a lack of consensus on methods for assessing the system level impacts of their exploitation [43,54,57].

One major obstacle is the broad scope of the biotic resource concept, and how to cover the many different types of resources it encompasses. Currently, the coverage of different biotic resources in LCA inventories is far from complete, and the level of aggregation is typically high. For instance, wood biomass, a highly versatile biotic resource, is being harvested from forests across the globe, originating from different tree species, and different ecosystems and habitats (managed and natural). However, in LCA inventories, these different flows of wood biomass are typically referred to as simply "wood", or at best, a distinction is made between "softwood" and "hardwood" [43]. Looking at biotic resources covered by established LCA databases (e.g., Ecoinvent 3.3 [58] and the European Reference Life Cycle Database, ELCD) [59]) confirms that this is not a problem unique for wood, but for most biotic resources. For example, in the ELCD inventory database, "biomass" is represented as a single aggregated category, without any distinction between different types of biomass, its origin, or what species it originates from. Similarly, the Ecoinvent inventory databases aggregate marine fish into a single resource flow, regardless of population or species. This level of aggregation is problematic, as it does not distinguish between biotic resources of the same category taken from different species or habitats (e.g., wood sourced from cosmopolitan vs. endemic species), and it does not account for important ecological aspects, such as variations in species vulnerability, species resilience, minimum viable population size, regeneration rate, etc. With the conditions for sustainable bio-based production systems in mind, this lack of information on ecological characteristics makes it impossible to develop reliable impact models for these resources, and to assess the system-level impacts from their exploitation [43]. Thus, there is an urgent need to improve the currently limited coverage of biotic resources in established life cycle impact (LCI) databases. Particularly urgent, in light of their role for the global economy, are categories such as topsoil, forest biomass, and commercial fish stocks [60].

The next challenge, after increasing the coverage of biotic resources in the assessment, is that of assessing the environmental impacts of their exploitation. To this end, indicator choice will have major implications and, currently, mass accounting is by far the most commonly used method in life cycle impact assessment (LCIA). On the one hand, accounting based on mass is simple and straightforward, extraction/harvest data are often readily available, and direct effects on the resource stock can be easily calculated. However, the approach has important limitations. For instance, characterization of environmental impacts based solely on mass is not straightforward, as the magnitude of impact will depend on the state of the resource stock and its supporting systems [43]. For example, the impact from harvesting 1000 tons of fish from a fish stock that is close to its Biomass for Maximum Sustainable Yield (BMSY) will be very different when compared to the same amount being harvested from a stock that is significantly below its BMSY [61]. In terms of mass, the impact is the same but, looking at regenerative capacity, the latter fish stock is likely to require a substantially longer time to recover from the extraction, because a lower regeneration rate is strongly correlated with population size. Another challenge is in how to account for quality aspects of the resource stock. For instance, the level of genetic diversity within the population will affect recovery rate after harvest, as well as how resilient the reduced stock is to external shocks. Recovery rate and resilience are likely to be greater if the genetic diversity within the population is high, compared to a population where genetic diversity is low [62,63].

Topsoil is another biotic resource in great need of better integration into the LCA methodology. Topsoil can be considered a primary biotic resource that can be depleted both by physical removal (e.g., through erosion or direct human interventions), or by quality degradation (e.g., the depletion of soil nutrients, changes in soil structure, salinization, etc.) [35]. However, similar to wood, the heterogeneity and variability in soil types and soil quality across the globe is significant, and different production systems have different soil requirements (e.g., pH requirements differ between coniferous and broad-leaved tree species). Therefore, assessing environmental impacts from soil degradation requires a comprehensive assessment methodology that takes this variability into account, rather than assuming impacts to be homogenous for different systems in different settings [35]. Further, the soil system is also one of the very important supporting systems involved in maintaining the regenerative capacity of many bio-based production systems. For instance, soil quality and soil productivity significantly affect forest regeneration [64,65], and studies in Poland have shown how variability in soil type, within the same forest, can increase or decrease tree recruitment by up to 300% [66]. Soil quality factors also have a significant impact on agricultural resilience and productivity (for some crops, the correlation coefficient between soil quality and yield has been documented to be as high as 0.9) [67,68].

According to the LCA standards provided by the International Organization for Standardization (ISO), the impact categories chosen in a given study should comprehensively cover environmental issues related to the targeted production system, while taking the goal and scope of the study into consideration [69]. However, as has been presented above, many environmental issues related to system-level impacts, and implications for the longevity and regenerative capacity of biotic resources, are currently not captured by the mainstream LCA methodology. This is alarming, as these constitute factors of great importance for making informed decisions regarding future biotic resource management [43,70].

Among existing efforts to overcome this gap, Langlois et al. presented alternative methods for LCIA of biotic resource depletion in fisheries where the Maximum Sustainable Yield (MSY) concept was incorporated into the impact assessment methodology, together with ecological aspects such as estimations of regeneration capacity [61]. Similar attempts have been made for terrestrial biotic resources [50,71], and Crenna et al. [43] presented an innovative approach where the renewal rate was calculated for a number of natural biotic resources (ranging from terrestrial to aquatic, and from mammals to algae), measured in years required for reproducing one kilogram of the resource after extraction. In general, however, a higher degree of case specificity (ecological features, local conditions, socio-economic structures, etc.) is needed for these approaches to be successful, as they currently do not take into consideration the significant variability in renewability rates—governed by the state of the resource and its interactions with the surroundings [43]. Schneider et al. [54] takes this further, suggesting that even the specific extraction site, and its surroundings, need to be explicitly modelled for an adequate assessment of impacts on biodiversity and ecosystems resulting from biotic resource depletion.

To summarize, assessing the environmental impacts from biotic resource use requires an improved coverage of different biotic resources than what is currently the case. Due to significant differences in renewal rate, geographical distribution, resilience to shocks, etc., between and within biotic resources, the current practice of aggregating these into broad categories, such as "wood" or "fish", makes impact assessment difficult. To really capture the environmental impacts from biotic resource depletion, the specific characteristics of the resource studied needs to be considered, and impacts should be studied from both a "resource perspective", focusing on the state of the resource stock itself, and from a "system perspective", focusing on environmental impacts caused through the interactions and interdependencies between the resource stock and its surrounding systems.

3.3.2. Freshwater Use

Freshwater is a key resource in terrestrial bio-based production systems, and a medium for different types of aquatic production (e.g., freshwater aquaculture and freshwater fisheries) [20]. As with biotic resource use, freshwater use has historically received limited attention in LCA [72]. Most existing impact assessment methods primarily use a volumetric approach [15], focusing on the volume of water extracted from a watershed by a studied activity over a given period of time. The result has often been that regions with a history of abundant water supplies have gradually disappeared from the environmental water debate [73]. This is unfortunate for several reasons. Firstly, freshwater supplies are dynamic and constantly changing and, hence, historically abundant supplies are not a guarantee of future water access. Furthermore, freshwater systems are complex and interconnected structures, often stretching over large geographical areas, and thus water extraction in abundant parts of the watershed can influence the water supply in other areas further downstream in the system [73]. Another important aspect is that freshwater is both an abiotic resource and an environmental compartment, and processes altering the hydrological compartment in one end of a watershed can have serious implications for areas further downstream. For instance, water-polluting substances can be released in low concentrations in one part of the system, without any detrimental impact on water quality, but cause degradation in water quality and restricted water access for distant users, as the pollutants accumulate over time further downstream in the same watershed (e.g., by rendering the water system unsuitable for aquaculture purposes). Based on these characteristics, assessments of the environmental impacts of freshwater use need to take at least three types of usage into account, consumptive use, degradative use, and in-stream use, and do so with indirect upstream and downstream impacts in mind [30,74]. In this review, the focus is on consumptive and in-stream use, as degradative use is typically covered by impact categories related to pollution (e.g., eutrophication and freshwater toxicity) [30].

Most LCIA methods focus on consumptive water use (that is, water that is withdrawn and not released back into its original source) [30]. Water To Availability (WTA) [72], Distance To Target [60], and the Water Stress Index (WSI) [74] are examples of approaches for assessing water consumption in LCIA that attempt to do so by incorporating relative freshwater availability in the impact assessment models. Even though much work has been done in developing these methods, several limitations still exist. Firstly, these methods typically are concerned with water withdrawal required for human activities, and limited attention is given to ecosystem needs. For bio-based production systems, this means that the assessment does not account for how the water consumed by the production system impacts the regeneration rate of the biological resource, nor the impacts on the supporting ecological systems involved in maintaining regeneration. For instance, studies have shown how water stress reduces pollination services in agriculture systems by limiting nectar production, flower development and by reducing habitat suitability for pollinators, thereby undermining the regenerative capacity of the system [75–77]. Canals, et al. [78] and Smakhtin, et al. [79] provided notable exceptions, as their methods not only accounts for human water needs, but also for environmental freshwater requirements (EFR). This is done by including impact pathways between freshwater consumption and the effects on surrounding ecosystems, e.g., by accounting for effects of changes in water availability for aquatic ecosystems, or effects on wetland habitats from lowered groundwater tables. Another limitation, when assessing water consumption, is that data on local water use and availability are often limited, and researchers often need to rely on regional average values, or extrapolate data from previous studies [30]. This typically causes high levels of uncertainty, as contextual and temporal variations in water availability and water needs are not fully captured [30,80]. For instance, studies have shown that river ecosystems can be highly sensitive to periodic droughts, and to the alteration of natural flow regimes caused by temporal peaks in water consumption [81]. Using yearly average values of water use when assessing ecosystem impacts from these activities evens out any inter-annual variations in the water withdrawal and water availability, thus masking potential alterations of the flow regime, and subsequent impacts on the ecosystem. It seems that really assessing the ecosystem impacts from freshwater use requires considerable knowledge regarding the spatial and temporal dynamics of the water extraction, as well as detailed information on the ecosystem's composition and hydrology [82]. In many cases, obtaining information at this level of detail is costly and resource demanding. Thus, a higher contextual resolution of the assessment needs to be balanced against the added costs and effort that this entails. If data collection and model development are too costly or time-consuming, the likelihood of application remains very low despite the potential improvements in model fidelity, and the subsequent quality of the study.

In contrast to the consumptive use, environmental impacts of in-stream water use, and the subsequent alteration of flow regimes, are rarely addressed in LCA studies, even though human structures and activities are known to affect hydrological systems, water resource availability, and ecosystem functions [30]. For example, water regulation, or drainage of wetlands for agricultural purposes, is a common phenomenon known to have affected more than 65% of natural wetlands in Europe and North America [75]. In the short term, wetland drainage might increase agriculture productivity by expanding the land available for agriculture. However, drainage also reduces many important regulating ESs, causing unintended outcomes, and hitting back on agricultural productivity, e.g., by increasing the vulnerability of the agricultural system to extreme weather, and by increased soil and nutrient runoff [75]—undermining the capacity of the system to regenerate. In this review, very few methods for assessing in-stream water use by LCA were identified. Two examples are Humbert and Maendly [83], who developed characterization factors for assessing impacts on aquatic biodiversity from hydropower production, and Gracey and Verones [84], who investigated the effects of hydropower production on aquatic and terrestrial biodiversity. Hydropower production was shown to have significant negative effects on biodiversity and ESs via its impacts on hydrological flows, geomorphology, water quality, and habitat fragmentation [83,84]. However, for many of these impact pathways, the assessment methodology is still not fully developed, or even non-existent, and thus there is a great need for further research and method development [84].

The state of freshwater resources can be a fundamental constraint or facilitator to bioeconomic growth and bio-based production, and efficient management is therefore of great importance [20]. However, the number of studies investigating how the current and future state of freshwater resources may influence the growth of the bioeconomy are few. Exceptions include Rosegrant et al. [20], who investigated future scenarios of how water scarcity might influence food production and food security, and concluded that the effect of bioeconomy development on future water availability and food security will depend on multiple factors. Technology development and adoption, crop selection, historical water-use efficiency, and governance structures for water management are all examples of factors in bioeconomy development that can be detrimental to future water availability. Berger et al. [50] conducted a study on the potential sustainability trade-offs between water use and the carbon footprint of European biofuels, and Ercin and Hoekstra [85] did a similar study on animal products. Furthermore, Veldkamp et al. [86] studied global data from the period 1971–2010, and concluded that human water interventions (land-use and land-cover changes, reservoir constructions, and water consumption) have historically contributed to changes in the geographical distribution of water-stressed regions, as well as to alterations in the dimensions of water scarcity in several of the studied areas. For most river basins, human interventions had an alleviating effect on water stress in the area of implementation, but an aggravating effect for areas further downstream from the intervention (increasing the level of stress for already water-scarce regions, or even pushing some areas into water stress). The overall trend observed, on regional and global levels, was that human interventions historically have caused water stress to travel downstream from the river basin [86]. These studies, the discussion above, and the documented high water use in many bio-based production systems, highlight the importance of thoroughly assessing the potential impacts the bioeconomy may have on freshwater scarcity. It is a possibility that bioeconomy development may become a driver of freshwater scarcity in some regions, and that water availability may become a limiting factor to bioeconomy development in other regions. Unless these impacts are carefully considered in the management of bio-based production systems, these systems may well undermine the long-term sustainability of the bioeconomy and contribute to water conflicts across regions.

Our findings highlight the importance of spatial differentiation and contextualization when assessing water use, and when translating it into impacts on the environment and the sustainability of bio-based production systems [50]. Several other studies support the need for more contextualized and dynamic impact assessment models as a complement to LCA, and the use of scenario analysis for more proactive environmental management [87,88]. The assessment of freshwater use in LCA needs to better acknowledge freshwater as being both a natural resource and a dynamic compartment in nature, and thereby broaden its scope to include not only impacts from consumptive use, but also in-stream and degradative use. For bio-based production systems, more focus should be on how different forms of water use may affect the regenerative capacity of the system. This requires the development of new impact models that include not only effects on human water needs, but also the water needs of surrounding ecosystems.

3.3.3. Biodiversity and ESs

Despite ambitious international and national targets for species conservation and habitat protection (e.g., the Strategic Plan for Biodiversity 2011–2020, following the Convention on Biological Diversity [89], and the Swedish environmental quality objectives [90]), biodiversity losses and ecosystem degradation continue in large parts of the world [45]. In the day-to-day debate, and in most assessment studies, the term biodiversity refers primarily to species diversity (the number of different species in a given area), and impact on biodiversity is measured as the change in species diversity resulting from a studied activity [45,91]. However, the term biodiversity denotes other features besides species level that are less often considered in the assessment, e.g., functional diversity (the function provided by a species or a combination of species in an area), genetic diversity (the genetic variation within a population), ecosystem diversity (the variety of different ecosystems within an area), etc. [45,92]. Additionally, there are qualitative aspects assigned to biodiversity (e.g., conservation targets, conservation status, species abundance, etc.) that are not fully considered when biodiversity impact assessment is limited to changes in species diversity [92]. These different dimensions of biodiversity are constantly interacting in ways that are not well understood. For instance, species diversity and composition influence functional diversity, and genetic diversity impacts on ecosystem dynamics [70].

The functions and ESs (including provisioning ESs, regulation and maintenance ESs, and cultural ESs [89]) provided by biodiversity through its different dimensions play a central role in the bioeconomy—both in terms of ecological sustainability and the intrinsic value of biodiversity [93], and also in ensuring high productivity, maintaining regenerative capacity, and ensuring the resilience of bio-based production systems. The most obvious link is seen when treating biodiversity as a resource, and an integral part of our natural capital [94]. Over-utilizing a species, causing its extinction, results in a loss in biodiversity, and subsequently a biotic resource that was previously utilized (or had the potential for future utilization) is no longer available for human use (losses in crop diversity and the extinction of commercial fish stocks being notable examples [95,96]). In effect, this means that the resource base of the bioeconomy is eroded, and the potential for bioeconomic growth is diminished. However, there are also other, subtler, ways in which biodiversity affects the regeneration, and long-term productivity, of bio-based production. For instance, in agriculture, biodiversity influences productivity and the rate of regeneration through pollination ESs. A higher diversity of wild pollinators can contribute to higher crop yields [97]. The richness and diversity of pollinators is furthermore affected by ecosystem diversity, where a more mixed and heterogenous landscape provides habitats for a greater diversity of pollinating insects, compared to a more homogenous, monoculture dominated, landscape [98]. Case studies have even shown that increasing ecosystem diversity, by preserving forest habitats as part of the agriculture landscape, can boost pollinator diversity, and improve crop productivity and farmers revenue by as much as 29% for smallholder farms in Tanzania [98]. In forestry, the effects of biodiversity on productivity have also been extensively studied, both in looking at the species diversity–productivity relationship, and also at the effects of forest structural diversity on stand productivity. Results from forestry suggest that both higher species diversity and structural diversity may increase production [99,100], and in fisheries, studies have shown that boosting population diversity can make the production system more stable and resilient to external disturbances [63]. Case studies on salmon fisheries even suggest that boosting population diversity can reduce the frequency unintended fisheries closures due to population collapse by up to ten times compared to a scenario with very low population diversity [63]. In other words, species and structural diversity can increase system resilience, strengthen ESs, and improve the regenerative capacity of the production system [99,101]. In agriculture, these positive effects of diversity on system stability is part of the reason for the growing interest in enhancing the crop genetic diversity in order to improve the climate resilience of agriculture systems [95]. On the soil level, recent studies have shown that soil microbial biodiversity (including species, functional and genetic diversity) plays a prominent role in governing plant productivity, supporting soil formation and nutrient cycling, improving plant resource-use efficiency, and enhancing plant stress resilience. More alarmingly, modern intensive management practices, of these systems, tend to reduce soil microbial diversity, thereby contributing to long-term erosion of system productivity [102].

Thus far, we have concluded that the dominant, and largely unidimensional, approach of measuring biodiversity, and changes in biodiversity, is too simplistic for assessing biodiversity impacts. We have also concluded that this approach can hide some of the environmental consequences of an activity and endanger the long-term sustainability of the studied system. For example, when assessing biodiversity with only a species diversity focus, losses in genetic diversity due to overharvesting can be masked by maintained levels of species diversity. Similarly, if endemic species are replaced by non-native species, the species diversity is unchanged, but impacts on the ecosystem, and the functions provided by the ecosystem, can be significant [103]. To truly assess the sustainability of bio-based production systems, and to avoid unintended negative impacts caused by reductions in one or more biodiversity dimensions, a significantly larger spectrum of biodiversity needs to be considered, rather than what is typically the case. This will require the identification of representative biodiversity indicators for the different dimensions, standardized methods of measuring these, and the development of new models for impact assessment [45,70,104]. Although multiple biodiversity indicators already exist (see for example www.bipindicators.net), the different dimensions of biodiversity are unevenly covered. Most indicators focus on species diversity, followed by ecosystem diversity, whereas genetic diversity and qualitative aspects of biodiversity are underrepresented [45]. Further, many existing indicators are only applicable to specific geographical regions, or they are calculated based on very location-specific indicator species. On the one hand, this potentially allows for very accurate assessments but, on the other hand, the indicators cannot easily be generalized and applied to areas outside their original region. The development of biodiversity indicators with a cosmopolitan representation, or indicators that can be adapted to the location of the assessment, is much needed [45].

Alongside the development of indicators for biodiversity and ESs, there is also a need to improve the impact assessment models, linking human activities to their effects on the studied indicators. The challenge here is that, for many ecosystems, these impact pathways are not fully understood and/or the data requirements for them to be calculated is not available at a sufficient spatial or temporal precision [45]. According to Chaplin-Kramer et al. [105], the most commonly used LCA methodologies lack the contextual resolution and detailed ecological information required to model the impact pathways from activity to biodiversity impact. In terms of ESs, most guidelines on LCA (e.g., ISO 14040 and 14044, and the International Reference Life Cycle Data System (ILCD) guidelines [1,106]) do not incorporate the ES concept, and the few studies that have done so predominantly focus on provisioning ESs, whereas regulation and maintenance ESs, and cultural ESs, are much less documented [41]. There is a clear risk that this imbalanced coverage of ESs will result in biased interpretations of LCA results, as some services are not covered in the assessment process [41].

One important obstacle to assessing functional biodiversity and ES impacts in LCA lies in the LCA methodology's limited capacity to manage multifunctionality and nonlinear relationships [70,105]. Many ESs are facilitated by the combined effects of multiple ecosystem functions, and their response to stress is often characterized by threshold and nonlinear behavior. This multifunctionality and, at times, counterintuitive behavior is hard to capture with conventional assessment methods [41,107]. For example, the ESs of soil formation are of great importance for the longevity of terrestrial bio-based production, and they are driven by several different processes (e.g., decay of organic matter and mineral weathering), and also feeds into a number of other ESs (primary production, mediation of flows, mediation of biota, etc.) [108]. The different drivers involved in providing ESs, and the interactions, interdependencies, and nonlinear relationships among these processes, are not fully understood, but are highly relevant when assessing the environmental impacts of bio-based production systems. Building this understanding will require research focused on entangling the cause–effect chains linking human activities to environmental impacts, and how these impacts affect ES values and services [41]. To this end, Teillard et al. [47] suggest the increased use of data, models, and modelling methods from the field of ecological science. Othoniel et al. [41] also stress the importance of increased multi- and interdisciplinary approaches to improve the impact assessment models of ESs in LCA. Focusing on improving the spatial and temporal resolution of these models is particularly important, as ESs are often the context-specific sum of multiple functions provided by the specific ecosystem [109]. Since ecosystems, and ecosystem composition, are dynamic and constantly evolving over time, the assessment of their response to impact need to take these dynamics into account as well [110]. For example, the two ESs flood regulation and climate regulation are strongly dependent on the dominating land cover in the studied area. In simple terms, grasslands tend to have a high flood regulation potential but modest climate regulation potential, forests tend to have a higher climate regulation potential compared to grasslands, and croplands, in general, have limited potential for both flood and climate regulation [111]. However, on a landscape level, the transition from one land cover type to another is rarely instantaneous but is rather a transitional process that gradually changes ES potential. Additionally, the potential of the different ESs is also influenced by the surrounding land cover matrix and thus, assessing the long-term impacts on ESs from changes in land cover requires that these interactions are also considered, accounting for the constant evolution of the landscape.

In general, the assessment of impacts on biodiversity and ESs in LCA needs better coverage. Both biodiversity and ESs are complex concepts, involving multiple dimensions and services provided to society. Thus, aggregating these into single impact categories, quantified by a limited set of indicators and characterization factors (as is standard in LCA methodology), may be an oversimplification of reality, with questionable value for environmental management [41]. Instead, indicators for multiple biodiversity dimensions and ESs are needed, as well as a better understanding of how these dimensions influence the long-term sustainability of bio-based production systems. It is of special importance to enhance the understanding of how human activities impact the genetic and functional dimensions of biodiversity, and how to use this knowledge to design bio-based production systems that support the necessary ESs for long-term productivity and overall sustainability. To achieve this, researchers have suggested closer collaboration with other disciplines, e.g., ecology, systems thinking, and ecosystem science, to improve the impact models, and to better account for the multifunctionality associated with several ESs [13,41,47,112].

4. Concluding Discussion

We have shown above that for bio-based production systems to be sustainable, they need to be managed so that the extraction of the resources they provide does not exceed the regeneration rate of the system, and the regenerative capacity of the system must not be diminished by the processes of production, extraction and resource utilization, or by other external factors. The regenerative capacity of the systems is often governed by interactions, and through interdependencies, with surrounding supporting systems. These supporting systems are involved in maintaining the necessary conditions, and providing the critical resources, for regeneration. With these requirements in mind, sustainability assessments of bio-based production systems require a perspective and scope where both the states of the primary system and supporting systems are accounted for. This means that impact categories need to be chosen so that effects on both the production system and the supporting systems are covered. In this study, we have focused on the impact categories "biotic resource depletion", "freshwater use", and impacts on "biodiversity and ESs", as these are particularly important aspects for many bio-based production systems and, also, these constitute aspects that have historically been underrepresented in sustainability assessment studies.

Based on the results of our assessment, we believe there is a need to develop and adapt existing sustainability assessment methods and frameworks to the requirements of bio-based production systems. We also believe there is a need to improve the capacity of the assessment methods to better account for the specific features of impact categories typical for bio-based production systems. Development in this direction poses challenges for several reasons.

First, as described above, the studied impact categories are characterized by being broad, multidimensional concepts but, in most assessments, these multiple dimensions are not accounted for. Instead, a simplistic approach is often taken, focusing on a single dimension or on highly aggregated indicators. To truly assess the environmental sustainability of bio-based production systems, these impact categories need to be disaggregated more, and assessed in their different dimensions and sub-categories. Only at a more disaggregated level can the multiple impact pathways between the impact category and the production and supporting systems be captured in the assessment. What level of disaggregation can be considered "enough" is a question beyond the scope of this review, but it probably depends on how many dimensions are represented in the target system, and the time and resources available to the study. If one accepts the need for disaggregation, this calls for an expanded, and a more detailed, set of indicators than what is often used in sustainability assessments of these systems [45,70,88]. Some indicators are more important for assessing direct impacts (e.g., amount of fish harvested from a fish stock directly affects the stock size), and other indicators are more important for assessing indirect impacts related to the regenerative capacity of the system (e.g., effects of reductions in genetic diversity on regeneration rate in fisheries). Currently, the latter type of indirect impact pathways is poorly covered, partly because it often involves impact pathways characterized by a high degree of nonlinearity, e.g., a sudden collapse of ecosystem functions when the ecosystem is pushed beyond a certain threshold/tipping point [70,88,113].

Covering impacts on both the primary production system and the supporting systems calls for an expansion of the system boundaries of the assessment beyond normal practice. Adopting such a broad system perspective is a challenge in terms of resource requirements, and also in terms of modelling capacity, data availability, and the often-limited conceptual understanding of several of the impact pathways governing the sustainability of bio-based production systems [70,113]. To better capture these impact pathways in the assessment requires targeted efforts towards several methodological challenges.

4.1. Accounting for Spatial Variation

More efforts need to be directed towards improving the level of spatial detail in the assessment, and how variations in geology, topography, land cover and other physical geographical features (also referred to as spatial variations [12]) affect the sustainability of bio-based production systems [12]. For instance, biodiversity and many ESs depend on local geography and landscape configuration [105]. Impacts on biodiversity and ESs from agriculture expansion have, for example, been shown to be strongly affected by the configuration of the surrounding landscape, as this influences the availability of suitable habitats for biodiversity and natural pollinators. Spatial variations in the surrounding landscape also affect local hydrology and cause potential soil erosion [105]. As agricultural systems are heavily dependent on these factors for their regeneration, and also heavily influence them, local geography and spatial variability clearly need to be accounted for in the sustainability assessment of these and other bio-based systems [26].

Increasing the level of spatial differentiation in the assessment requires location-explicit data, and the development of geographically tailored impact models [46]. Typically, however, the necessary data to do this at the local and sub-regional scale are missing, and researchers are left to rely on the extrapolation of national or regional averages [40,51,114]. This can strongly reduce the representativeness of the results of the assessment, especially if the input values are based on national averages for a large and heterogeneous country, encompassing large variations in geology, topography, land cover etc. In such a case, assuming the impact pathways will be homogenous for the entire reference area might be an oversimplification, as these can vary significantly with spatial variations and geographical heterogeneity [14,115].

Accounting for spatial variability to a greater extent than is the case today is likely to be of great importance for ensuring the sustainable development of the growing bioeconomy. The ongoing transformation towards bio-based production, and the increasing use of biomass, is likely to lead to an economy where feedstock is produced, sourced, and processed in a variety of geographically diverse locations—even more so than in the case of current fossil supply chains [116,117]. Transformation towards a more distributed and regionalized, or even localized, economy entails an increasing degree of spatial variation across production systems that needs to be accounted for, in order to ensure that the sustainability of these systems is reliably assessed [9]. In order to achieve this, several researchers have proposed approaches where multiple assessment methods, or features from different assessment approaches, are used in combination [13,46,112,118]. For example, Jeswani et al. [118], and Ford et al. [39] suggested using LCA in combination with Environmental Impact Assessment (EIA), as the EIA approach is designed to take local geography, and potential background pressures that are typically not considered in LCA, into account. The regionalization of LCA, using Geographical Information Systems (GIS), is another promising approach under development [46]. For example, the LCA software OpenLCA [119] allows the use of GIS data for location-specific inventory development, and the defining of regionalized impact factors in the assessment process [120]. However, challenges remain, both in the collection and availability of spatially explicit data, and in how to incorporate necessary spatial features, such as landscape configuration, into the impact models. These are important aspects for ensuring meaningful impact assessments at the regional and sub-regional scale [105], and require an increased use of primary and secondary local data (e.g., with the help of GIS and satellite and image analysis technologies), and for an increased use of local knowledge that would be integrated into the impact models [13,105]. Development in this direction could increase the accuracy of the assessment, but also require more time and effort devoted to data collection, processing, and analysis, as well as increasing the level of complexity of the assessment.

4.2. Local Environmental Uniqueness

Closely related to spatial variations are what Reap et al. [12] describes as features of "local environmental uniqueness". This denotes non-physical, spatially varying, parameters and characteristics of a system that influence how sensitive the system is to external pressures. Examples include soil quality factors, soil buffering capacity, population density, etc. [12], but environmental uniqueness can also refer to qualitative aspects, such as the type of farming practices used, or variations in qualitative and genetic aspects of biodiversity (e.g., occurrence of endemic or red-listed species) [12,41]. These factors can strongly influence the sustainability of a production system and its environmental impacts. For bio-based production systems, these, often intangible, factors can be particularly important to consider in the assessment, as they influence the production system, its surrounding supporting systems, and the shape and magnitude of interlinkages between the two [13,26]. For example, soil quality properties, such as soil organic carbon (SOC) content, water holding capacity, texture, chemistry, microbiology, etc. are examples of local environmental uniqueness of great importance for soil productivity and resilience [121,122]. In an agricultural production system, productivity and environmental impact are both strongly affected by these soil quality parameters [41]. High-quality soils can give greater yields per unit effort than low-quality soils, and lower quality soils may require more intensive farming practices to be economically productive, e.g., in the form of additional fertilizer use and intensified tillage, increased nutrient runoff, and subsequent environmental impacts [123]. Since soil quality can vary significantly within and between regions, this aspect of local environmental uniqueness can strongly influence farming practices and, subsequently, the sustainability of seemingly very similar production systems [26].

It is also important to keep in mind that many aspects of local environmental uniqueness are not static but are rather highly dynamic parameters that are continuously changing in response to external pressures. For instance, SOC contributes to several beneficial soil functions, including soil productivity, carbon sequestration, and water and nutrient retention [124]. Soil tillage practices can also increase agricultural productivity by improving soil structure. However, long term, intensive tillage can also cause the depletion of SOC by accelerating decomposition [124]. With losses of SOC beyond a certain threshold, the benefits previously provided by SOC start to decline, and soil productivity is eroded, further increasing the need for an intensification of tillage and other farming practices, in order to maintain productivity. The result is a reinforcing feedback loop of decreasing soil quality, leading to reductions in soil productivity. This simple example illustrates that soil quality, and other parameters of environmental uniqueness, cannot be treated as spatial and/or temporal constants in the sustainability assessment of these systems. At a high level of SOC, or any other parameter of environmental uniqueness, the environmental impacts from a production system (e.g., tillage farming) may be negligible, or even beneficial, for productivity. From a long-term perspective, however, if the disturbance continues, and the parameter decreases beyond a certain level, this can trigger feedback loops causing the environmental impacts of the previously sustainable production system to escalate.

Most methods for environmental assessment, including state-of-the-art LCA, tend not to capture many important aspects of local environmental uniqueness. At best, a simple characterization is made where the dimensions of environmental uniqueness are organized into discrete categories (e.g., farming practices are commonly categorized into conventional vs. organic [26]), but, in many cases, such differentiation is non-existent (e.g., in biodiversity assessments, species vulnerability or endemism are rarely accounted for [43]). The reality is that dimensions of environmental uniqueness cannot be treated as discrete categories, but should rather be seen as a range, or spectra [26]. The studied system is constantly moving along these spectra, shaped by synergies and feedback interactions triggered by its own management, and by influences from surrounding systems [26].

Accounting for environmental uniqueness, and for the interactions and emergent system properties these create (feedback relationships, system thresholds, etc.), is a significant challenge that needs to be addressed in order to further improve the sustainability assessment of bio-based production systems [112,113,125]. While significant research is being conducted on developing more regionalized, or even location specific, environmental assessments [126,127], the limited availability of impact models accounting for the unique features of the local environment remains an obstacle. Instead, practitioners are often forced to rely on more readily available, site-generic impact models, due to the lack of detailed knowledge regarding the local uniqueness of the territory studied [15,26,126,128]. Adding the necessary level of detail to the assessment will often require extensive data collection, and close collaboration with local stakeholders and experts from multiple disciplines and sectors [112,126,129]. This approach, building on local knowledge as a central component in the assessment and management planning, has been advocated in several studies, including case studies on coastal and freshwater fisheries in Sweden, the Mediterranean, Brazil and Southeast Asia [130–132], forest biodiversity management in Europe [133], and on sustainable agriculture development in the UK [134]. Furthermore, examples of collaborations across scientific disciplines and methods to capture and analyze this environmental uniqueness have been documented in studies where, e.g., LCA methods have been combined with GIS [120,126], System Dynamics modelling [88], Ecosystem modelling [105], and Group Model Building [135].

There is a risk that important environmental impacts may be overlooked if environmental uniqueness is not considered in the assessment [92,105]. However, adding levels of detail to the assessment exemplified above is resource intensive, and requires a greater degree of cross-discipline collaboration and stakeholder involvement than what is common in conventional assessment methods [105].

4.3. Environmental Dynamics

The importance of accounting for environmental dynamics has been introduced in Section 4.2., exemplified with the change in the decomposition rate of SOC in response to tillage. More generally, environmental dynamics refer to temporally changing aspects that influence the state of the studied system, its interactions with surrounding systems, and the magnitude of its environmental impact. This includes, for example, the timing and rate of release of emissions, timing and rate of resource extraction, temporal delays, seasonal variations, etc. [12]. Bio-based production systems are particularly sensitive to these factors, and they can significantly influence their environmental performance. For instance, seasonal food web dynamics can significantly affect biodiversity impacts from fish harvesting [9], and provisioning of ESs can be more or less affected by a stressor depending on its timing over the year [41,107]. There can be temporal delays between the time of environmental impact and the observed effect on ESs (e.g., due to variations in supply and demand for the service over time [107]), and the productivity and input requirements of a production systems can change with its age (e.g., perennial cropping systems exhibit nonlinear patterns of increasing and then decreasing yields per unit effort over their lifetime [136]).

Environmental dynamics can also indirectly influence the long-term sustainability of bio-based production systems via their effects on key supporting systems. The hydrological system is one example. As described above, freshwater is a critical resource for most terrestrial bio-based systems, and in many areas, groundwater is the dominating source [137]. Groundwater availability is determined by the local hydrology, influenced by several factors, such as percolation rate, soil type, temperature, etc. However, the hydrological system is also characterized by temporal and spatial delays, meaning that it takes time before the full effect of a disturbance at one part of the system is experienced in other parts of the system [138]. For instance, a farmer may want to increase productivity by investing in irrigation. Therefore, the farmer decides to increase groundwater use by drilling a new well and increase extraction. Due to the temporal delays in the hydrological system, an immediate effect on the groundwater level in neighboring wells may not be experienced. Depending on the local hydrology, it may take one, or several, years before the increased water extraction affects neighboring wells further downstream. Once the effect has reached neighboring areas, and the farmer cuts extraction back to its original value, the decline in groundwater level will continue for some time before it slowly starts increasing back to its original level [138]. The dynamics of the hydrological system makes sustainability assessment and management of any bio-based production system relying on the water resource very difficult, as the true effect of altered water use is only seen after a considerable time delay. If, during this delay, the farmer has made capital investments in an irrigation system, and expanded irrigation to finance these investments, he may now find himself in a lock-in, relying on an unsustainable exploitation of water resources in order to maintain productivity and economic profitability.

Environmental dynamics are often overlooked in environmental assessment studies. In LCA, a steady-state approach is typically taken, where emissions and resource consumption occurring throughout the studied lifecycle are aggregated into a single value and assigned to a given point in time. The potential environmental impacts are characterized using predominantly linear impact assessment models and thus, time-dependent changes in environmental processes, in the production system, or in the environment responsiveness to stress, are not considered [139,140]. This can be problematic for the reliability of the assessment, as environmental stressors are often stochastically spread out in time, and the magnitude of their impact fluctuates with the development of the receiving system, and with the accumulation of environmental pressure [10,26,88,104,139]. This approach reduces the reliability of the assessment as a tool for policy development and scenario-based planning. With growing interest in using LCA for predictive modelling, bioeconomy strategic planning, and the assessment of emerging technologies, several authors stress the need to either develop the LCA methodology, or to complement it with other methods in order to better integrate temporal dynamics [10,13,47,112,141,142]. A challenge, though, is that most operational LCA methods do not incorporate the necessary temporal information, or the required case-specific data are not available, to account for environmental dynamics in the impact assessment [41,140]. On the one hand, deepening the assessment to include this information could result in a substantial level of detail and an increase of depth to the assessment. On the other hand, Almeida et al. [143] stresses that it is not always the case that the extra effort required to increase the temporal resolution of the LCA matches the potential gain in results. Whether or not this is the case depends on the context and objectives of the study. It is likely that short-lived processes are less affected by temporal dynamics than long-lived processes, and it is also likely that sensitivity to temporal dynamics varies between different systems [104,143].

In conclusion, adding temporal dynamics to the assessment of bio-based production systems could, in many cases, allow for a more accurate impact assessment. These systems are typically strongly affected by seasonal variations, and the production system and its supporting systems are constantly evolving, causing changes in their environmental performance and response to environmental stressors. Unless these dynamics are taken into consideration in the assessment process, the results, and the following policy actions, are likely to be based on a static system perspective when, in fact, the system is highly dynamic. However, with the added effort this type of assessment might entail, it is up to the practitioner executing the assessment to evaluate how significant environmental dynamics are to the objectives of the study, and for which aspects of the impact model it is worthwhile to invest in temporal differentiation [104,144].

5. The Way Forward

5.1. Expanding System Boundaries

Ensuring the sustainability of bio-based production systems requires assessments methods that are tailored to the specific characteristics of these systems. To ensure long-term sustainability, the rate of resource extraction must not exceed the rate of regeneration, and the regeneration capacity of the system must not be diminished [24]. In an environmental sustainability assessment, this entails expanding the system boundaries beyond conventional practice so that both the primary system and its supporting systems are covered, and the cross-system interactions involved in providing the necessary conditions for regeneration are accounted for [23]. “What are the conditions necessary for maintaining the regeneration capacity of the systems?” and, “what are the supporting systems necessary for maintaining these conditions?" are two helpful questions for practitioners to consider when deciding upon the system boundaries at an early stage of an assessment study.

5.2. Rethinking Impact Categories

Broader system boundaries call for a subsequent expansion of the environmental impact categories covered, so that impacts on both the primary system and its supporting systems are considered. More specifically, efforts must be targeted towards improving the coverage, and impact models, of categories affecting the regeneration capacity of bio-based systems. In this study, we have focused on a select few, historically underrepresented, impact categories, that are generally agreed to be of great importance for the long-term sustainability of many bio-based production systems (biotic resource depletion, water use and biodiversity and ESs).

These impact categories are challenging to assess because they are characterized by being multidimensional and complex concepts, often with multiple factors affecting the primary production system and surrounding supporting systems. In most assessments, however, these impact categories are represented in a highly aggregated format. In order to truly model these impact categories, they need to be disaggregated into their different dimensions, accompanied by suitable indicators for each dimension. For example, biodiversity needs to be differentiated into dimensions including species diversity, genetic diversity and functional diversity, each accompanied by representative indicators to measure their impacts on both the primary production system and its supporting systems.

5.3. Contextualizing the Impact Models

Once sufficiently broad system boundaries have been set (including both the primary system and its supporting systems), key impact categories have been identified, and their different dimensions and indicators established, the impact pathways connecting the impact categories with the bio-based production systems need to be understood and modelled. Entangling these impact pathways requires targeted efforts towards moving away from generalized assessments, and instead moving towards more context specific impact models. By increasing the level of spatial- and temporal differentiation in the assessment, including more details on geographical variations, environmental uniqueness, and environmental dynamics of the system, a more representative assessment of its long-term sustainability can be achieved. This entails several methodological challenges, some of which were discussed in Sections 4.1–4.3. To overcome these challenges, we advocate increased collaboration with other research fields, such as ecological science, system theory, risk modelling, scenario analysis, etc., increased collaboration with local stakeholders and actors with local ecological knowledge, and the use of approaches where multiple modelling methods are applied in combination (e.g., LCA is combined with GIS, ecological modelling methods, and System Dynamics modelling).

Before initializing this type of cross-discipline and multi-modelling approach, one should be aware that the efforts and resources required for such an assessment can be significant. If reliable location-specific data and impact models are not available, these need to be collected and developed as part of the assessment process. With the inherent complexity of many bio-based systems, and the often incomplete understanding of the processes and feedback loops connecting them to their supporting systems, making assumptions regarding the structure and dynamics of the impact pathways may be required in the modelling process. Some would argue that expanding the assessment as advocated above will only lead to added uncertainty and costs, while being of little help in guiding the transition to a sustainable bioeconomy. We argue differently. As presented in this review, taking a systems approach in sustainability assessments, tailoring it to the context of the study by including a significant degree of case-specific information and impact models, is a necessity to ensure the criteria for the sustainability of these systems are met. It is true that the models developed for such an assessment will never constitute perfect representations of reality, and a certain degree of uncertainty is unavoidable. However, the complex nature of bio-based systems cannot be ignored. Even an assessment based on imperfect models, as long as it is built on best-available knowledge and transparent assumptions, is likely to be better guidance for sustainable development than an approach where the complexity of the issue is marginalized, and significant drivers of environmental degradation are intentionally left out due to limitations in modelling capacity.

Funding: This research received no external funding.

Acknowledgments: We would like to acknowledge Liisa Fransson from RISE Research Institutes of Sweden for providing valuable input and supervision throughout the review and writing process.

Conflicts of Interest: The author declares no conflict of interest.

References

1. EC. *Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committe and the Committee of the Regions: A Bioeconomy for Europe*; European Commission: Brussels, Belgium, 2012.
2. EC. *A Sustainable Bioeconomy for Europe: Strengthening the Connection between Economy, Society and the Environment. Updated Bioeconomy Strategy*; Publications Office of the European Union: Luxembourg, 2018; p. 107.
3. Ronzon, T.; Lusser, M.; Klinkenberg, M.; Landa, L.; Sanchez Lopez, J.; M'Barek, R.; Hadjamu, G.; Belward, A.; Camia, J.G.; Cristobal, J.; et al. *Bioeconomy Report 2016*; Joint Research Centre: Brussels, Belgium, 2017.
4. Priefer, C.; Jorissen, J.; Fror, O. Pathways to Shape the Bioeconomy. *Resources* **2017**, *6*, 23. [CrossRef]
5. Dietz, T.; Borner, J.; Forster, J.J.; von Braun, J. Governance of the Bioeconomy: A Global Comparative Study of National Bioeconomy Strategies. *Sustainability* **2018**, *10*, 20. [CrossRef]
6. Bugge, M.M.; Hansen, T.; Klitkou, A. What Is the Bioeconomy? A Review of the Literature. *Sustainability* **2016**, *8*, 22. [CrossRef]
7. Meyer, R. Bioeconomy Strategies: Contexts, Visions, Guiding Implementation Principles and Resulting Debates. *Sustainability* **2017**, *9*, 1031. [CrossRef]
8. Ronzon. *Bioeconomy Report 2016*; Publications Office of the European Union: Luxembourg, 2017.
9. Pettersen, J.B.; Song, X.Q. Life Cycle Impact Assessment in the Arctic: Challenges and Research Needs. *Sustainability* **2017**, *9*, 20. [CrossRef]
10. McManus, M.C.; Taylor, C.M. The changing nature of life cycle assessment. *Biomass Bioenerg.* **2015**, *82*, 13–26. [CrossRef]
11. Garnett, T. Three perspectives on sustainable food security: Efficiency, demand restraint, food system transformation. What role for life cycle assessment? *J. Clean Prod.* **2014**, *73*, 10–18. [CrossRef]
12. Reap, J.; Roman, F.; Duncan, S.; Bras, B. A survey of unresolved problems in life cycle assessment. *Int. J. Life Cycle Assess.* **2008**, *13*, 374–388. [CrossRef]
13. Sala, S.; Anton, A.; McLaren, S.J.; Notarnicola, B.; Saouter, E.; Sonesson, U. In quest of reducing the environmental impacts of food production and consumption. *J. Clean Prod.* **2017**, *140*, 387–398. [CrossRef]
14. Bjorn, A.; Hauschild, M.Z. Absolute versus Relative Environmental Sustainability What can the Cradle-to-Cradle and Eco-efficiency Concepts Learn from Each Other? *J. Ind. Ecol.* **2013**, *17*, 321–332. [CrossRef]
15. Finnveden, G.; Hauschild, M.Z.; Ekvall, T.; Guinee, J.; Heijungs, R.; Hellweg, S.; Koehler, A.; Pennington, D.; Suh, S. Recent developments in Life Cycle Assessment. *J. Environ. Manag.* **2009**, *91*, 1–21. [CrossRef] [PubMed]
16. Reap, J.; Bras, B.; Newcomb, P.J.; Carmichael, C. Improving life cycle assessment by including spatial, dynamic and place-based modeling. In Proceedings of the ASME 2003 Design Engineering Technical Conferences and Computer and Information in Engineering Conference, Chicago, IL, USA, 2–6 September 2003.
17. Reap, J.; Roman, F.; Duncan, S.; Bras, B. A survey of unresolved problems in life cycle assessment-Part 1: Goal and scope and inventory analysis. *Int. J. Life Cycle Assess.* **2008**, *13*, 290–300. [CrossRef]
18. Cristobal, J.; Matos, C.T.; Aurambout, J.P.; Manfredi, S.; Kavalov, B. Environmental sustainability assessment of bioeconomy value chains. *Biomass Bioenerg.* **2016**, *89*, 159–171. [CrossRef]

19. Pursula, T.; Aho, M.; Rönnlund, I.; Päällysaho, M. Environmental Sustainability Indicators for the Bioeconomy. In *Towards a Sustainable Bioeconomy: Principles, Challenges and Perspectives*; Leal Filho, W., Pociovălișteanu, D.M., Borges de Brito, P.R., Borges de Lima, I., Eds.; Springer International Publishing: Cham, Switzerland, 2018; pp. 43–61.
20. Rosegrant, M.W.; Ringler, C.; Zhu, T.J.; Tokgoz, S.; Bhandary, P. Water and food in the bioeconomy: Challenges and opportunities for development. *Agric. Econ.* **2013**, *44*, 139–150. [CrossRef]
21. Bennich, T.; Belyazid, S. The Route to Sustainability-Prospects and Challenges of the Bio-Based Economy. *Sustainability* **2017**, *9*, 887. [CrossRef]
22. Szekacs, A. Environmental and Ecological Aspects in the Overall Assessment of Bioeconomy. *J. Agric. Environ. Ethics* **2017**, *30*, 153–170. [CrossRef]
23. Ge, L.; Anten, N.P.R.; van Dixhoorn, I.D.E.; Feindt, P.H.; Kramer, K.; Leemans, R.; Meuwissen, M.P.M.; Spoolder, H.; Sukkel, W. Why we need resilience thinking to meet societal challenges in bio-based production systems. *Curr. Opin. Environ. Sustain.* **2016**, *23*, 17–27. [CrossRef]
24. Goodland, R.; Daly, H. Environmental sustainability: Universal and non-negotiable. *Ecol. Appl.* **1996**, *6*, 1002–1017. [CrossRef]
25. Sillanpää, M.; Ncibi, M.C. *Bioeconomy: The Path to Sustainability*; Springer: Cham, Switzerland, 2017.
26. Notarnicola, B.; Sala, S.; Anton, A.; McLaren, S.J.; Saouter, E.; Sonesson, U. The role of life cycle assessment in supporting sustainable agri-food systems: A review of the challenges. *J. Clean Prod.* **2017**, *140*, 399–409. [CrossRef]
27. Ostrom, E. A General Framework for Analyzing Sustainability of Social-Ecological Systems. *Science* **2009**, *325*, 419–422. [CrossRef]
28. Evans, M.R. Modelling ecological systems in a changing world. *Philos. Trans. Royal Soc. Lond. Ser. B Biol. Sci.* **2012**, *367*, 181–190. [CrossRef] [PubMed]
29. Karnosky, D.F. Impacts of elevated atmospheric CO2 on forest trees and forest ecosystems: Knowledge gaps. *Environ. Int.* **2003**, *29*, 161–169. [CrossRef]
30. Pfister, S. Water Use. In *Life Cycle Impact Assessment*; Hauschild, M.Z., Huijbregts, M.A.J., Eds.; Springer: Berlin, Germany, 2015; pp. 223–245.
31. Ugedal, O.; Næsje, T.F.; Thorstad, E.B.; Forseth, T.; Saksgård, L.M.; Heggberget, T.G. Twenty years of hydropower regulation in the River Alta: Long-term changes in abundance of juvenile and adult Atlantic salmon. *Hydrobiologia* **2008**, *609*, 9–23. [CrossRef]
32. Norgate, T.E.; Jahanshahi, S.; Rankin, W.J. Assessing the environmental impact of metal production processes. *J. Clean Prod.* **2007**, *15*, 838–848. [CrossRef]
33. Sonter, L.J.; Moran, C.J.; Barrett, D.J.; Soares-Filho, B.S. Processes of land use change in mining regions. *J. Clean Prod.* **2014**, *84*, 494–501. [CrossRef]
34. Paull, D.; Banks, G.; Ballard, C.; Gillieson, D. Monitoring the Environmental Impact of Mining in Remote Locations through Remotely Sensed Data. *Geocarto Int.* **2006**, *21*, 33–42. [CrossRef]
35. Vidal Legaz, B.; Maia De Souza, D.; Teixeira, R.F.M.; Antón, A.; Putman, B.; Sala, S. Soil quality, properties, and functions in life cycle assessment: An evaluation of models. *J. Clean Prod.* **2017**, *140*, 502–515. [CrossRef]
36. Lazarevic, D.; Martin, M. Life cycle assessments, carbon footprints and carbon visions: Analysing environmental systems analyses of transportation biofuels in Sweden. *J. Clean Prod.* **2016**, *137*, 249–257. [CrossRef]
37. Klöpffer, W. *Background and Future Prospects in Life Cycle Assessment*; Springer Netherlands: Dordrecht, The Netherlands, 2014.
38. Bjørn, A.; Owsianiak, M.; Molin, C.; Laurent, A. Main Characteristics of LCA. In *Life Cycle Assessment: Theory and Practice*; Hauschild, M.Z., Rosenbaum, R.K., Olsen, S.I., Eds.; Springer International Publishing: Cham, Switzerland, 2018; pp. 9–16.
39. Ford, J.S.; Pelletier, N.L.; Ziegler, F.; Scholz, A.J.; Tyedmers, P.H.; Sonesson, U.; Kruse, S.A.; Silverman, H. Proposed Local Ecological Impact Categories and Indicators for Life Cycle Assessment of Aquaculture A Salmon Aquaculture Case Study. *J. Ind. Ecol.* **2012**, *16*, 254–265. [CrossRef]
40. Sala, S.; Farioli, F.; Zamagni, A. Life cycle sustainability assessment in the context of sustainability science progress (part 2). *Int. J. Life Cycle Assess.* **2013**, *18*, 1686–1697. [CrossRef]

41. Othoniel, B.; Rugani, B.; Heijungs, R.; Benetto, E.; Withagen, C. Assessment of Life Cycle Impacts on Ecosystem Services: Promise, Problems, and Prospects. *Environ. Sci. Technol.* **2016**, *50*, 1077–1092. [CrossRef] [PubMed]
42. McClellande, S.C.; Arndt, C.; Gordon, D.R.; Thoma, G. Type and number of environmental impact categories used in livestock life cycle assessment: A systematic review. *Livest. Sci.* **2018**, *209*, 39–45. [CrossRef]
43. Crenna, E.; Sozzo, S.; Sala, S. Natural biotic resources in LCA: Towards an impact assessment model for sustainable supply chain management. *J. Clean Prod.* **2018**, *172*, 3669–3684. [CrossRef] [PubMed]
44. Tufvesson, L.M.; Tufvesson, P.; Woodley, J.M.; Borjesson, P. Life cycle assessment in green chemistry: Overview of key parameters and methodological concerns. *Int. J. Life Cycle Assess.* **2013**, *18*, 431–444. [CrossRef]
45. Winter, L.; Lehmann, A.; Finogenova, N.; Finkbeiner, M. Including biodiversity in life cycle assessment—State of the art, gaps and research needs. *Environ. Impact Assess. Rev.* **2017**, *67*, 88–100. [CrossRef]
46. Hellweg, S.; Canals, L.M.I. Emerging approaches, challenges and opportunities in life cycle assessment. *Science* **2014**, *344*, 1109–1113. [CrossRef] [PubMed]
47. Teillard, F.; de Souza, D.M.; Thoma, G.; Gerber, P.J.; Finn, J.A. What does Life-Cycle Assessment of agricultural products need for more meaningful inclusion of biodiversity? *J. Appl. Ecol.* **2016**, *53*, 1422–1429. [CrossRef]
48. Lorilla, R.S.; Poirazidis, K.; Kalogirou, S.; Detsis, V.; Martinis, A. Assessment of the Spatial Dynamics and Interactions among Multiple Ecosystem Services to Promote Effective Policy Making across Mediterranean Island Landscapes. *Sustainability* **2018**, *10*, 28. [CrossRef]
49. Alroe, H.F.; Moller, H.; Laessoe, J.; Noe, E. Opportunities and challenges for multicriteria assessment of food system sustainability. *Ecol. Soc.* **2016**, *21*, 8. [CrossRef]
50. Berger, M.; Pfister, S.; Bach, V.; Finkbeiner, M. Saving the Planet's Climate or Water Resources? The Trade-Off between Carbon and Water Footprints of European Biofuels. *Sustainability* **2015**, *7*, 6665–6683. [CrossRef]
51. Laurin, L.; Amor, B.; Bachmann, T.M.; Bare, J.; Koffler, C.; Genest, S.; Preiss, P.; Pierce, J.; Satterfield, B.; Vigon, B. Life cycle assessment capacity roadmap (section 1): Decision-making support using LCA. *Int. J. Life Cycle Assess.* **2016**, *21*, 443–447. [CrossRef]
52. Pelletier, N.L.; Ayer, N.W.; Tyedmers, P.H.; Kruse, S.A.; Flysjo, A.; Robillard, G.; Ziegler, F.; Scholz, A.J.; Sonesson, U. Impact categories for life cycle assessment research of seafood production systems: Review and prospectus. *Int. J. Life Cycle Assess.* **2007**, *12*, 414–421. [CrossRef]
53. Bare, J.C. Life cycle impact assessment research developments and needs. *Clean Technol. Environ. Policy* **2010**, *12*, 341–351. [CrossRef]
54. Schneider, L.; Bach, V.; Finkbeiner, M. LCA Perspectives for Resource Efficiency Assessment. In *Special Types of Life Cycle Assessment*; Finkbeiner, M., Ed.; Springer: Dordrecht, The Netherlands, 2016; pp. 179–218.
55. Klinglmair, M.; Sala, S.; Brandão, M. Assessing resource depletion in LCA: A review of methods and methodological issues. *Int. J. Life Cycle Assess.* **2014**, *19*, 580–592. [CrossRef]
56. Sonnemann, G.; Gemechu, E.D.; Adibi, N.; De Bruille, V.; Bulle, C. From a critical review to a conceptual framework for integrating the criticality of resources into Life Cycle Sustainability Assessment. *J. Clean Prod.* **2015**, *94*, 20–34. [CrossRef]
57. Dewulf, J.; Benini, L.; Mancini, L.; Sala, S.; Blengini, G.A.; Ardente, F.; Recchioni, M.; Maes, J.; Pant, R.; Pennington, D. Rethinking the Area of Protection "Natural Resources" in Life Cycle Assessment. *Environ. Sci. Technol.* **2015**, *49*, 5310–5317. [CrossRef] [PubMed]
58. Ecoinvent. Ecoinvent Database. Available online: https://www.ecoinvent.org/database/ecoinvent-33/new-data-in-ecoinvent-33/new-data-in-ecoinvent-33.html (accessed on 6 May 2018).
59. European Commission, J.E. ELCD, Version 3.2. Available online: https://nexus.openlca.org/database/ELCD (accessed on 6 May 2018).
60. Sala, S.; Benini, L.; Castellani, V.; Vidal Legaz, B. *Environmental Footprint—Update of Life Cycle Impact Assessment Methods*; DRAFT for TAB; EC-JRC: Brussels, Belgium, 2016.
61. Langlois, J.; Freon, P.; Delgenes, J.-P.; Steyer, J.-P.; Helias, A. New methods for impact assessment of biotic-resource depletion in life cycle assessment of fisheries: Theory and application. *J. Clean Prod.* **2014**, *73*, 63–71. [CrossRef]
62. Johnson, D.W.; Freiwald, J.; Bernardi, G. Genetic diversity affects the strength of population regulation in a marine fish. *Ecology* **2016**, *97*, 627–639. [CrossRef]

63. Schindler, D.E.; Hilborn, R.; Chasco, B.; Boatright, C.P.; Quinn, T.P.; Rogers, L.A.; Webster, M.S. Population diversity and the portfolio effect in an exploited species. *Nature* **2010**, *465*, 609. [CrossRef]
64. De Groote, S.R.E.; Vanhellemont, M.; Baeten, L.; Caron, M.M.; Martel, A.; Bonte, D.; Lens, L.; Verheyen, K. Effects of Mineral Soil and Forest Floor on the Regeneration of Pedunculate Oak, Beech and Red Oak. *Forests* **2018**, *9*, 21. [CrossRef]
65. Ross, C.W.; Watt, M.S.; Parfitt, R.L.; Simcock, R.; Dando, J.; Coker, G.; Clinton, P.W.; Davis, M.R. Soil quality relationships with tree growth in exotic forests in New Zealand. *For. Ecol. Manag.* **2009**, *258*, 2326–2334. [CrossRef]
66. Kuijper, D.P.J.; Cromsigt, J.P.G.M.; Jędrzejewska, B.; Miścicki, S.; Churski, M.; Jędrzejewski, W.; Kweczlich, I. Bottom-up versus top-down control of tree regeneration in the Białowieża Primeval Forest, Poland. *J. Ecol.* **2010**, *98*, 888–899. [CrossRef]
67. Palansooriya, K.N.; Ok, Y.S.; Awad, Y.M.; Lee, S.S.; Sung, J.K.; Koutsospyros, A.; Moon, D.H. Impacts of biochar application on upland agriculture: A review. *J. Environ. Manag.* **2019**, *234*, 52–64. [CrossRef] [PubMed]
68. Obade, V.D. Integrating management information with soil quality dynamics to monitor agricultural productivity. *Sci. Total Environ.* **2019**, *651*, 2036–2043. [CrossRef] [PubMed]
69. ISO. *ISO 14044:2006 Environmental Management—Life Cycleassessment—Requirements and Guidelines*; International Organization for Standardization: Geneva, Switzerland, 2006.
70. Curran, M.; de Baan, L.; De Schryver, A.M.; van Zelm, R.; Hellweg, S.; Koellner, T.; Sonnemann, G.; Huijbregts, M.A.J. Toward Meaningful End Points of Biodiversity in Life Cycle Assessment. *Environ. Sci. Technol.* **2011**, *45*, 70–79. [CrossRef] [PubMed]
71. Bach, V.; Berger, M.; Finogenova, N.; Finkbeiner, M. Assessing the Availability of Terrestrial Biotic Materials in Product Systems (BIRD). *Sustainability* **2017**, *9*, 35. [CrossRef]
72. Berger, M.; Pfister, S.; Motoshita, M. Water Footprinting in Life Cycle Assessment: How to Count the Drops and Assess the Impacts? In *Special Types of Life Cycle Assessment*; Finkbeiner, M., Ed.; Springer: Dordrecht, The Netherlands, 2016; pp. 73–114.
73. Hoekstra, A.Y. A critique on the water-scarcity weighted water footprint in LCA. *Ecol. Indic.* **2016**, *66*, 564–573. [CrossRef]
74. Pfister, S.; Koehler, A.; Hellweg, S. Assessing the Environmental Impacts of Freshwater Consumption in LCA. *Environ. Sci. Technol.* **2009**, *43*, 4098–4104. [CrossRef]
75. Gordon, L.J.; Finlayson, C.M.; Falkenmark, M. Managing water in agriculture for food production and other ecosystem services. *Agric. Water Manag.* **2010**, *97*, 512–519. [CrossRef]
76. Tsalkatis, P.; Petanidou, T.; Tscheulin, T.; Takkis, K. Climate change reduces nectar secretion in two common Mediterranean plants. *AoB PLANTS* **2015**, *7*. [CrossRef]
77. Gallagher, M.K.; Campbell, D.R. Shifts in water availability mediate plant–pollinator interactions. *New Phytol.* **2017**, *215*, 792–802. [CrossRef] [PubMed]
78. Canals, L.M.I.; Chenoweth, J.; Chapagain, A.; Orr, S.; Anton, A.; Clift, R. Assessing freshwater use impacts in LCA: Part I-inventory modelling and characterisation factors for the main impact pathways. *Int. J. Life Cycle Assess.* **2009**, *14*, 28–42. [CrossRef]
79. Smakhtin, V.; Revenga, C.; Doll, P. A pilot global assessment of environmental water requirements and scarcity. *Water Int.* **2004**, *29*, 307–317. [CrossRef]
80. Ridoutt, B.G.; Huang, J. Environmental relevance-the key to understanding water footprints. *Proc. Natl. Acad. Sci. USA* **2012**, *109*, E1424. [CrossRef] [PubMed]
81. Lake, P.S. Ecological effects of perturbation by drought in flowing waters. *Freshwater Biol.* **2003**, *48*, 1161–1172. [CrossRef]
82. Kounina, A.; Margni, M.; Bayart, J.B.; Boulay, A.M.; Berger, M.; Bulle, C.; Frischknecht, R.; Koehler, A.; Canals, L.M.I.; Motoshita, M.; et al. Review of methods addressing freshwater use in life cycle inventory and impact assessment. *Int. J. Life Cycle Assess.* **2013**, *18*, 707–721. [CrossRef]
83. Humbert, S.; Maendly, R. Characterisation factors for damage to aquatic biodiversity causedby water use especially from dams used for hydropower. In Proceedings of the 35th LCA Discussion Forum, Zurich, Switzerland, 5 June 2008.
84. Gracey, E.O.; Verones, F. Impacts from hydropower production on biodiversity in an LCA framework-review and recommendations. *Int. J. Life Cycle Assess.* **2016**, *21*, 412–428. [CrossRef]

85. Ercin, A.E.; Hoekstra, A.Y. Water footprint scenarios for 2050: A global analysis. *Environ. Int.* **2014**, *64*, 71–82. [CrossRef]
86. Veldkamp, T.I.E.; Wada, Y.; Aerts, J.; Doll, P.; Gosling, S.N.; Liu, J.; Masaki, Y.; Oki, T.; Ostberg, S.; Pokhrel, Y.; et al. Water scarcity hotspots travel downstream due to human interventions in the 20th and 21st century. *Nat. Commun.* **2017**, *8*, 12. [CrossRef]
87. Winz, I.; Brierley, G.; Trowsdale, S. The Use of System Dynamics Simulation in Water Resources Management. *Water Resour. Manag.* **2009**, *23*, 1301–1323. [CrossRef]
88. Onat, N.C.; Kucukvar, M.; Tatari, O.; Egilmez, G. Integration of system dynamics approach toward deepening and broadening the life cycle sustainability assessment framework: A case for electric vehicles. *Int. J. Life Cycle Assess.* **2016**, *21*, 1009–1034. [CrossRef]
89. CBD. Aichi Biodiversity Targets. Available online: https://www.cbd.int/sp/targets/ (accessed on 16 May 2019).
90. Swedish Environmental Protection Agency. *Sweden's Environmental Objectives*; Swedish Environmental Protection Agency: Stockholm, Sweden, 2012; p. 26.
91. Souza, D.M.; Teixeira, R.F.M.; Ostermann, O.P. Assessing biodiversity loss due to land use with Life Cycle Assessment: Are we there yet? *Glob. Chang. Biol.* **2015**, *21*, 32–47. [CrossRef] [PubMed]
92. Gabel, V.M.; Meier, M.S.; Kopke, U.; Stolze, M. The challenges of including impacts on biodiversity in agricultural life cycle assessments. *J. Environ. Manag.* **2016**, *181*, 249–260. [CrossRef] [PubMed]
93. Pearson, R.G. Reasons to Conserve Nature. *Trends Ecol. Evol.* **2016**, *31*, 366–371. [CrossRef] [PubMed]
94. Teytelboym, A. Natural capital market design. *Oxf. Rev. Econ. Policy* **2019**, *35*, 138–161. [CrossRef]
95. Cheng, A. Shaping a sustainable food future by rediscovering long-forgotten ancient grains. *Plant Sci.* **2018**, *269*, 136–142. [CrossRef] [PubMed]
96. Pitcher, T.J.; Cheung, W.W.L. Fisheries: Hope or despair? *Mar. Pollut. Bull.* **2013**, *74*, 506–516. [CrossRef] [PubMed]
97. Garibaldi, L.A.; Carvalheiro, L.G.; Vaissière, B.E.; Gemmill-Herren, B.; Hipólito, J.; Freitas, B.M.; Ngo, H.T.; Azzu, N.; Sáez, A.; Åström, J.; et al. Mutually beneficial pollinator diversity and crop yield outcomes in small and large farms. *Science* **2016**, *351*, 388. [CrossRef] [PubMed]
98. Tibesigwa, B.; Siikamaki, J.; Lokina, R.; Alvsilver, J. Naturally available wild pollination services have economic value for nature dependent smallholder crop farms in Tanzania. *Sci. Rep.* **2019**, *9*, 10. [CrossRef]
99. Ammer, C. Diversity and forest productivity in a changing climate. *New Phytol.* **2019**, *221*, 50–66. [CrossRef]
100. Bohn Friedrich, J.; Huth, A. The importance of forest structure to biodiversity–productivity relationships. *R. Soc. Open Sci.* **2017**, *4*, 160521. [CrossRef] [PubMed]
101. Mimura, M.; Yahara, T.; Faith, D.P.; Vazquez-Dominguez, E.; Colautti, R.I.; Araki, H.; Javadi, F.; Nunez-Farfan, J.; Mori, A.S.; Zhou, S.L.; et al. Understanding and monitoring the consequences of human impacts on intraspecific variation. *Evol. Appl.* **2017**, *10*, 121–139. [CrossRef] [PubMed]
102. El Mujtar, V.; Munoz, N.; Mc Cormick, B.P.; Pulleman, M.; Tittonell, P. Role and management of soil biodiversity for food security and nutrition; where do we stand? *Glob. Food Secur. Agric. Policy* **2019**, *20*, 132–144. [CrossRef]
103. Vanbergen, A.J.; Espindola, A.; Aizen, M.A. Risks to pollinators and pollination from invasive alien species. *Nat. Ecol. Evol.* **2018**, *2*, 16–25. [CrossRef] [PubMed]
104. Pinsonnault, A.; Lesage, P.; Levasseur, A.; Samson, R. Temporal differentiation of background systems in LCA: Relevance of adding temporal information in LCI databases. *Int. J. Life Cycle Assess.* **2014**, *19*, 1843–1853. [CrossRef]
105. Chaplin-Kramer, R.; Sim, S.; Hamel, P.; Bryant, B.; Noe, R.; Mueller, C.; Rigarlsford, G.; Kulak, M.; Kowal, V.; Sharp, R.; et al. Life cycle assessment needs predictive spatial modelling for biodiversity and ecosystem services. *Nat. Commun.* **2017**, *8*, 15065. [CrossRef] [PubMed]
106. EC. *General guide for Life Cycle Assessment—Detailed Guidance*, 1st ed.; Publications Office of the European Union: Luxembourg, 2010.
107. Koch, E.W.; Barbier, E.B.; Silliman, B.R.; Reed, D.J.; Perillo, G.M.; Hacker, S.D.; Granek, E.F.; Primavera, J.H.; Muthiga, N.; Polasky, S.; et al. Non-linearity in ecosystem services: Temporal and spatial variability in coastal protection. *Front. Ecol. Environ.* **2009**, *7*, 29–37. [CrossRef]
108. Robinson, D.A.; Seaton, F.; Sharps, K.; Thomas, A.; Roberts, F.P.; van der Ploeg, M.; Jones, L.; Stolte, J.; Puig de la Bellacasa, M.; Harrison, P.; et al. Soil Resources, the Delivery of Ecosystem Services and Value. *Oxf. Res. Encycl. Environ. Sci.* **2017**, 1–43. [CrossRef]

109. Crossman, N.D.; Burkhard, B.; Nedkov, S.; Willemen, L.; Petz, K.; Palomo, I.; Drakou, E.G.; Martín-Lopez, B.; McPhearson, T.; Boyanova, K.; et al. A blueprint for mapping and modelling ecosystem services. *Ecosyst. Serv.* **2013**, *4*, 4–14. [CrossRef]
110. de Groot, R.S.; Alkemade, R.; Braat, L.; Hein, L.; Willemen, L. Challenges in integrating the concept of ecosystem services and values in landscape planning, management and decision making. *Ecol. Complex.* **2010**, *7*, 260–272. [CrossRef]
111. Stürck, J.; Schulp, C.J.E.; Verburg, P.H. Spatio-temporal dynamics of regulating ecosystem services in Europe—The role of past and future land use change. *Appl. Geogr.* **2015**, *63*, 121–135. [CrossRef]
112. Onat, N.C.; Kucukvar, M.; Halog, A.; Cloutier, S. Systems Thinking for Life Cycle Sustainability Assessment: A Review of Recent Developments, Applications, and Future Perspectives. *Sustainability* **2017**, *9*, 25. [CrossRef]
113. Moore, J.C. Predicting tipping points in complex environmental systems. *Proc. Natl. Acad. Sci. USA* **2018**, *115*, 635. [CrossRef]
114. Yang, Y. Toward a more accurate regionalized life cycle inventory. *J. Clean Prod.* **2016**, *112*, 308–315. [CrossRef]
115. Yang, Y.; Campbell, J.E. Improving attributional life cycle assessment for decision support: The case of local food in sustainable design. *J. Clean Prod.* **2017**, *145*, 361–366. [CrossRef]
116. McCormick, K.; Kautto, N. The Bioeconomy in Europe: An Overview. *Sustainability* **2013**, *5*, 2589–2608. [CrossRef]
117. Hong, B.H.; How, B.S.; Lam, H.L. Overview of sustainable biomass supply chain: From concept to modelling. *Clean Technol. Environ. Policy* **2016**, *18*, 2173–2194. [CrossRef]
118. Jeswani, H.K.; Azapagic, A.; Schepelmann, P.; Ritthoff, M. Options for broadening and deepening the LCA approaches. *J. Clean Prod.* **2010**, *18*, 120–127. [CrossRef]
119. GreenDelta. OpenLCA. Available online: http://www.openlca.org/ (accessed on 7 August 2018).
120. Rodríguez, C.; Greve, S. *Regionalized LCIA in Open LCA*; GreenDelta GmbH: Berlin, Germany, 2016.
121. Brandão, M.; Canals, L.i.M. Global characterisation factors to assess land use impacts on biotic production. *Int. J. Life Cycle Assess.* **2013**, *18*, 1243–1252. [CrossRef]
122. Ludwig, M.; Wilmes, P.; Schrader, S. Measuring soil sustainability via soil resilience. *Sci. Total Environ.* **2018**, *626*, 1484–1493. [CrossRef]
123. Oldfield, E.E.; Bradford, M.A.; Wood, S.A. Global meta-analysis of the relationship between soil organic matter and crop yields. *Soil* **2019**, *5*, 15–32. [CrossRef]
124. Haddaway, N.R.; Hedlund, K.; Jackson, L.E.; Kätterer, T.; Lugato, E.; Thomsen, I.K.; Jørgensen, H.B.; Isberg, P.-E. How does tillage intensity affect soil organic carbon? A systematic review. *Environ. Evid.* **2017**, *6*, 30. [CrossRef]
125. Leenhardt, P.; Teneva, L.; Kininmonth, S.; Darling, E.; Cooley, S.; Claudet, J. Challenges, insights and perspectives associated with using social-ecological science for marine conservation. *Ocean Coast. Manag.* **2015**, *115*, 49–60. [CrossRef]
126. Loiseau, E.; Aissani, L.; Le Feon, S.; Laurent, F.; Cerceau, J.; Sala, S.; Roux, P. Territorial Life Cycle Assessment (LCA): What exactly is it about? A proposal towards using a common terminology and a research agenda. *J. Clean Prod.* **2018**, *176*, 474–485. [CrossRef]
127. Loiseau, E.; Roux, P.; Junqua, G.; Maurel, P.; Bellon-Maurel, V. Adapting the LCA framework to environmental assessment in land planning. *Int. J. Life Cycle Assess.* **2013**, *18*, 1533–1548. [CrossRef]
128. Sandin, G.; Peters, G.; Svanström, M. *Life Cycle Assessment of Forest Products*; Springer: Berlin/Heidelberg, Germany, 2016.
129. Zamagni, A.; Guinee, J.; Heijungs, R.; Masoni, P.; Raggi, A. Lights and shadows in consequential LCA. *Int. J. Life Cycle Assess.* **2012**, *17*, 904–918. [CrossRef]
130. Berkes, F.; Colding, J.; Folke, C.e. *Navigating Social-Ecological Systems: Building Resilience for Complexity and Change*; Berkes, F., Colding, J., Folke, C., Eds.; Cambridge University Press: Cambridge, UK, 2002.
131. Silvano, R.A.M.; Valbo-Jørgensen, J. Beyond fishermen's tales: Contributions of fishers' local ecological knowledge to fish ecology and fisheries management. *Environ. Dev. Sustain.* **2008**, *10*, 657. [CrossRef]
132. Raicevich, S.; Alegret, J.L.; Frangoudes, K.; Giovanardi, O.; Fortibuoni, T. Community-based management of the Mediterranean coastal fisheries: Historical reminiscence or the root for new fisheries governance? *Reg. Stud. Marine Sci.* **2018**, *21*, 86–93. [CrossRef]

133. Joa, B.; Winkel, G.; Primmer, E. The unknown known—A review of local ecological knowledge in relation to forest biodiversity conservation. *Land Use Pol.* **2018**, *79*, 520–530. [CrossRef]
134. Oliver, D.M.; Fish, R.D.; Winter, M.; Hodgson, C.J.; Heathwaite, A.L.; Chadwick, D.R. Valuing local knowledge as a source of expert data: Farmer engagement and the design of decision support systems. *Environ. Model. Softw.* **2012**, *36*, 76–85. [CrossRef]
135. Laurenti, R.; Lazarevic, D.; Poulikidou, S.; Montrucchio, V.; Bistagnino, L.; Frostell, B. Group Model-Building to identify potential sources of environmental impacts outside the scope of LCA studies. *J. Clean Prod.* **2014**, *72*, 96–109. [CrossRef]
136. Bessou, C.; Basset-Mens, C.; Tran, T.; Benoist, A. LCA applied to perennial cropping systems: A review focused on the farm stage. *Int. J. Life Cycle Assess.* **2013**, *18*, 340–361. [CrossRef]
137. Sears, L.; Caparelli, J.; Lee, C.; Pan, D.; Strandberg, G.; Vuu, L.; Lawell, C. Jevons' Paradox and Efficient Irrigation Technology. *Sustainability* **2018**, *10*, 12. [CrossRef]
138. Bredehoeft, J.D. Monitoring regional groundwater extraction: The problem. *Ground Water* **2011**, *49*, 808–814. [CrossRef]
139. Beloin-Saint-Pierre, D.; Levasseur, A.; Margni, M.; Blanc, I. Implementing a Dynamic Life Cycle Assessment Methodology with a Case Study on Domestic Hot Water Production. *J. Ind. Ecol.* **2017**, *21*, 1128–1138. [CrossRef]
140. Levasseur, A.; Lesage, P.; Margni, M.; Deschenes, L.; Samson, R. Considering Time in LCA: Dynamic LCA and Its Application to Global Warming Impact Assessments. *Environ. Sci. Technol.* **2010**, *44*, 3169–3174. [CrossRef]
141. Reale, F.; Cinelli, M.; Sala, S. Towards a research agenda for the use of LCA in the impact assessment of policies. *Int. J. Life Cycle Assess.* **2017**, *22*, 1477–1481. [CrossRef]
142. Arvidsson, R.; Tillman, A.-M.; Sandén, B.A.; Janssen, M.; Nordelöf, A.; Kushnir, D.; Molander, S. Environmental Assessment of Emerging Technologies: Recommendations for Prospective LCA. *J. Ind. Ecol.* **2018**, *22*, 1286–1294. [CrossRef]
143. Almeida, J.; Degerickx, J.; Achten, W.M.J.; Muys, B. Greenhouse gas emission timing in life cycle assessment and the global warming potential of perennial energy crops. *Carbon Manag.* **2015**, *6*, 185–195. [CrossRef]
144. Guerry, A.D.; Polasky, S.; Lubchenco, J.; Chaplin-Kramer, R.; Daily, G.C.; Griffin, R.; Ruckelshaus, M.; Bateman, I.J.; Duraiappah, A.; Elmqvist, T.; et al. Natural capital and ecosystem services informing decisions: From promise to practice. *Proc. Natl. Acad. Sci. USA* **2015**, *112*, 7348–7355. [CrossRef]

Appendix 1

Supplementary material

Bio-Based Production Systems: Why Environmental Assessment Needs to Include Supporting Systems

Andreas Nicolaidis Lindqvist [1,2,*] , Sarah Broberg [1], Linda Tufvesson [2], Sammar Khalil [2] and Thomas Prade [2]

[1] RISE Research Institutes of Sweden, Ideon Beta5, Scheelevägen 17, 22370 Lund, Sweden
[2] Swedish University of Agricultural Sciences, Department of Biosystems and Technology, P.O. Box 103, SE-230 53 Alnarp, Sweden
[*] Correspondence: andreas.nicolaidis@ri.se

Content

- PRISMA Author Checklist
- PRISMA review flow diagram

PRISMA 2009 Checklist

Section/topic	#	Checklist item	Reported on page #
TITLE			
Title	1	**Identify the report as a systematic review, meta-analysis, or both.** The paper is a qualitative study applying a systematic literature review approach.	1
ABSTRACT			
Structured summary	2	**Provide a structured summary including, as applicable: background; objectives; data sources; study eligibility criteria, participants, and interventions; study appraisal and synthesis methods; results; limitations; conclusions and implications of key findings; systematic review registration number.**	1
INTRODUCTION			
Rationale	3	**Describe the rationale for the review in the context of what is already known.** Transformation towards a bioeconomy is of high political priority, aimed towards delivering on several sustainability objectives. LCA is considered a key tool for sustainability assessment and planning of the bioeconomy. However, researchers have expressed their concerns regarding existing weaknesses and limitations of the method, non the least when applied on complex bio-based systems. Reap et al. (2008) presented seven problem areas in the LCIA and LCA interpretation phases in 2008, all with potentially significant implications to LCA reliability and usefulness as a tool for sustainable development. The relevance of these problem areas are here reassessed, in light of the last decade of LCA development and in light of the current bioeconomy discourse.	1-2
Objectives	4	**Provide an explicit statement of questions being addressed with reference to participants, interventions, comparisons, outcomes, and study design (PICOS).** The objectives of the study are to reassess the methodological limitations and problem areas in the LCIA and LCA interpretation phase presented by Reap et al. (2008), and to investigate what implications these limitations have to the reliability and usefulness of LCA as a tool for sustainability assessment and policy development in the bioeconomy. The objective was also to provide recommendations for future research to address the studied limitations. Research questions: • Are the methodological limitations in LCIA and LCA interpretation presented by Reap et al. [7] applicable to/relevant for sustainability assessment and planning of bioeconomy systems (or bio-based systems)? • How do the limitations affect the reliability of LCA as a tool for sustainability assessment and for policy development in the bioeconomy transition? • What should be the focus of future research to manage these limitation?	3
METHODS			
Protocol and registration	5	**Indicate if a review protocol exists, if and where it can be accessed (e.g., Web address), and, if available, provide registration information including registration number.**	

		No review protocol exist.	
Eligibility criteria	6	**Specify study characteristics (e.g., PICOS, length of follow-up) and report characteristics (e.g., years considered, language, publication status) used as criteria for eligibility, giving rationale.** Studies reporting on LCA methodology, LCA limitations/weaknesses/research needs, bioeconomy and/or bio-based economy and sustainability assessment methods were selected. Publications in English and/or Swedish primarily from the years 2008 – 2018 were considered. Primarily peer reviewed scientific articles and reviews were considered but additional “grey literature” (e.g. reports, NGO publications, books and book sections) were also included when relevant.	3
Information sources	7	**Describe all information sources (e.g., databases with dates of coverage, contact with study authors to identify additional studies) in the search and date last searched.** Scientific journal articles and review papers were obtained from primarily the Web of Science databases. Google Scholar and Google were also utilized for retrieving “grey literature”, such as reports, government documents or NGO publications, and for finding references not covered by the Web of Science databases. When appropriate, key references in articles from the literature searches were also retrieved and included in the literature review and additional sources were achieved through consultation with peer researchers with expertise in LCA and the bioeconomy fields.	3
Search	8	**Present full electronic search strategy for at least one database, including any limits used, such that it could be repeated.** The main search terms included were: *LCA AND Weakness*, LCA AND Limitation*, LCA AND “Research need*”*. The search terms were topic based, no geographical restrictions were applied and only studies published in English or Swedish were considered. Additionally, a reviews-only search was conducted with the terms *sustainability assessment AND methods.*	3
Study selection	9	**State the process for selecting studies (i.e., screening, eligibility, included in systematic review, and, if applicable, included in the meta-analysis).** First titles and then abstracts of the search results were scanned. Articles considered relevant for the study were downloaded and read in full text. Selection criteria used for identifying relevant articles were based on the objectives of the study: (1) a focus on limitations and problem areas in LCA, and especially the LCIA and interpretation phase; (2) a focus on bioeconomy and/or food system sustainability; and (3) a focus on approaches to improve the LCA methodology and potentials for future research. Studies considered relevant for one or more of the stated objectives were included in the review. When appropriate, key references of selected articles were also retrieved and included.	3
Data collection process	10	**Describe method of data extraction from reports (e.g., piloted forms, independently, in duplicate) and any processes for obtaining and confirming data from investigators.** Only qualitative data was considered for the review. Data extraction and analysis was conducted by the review author and results were presented in a draft manuscript format to a team of four peer researchers with subject expertise for feedback and quality assurance. This was an iterative and continuous process throughout the duration of the study.	3
Data items	11	**List and define all variables for which data were sought (e.g., PICOS, funding sources) and any assumptions and simplifications made.** Data collection was based on the study objectives, scope and predefined problem areas as presented by Reap et al.	1-3

		(2008).	
Risk of bias in individual studies	12	**Describe methods used for assessing risk of bias of individual studies (including specification of whether this was done at the study or outcome level), and how this information is to be used in any data synthesis.** Contrasting views in terms of bioeconomy and LCA methodology were indicated in the “Background” and “Problem areas” sections.	4 -18
Summary measures	13	**State the principal summary measures (e.g., risk ratio, difference in means).** No statistical analysis was conducted. Results were analysed qualitatively.	
Synthesis of results	14	**Describe the methods of handling data and combining results of studies, if done, including measures of consistency (e.g., I^2) for each meta-analysis.** No statistical analysis was conducted. Results were analysed qualitatively.	

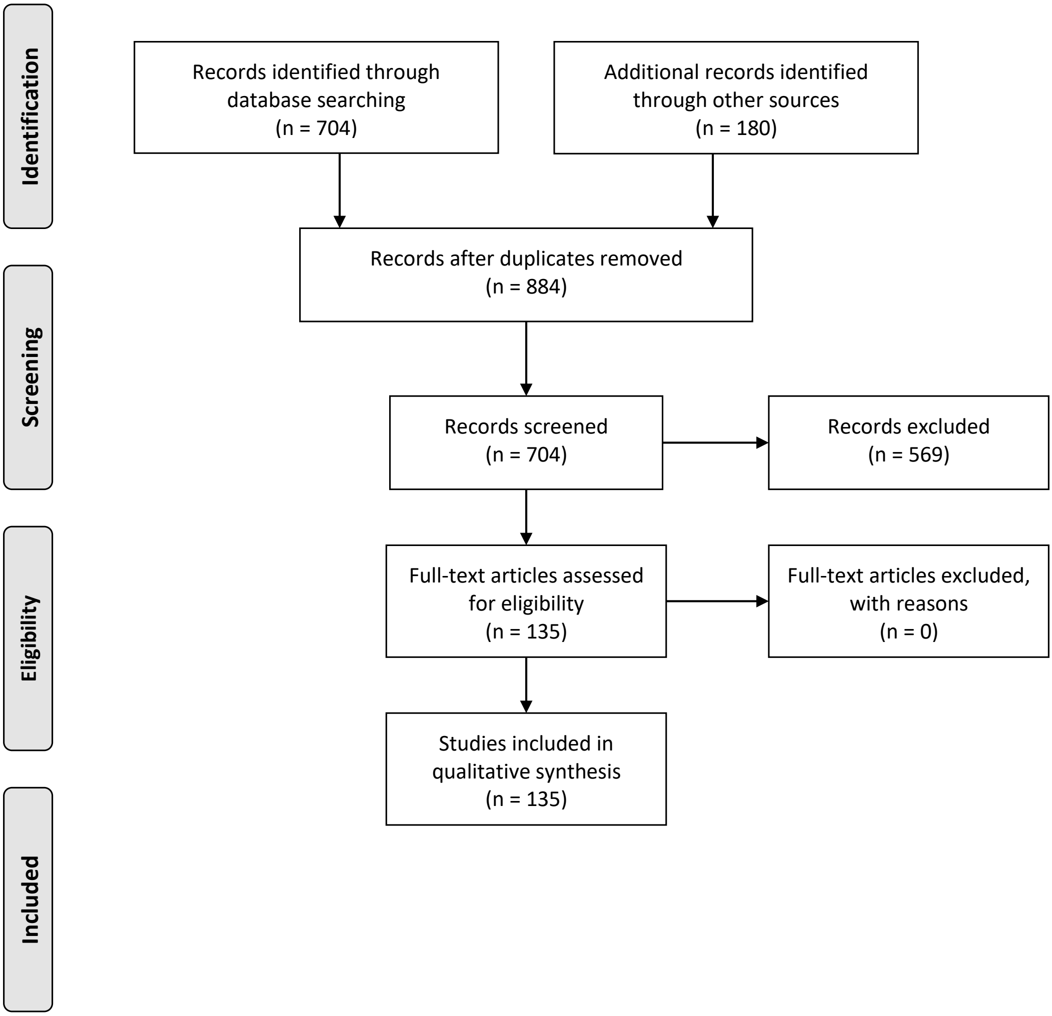
Section/topic	#	Checklist item	Reported on page #
Risk of bias across studies	15	**Specify any assessment of risk of bias that may affect the cumulative evidence (e.g., publication bias, selective reporting within studies).** No assessment was conducted. Efforts to minimize bias included assuring a broad and diverse range of scientific sources were assessed in the literature studies.	
Additional analyses	16	**Describe methods of additional analyses (e.g., sensitivity or subgroup analyses, meta-regression), if done, indicating which were pre-specified.** No further statistical analyses of data were conducted.	
RESULTS			
Study selection	17	**Give numbers of studies screened, assessed for eligibility, and included in the review, with reasons for exclusions at each stage, ideally with a flow diagram.**	3
Study characteristics	18	**For each study, present characteristics for which data were extracted (e.g., study size, PICOS, follow-up period) and provide the citations.** Data extraction was based on the objectives and research questions of the review and the pre-defined LCA problem areas as presented by Reap et al. (2008).	
Risk of bias within studies	19	**Present data on risk of bias of each study and, if available, any outcome level assessment (see item 12).** There were no statistical analyses.	
Results of individual studies	20	**For all outcomes considered (benefits or harms), present, for each study: (a) simple summary data for each intervention group (b) effect estimates and confidence intervals, ideally with a forest plot.** There were no statistical analyses.	
Synthesis of results	21	**Present results of each meta-analysis done, including confidence intervals and measures of consistency.**	

		There were no statistical analyses.	
Risk of bias across studies	22	**Present results of any assessment of risk of bias across studies (see Item 15).** There were no statistical analyses.	
Additional analysis	23	**Give results of additional analyses, if done (e.g., sensitivity or subgroup analyses, meta-regression [see Item 16]).** There were no statistical analyses.	
DISCUSSION			
Summary of evidence	24	**Summarize the main findings including the strength of evidence for each main outcome; consider their relevance to key groups (e.g., healthcare providers, users, and policy makers).**	18-19
Limitations	25	**Discuss limitations at study and outcome level (e.g., risk of bias), and at review-level (e.g., incomplete retrieval of identified research, reporting bias).**	18
Conclusions	26	**Provide a general interpretation of the results in the context of other evidence, and implications for future research.**	18-19
FUNDING			
Funding	27	**Describe sources of funding for the systematic review and other support (e.g., supply of data); role of funders for the systematic review.** This research received no external funding.	19

From: Moher D, Liberati A, Tetzlaff J, Altman DG, The PRISMA Group (2009). Preferred Reporting Items for Systematic Reviews and Meta-Analyses: The PRISMA Statement. PLoS Med 6(7): e1000097. doi:10.1371/journal.pmed1000097

For more information, visit: **www.prisma-statement.org**.

PRISMA 2009 Flow Diagram



From: Moher D, Liberati A, Tetzlaff J, Altman DG, The PRISMA Group (2009). *P*referred *R*eporting *I*tems for *S*ystematic Reviews and *M*eta-*A*nalyses: The PRISMA Statement. PLoS Med 6(7): e1000097. doi:10.1371/journal.pmed1000097

For more information, visit www.prisma-statement.org.

II

Water Resources Management
https://doi.org/10.1007/s11269-021-02819-1

Human-Water Dynamics and their Role for Seasonal Water Scarcity – a Case Study

Andreas Nicolaidis Lindqvist[1,2] · Rickard Fornell[1] · Thomas Prade[2] · Linda Tufvesson[2] · Sammar Khalil[2] · Birgit Kopainsky[3]

Received: 5 October 2020 / Accepted: 21 March 2021/Published online: 17 July 2021

Abstract
Ensuring sustainable management and an adequate supply of freshwater resources is a growing challenge around the world. Even in historically water abundant regions climate change together with population growth and economic development are processes that are expected to contribute to an increase in permanent and seasonal water scarcity in the coming decades. Previous studies have shown how policies to address water scarcity often fail to deliver lasting improvements because they do not account for how these processes influence, and are influenced by, human-water interactions shaping water supply and demand. Despite significant progress in recent years, place-specific understanding of the mechanisms behind human-water feedbacks remain limited, particularly in historically water abundant regions. To this end, we here present a Swedish case study where we, by use of a qualitative system dynamics approach, explore how human-water interactions have contributed to seasonal water scarcity at the local-to-regional scale. Our results suggest that the current approach to address water scarcity by inter-basin water transports contributes to increasing demand by creating a gap between the perceived and actual state of water resources among consumers. This has resulted in escalating water use and put the region in a state of systemic lock-in where demand-regulating policies are mitigated by increases in water use enabled by water transports. We discuss a combination of information and economic policy instruments to combat water scarcity, and we propose the use of quantitative simulation methods to further assess these strategies in future studies.

Keywords Water · Resource management · Socio-hydrology · Systems thinking · System dynamics

✉ Andreas Nicolaidis Lindqvist
andreas.nicolaidis@ri.se

Extended author information available on the last page of the article

1 Introduction

Water scarcity is a growing problem impacting human health, economic development and ecological systems in many regions around the world (Wimmer et al. 2015). Pressure on global freshwater resources, driven by population growth, expansion and intensification of agriculture, urbanization, industrial development and climate change, are expected to push up to 50% of the world's population into a state of permanent or periodic water insecurity by 2050 (United Nations 2018).

Addressing this challenge requires integrated approaches that account for how water acts as a link between different parts of society and nature. Better understanding of how actions in one part of the interconnected social, hydrological and ecological system, may have cascading effects across space and time is pressingly needed (United Nations 2018). Integrated Water Resources Management (IWRM) (Gorre-Dale 1992) is the dominating approach used in the management and planning of water resources. However, the IWRM approach has gained critique for treating the social and hydrological sectors as isolated subsystems that to a large extent develop independently from one another (Blair and Buytaert 2016). This approximation may be sufficient for short-term management but for long-term planning and policy making, failing to account for the bidirectional human-water feedbacks can lead to unintended consequences and "policy resistance" (Di Baldassarre et al. 2019; Sterman 2000a).

Policy resistance is the phenomenon where well-intended policy solutions fail to produce their desired outcomes due to unanticipated feedback effects, triggered endogenously by the causal structure of the targeted system (Sterman 2000a). Two well-documented examples of policy resistance with respect to socio-hydrological interactions are "Water Rebound Effects" (Beal et al. 2014), where improvements in efficiency lead to higher total consumption, in addition to "Supply-Demand Cycles" (Kallis 2010), where increases in water supply capacity enable growth that generate further capacity demand. These phenomena occur because policies are designed and implemented without taking into account the two-way feedbacks between the physical, technical and social dimensions of the human-water system, leading to counterintuitive changes in water demand (Di Baldassarre et al. 2019).

Over the last years, considerable progress has been made in building macro-level theories on how socio-hydrological interactions influence water system behavior, and in the scientific community there is a strong consensus on the importance of accounting for these interactions in water management and planning (Di Baldassarre et al. 2019; Langarudi et al. 2019). Despite these advancements, in-depth and place-specific understanding of the mechanisms behind human-water feedbacks remain limited (Xu et al. 2018). Among practitioners the water management and planning process still relies heavily on hydrological and socio-economic forecasts largely conducted in isolation from one another (Di Baldassarre et al. 2019). To address this knowledge gap, further case-based studies are needed that can generate insights on the role of human-water feedbacks in different social and hydrological settings (Blair and Buytaert 2016).

Among published socio-hydrological case studies, regions with a long history of water scarcity (including the Mediterranean, the Middle East, Australia, parts of the US and parts of Africa) are relatively well-represented (Blair and Buytaert 2016). In contrast, Sweden and other historically water-abundant regions are poorly represented. However, unusually dry weather conditions in recent years have caused local-to-regional seasonal water scarcity to become a growing problem even in these typically water-abundant areas (Ahopelto et al. 2019). With the effects of climate change, this development is likely to continue in the coming decades (Asp et al. 2015). Thus, understanding how socio-hydrological interactions influence water scarcity, and how to manage these interactions, is pressingly needed to guarantee a sustainable future water supply also in hitherto water secure regions.

To this end, we here present a case study from the Swedish island Fårö where we investigate in what ways human-water interactions have contributed to policy resistance, leading to reoccurring and increasingly severe, periods of seasonal water scarcity over the last two decades. We apply a systems thinking (ST) approach, using qualitative system dynamics (SD) to identify the key human-water feedbacks contributing to seasonal water scarcity. The SD method is well-established in the field of social systems modeling in general and socio-hydrological modeling in particular (Bahaddin et al. 2018; Di Baldassarre et al. 2019), and focuses on capturing how the interactions of biophysical and social processes drive overall system behavior. The strength of the method is in its flexibility to model both physical and behavioral processes, and its transparency and ability to shed light on the dynamics emerging from interacting processes in the studied system (Di Baldassarre et al. 2015). We first present the methodology applied to assess the links between the society and the water systems and the logic we use to connect these. Specifically, the identified feedbacks are synthesized into a causal map, providing a conceptual model of the socio-hydrological processes governing water supply and demand on Fårö. As a second step, the conceptual model is used to analyze why historic policies to combat water scarcity have turned out ineffective, and directions for future water management are suggested based on the causal structure of the system. Findings from the study will contribute to building well-needed conceptual understanding of how socio-hydrological dynamics can influence water supply and demand at the local-to-regional scale and push previously water secure areas into water scarcity. This knowledge is important for assisting communities and practitioners in proactive water management and planning. The causal map developed in this paper will be used as a basis for further developing a quantitative simulation model that allows assessing the direct and indirect, short- as well as long-term, effects, synergies and tradeoffs, of different policy measures on the availability of water and socio-economic development, on Fårö and other regions. The quantitative model will be presented in a forthcoming paper by the authors.

2 Materials and Methods

2.1 Study Area

Fårö island (57.9°N, 19.1°E) is located in the Baltic Sea and belongs to the Swedish municipality of Gotland. The area is approximately 114 km^2 with a yearly precipitation of about 500–600 mm and average summer and winter temperatures of 16 °C and − 2 °C, respectively (SMHI, retrieved 2021-02-03). The island has about 300 permanent households and 725 part-time households, used mainly in the summer period, and tourism and agriculture are the dominating industries. The geology is dominated by limestone bedrock covered by a thin layer (0–1 m) of postglacial sediments and sedimentary rock. Due to the geological features, most of the groundwater aquifers are small and respond quickly to changes in weather and/or extraction rates. The only exception is a comparatively large aquifer located in the northeastern part of the island where deep layers of aeolian sand sediments (up to 20 m in depth) allows for considerable groundwater extraction and storage (SGU geokartan, 2021-02-03). This is where the only municipal water plant on the island is located and from here a public grid supplies water to the majority of the tourist facilities, and about 50 residential households. Households outside the public grid rely on private wells for their water supply (Region Gotland 2014; Sjöstrand et al. 2014).

Over the last two decades, Fårö has developed into a popular tourist destination. During peak season (June – August) about 10,000 tourists and part-time residents visit the island (Region Gotland 2014). The visitors are concentrated in the area around the public grid where most of the tourist attractions are located, creating a sharp increase in water consumption that coincides with the seasonal low point in groundwater generation, putting a lot of pressure on the municipal water system.

The municipal water plant started experiencing problems keeping up with demand in the early 2000's. Since 2006, water demand has exceeded supply capacity every summer and the municipality has been supplementing the local plant with water transported by truck from neighboring regions of Gotland. Over the years, several policy measures have been introduced by the municipality to reduce the reliance on transported water (Table 1).

Despite the abovementioned efforts, the extent of the transports has grown from about 1500 m^3 in 2006 to more than 3000 m^3 in 2019, with a record peak in 2016 when 5500 m^3 of water was delivered. The only exception to the trend was 2017 when transports were reduced due to the exploitation of a new aquifer that was later terminated (Table 1). To meet peak season demand (approximately eight weeks every summer) the municipal water services are at present relying on daily water deliveries (Region Gotland personal communication, 2020-05-11). This is not only economically costly for the municipality; low water self-sufficiency is also a significant risk for the region if the water supply-chain would be disrupted. Furthermore, according to recent economic and climate projections for the region, water demand is likely to continue to increase in the coming 30 years, and supply is expected to become increasingly unpredictable. Together, these two trends are likely to further increase the pressure on the water supply systems on Fårö (Eklund 2018).

Table 1 Public policies adopted to reduce reliance on water transports

Policy	Year of introduction	Description
Restrictions on new connections to the public grid	2000	A full stop on new connections to the public water grid is enacted. No new requests are accepted until local water self-sufficiency can be ensured. Exceptions are made to communities of households where inadequate drinking water supply or quality poses a threat to human health.
Water use restrictions	2007	Consumers connected to the public grid are prohibited to use water from the municipal grid for gardening and swimming pools. In 2007 the restrictions applied from June to September, but the duration was gradually extended, and since 2016, restrictions apply from April to October.
Information campaigns	2007	Information on the state of groundwater resources starts being communicated by the municipality on their website.
Minimum well-capacity requirements	2008	Documentation of a minimum well-capacity of 600 l per hour becomes a requirement for building permits to be issued to new off-grid house construction projects.
New aquifer exploitation	2016	A new municipal aquifer is identified and taken into use in 2016 to supplement the existing aquifer. Exploitation of the aquifer is terminated in 2018 due to unsatisfactory water quality.
Information campaigns	2017	An information campaign to encourage water savings in households and among tourists is launched. Information and encouragement to use water more efficiently is communicated in media, on tourist resorts and on the ferry to the island.

2.2 Methods

The assessment in this paper was conducted in two steps. First, a conceptual model, based on participatory modelling exercises and municipal reports and planning documents, was designed and validated (section 2.2.1.). Then, the model was used to identify and analyze potential feedback mechanisms responsible for the increase in water scarcity on Fårö in the past two decades (section 2.2.2.).

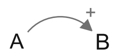
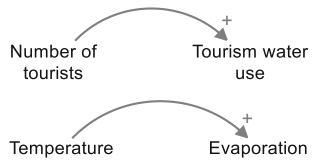
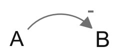
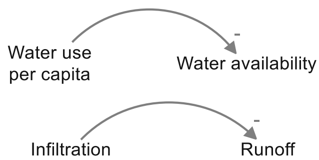
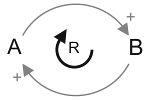
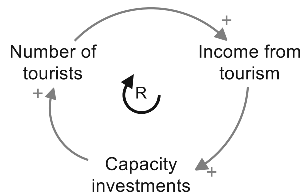
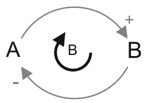
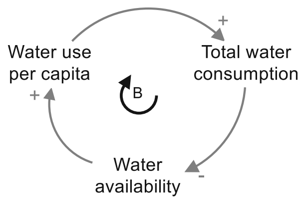
2.2.1 Model Development

To model the key human-water interactions on Fårö we adopted an approach grounded in qualitative SD modeling, utilizing and triangulating a variety of different information and data sources including participatory modeling, literature, statistical data and expert knowledge (Martinez-Moyano and Richardson 2013). The entire process was conducted in close collaboration with the Department of Water Management at Region Gotland (RG), together with representatives from Geological Survey of Sweden (SGU) and the Gotland County Administrative Board. A total of 14 participants, including water utility engineers, technicians, water and environment strategists, hydrogeologists and county water administrators participated in the model development process (see supplementary information for details). The participants had no prior experiences in SD modeling but were given an introduction to the concepts of positive and negative causal relationships, causal mapping, and how circular chains of causality can form feedback loops (Table 2), at the start of each modeling activity. Meetings, workshops and modeling sessions were all conducted online using the Microsoft Teams video meeting function.

Model development started by semi-structured group discussions with representatives from RG. The questions had been prepared beforehand and during the meeting the researchers acted as facilitators, presenting the questions, taking notes, and moderating the discussions. First, the participants were asked to describe how public and private water supply, water demand and water sufficiency had been changing on Fårö over the period from 2000 to 2019. Based on the descriptions, the researchers sketched the behavior of the described variables on "behavior over time" (BOT) graphs (Andersen and Richardson 1997). Time was represented on the horizontal axis and the state of the factor of interest, represented by the vertical axis, was sketched as a continuous variable changing over time according to the participant descriptions. From the BOT graphs, general trends in behavior were elicited together with the participants (e.g. accelerating increase, accelerating decline, oscillations, etc.). These trends were described as the overarching behavior modes, the problem reference modes, of the Fårö human-water system (Sterman 2000b). The participants were then asked to describe: (I) what they conceived as the underlying causes, the drivers, to the behaviors presented in the elicited graphs; (II) what effects these changes in water supply, demand and sufficiency had triggered (public policies, consumer behavior changes, etc.); and (III) if there were other socio-economic or biophysical trends they had witnessed during the same time period that could have influenced water supply, demand and/or sufficiency. Results from the session were documented to be used in the forthcoming modeling process before the meeting was closed.

After the group discussions, the behaviors elicited from the participants were validated by comparing them to statistical data from Statistics Sweden (https://www.scb.se/en/) and RG. The validation of the suggested cause-effect relationships was conducted by structure examination tests (Schwaninger and Groesser 2016); suggested causal connections were cross-compared against findings from previous studies on Fårö (Brunner 2014; Rivera et al. 2011;

Table 2 Top section: graphical notation and polarity of causal relationships used in model development. Bottom section: examples and behavior of reinforcing and balancing feedbacks. Adapted from Mirchi et al. (2012)

Connection	Causal relationship	Examples of causal relationships
	A change in the state of A causes the state of B to change in the same direction; all else equal, if A increases/decreases, B increases/decreases to a state above/below what it would otherwise have been.	
	All else equal, a change in the state of A causes the state of B to change in the opposite direction; all else equal, if A increases/decreases, B decreases/increases to a state below/above what it would otherwise have been.	
Feedback loop	**Behavior**	**Examples of feedback loops**
	Reinforcing (positive) feedback. If the state of A changes, this causes a change in B that feeds back to amplify the change in A. Behavior: accelerating growth or decline. Notation used in causal maps: a curved arrow with a capital R.	
	Balancing (negative) feedback. If the state of A changes, this causes a change in B that feeds back to negate/dampen the change in A. Behavior: balancing change, stabilizing around a reference or goal level. Notation used in causal maps: a curved arrow with a capital B.	

Sjöstrand et al. 2014), and the perspectives of subject experts to assess how well they matched established understanding of the system. For instance, hydrogeologists from SGU were consulted for validation of statements regarding hydrology and groundwater processes. When no data or previous studies were available to confirm a causal statement or trend, it was cross-checked for consistency with the statements from other participants in the study. The suggested trend was assumed to be substantiated if there was uniform agreement about its overall behavior (e.g. increasing, decreasing, oscillating) among the participants. If there was disagreement, the suggested trend was further discussed in subsequent modeling sessions until consensus could be reached.

Following validation, the BOT graphs and elicited drivers were used as a starting point from which the chains of cause and effect were modeled backwards, striving towards providing an endogenous explanation to the elicited trends according to methods described by Martinez-Moyano and Richardson (2013) and Sterman (2000b). To achieve a consistent causal explanation, the driving variables provided by the study participants were complemented by additional auxiliary variables from previous studies (Brunner 2014; Eklund 2018; Rivera et al. 2011; Sjöstrand et al. 2014) and follow-up discussions with the participants. From this process the first draft of the causal map was developed by the modeling team.

Structural validation and further refinement of the causal map were conducted through a modeling workshop with the project participants. The draft model was presented on screen and in a step-by-step fashion the researchers guided the participants through the entire model, explaining the logic and assumptions of each causal link. The participants were asked to critically review each link presented and indicate if they agreed or disagreed with the suggested causation and polarity. The participants were also prompted to provide suggestions for changes and improvements to the presented model structure. Suggested changes were discussed within the group until consensus regarding their validity and place in the model structure was reached. Structural adjustments suggested were documented and implemented to the model structure by the research team after the workshop, generating an updated model draft. This cycle of participatory validation and adjustments was repeated twice at which point no further changes to the model structure were voiced. The result was a final conceptual model of the socio-hydrological processes regulating water supply, demand, and sufficiency on Fårö.

2.2.2 Model Analysis

Being able to distinguish which feedback loops in a system are responsible for generating an observed behavior can provide qualitative information about suitable direction and design of future policy interventions (Mirchi et al. 2012; Sterman 2000a). To this end, the conceptual model developed in 2.2.1. was analyzed for feedback loops and these were labeled according to the notations described in Table 2. By comparing the reference modes elicited in 2.2.1. with the feedback structure in the model, initial hypotheses about feedback loops that drive system behavior at any point in time in the past could be identified (Bahaddin et al. 2018).

Results from the feedback loop analysis were used to examine why historic policy interventions to mitigate water scarcity had been ineffective. Lastly, the feedback structure was used to provide directions for future water management policies by identifying intervention points in the system that might help to shift loop dominance and loop direction towards more desirable outcomes.

3 Results

Results from the model development and the model analysis process are presented in section 3.1. and section 3.2. respectively.

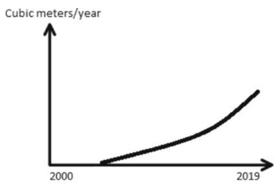
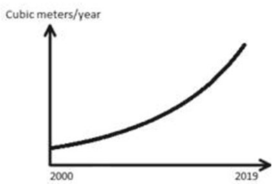
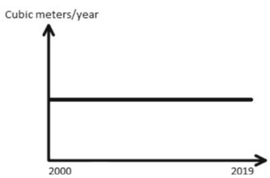
3.1 Historic Behavior and Model Structure

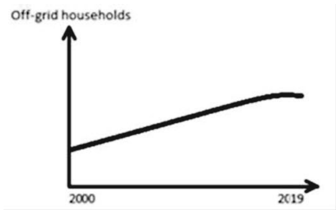
BOT graphs of problem variables and trends elicited in the initial group discussions are presented in Table 3 and the individual model variables in the final model are presented in Table 4 together with their causal relationships. The full conceptual model is presented in Fig. 1.

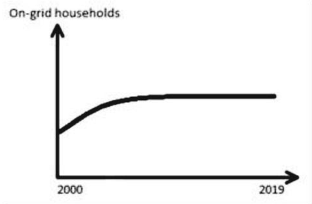
3.2 Results from Feedback Loop Analysis

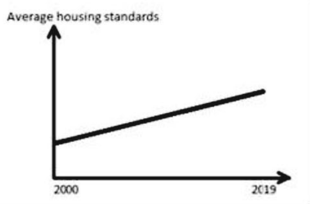
The final model consisted of a total of 14 feedbacks loops (Fig. 1 and Table 5). Dynamic hypotheses derived from comparing the feedback structure of the model with the reference modes in Table 3 are presented below.

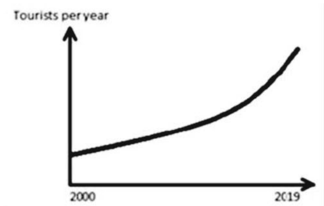
Table 3 Dynamic behavior of key problem variables and trends elicited and validated during the model development process. Modes of validation used include comparison to statistical data provided by Statistics Sweden [A] or RG [B], literature [C] (reference in brackets), expert judgement by hydrogeologists from SGU [D], and agreement within the project group [E]

Variable	Behavior over time (BOT)	Description
Water transports		Water transports was used as a proxy for water scarcity. Transportation of water started in 2006 and has been increasing since then except for the years 2017 and 2018 when a backup aquifer was temporally taken into use by the municipality. Water transports are concentrated to June, July and August as the coincides with the tourist season. Validated by: [B]
On-grid water use		On-grid water use has been increasing since year 2000. The rate of increase has been greatest in the last 4-5 years. Validated by: [B]
Water supply capacity of the public plant		The water supply capacity has remained stable over the study period. Validated by: [E]

<table>
<tr><td>Off-grid households</td><td>
</td><td>The number of off-grid households have been steadily increasing for most of the study period but in the last years the growth shows tendencies of slowing down due to difficulties in finding housing sites with sufficient groundwater supplies.

Validated by: [B, C, E]
(Brunner 2014)</td></tr>
<tr><td>On-grid households</td><td>
</td><td>The number of on-grid households increased up to 2006 and have since then remained stable due to restrictions on new connections to the public grid.

Validated by: [E]</td></tr>
<tr><td>Housing standards</td><td>
</td><td>Average housing standards have been increasing over the study period.

Validated by: [E]</td></tr>
<tr><td>Tourists per year</td><td>
</td><td>The number of tourists per year have been increasing over the study period. The rate of increase has been accelerating in the last ten years.

Validated by: [A, C, E]
(Brunner 2014)</td></tr>
</table>

4 Discussion

4.1 What Drives the Increasing Water Transports and why Have Previous Policies Been Ineffective?

To explain the historic growth in water transports illustrated in Table 3 one needs to understand the combined effects of the feedback loops in the system. *Water transports* is a response to the *on-grid water gap* and occurs when *total-on-grid water use* exceeds the *on-grid water supply*. This is a supply-targeting policy and through *B1* it can quickly close the gap by supplementing the local water system with water from an exogenous source. Loop *B4-B6* and *B7* on the other hand reduce the gap by lowering water use by imposing physical constraints, increasing awareness among consumers, and/or by slowing down growth in tourism and housing standards. These balancing processes (increasing supply or reducing consumption) can both individually stabilize *water transport*s, but in combination they can, counterintuitively, cause it to

Table 4 Variables and causal relationships included in the final model. Variable definition is provided in italic under the variable name in the left most column. Modes of validation follow the same logic as described in Table 3

Variable (definition in *italics*)		Causal relationships	Polarity	Validation
Actual capacity gap *The difference between the local water supply capacity (excluding transported water) and the total on-grid water use*	→	Water use restrictions Restrictions on new connections to public grid	+ +	[E] [E]
Aquifer extraction capacity *The physical extraction capacity from the municipal aquifer*	→	Local water supply capacity	+	[E]
Average well capacity *The average water supply capacity of private wells*	→	Off-grid water supply	+	[E]
Connections to public grid *The number of new consumers connecting to the public grid per year*	→	On-grid households Off-grid households	+ -	[E] [E]
Destination attractiveness *The relative attractiveness of Fårö as a holiday destination*	→	Tourists Property prices New constructions	+ + +	[C, E] (Formica and Uysal 2006) [E] [E]
Fraction of aquifer exploited *The fraction of suitable locations for private wells already exploited*	→	New constructions Average well capacity	- -	[C, E] (Bardi and Lavacchi 2009) [B, E]
Household water use *The total amount of water consumed by households on the public grid per year*	→	Total on-grid water use	+	[E]
Housing standards *The property size, number of bathrooms, swimming pools and garden size*	→	Water use per capita	+	[C] (Bich-Ngoc and Teller 2018)
Local water supply capacity *The water supply capacity of the local municipal water system*	→	Actual capacity gap On-grid water supply	- +	[E] [E]
New Constructions *The number of new house constructions per year*	→	Off-grid households	+	[E]
Off-grid households *The number of households with private wells outside the public grid*	→	Off-grid water use Private wells Connections to public grid	+ + +	[E] [B, E] [E]
Off-grid water gap *The difference between the off-grid water use and the off-grid water supply plus water extracted at the public tap station by off-grid households*	→	Water extraction at tap station Off-grid water use	+ -	[E] [E]
Off-grid water supply *The supply of water to off-grid households from private wells*	→	Off-grid water gap	-	[E]
Off-grid water use *The water use by off-grid households*	→	Off-grid water gap	+	[E]
On-grid households	→	Household water use	+	[E]
The number of households connected to the public grid		Connections to public grid	-	[E]
On-grid water gap *The difference between the total on-grid water use and the on-grid water supply*	→	Water transports Perceived water sufficiency Water use per capita	+ - -	[B] [C] (Di Baldassarre et al. 2018; Kallis 2010) [E]
On-grid water supply *The total water available to consumers on the public grid*	→	On-grid water gap	-	[E]
Perceived water sufficiency *The perceived level of water self-sufficiency among consumers on the public grid*	→	Destination attractiveness Water use per capita	+ +	[C, E] (Formica and Uysal 2006) [E]
Private wells *The number of off-grid private wells*	→	Fraction of aquifer exploited	+	[E]
Property prices *The average house price on Fårö*	→	Housing standards	+	[C, E] (Bich-Ngoc and Teller 2018)

Restrictions on new connections to public grid *The extent and duration of restrictions issued by the municipality, limiting the possibilities for new households to connect to the public grid*	→	Connections to public grid	-	[E]
Total on-grid water use *The total water use on the public grid*	→	On-grid water gap	+	[E]
Tourists *The number of tourists visiting Fårö every year*	→	Total on-grid water use Water extraction at tap station	+ +	[E] [E]
Water extraction at tap station *The amount of water extracted from public tap stations*	→	Total on-grid water use Off-grid water gap	+ -	[E] [E]
Water plant capacity *Technical extraction capacity of the municipal water plant*	→	On-grid water supply	+	[E]
Water transports *The amount of water transported by truck to supplement the public water supply*	→	On-grid water supply	+	[E]
Water use per capita *The average water use per person and year*	→	Household water use	+	[E]
Water use restrictions *The extent and duration of water use restrictions applying to consumers on the public grid*	→	Water use per capita	-	[E]

escalate: When *B1* closes the *on-grid water gap* by increasing supply, this contributes to maintaining a perception of water sufficiency among consumers. Incentives to save water erode (*B4*), and in the longer-term this drives investments in water-demanding capital, e.g. the expansion of tourist capacity and the improvement of housing standards (*B5-B7*). This combination of balancing feedback loops can help explain why historic policies to reduce consumption (see Table 1) have been ineffective in providing lasting reductions on water use.

Many of the investments in water-demanding capital that are made possible thanks to the *water transports* are long-lived. For instance, a new hotel may have a lifetime of several decades during which it will require a steady water supply to operate. Therefore, *water transports* indirectly contribute to slowly increasing the water supply necessary for the island to meet the minimum requirements of its businesses and households. This phenomenon of increased supply causing an increase in demand, also known as supply-demand cycles (Kallis 2010), can help explain the growth in *on-grid water use*, *tourists* and *water transports* presented by the BOT graphs in Table 3.

The long lifetime of the water-consuming capital can also help explain why historic policies to decrease water use have been ineffective in reducing water transports. New investment decisions are made with the expectation that *water transports* will continue and water supply will remain high. Once the investments have been made it is very difficult for the municipality to phase out *water transports*, thereby reducing water availability back to its previous level, without negatively impacting investors (Greve et al. 2018). This results in a *systemic lock-in*, a phenomenon where historic events determine the future behavior of the system. These effects are well-documented in studies on human-energy systems (Seto et al. 2016), and our results suggest *system lock-ins* can also arise in human-water systems where they can greatly interfere with future water management policies. These findings are in line with previous studies (Markolf et al. 2018) and illustrate the importance of understanding and assessing the potential the systemic impacts of water management strategies.

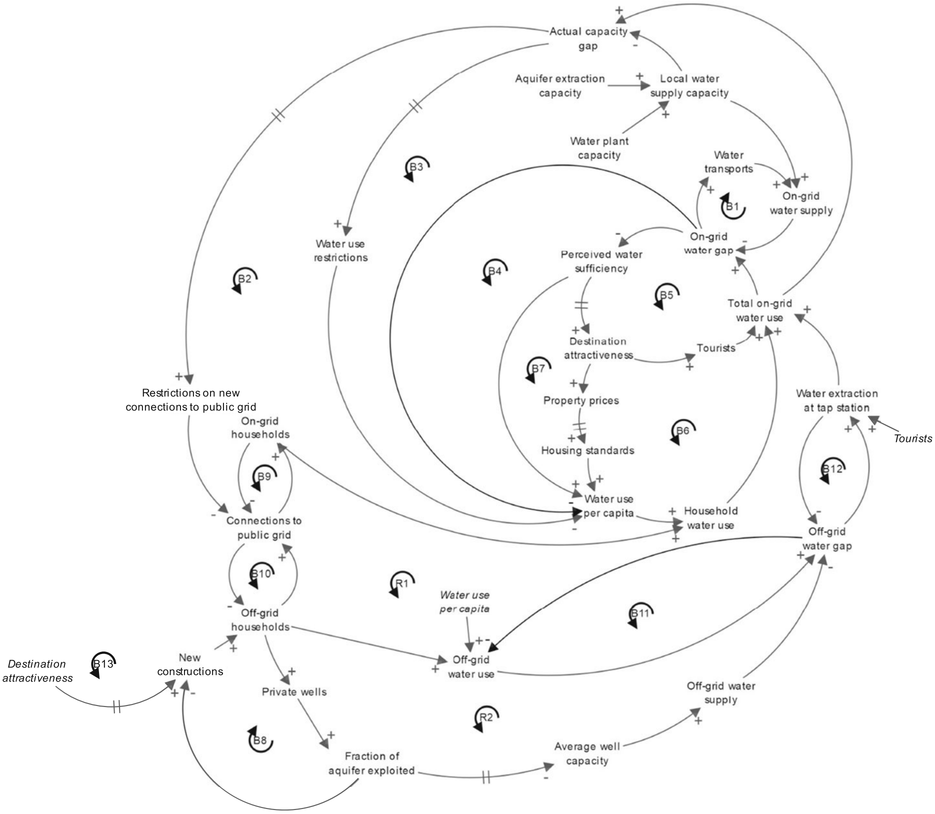


Fig. 1 Final conceptual model derived from the causal relationships described in Table 4. Causal connections with double dashed bars indicate that there is a time delay between cause and effect. Curved arrows with a capital B/R represent balancing and reinforcing feedback loops respectively according to the notation explained in Table 2

As illustrated in Fig. 1, there is a distinction between the *on-grid water gap* and the *actual capacity gap.* The *on-grid water gap* puts a physical limit to consumer water use (if the gap grows too big, supply failures start occurring) but the *actual capacity gap* is the difference between the *local water supply capacity* of the public system and the *total on-grid water use*. In contrast to the *on-grid water gap* that can be periodically closed by supplementing supply with transported water, the *actual capacity gap* has been growing throughout the study period as water use has increased but local supply capacity has remained steady (see Table 3). The growing *actual capacity gap* has caused municipal *water use restrictions* to increase in both scope and duration during the study period (*B3*) and caused a decline in the number of new *connections to the public grid* (*B2*). Even though small reductions in water use have been attributed to these restrictions (about 10–15% decrease, Region Gotland personal communication 2020-05-11) and the number of on-grid households have stabilized (Table 3), the reductions in *total on-grid water use* have not been permanent. After a 12–24 month delay following observed effects of restrictions, water use has tended to return to, or above, its previous levels (Region Gotland personal communication, 2020-05-11). This suggest that loop *B2* and *B3* are insufficient to counteract the growth in water consumption caused by the supply-demand cycles described above.

Table 5 Identified feedback loops and a description of the dynamic behavior they generate in the context of the study. Loop numbers with the prefix B/R represent balancing and reinforcing feedback loops respectively according to the notation explained in Table 2

Feedback loop	Description
B1	Causal chain: *on-grid water gap* ➔ *on-grid water supply* ➔ *water transports* ➔ *on-grid water gap*
	Behavior: as the *on-grid water gap* expands this trigger more *water transports* which closes the gap by increasing the *on-grid water supply*.
B2	Causal chain: *total on-grid water use* ➔ *actual capacity gap* ➔ *restrictions on new connections to public grid* ➔ *connections to public grid* ➔ *on-grid households* ➔ *household water use* ➔ *total on-grid water use*
	Behavior: when *total on-grid water use* grows, this expands the *actual capacity gap,* which makes the municipality enforce stronger *restrictions on new connections to public grid.* The restrictions reduce the number of *connections to the public grid*, thereby maintaining the number of *on-grid households* and *household water use* lower than and they would otherwise have been which reduces *total on-grid water use*.
B3	Causal chain: *actual capacity gap* ➔ *water use restrictions* ➔ *water use* per capita ➔ *household water use* ➔ *total on-grid water use* ➔ *actual capacity gap*
	Behavior: if *household water use increases, total on-grid water use* will also increase, and *the actual capacity gap* expands. The expanded gap causes the municipality to enforce stronger *water use restrictions* which drives down *water use* per capita and makes *household water use* decline again.
B4	Causal chain: *on-grid water gap* ➔ *water use* per capita ➔ *household water use* ➔ *total on-grid water use* ➔ *on-grid water gap*
	Behavior: if the *on-grid water gap* grows very large this will eventually cause *water use* per capita to decrease due to supply failures. The reduction in *water use* per capita will cause *household water use* and *total on-grid water* use to decrease, eventually reducing the *on-grid water gap.*
B5	Causal chain: *on-grid water gap* ➔ *perceived water sufficiency* ➔ *destination attractiveness* ➔ *tourists* ➔ *total on-grid water use* ➔ *on-grid water gap*
	Behavior: if the *on-grid water gap* increases to a level where it starts influencing water supply to tourist facilities this will cause the *perceived water sufficiency* to decline, reducing the *destination attractiveness* of Fårö. Falling *destination attractiveness* will cause the number of *tourists* to decline and *total on-grid water use* to go down, making the *on-grid water gap shrink* due to lower demand.
B6	Causal chain: *on-grid water gap* ➔ *perceived water sufficiency* ➔ *destination attractiveness* ➔ *property prices* ➔ *housing standards* ➔ *water use* per capita ➔ *household water use* ➔ *total on-grid water use* ➔ *on-grid water gap*
	Behavior: if the *on-grid water gap* is closed, the *perceived water sufficiency* goes up and *destination attractiveness* increases. Higher *destination attractiveness* leads to higher *property prices* which over time drives up *housing standards*. Higher *housing standards* increases average *water use* per capita, increasing *household water use* and *total water use*, causing the *on-grid water gap* to expand again.
B7	Causal chain: *on-grid water gap* ➔ *perceived water sufficiency* ➔ *water use* per capita ➔ *household water use* ➔ *total on-grid water use* ➔ *on-grid water gap*
	Behavior: if the *perceived water sufficiency* increase, this causes an increase in *water use* per capita, higher *household water use* and a growing *total on-grid water use*. The increased water use causes the *on-grid water gap* to expand and *perceived water sufficiency* among consumers declines again.
B8	Causal chain: *off-grid households* ➔ *private wells* ➔ *fraction of aquifer exploited* ➔ *new constructions* ➔ *off-grid households*
	Behavior: When the number of *off-grid households* increase, this leads to more *private wells* being drilled and the *fraction of aquifer exploited* increases. This pushes down the number of *new constructions* because it gets progressively harder to find new housing sites with sufficient aquifer capacity for exploitation, thereby reducing further growth in *off-grid households*.
B9	Causal chain: *on-grid households* ➔ *connections to public grid* ➔ *on-grid households*
	Behavior: If the number of *on-grid households* increase fewer *new connections to public grid* will be allowed by the municipality, keeping the number of *on-grid households* below what they would otherwise have been.

Table 5 (continued)

Feedback loop	Description
B10	Causal chain: *off-grid households* ➔ *connections to public grid* ➔ *off-grid households* Behavior: The municipality generally do not approve new households to connect to the public grid because of the current low level of water self-sufficiency. However exceptions are sometimes made and thus, if the number of *off-grid households* increase the number of new *connections to the public grid* will also increase, thereby reduce the number of *off-grid households* below what it would otherwise have been.
B11	Causal chain: *off-grid water use* ➔ *off-grid water gap* ➔ *off-grid water use* Behavior: If the *off-grid water use* increases this will increase the *off-grid water gap*, and if the gap grows big enough it will reduce the *off-grid water use* below what it would otherwise have been.
B12	Causal chain: *water extraction at tap station* ➔ *off-grid water gap* ➔ *water extraction at tap station* Behavior: If the *off-grid water gap* increases, e.g. because demand exceeds the supply capacity of the private wells, this will lead to more households having to go to the municipal tap station to get their water. W*ater extraction at the public tap station* will increase and the *off-grid water gap* to be momentarily reduced.
B13	Causal chain: *destination attractiveness* ➔ *new constructions* ➔ *off-grid households* ➔ *off-grid water gap* (either via *off-grid water use* or via *private wells)* ➔ *water extraction at tap station* ➔ *total on-grid water use* ➔ *on-grid water gap* ➔ *perceived water sufficiency* ➔ *destination attractiveness* Behavior: If the *destination attractiveness* of Fårö increases this leads to more *new constructions* and therefore a growth in the number of *off-grid households*. More households lead to a greater *off-grid water use* and a growing *off-grid water gap*. The off-grid gap is closed by increasing *water extraction at tap station,* which increases the *total on-grid water use.* This leads to a *greater on-grid water gap* (effectively moving the gap from off-grid to on-grid). The growing *on-grid water gap* contributes to a lower *perceived water sufficiency*, dampening further growth in *destination attractiveness.*
R1	Causal chain: *actual capacity gap* ➔ *restrictions on new connections to public grid* ➔ *connections to public grid* ➔ *off-grid households* ➔ *off-grid water use* ➔ *off-grid water gap* ➔ *water extraction at tap station* ➔ *total on-grid water use* ➔ *actual capacity gap* Behavior: When the *actual capacity gap* grows the municipality imposes more *restrictions on new connections to the public grid*. More restrictions cause fewer *connections to the public grid*, in effect reducing the movement of households from off-grid to on-grid, keeping the number of *off-grid households* above what they would otherwise have been. More *off-grid households* increase the pressure on the off-grid water system, resulting in more *off-grid households* that will need to use the public tap station. This increases the *total public water demand* and making the *actual capacity gap* grow even further.
R2	Causal chain: *actual capacity gap* ➔ *restrictions on new connections to public grid* ➔ *connections to public grid* ➔ *off-grid households* ➔ *private wells* ➔ *fraction of aquifer exploited* ➔ *average well capacity* ➔ *off-grid water gap* ➔ *water extraction at tap station* ➔ *total on-grid water use* ➔ *actual capacity gap* Behavior: Following the same logic as R1, an increase in *actual capacity gap* leads to a build up of *off-grid households* beyond what otherwise would have been. More households lead to more *private wells* and a greater *fraction of aquifer exploited*. Over time this leads to a decline in *average well capacity* because there are fewer and fewer high-capacity well sites left to exploit. This increases the *off-grid water gap* and, by the same logic as in R1, increases *extraction at tap station*, rises the *total on-grid water use* and expands the *actual capacity gap* even more.

The remaining loops influencing water transports are *R1*, *R2*, *B8* (indirectly) and *B13*. *R1* and *R2* will both, in theory, contribute to the *on-grid water gap* by increasing the number of *off-grid households* that utilizes the public tap station, thereby increasing the *total on-grid water use*. However, since no data is available on the use of the public tap station, the contribution of these loops to the historic water transports cannot be determined. That said, it is likely that if *destination attractiveness* is maintained high, e.g. by means of *water transports*, this will attract more *new constructions* off-grid and thereby increase the *extraction* *at the public tap station* (*B13*). At least during dry years, when the probability of private wells running dry is high, this can contribute to future water transports, in effect shifting the water gap from the private to the public water system. The potential magnitude of this shift is largely governed by loop *B8*; growth in *off-grid households* will eventually be limited by the availability of housing sites with sufficient aquifer capacity for building permits to be issued. As described in Table 1, minimum well-capacity requirements for new constructions have been imposed by the municipality, strengthening the effect of loop *B8*, causing a slowdown in the growth of *off-grid households* in some parts of the island (Table 3). It thus seems that minimum capacity requirements for off-grid households can both reduce the risk for water scarcity among off-grid consumers, and reduce the need for future water transports. However, since these minimum requirements only apply for new constructions and not for upgrading or expansion of existing houses, off-grid *water use* per capita and total *off-grid water use* may continue to increase and contribute to *water transports*.

To summarize, our findings suggest that the growing need for water transports is a result of the supply-demand cycles created when an increase in water supply contributes to a further increase in demand. In the short run, increasing *water transports* addresses the symptom of the problem (the *on-grid water gap*), but the policy fails to address, and may even enforce, the underlying human-water interactions that drive the demand cycles and the lock-in effects they create in the long run.

4.2 How Can Future Water Scarcity and Increasing Reliance on Water Transports Be Mitigated?

As our findings suggest, and as supported by previous studies, improvements in regional water self-sufficiency achieved by supply-targeted policies (e.g., inter-basin water transfers or expansion of water reservoirs) will quickly be offset by increased water consumption unless complemented by sufficiently rigorous policies on the demand-side as well (Kallis 2010). Some of the demand targeting policies implemented by RG have contributed to reducing the number of consumers (e.g. by reducing the number of on-grid households and limiting new off-grid constructions to areas with sufficient water supplies) but lasting reductions in water use among already established consumers have not been achieved. We hypothesize this is mainly due to the perception of water sufficiency among consumers that is maintained high due to the reoccurring water transports. Thus, future policies need to be directed towards bringing perceptions of the *on-grid water gap* closer to the *actual water gap,* combined with policies that weaken the reinforcing effects increasing supply has on water demand.

Suggestions for suitable policies may be found in previous studies. For instance, Mini et al. (2015) conducted a study on the effectiveness of water conservation measures in California, highlighting that mandatory restrictions, combined with pricing measures, can be effective to reduce household water consumption. On Fårö, consumption tariffs on public water have not been extensively utilized as a policy measure to reduce consumption. Introducing a pricing model where water tariffs are correlated with the level of the actual capacity gap could create incentives to reduce water use. Applying this type of pricing to both on-grid households and to the public tap station (which is currently free of charge) is a possible policy to both bring perceived water sufficiency closer to the real state of water sufficiency, and generate additional income for the municipality to invest in the water supply system. On the other hand, Lu et al. (2019) conducted a study comparing the effect of price interventions and behavior interventions on household water consumption in the UK. Results show that behavioral interventions may be more effective than price interventions in regions where the household water bill is relatively small in relation to household income. With the majority of houses on Fårö being part-time holiday houses, typically belonging to high-income consumers (Region Gotland personal communication, 2020-05-11), interventions focusing on behavior rather than price may be more suitable for the region. Also, tourist water use would not be directly addressed by water pricing schemes and therefore campaigns to increase awareness about the fragility of local water resources may be a more effective strategy to reduce tourist water consumption (Gabarda-Mallorquí and Ribas Palom 2016).

Because of the *lock-in effects* described in section 3.2.1., significant reductions in water supplies is not a realistic policy solution for Fårö. However, gradually increasing local water supply capacity and successively replacing water transports, whilst at the same time controlling total on-grid water use by means of the fiscal and information policy measures described above, could allow for a transition towards water self-sufficiency. Artificial groundwater infiltration, wastewater recycling, seasonal water storage and stormwater utilization are all examples of potential solutions to increase local water supply, see e.g. (Pincetl et al. 2019). Falco and Webb (2015) present the use of "water microgrids" as a promising solution to contribute to both consumer behavioral change and increase the resilience of water supply systems. A distributed system for rainwater collection, storage and distribution could provide significant redundancy as precipitation could be collected and stored during the winter season when many part-time houses are not in use. In the summer, when water demand is high, the collected rainwater can supplement the public grid and greatly reduce the stress on the municipal groundwater aquifers. This would reduce the need for water transports and cut some of the associated logistical costs. The money could instead be directed towards subsidizing household water collection and storage infrastructure. Additionally, turning water consumers to small-scale water producers would make households part of the water supply system. Consumers could monitor the water level of their storage cisterns, constantly maintaining an updated level of *perceived water sufficiency*, and therefore make more informed decisions regarding their own consumption. Being more responsible for their own water supply, consumers are less likely to make new investments in water-intensive capital, thereby reducing the risk for unintended lock-in effects to occur.

5 Conclusions

In this study we have explored how human-water interactions can influence water supply and demand at the local-to-regional scale. We have developed a conceptual model of how water transfers can lead to supply-demand cycles and cause system lock-in effects, pushing previously water-secure regions into a state of escalating water scarcity that is resistant to policy interventions. The case study presented provides a detailed account of some of the systemic feedbacks contributing to this phenomenon and does so in a geographical region largely underrepresented in the socio-hydrological literature. To address the growing reliance on water transports on Fårö, we suggest future policies to focus on a combination of information and economic policy interventions (e.g. demand correlated water tariffs) to incentivize reductions in water use, possibly in combination with a distributed system for rainwater collection, treatment and storage. These policies would contribute to consumer perceptions of the state of water resources being more aligned with their actual state, thereby reducing the risk of escalating water use. If these demand-side policies are effective, incomes from the water tariffs could help finance the investments needed for establishing a distributed water supply system, or other measures to increase the local water supply capacity, thereby reducing the need for further water transports.

We want to emphasize that the scope of this study, and the qualitative model developed, do not allow for detailed predictions to be made about what is the “optimal” suit of policies for the studied region. Detailed policy assessments and recommendations would require the development of a quantitative simulation model, which is the next step of this study. That said, we believe that with the anticipated effects of climate change, and the growing demand for water resources, many other regions worldwide are likely to face similar challenges as Fårö in the coming decades. We hope that the findings from this study can support water resources managers in these regions to anticipate the systemic impacts of their strategic choices, and help them account for human-water interactions in the assessment and planning of future water supply systems.

Supplementary Information The online version contains supplementary material available at https://doi.org/10.1007/s11269-021-02819-1.

Acknowledgments Thanks to representatives from the Department of Water Management at Region Gotland for their contributions in development and validation of the system map.

Authors’ Contributions All authors contributed to the study conception and design. Material preparation, data collection and analysis were performed by A. Nicolaidis Lindqvist and B. Kopainsky, R. Fornell and T. Prade. The first draft of the manuscript was written by A. Nicolaidis Lindqvist and all authors commented on previous versions of the manuscript. All authors read and approved the final manuscript.

Funding This research was partially funded by SMHI – Swedish Meteorological and Hydrological Institute, grant “Ansökan for utveckling av verktyg till stöd for samhällets klimatanpassningsarbete, 2019”.

Declarations

Conflict of Interest The authors declare no conflict of interest.

References

Ahopelto L, Veijalainen N, Guillaume JHA, Keskinen M, Marttunen M, Varis O (2019) Can there be water scarcity with abundance of water? Analyzing Water Stress during a Severe Drought in Finland. Sustainability 11:1548

Andersen DF, Richardson GP (1997) Scripts for group model building. Syst Dyn Rev 13:107–129. https://doi.org/10.1002/(sici)1099-1727(199722)13:2<107::Aid-sdr120>3.0.Co;2-7

Asp M et al. (2015) Framtidsklimat i Gotlands län - enligt RCP-scenarier vol 31. SMHI,

Bahaddin B, Mirchi A, Watkins D, Ahmad S, Rich E, Madani K (2018) System archetypes in water resource management. In: World Environmental and Water Resources Congress 2018. pp 130-140. https://doi.org/10.1061/9780784481400.012

Bardi U, Lavacchi A (2009) A Simple Interpretation of Hubbert's Model of Resource Exploitation Energies 2: 646–661

Beal C, Makki A, Stewart R (2014) What does reboundingwater use look like? An examination of post-drought and post-flood water end-use demand in Queensland, Australia. Water Sci Technol Water Supply 14:561–568. https://doi.org/10.2166/ws.2014.008

Bich-Ngoc N, Teller J (2018) A review of residential water consumption determinants. In pp:685–696. https://doi.org/10.1007/978-3-319-95174-4_52

Blair P, Buytaert W (2016) Socio-hydrological modelling: a review asking "why, what and how?". Hydrol Earth Syst Sci 20:443–478. https://doi.org/10.5194/hess-20-443-2016

Brunner F (2014) FÖP Fårö - Fördjupad översiktsplan 2025. Region Gotland

Di Baldassarre G, Viglione A, Carr G, Kuil L, Yan K, Brandimarte L, Blöschl G (2015) Debates—perspectives on socio-hydrology: capturing feedbacks between physical and social processes. Water Resour Res 51: 4770–4781. https://doi.org/10.1002/2014WR016416

Di Baldassarre G et al (2018) Water shortages worsened by reservoir effects. Nature Sustainability 1:617–622. https://doi.org/10.1038/s41893-018-0159-0

Di Baldassarre G et al (2019) Sociohydrology: scientific challenges in addressing the sustainable development goals. Water Resour Res 55:6327–6355. https://doi.org/10.1029/2018wr023901

Eklund F (2018) Regional vattenförsörjningsplan för Gotlands län. Visby

Formica S, Uysal M (2006) Destination attractiveness based on supply and demand evaluations: an analytical framework. J Travel Res 44:418–430. https://doi.org/10.1177/0047287506286714

Gabarda-Mallorquí A, Ribas Palom A (2016) Understanding reductions in water consumption in tourist areas: a case study of the Costa Brava, Spain. International Journal of Water Resources Development 32:912–930. https://doi.org/10.1080/07900627.2016.1142861

Gorre-Dale E (1992) The Dublin statement on water and sustainable development. Environ Conserv 19:181–181

Gotland R (2014) Fördjupad översiktsplan för Fårö:2025

Greve P et al (2018) Global assessment of water challenges under uncertainty in water scarcity projections. Nature Sustainability 1:486–494. https://doi.org/10.1038/s41893-018-0134-9

Kallis G (2010) Coevolution in water resource development: the vicious cycle of water supply and demand in Athens. Greece Ecol Econ 69:796–809. https://doi.org/10.1016/j.ecolecon.2008.07.025

Langarudi SP, Maxwell CM, Bai YN, Hanson A, Fernald A (2019) Does socioeconomic feedback matter for water models? Ecol Econ 159:35–45. https://doi.org/10.1016/j.ecolecon.2019.01.009

Markolf SA et al (2018) Interdependent infrastructure as linked social, ecological, and technological systems (SETSs) to address lock-in and enhance resilience. Earth's Future 6:1638–1659. https://doi.org/10.1029/2018ef000926

Martinez-Moyano I, Richardson G (2013) Best practices in system dynamics modeling. Syst Dyn Rev 29:29–123. https://doi.org/10.1002/sdr.1495

Mirchi A, Madani K, Watkins D, Ahmad S (2012) Synthesis of system dynamics tools for holistic conceptualization of water resources problems. Water Resour Manag 26:2421–2442. https://doi.org/10.1007/s11269-012-0024-2

Nations U (2018) SDG 6 synthesis report 2018 on water and sanitation. United Nations. https://doi.org/10.18356/e8fc060b-en

Pincetl S et al (2019) Adapting urban water systems to manage scarcity in the 21st century: the case of Los Angeles. Environ Manag 63:293–308. https://doi.org/10.1007/s00267-018-1118-2

Rivera P, Ridderstolpe, Djurberg (2011) VA-utredning för Fårö inför fördjupad översiktsplanering

Schwaninger M, Groesser S (2016) System dynamics modeling: validation for quality assurance. In pp 1–20. https://doi.org/10.1007/978-3-642-27737-5_540-3

Seto K, Davis S, Mitchell R, Stokes E, Unruh G, Ürge-Vorsatz D (2016) Carbon lock-in: types, causes, and policy implications. Annu Rev Environ Resour 41:41–452. https://doi.org/10.1146/annurev-environ-110615-085934

Sjöstrand Y, Kärrman S, Strömberg V (2014) Metod för val av vattenförsörjning i områden med vattenbrist – en handbok. SP Technical Research Institute of Sweden. Borås, Sweden

Sterman J (2000a) Business dynamics, system thinking and modeling for a complex world https://www.lst-iiepiiep-unescoorg/cgi-bin/wwwi32exe/[in=epidoc1in]/?t2000=013598/(100) 19

Sterman J (2000b) Business dynamics: systems thinking and modeling for a complex world. McGraw-Hill

Wimmer F et al (2015) Modelling the effects of cross-sectoral water allocation schemes in Europe. Clim Chang 128:229–244. https://doi.org/10.1007/s10584-014-1161-9

Xu L, Gober P, Wheater HS, Kajikawa Y (2018) Reframing socio-hydrological research to include a social science perspective. J Hydrol 563:76–83. https://doi.org/10.1016/j.jhydrol.2018.05.061

Affiliations

Andreas Nicolaidis Lindqvist[1,2] • Rickard Fornell[1] • Thomas Prade[2] • Linda Tufvesson[2] • Sammar Khalil[2] • Birgit Kopainsky[3]

[1] RISE Research Institutes of Sweden, Ideon Beta5, Scheelevägen 17, 22370 Lund, Sweden

[2] Department of Biosystems and Technology, Swedish University of Agricultural Sciences, POO Box 103, SE-230 53 Alnarp, Sweden

[3] Department of Geography, System Dynamics Group, University of Bergen, POO Box 7800, 5020 Bergen, Norway

Appendix 2

Supplementary material

Human-Water Dynamics and their Role for Seasonal Water Scarcity – a Case Study

Andreas Nicolaidis Lindqvist [1,2,*] , Rickard Fornell [1], Thomas Prade [2],
Linda Tufvesson [2], Sammar Khalil [2] & Birgit Kopainsky [3]

[1] RISE Research Institutes of Sweden, Ideon Beta5, Scheelevägen 17, 22370 Lund, Sweden

[2] Department of Biosystems and Technology, Swedish University of Agricultural Sciences, POO Box 103, SE-230 53 Alnarp, Sweden

[3] Department of Geography, System Dynamics Group, University of Bergen, POO Box 7800, 5020 Bergen, Norway

[*] Correspondence: andreas.nicolaidis@ri.se

Table SI1. Organizations and professional roles of study participants in the model development process.

Organization	Professional role	Number of participants
Region Gotland	Water resources developer	5
	Water utility engineer	2
	Water plant technician	1
	Environmental strategist	1
	Environmental resources inspector	1
Geological Survey of Sweden	Hydrogeologist	3
Gotland County Administrative Board	Water administrator	1

III

Journal of Hydrology: Regional Studies 41 (2022) 101066

Contents lists available at ScienceDirect

Journal of Hydrology: Regional Studies

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

Impacts of future climate on local water supply and demand – A socio-hydrological case study in the Nordic region

Andreas Nicolaidis Lindqvist [a,b,*], Rickard Fornell [a], Thomas Prade [b], Sammar Khalil [b], Linda Tufvesson [b], Birgit Kopainsky [c]

[a] RISE Research Institutes of Sweden, Ideon Beta5, Scheelevägen 17, 22370 Lund, Sweden
[b] Department of Biosystems and Technology, Swedish University of Agricultural Sciences, POO Box 103, SE-230 53 Alnarp, Sweden
[c] Department of Geography, System Dynamics Group, University of Bergen, POO Box 7800, 5020 Bergen, Norway

A R T I C L E I N F O

Keywords:
Climate change
Groundwater
Water scarcity
System Dynamics
Socioeconomic impact

A B S T R A C T

Study region: Fårö island, part of Region Gotland, Sweden.
Study focus: Despite its importance for proactive planning and management, understanding of how future climate and socioeconomic trends may interact to influence water supply and demand at sub-regional scale remains limited for the Nordic region. We aim to close this knowledge gap by developing a combined social and hydrological simulation model for Fårö island in the Baltic Sea. We use multivariate Monte Carlo simulations to explore the effects of future climate scenarios (RCP4.5 and RCP8.5) on local groundwater supplies, and subsequent impacts on the housing sector, tourism sector, and municipal water supply system in the period 2020–2050.
New hydrological insights for the region: Our results suggest that groundwater storage will remain critically low in the coming 30 years, with a 60–70% probability of the groundwater head falling to lower levels than experienced in the past 60 years. Low water availability and widespread saltwater intrusion will constrain housing and tourism development by up to 11% and 30% respectively. To sustain growth, the tourist sector will become increasingly reliant on water from private wells, and supplementary water deliveries from neighboring regions will be required to meet water demand on the municipal grid.

1. Introduction

Water scarcity is a problem with impacts for human health, economic development, and ecosystems in many regions around the world (Rijsberman, 2006; Wimmer et al., 2015). It is estimated that up to 50% of the global population will experience seasonal or permanent water insecurity by 2050, caused by a combination of changes in climate, urban and rural development, and population growth (United Nations, 2018). Understanding how trends in climate and socioeconomic development interact to influence water supply and demand across space and time is of great importance to support mitigation and adaptation to water scarcity (United Nations, 2018). Building relevant knowledge is challenging, however, as climate-driven impacts on water resources have been shown to differ substantially both between and within geographical regions (Bessah et al., 2020; Wu et al., 2020). Furthermore, studies in recent years have demonstrated that understanding the interplay between social and hydrological systems is an important component in long-term sustainable management of water resources (Di Baldassarre et al., 2019). Although much progress has been made in

* Corresponding author at: RISE Research Institutes of Sweden, Ideon Beta5, Scheelevägen 17, 22370 Lund, Sweden.
E-mail address: andreas.nicolaidis@ri.se (A. Nicolaidis Lindqvist).

https://doi.org/10.1016/j.ejrh.2022.101066
Received 20 December 2021; Received in revised form 9 March 2022; Accepted 22 March 2022
Available online 30 March 2022

developing macro-level theories about the mechanisms of socio-hydrological dynamics (Di Baldassarre et al., 2019; Sivapalan et al., 2012), place-specific understanding of human-water systems remains limited (Xu et al., 2018).

Most previous studies assessing the local interplay between climate, hydrology, and the social system have focused on regions with a long history of water scarcity, such as the Mediterranean region (Darvini and Memmola, 2020; Fabre et al., 2015), the Middle East (Gohari et al., 2013), Australia (van Emmerik et al., 2014), parts of the US (Fernald et al., 2012), and different parts of Africa (Bessah et al., 2020; Fraser et al., 2011). The Nordic region is poorly represented in such research, partly because freshwater has historically been a plentiful resource in the region. However, with the anticipated effects of climate change, there is reason to suspect that the Nordic region will not be spared from water scarcity for much longer. Indeed, unusually dry summers in recent years have caused periods of local to regional seasonal water scarcity, even in typically water-abundant areas (Ahopelto et al., 2019; Stensen et al., 2019). Moreover, recent reports project that the frequency, intensity, and duration of seasonal water shortages will continue to increase in the coming decades (Asp et al., 2015). To enable proactive and robust water management strategies to be developed for the Nordic region, improved local understanding of the combined effects of climate and socioeconomic change on water supply and demand is essential.

In this paper, we contribute to this end by presenting results from a case study exploring how climate and socioeconomic processes interact to influence supply and demand for drinking water on the Swedish island of Fårö. We develop a simulation model of the hydrological and socioeconomic mechanisms governing water supply and demand, drawing on a combination of qualitative and quantitative data sources. The model, calibrated to 20 years of historical data, is used in simulation experiments investigating how projected changes in climate are likely to influence water supply and water quality in the coming 30 years. We then explore the implications of these changes for the three largest water-dependent stakeholders on the island: the municipality, the tourism sector, and the housing sector.

The remainder of the paper is organized as follows: in Section 2, we briefly describe the study area and provide an account of how challenges related to drinking water supply and demand have developed in the past 20 years. We present the expected changes in regional climate in the coming 30 year, and we define the aim of this study. In Section 3, we outline the model development process and provide a high-level description of the model structure. We also present the model calibration process, and we describe the experimental set-up used to explore the hydrological and socioeconomic implications of future climate scenarios. In Section 4, calibration results are summarized and we present and discuss the results from the simulation experiments. We first describe climate effects on groundwater supply and then the impact of these effects on households, tourism, and the municipal water supply system. We also highlight some limitations of the study and consider areas for future research to support local water resources management and planning. In Section 5, we summarize our key findings.

2. The study area

Fårö is a small island (114 km^2) in the Baltic Sea (57.9°N, 19.1°E) belonging to the Swedish municipality of Gotland (Fig. 1). The island has an average summer and winter temperature of 16 °C and − 2 °C, respectively, and yearly precipitation of 500–600 mm (SMHI, retrieved 3 February, 2021). The main industries are tourism and agriculture, and the population consists of about 300 permanent households and 1000 holiday households (Lantmäteriet, 2021). Drinking water is obtained exclusively from groundwater sources, mostly from private wells drilled into the limestone-dominated bedrock. The soil layer is shallow (0–2 m) across most of the island except for an area in the northeast, where layers of coarse-grained sandy soil up to 20 m deep make up one of the few large aquifers in the region. This aquifer provides water to the municipal grid, which serves most tourist facilities and about 50 residential households (Brunner, 2014; Rivera and Ridderstolpe, 2011; Sjöstrand et al., 2014).

In recent decades, Fårö has become a popular holiday destination with a growing tourism sector (Fig. 2A). In the high season (June-August), about 10,000 tourists and part-time residents visit the island (Brunner, 2014). Water use has thus increased over time[1] (Fig. 2B). Between 2000 and 2020 water use on the municipal grid increased from about 5500–9500 cubic meters per year. In fact, since 2006 water supply from the municipal aquifer has been insufficient to meet demand in the summer period, requiring supplementary transport of water from other regions of Gotland to secure supply on the municipal grid[1] (Fig. 2C). In 2006 in total 1500 cubic meters of supplementary drinking water was transported to Fårö and by 2019 the figure had doubled to just above 3000 cubic meters. Within the private water sector, the number of holiday homes has also increased over time and increasing incidence of saltwater intrusion into private wells has been detected (Magnus Pettersson, Region Gotland, personal communication 25 January 2021). The growing reliance on transported water and the problem of saltwater intrusion create challenges for the municipality, the tourist sector, and private households. For the municipality, reliance on transported water is a risk as it makes the island vulnerable to disturbances, such as delivery delays, strikes, or unexpected peaks in water consumption. The municipal water supply has already come close to running out on several occasions, because of fluctuations in demand and delivery delays. For the tourist sector, water supply constraints can limit growth and development. In the past ten years, establishment of new tourist facilities has been delayed, or even canceled, because of insufficient water supplies (Rolf Lindvall, Sudersand Resort, personal communication 15 October 2020). For the housing sector, salt contamination of groundwater sources restricts new housing developments, as building permits are not issued in locations with elevated chloride levels (Gotlands Kommun, 2008). Further, if salt intrusion becomes widespread among households outside the public grid, the municipality may become legally required to extend its water management area and provide water services

[1] Monthly data on groundwater extraction, water supply capacity and water transport volumes are classified information and can therefore not be displayed in the paper (Mikael Tiouls, Region Gotland, personal communication 7 December 2021). For inquiries about the data please contact Region Gotland.

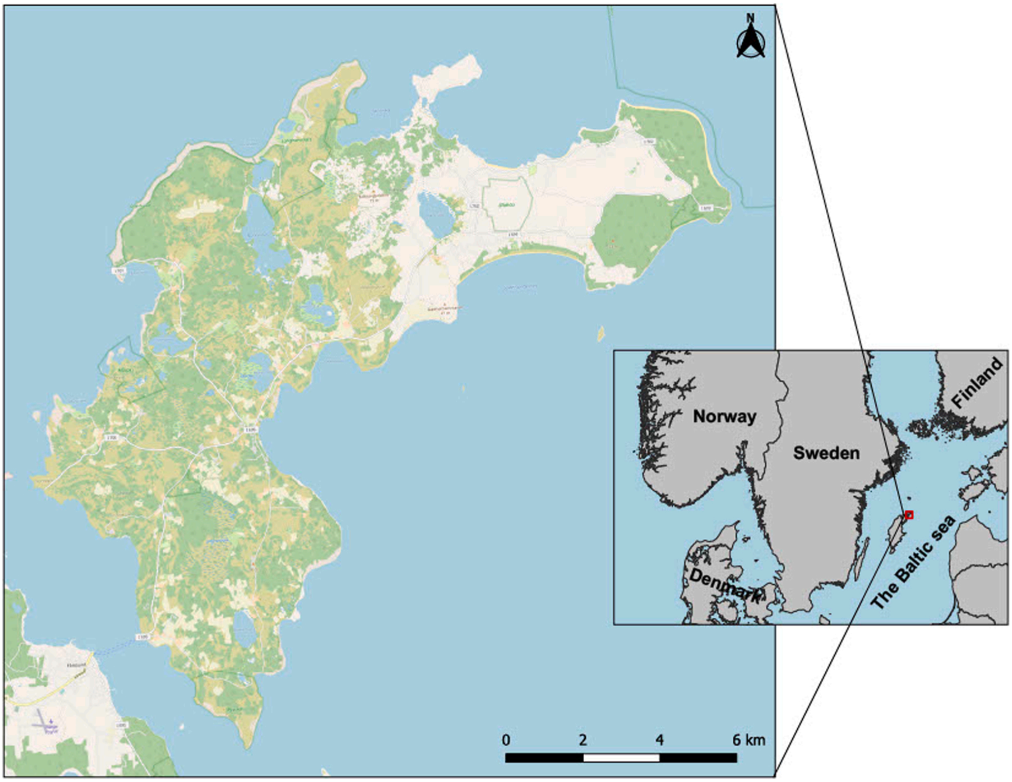


Fig. 1. Map of Fårö island. Location in the Baltic Sea indicated by red box in the small map of the Nordic region.
Source: Open Street Map & Eurostat, 19 November 2021.

to communities currently outside the public grid (Swedish Environment and Energy department, 2007). This could lead to a significant increase in demand for municipal water and could potentially require substantial investments in water transport or alternative water supply technologies.

2.1. Aim of study

Between 2020 and 2050, climate change is expected to increase regional mean temperature by approximately 1.0–1.3 °C and increase precipitation by 2–10% compared with the past 20-year period (Asp et al., 2015). In this study, we investigate how these changes are likely to influence local water supply and water quality, and explore the interplay with existing water supply challenges on Fårö island. The aim is to improve understanding of the local-level impacts of climate change and provide input for proactive water resources planning and management in a hitherto poorly studied region.

3. Material & methods

3.1. Model development

We develop a combined social and hydrological simulation model of the key mechanisms driving water supply and demand on Fårö in Stella Architect by ISEE Systems, Lebanon USA, following the System Dynamics modeling method (Pruyt, 2006; Sterman, 2000). The model consists of six interconnected submodules: *Climate, Public Water Supply, Private Water Supply, Household Water Use, Tourism Water Use,* and *Public Water Supply Demand Balance* (Fig. 3) and simulates from 2000 to 2050 at time units of one month. The causal structure of the model is based on a qualitative modeling study conducted by Lindqvist et al. (2021), exploring the drivers of water scarcity on Fårö. Additional scrutiny of the scientific literature and municipal reports, and repeated consultations and validation meetings with the Department of Water Management at Region Gotland throughout the modeling process, are used to cross-validate the structural and operational representation of the water management system in the model.

An overview of the structure, data inputs, and data outputs for each submodule is presented in Sections 3.1.1–3.1.6. For full model

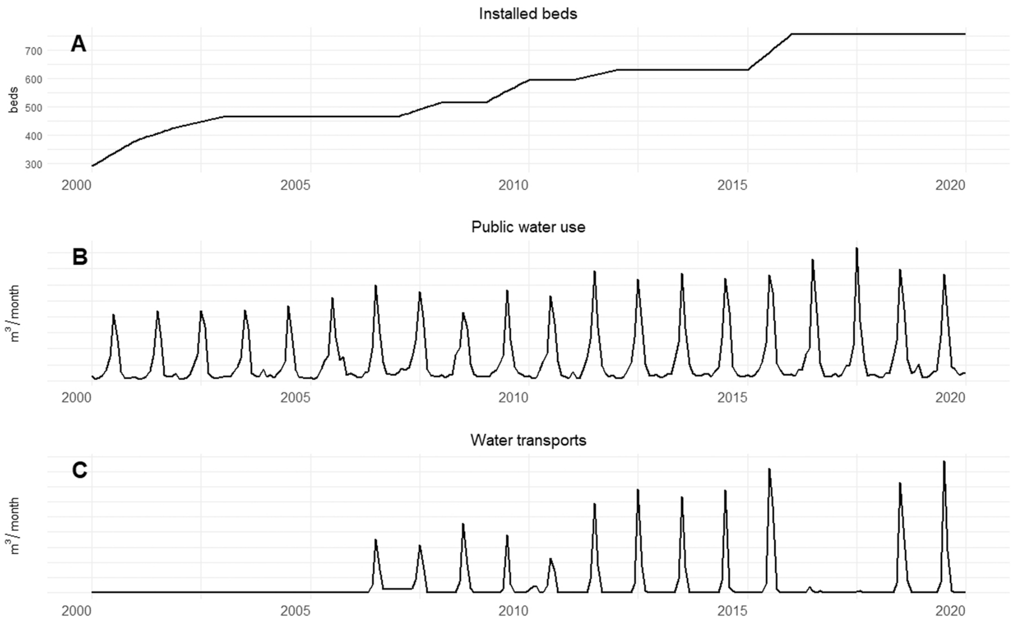


Fig. 2. Installed beds (A) (Region Gotland, 2021a), water use on the municipal grid (B) (Region Gotland, 2021b), and water transported to Fårö (C) (Region Gotland, 2021b) in the period 2000–2020. Water use and water transport volumes is classified information, scales intentionally left blank.

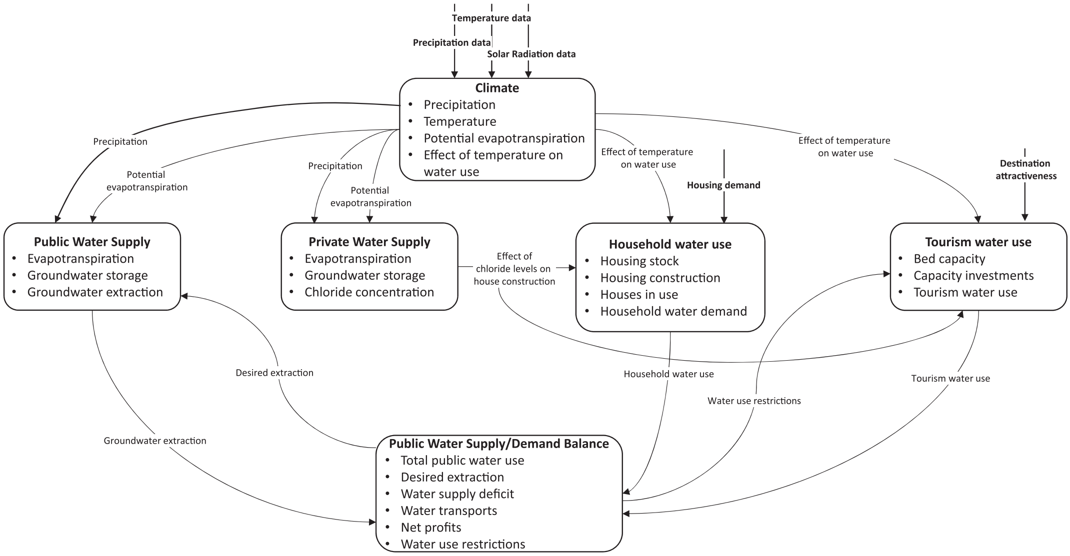


Fig. 3. Diagram of the full model. Boxes represent the six submodules with their key processes, and stock variables indicated. Arrows represent exogenous data inputs (bold) and information exchange between modules.

documentation, see the appendix A.

3.1.1. Climate

The Climate module (Fig. 4) imports monthly data on temperature, precipitation, solar radiation, and wind speed, and calculates *potential evapotranspiration* (*PET*) using the Penman-Monteith method (Penman and Keen, 1948). The *effect of temperature on per capita water use* is also calculated, assuming that water use increases by 2% for every °C by which the daily maximum temperature of the month exceeds 15 °C (Dimkić, 2020). Outputs to other submodules are *precipitation*, *PET*, and *effect of temperature on per capita water use*.

For the historical period 2000–2020, we use observation data from local weather stations (SMHI, 2021) as climate inputs. For the future period 2020–2050, we used projected values provided by SMHI (Asp et al., 2015). These projections were produced by the regional climate model RCA4 (Strandberg et al., 2014), by downscaling and averaging across an ensemble of climate scenarios produced by nine global climate models (CanESM2, CNRM-CM5, GFDL-ESM2M, EC-EARTH, IPSL-CM5A-MR, MIROC5, MPI-EMS-LR, NorESMI1-M and HadGEM2-ES) (Sjökvist et al., 2015) for the two Representative Concentration Pathways RCP4.5 and RCP8.5 (IPCC, 2013) (see appendix A for details). To simulate between-year variations in future climate inputs, we sample future precipitation, temperature, and solar radiation values from their respective probability density functions, parameterized using the SMHI projected mean and historic standard deviation values.

3.1.2. Public Water Supply

The Public Water Supply module (Fig. 5) simulates the dynamics of the public water supply system, including the hydrology of the municipal aquifer and groundwater pumping in municipal wells. The aquifer is modeled using Budyko-based methods for water balance modeling as described by Zhang et al. (2008), adapted to meet the requirement of unit consistency under System Dynamics modeling conventions (Sterman, 2000).

In the model, the aquifer consists of two connected cylindrical stocks, the top representing *soil storage* and the bottom representing *groundwater storage*. The dynamics of the stocks are governed by the flows of *infiltration*, *evapotranspiration*, *recharge*, *deep evapotranspiration*, *base flow*, *horizontal groundwater flow*, and *extraction. Infiltration* is calculated as a nonlinear function of the demand/supply relationship between the level of the *soil storage* stock (demand) and the incoming *precipitation* (supply) (Zhang et al., 2008). As the saturation level of the *soil storage* stock decreases, the proportion of *precipitation* partitioned to *infiltration* asymptotically approaches one. The shape of the partitioning curve is governed by the *rainfall retention capacity* of the catchment, a model parameter representing the physical capacity of the soil and vegetation of the aquifer to retain water (Zhang et al., 2008). The outflow from *soil storage* is partitioned between *evapotranspiration* and *recharge* according to similar functions as applied for *infiltration*. *Evapotranspiration* is calculated as a function of the relationship between *soil storage* and *PET*, and *recharge* as function of the relationship between *soil storage* and the *storage capacity of the aquifer.* As *soil storage* increases, *recharge* will also increase and *evapotranspiration* will approach *PET.* The relative proportion of available water that goes to *recharge* or *evapotranspiration* is controlled by the *evapotranspiration efficiency* of the catchment. Higher *evapotranspiration efficiency* means that more available water goes to *evapotranspiration* and less to *recharge* (Zhang et al., 2008). The *deep evapotranspiration* flow ensures that if the *groundwater level* approaches the shallow soil layer, groundwater also becomes available for *evapotranspiration* (Yeh and Famiglietti, 2009).

The groundwater *base flow* is modeled as the product of the *groundwater storage* and a constant discharge factor, and the *extraction* flow represents groundwater extraction by pumping by municipal wells. *Extraction* is set equal to *desired extraction* (calculated in the *Public Water Supply Demand Balance* module) if the *groundwater level* in the aquifer remains above the average depth in the municipal wells. As the *groundwater level* approaches the depth in the municipal wells*, extraction* declines linearly. Lastly, the *horizontal groundwater flow* represents the exchange of water between the municipal aquifer and its surroundings. This allows *groundwater storage* to adjust to the groundwater level in surrounding catchments, and is calculated using Darcy's flow equation (Hillel, 2004).

3.1.3. Private Water Supply

The Private Water Supply module (Fig. 6) simulates the groundwater dynamics and the chloride concentration in aquifers outside the public water system. The water balance structure is similar to, and uses the same climate inputs, as that presented in Section 3.1.2, but it only accounts for natural water fluxes (*infiltration, evapotranspiration, recharge, deep evapotranspiration* and *baseflow)*. *Extraction* is

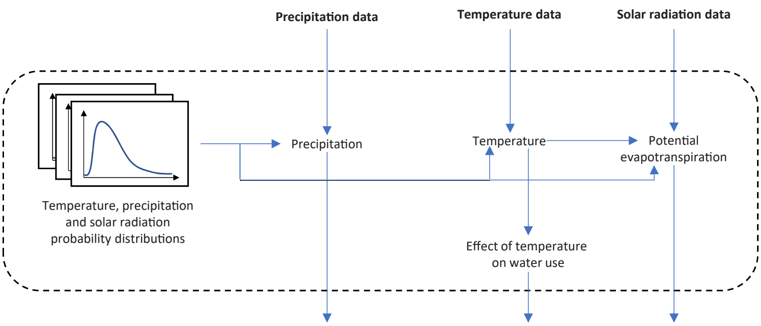


Fig. 4. Graphical representation of the Climate module. The dashed box represents the module boundary. Blue arrows represent information flows, variables in bold are exogenous data inputs, variables in normal font are endogenously calculated, and boxes with distribution curves represent the probability distribution functions used in simulation of inter-annual variations in climate inputs.

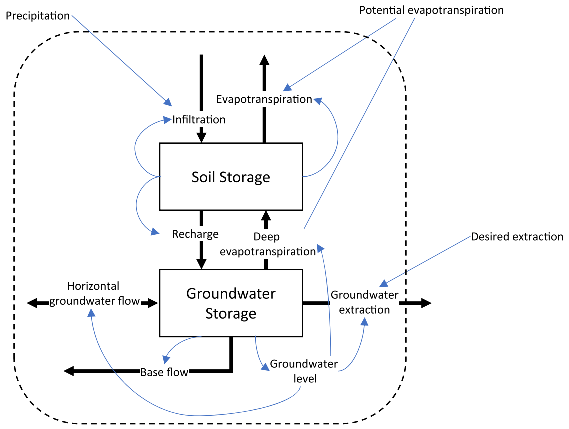


Fig. 5. Graphical representation of the Public Water Supply module. Boxes are stock variables, representing accumulation and storage of water in the shallow soil (*Soil Storage*) and deep groundwater layer of the aquifer (*Groundwater Storage*). Black arrows represent the flow of water within the aquifer and exchange of water between the aquifer and the surroundings. *Precipitation* and *potential evapotranspiration* are inputs from the Climate module, *desired extraction* is an input from the Public Water Supply/Demand Balance module, and *extraction* is an output to the Public Water Supply/ Demand Balance module.

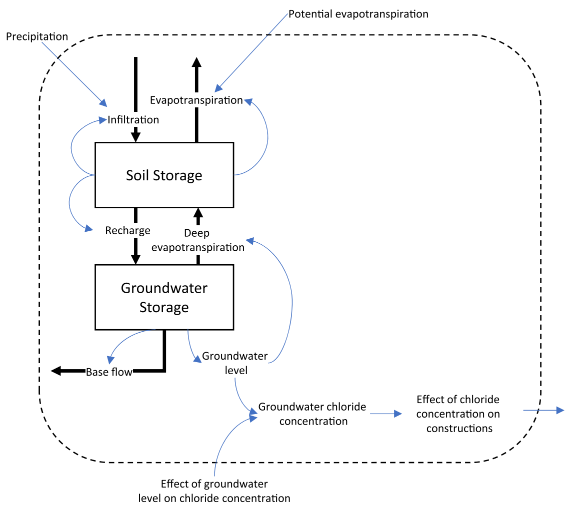


Fig. 6. Graphical representation of the Private Water Supply module. *Precipitation* and *potential evapotranspiration* are inputs from the Climate module, *effect of groundwater level on chloride concentration* is an exogenous constant, and *effect of chloride concentration on constructions* is an output to the Household Water Use module**.**

deliberately excluded because of lack of reliable data on historical extraction rates, and because water quality (measured by chloride concentration), rather than water quantity, has historically been the determining factor of household water supply (Magnus Pettersson, Region Gotland personal communication 25 October 2021). *Horizontal groundwater flow* is also excluded from the private water supply module, assuming homogeneous groundwater levels across the island.

Groundwater chloride concentration represents the average chloride level in groundwater across the island and is calculated as a linear function of the groundwater level using the basicTrendline package (Mei and Yu, 2020) in RStudio (R Development Core Team, 2019). The linear model was calibrated using five years of groundwater level data and data on chloride levels in 328 water samples from across the island, yielding a statistically significant negative *effect of groundwater level on chloride concentration* ($P < 0.01$). The *groundwater chloride concentration* is used to estimate the proportion of well sites with chloride levels exceeding the recommended limit values set by the Swedish Food Agency (2017) (100 and 300 mg/L chloride (Cl)) as limit values for technical and drinking use, respectively) and the maximum permissible chloride concentration when granting building permits (100 mg/L Cl). The proportion of well sites exceeding the limiting value at any given chloride level is obtained by linear (100 mg/L Cl) and nonlinear (300 mg/L Cl) regression (Mei and Yu, 2020), with the *groundwater chloride concentration* as the independent variable, and the fraction of samples above 100 mg/L Cl ($P < 0.05$) and 300 mg/L Cl ($P < 0.01$) as dependent variables. The *effect of chloride concentration on house construction* represents the limiting effect of groundwater chloride levels on new construction in the housing and tourism sector. If *groundwater chloride concentration* increases, fewer building permits will be issued and construction rates will decline.

3.1.4. Household Water Use

In the Household Water Use module (Fig. 7), the size of the household sector and total *household water use* is calculated. The total *housing stock* is divided into two groups (k) based on the water source (public water or private well), and each group is further segmented into two types (j) based on utilization (permanent or part-time). New houses are added to the system by the flow *construction starts*, which adds to the stock of *houses under construction.* The rate of *construction starts* is influenced positively by higher *housing prices* and negatively by the *effect of chloride concentration on house construction*. After a 12-month time delay (representing the house construction time), houses flow from the *houses under construction* stock to the *housing stock* via the *construction finalization* flow.

Construction starts is governed by a simple supply-demand structure where a stock of *potential buyers* is compared to the number of houses finalized each month to give a demand-supply ratio that is used to set the *housing price*. The base rate at which new buyers are added to the *potential buyers* stock is set using exogenous data on *housing demand* (based on data on building permits issued between 2000 and 2020, (SCB, 2021), and the stock is drained at the rate of *construction finalization*. The *housing price* is used as an indicator of *household affluence* (Englund, 2011), and it influences the quantity demand for houses by regulating the flow of new *buyers added* to the *potential buyers* stock and the number of *construction starts.* We use a price elasticity of demand and supply of − 0.5 (Englund, 2011) and 0.1 (International Monetary Fund. European Dept, 2015), respectively.

The number of *houses in use* at any given point in time depends on the household type and the duration of the tourist season. Permanent households are in constant use, while the proportion of part-time households in use is determined by multiplying the number of part-time households by a *tourist season* utilization factor. This factor takes values between zero and one depending on the time of the year (here based on estimates by Region Gotland). *Water use per household* is the product of the number of *residents per household* and *normal water use per capita* (140 L/person/day, (Swedish Water, 2020)), and responds dynamically to changes in temperature through the *effect of temperature on water use* from the Climate module (Dimkić, 2020), and the level of *household affluence* (Höglund, 1999; Wiedmann et al., 2020). The total *household water use* is the product of the number of *houses in use* and the *water use per household,* and provides input to the Public Water Supply Demand Balance module.

3.1.5. Tourism Water Use

The Tourism Water Use module (Fig. 8) simulates the development of the tourist sector and its total water use. The size of the tourist sector is measured by its bed capacity and it is modeled by a three-compartment aging chain (Sterman, 2000) consisting of the stocks

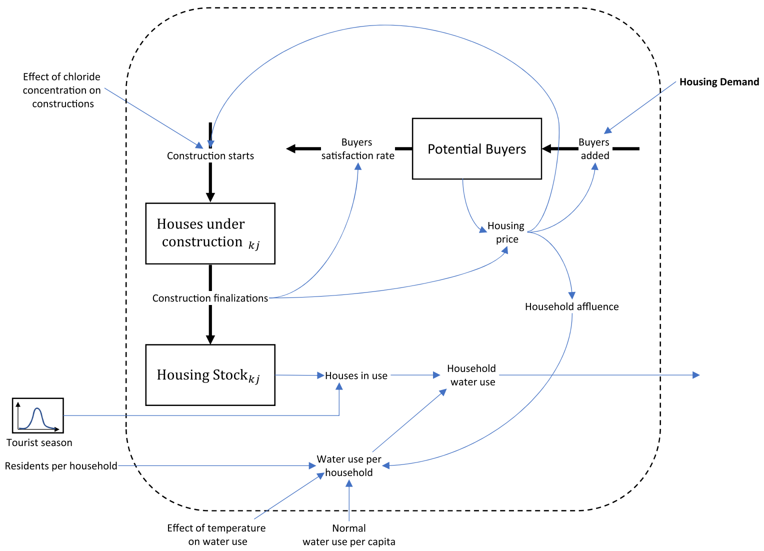


Fig. 7. Graphical representation of the Household Water Use module. *Housing demand* is an exogenous data input representing the demand for housing. *Effect of chloride concentration on construction* and *effect of temperature on water use* are inputs from the Private Water Supply module and the Climate module, respectively. *Residents per household* and *normal water use per capita* are model constants, and *tourist season* is a lookup function adjusting the number of *houses in use* according to the duration of the tourist season. *Household water use* is an output to the Public Water Supply Demand Balance module.

beds planned, *beds on order*, and installed *beds*, linked by the flows *adding beds to plan*, *ordering new beds*, and *installing new beds*. The number of *tourists* is modeled as an additional stock that increases or decreases with the flow of *net arrivals*. *Net arrivals* fluctuate with the tourist season using the same *tourist season* utilization factor as in Section 3.1.4. and respond to changes in *destination attractiveness* (assumed constant in the base case scenario). The number of beds added to the system each month is controlled by a goal-gap function where the level of *capacity utilization* (the ratio of *tourists* to installed *beds*) is compared to a *desired capacity utilization*. If *capacity utilization* exceeds *desired capacity utilization*, this leads to an increase in *desired expansion* and an inflow to *beds planned*. Beds move in batches from *beds planned*, through *beds on order,* to installed *beds* with a total planning and construction delay of 24 months. Bed capacity investments are bounded by the water self-sufficiency of the public grid and the possibility of tourist facilities to drill their own wells. If public water self-sufficiency is low, *water use restrictions* (calculated in the Public Water Supply Demand Balance module, Section 3.1.6.) will limit planning and investment in new bed capacity. This will force tourist facilities to search for private water supply sources by drilling new wells, making the *groundwater chloride concentration* the limiting factor for tourism growth.

Total tourism water use is calculated as the product of *normal water use per tourist,* the number of *tourists,* and the *effect of temperature on water use* imported from the Climate module, and it provides input to the Public Water Supply Demand module.

3.1.6. Public Water Supply Demand Balance

The Public Water Supply Demand Balance module (Fig. 9) sums up the total water use from the tourist sector with the total public water use in the household sector to calculate the *total public water use*. The *total public water use* dictates the *desired pumping* from the public aquifer (in the Public Water Supply module, Section 3.1.2.) and, when multiplied by the consumer *water price*, the *municipal revenues from water tariffs*. *Desired extraction* is compared to the actual *groundwater extraction* from the Public Water Supply module, to calculate a *water supply deficit*. The deficit triggers *water use restrictions* that limit further expansion of the tourist sector (Section 3.1.5.) and increases the volume and costs of *water transports.* The difference between revenues and costs of the water supply system is used as an estimate of the *net profits* of the municipal drinking water system.

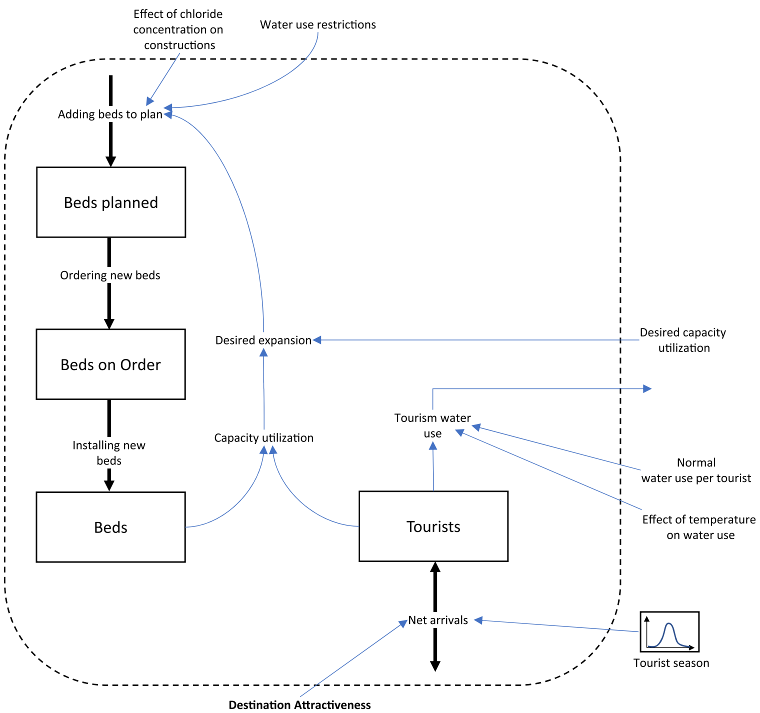


Fig. 8. Graphical representation of the Tourism Water Use module. *Destination attractiveness* is an exogenous data input. *Effect of chloride concentration on construction, water use restrictions*, and *effect of temperature on water use* are inputs from the Private Water Supply module, the Public Water Supply Demand Balance module, and the Climate module, respectively. *Desired capacity utilization* and *normal water use per tourist* are model constants, and *tourist season* is a lookup function adjusting the number of *net arrivals* of tourists according to the duration of the tourist season. *Tourism water use* is an output to the Public Water Supply Demand Balance module.

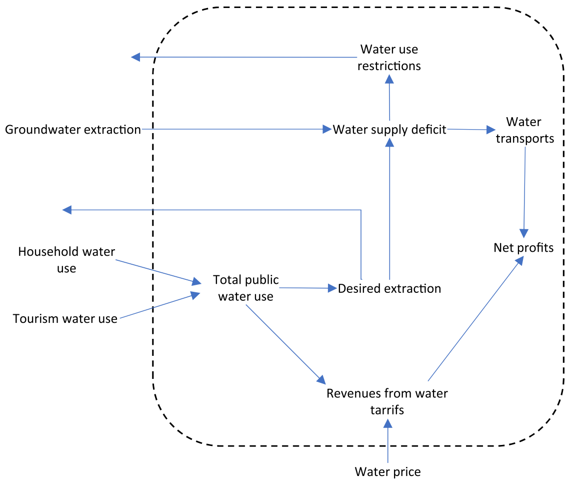


Fig. 9. Graphical representation of the Public Water Supply Demand Balance module. *Groundwater extraction, household water use,* and *tourism water use* are inputs from the Public Water Supply, Household Water Use, and Tourism Water Use modules, respectively. *Water price* is a model constant, and *desired extraction* and *water use restrictions* are outputs to the Public Water Supply, Household Water Use, and Tourism Water Use modules.

3.2. Model calibration

Calibration is conducted by varying module parameter inputs to optimize the fit of the simulation outputs to historical data on groundwater levels, water use, tourism and housing development, provided by Swedish Metrological and Hydrological Institute (SMHI, 2021), Geological Survey of Sweden (SGU, 2021a, 2021b), Statistics Sweden (SCB, 2021), and Region Gotland (Region Gotland, 2021a, 2021b). Each submodule is first calibrated individually, adhering to strategies for partial model calibration described by Homer (2012), followed by a final round of full model calibration and evaluation to ensure consistency with historical data is maintained with the complete set of between-module feedbacks active. Parameter estimates are selected based on literature studies, expert opinions, local empirical data or, if beforementioned information sources are not available, best estimates by the modelers. Parameters with high uncertainty regarding their true values, and with high impact on simulation results, were numerically estimated using Powell optimization (Powell, 2009). The squared error between simulated and observed timeseries is used as payoff function in the parameter estimation process, and results are evaluated quantitatively, using Theil Inequality Coefficients (Sterman, 1984), and qualitatively by comparing the derived parameter estimates with ranges suggested in the literature and by local experts. For a complete list of calibration inputs and outputs see appendix B.

Due to lack of data on historical groundwater levels in the municipal aquifer, a two-step procedure is used for groundwater calibration. First, the aquifer structure presented in Section 3.1.2 is calibrated to 25 years of data (1971–1996) on historical groundwater levels from an aquifer in southern Sweden (SGU, 2020) that has similar geological and landcover characteristics as those found on Fårö. This step ensures that the structure can replicate the general groundwater dynamics of the aquifer. In a second round of calibration, the pre-calibrated structure is finetuned to represent the municipal aquifer on Fårö by optimizing its fit to available data on municipal groundwater extraction between 2000 and 2020. Calibration results are presented in Section 4.1.

3.3. Experimental set-up

For future climate, two climate scenarios are considered, Representative Concentration Pathway (RCP) 4.5 and 8.5 (IPCC, 2013). In the RCP4.5 scenario we assume, depending on season, a 2–11% increase in monthly mean precipitation and a 0.96–1.24 °C increase in monthly mean temperature between 2020 and 2050. In the RCP8.5 scenario, the corresponding values are a 5–11% increase in precipitation and a 1.6–1.32 °C increase in temperature (Asp et al., 2015). For future solar radiation we use monthly averages from the past 12 years (SMHI, 2021), and for future mean monthly wind speed we use historic averages (Alexandersson, 2006). We assume new house constructions will continue to occur primarily outside the public water grid, and in the tourist sector we expect the growth in demand for hotels and other tourist facilities to continue at the same rate as seen in the past 20 years. Lastly, we assume that water transports will continue to be the main municipal strategy to cover seasonal peaks in water demand.

To handle the uncertainty embedded in long-term policy and strategy planning, it is necessary to explore a wide ensemble of plausible futures and let the full outcome space inform the decision-making process (Bankes, 1993). To this end, we carry out multivariate Monte-Carlo (MC) simulations, varying model parameters governing future climate (precipitation and temperature), housing supply and demand (future demand for houses, price elasticity of demand, price elasticity of supply, and price sensitivity of the ratio between demand and supply), groundwater chloride levels (effect of groundwater level on chloride concentration), the effect of chloride levels on tourism and house expansion (effect of chloride concentration on house construction), and per capita water use (affluence effect on water use). The parameters included in the MC analysis, and their associated ranges, are selected based on extensive partial model sensitivity testing, and the availability of reliable empirical or literature-based estimates. In other words, parameters that show a significant effect on simulation outputs and a high uncertainty with regards to their true values were included in the MC analysis (see appendix C for a full list of parameters, distributions, and ranges chosen).

We simulate the model 1000 times with randomly selected parameter values taken from predefined probability distributions within specified ranges. Results are reported as outcome ranges, bounded by the 95% confidence intervals, and the mean of the 1000 simulations is used to study long-term trends in groundwater levels using the seasonally adjusted Mann-Kendall trend test (McLeod, 2011). To assess the effect of future climate on groundwater levels, we compare the results from our MC simulations to the simulated groundwater regimes in two reference periods (P1, P2). P1 is the period 1961–1990, a commonly used reference in climate impact assessments (Asp et al., 2015; Sjökvist et al., 2015). P2 is the more recent period 2000–2020. Comparison against Pl gives a long-term perspective of the groundwater regime in the future and makes our results comparable to those of other studies, but it is less relevant for planning and policy purposes. P2 is the period during which water scarcity has developed into a problem on Fårö and is therefore a more relevant reference for policy makers when assessing the impacts of future groundwater levels.

Additionally, for planning and management purposes, an indication of variations and the risk of extreme events is equally, if not more, important than the average trajectory suggested by the MC ensemble (McCollum et al., 2020). We therefore calculate the probability of future extreme groundwater drawdowns, defined as the fraction of simulated scenarios where the groundwater head

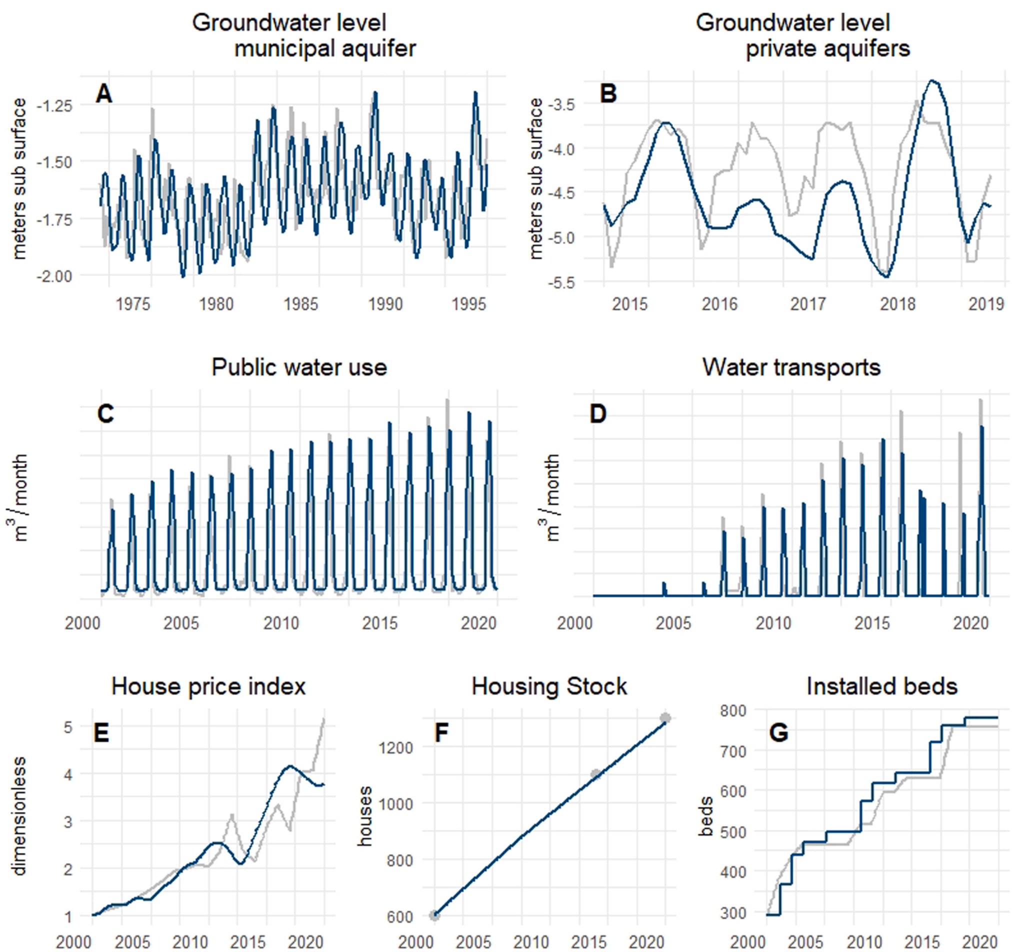


Fig. 10. Observed (grey) and simulated (blue) values for historic groundwater levels (A & B), public water use (C), municipal water transports (D), house price index (E), housing stock (F), and installed beds (G) in 2000–2020. Public water use and historic water transports is classified information and scales have intentionally been left blank.

reaches a level more extreme than the 2.5th percentile of its range in P1 and P2, or more extreme than the lowest groundwater level experienced in either P1 or P2.

To assess the implications of changes in future groundwater quality and availability on the housing and tourism sectors, we compare the results from the MC analysis with a simulated scenario where housing and tourism development is not constrained by water availability. In other words, we simulate a scenario where the growth of the two sectors is allowed to reach its full potential, and we use this as our baseline for assessing the impact of water scarcity on socioeconomic development in the region.

4. Results and discussion

4.1. Calibration results

Our calibration results show an overall acceptable fit to available hydrological and socioeconomic data (Fig. 10). Most importantly, the model captures the general trends of increasing summer water use and growing reliance on water transports. Furthermore, the mean absolute error (MAE) of the groundwater simulations is low (12 and 48 cm for the municipal and private aquifers respectively), with most of the error caused by unequal covariance between the observed and simulated timeseries (98% and 56% for the municipal and private aquifers respectively). This indicates a low level of systematic error and provides confidence that the model is capable of replicating the dominating behavior trends in the hydrological system (Sterman, 1984). Summary statistics for the calibration results are presented in Table 1.

4.2. Effects of future climate on groundwater storage and groundwater quality

The mean of our MC ensemble shows that groundwater levels in both public and private aquifers on Fårö are likely to remain within the range seen in the previous 20-year period (P2) (Fig. 11). A slight, but statistically significant ($P < 0.05$), increasing trend in groundwater storage in both aquifer types can be seen from 2030. These trends aside, compared to the reference period P1, the projected groundwater levels remain critically low, suggesting continuation of the decades-long regime of low groundwater storage. These results are in line with findings in monitoring studies conducted by SGU that most aquifers on Gotland have been at historically low levels for most of the time since the turn of the millennium (SGU, 2021a). Therefore, the slight increase in groundwater storage suggested by our simulations is from a historically low level and should not be interpreted as a return to some long-term historical normal.

Both aquifer types show substantial variation in groundwater levels between the upper and lower bound of the simulated outcome space (Fig. 11). The difference between the higher and lower confidence interval is up to 90 cm in the public aquifer and about 180 cm in private aquifers. It is important to acknowledge that the confidence bounds do not represent individual scenarios from the MC analysis. Rather, they mark extreme values taken by any of the 1000 independent simulations, and should therefore be interpreted as plausible ranges within which groundwater levels are likely to fluctuate in the coming 30 years. Analyzing the extremes of the outcome space makes this clear (Table 2). Between 2020 and 2050, a groundwater level more extreme than the lowest level ever experienced since 1961 occurs at least once in between 60% and 70% of the simulated scenarios. On average, such an extreme month occurs 14 times for the public aquifer and four times for private aquifers in the 30-year period. Months with groundwater levels lower than the 95% range of P1 and P2 occur at least once in more than 80% of the scenarios, or on average 211 and 36 times for the public aquifer, and 31 and 10 times for private aquifers.

Like the groundwater level, the ensemble mean suggests no significant change in chloride concentrations compared with the P2 period. However, because of the high probability of recurring periods with low groundwater levels it is likely that the number of households experiencing occasional water quality issues will increase in the coming decades. Likewise, between-year variation in groundwater chloride (SD = 45.9 mg/L Cl) can result in some locations, in years with high groundwater levels, shifting from being just above to just below the building permit threshold (100 mg/L Cl), and thereby increase the potential for new housing projects. The low spatial resolution of available data does not allow us to identify in what locations on Fårö large fluctuations in chloride levels are most likely. However, previous studies by Dahlqvist et al. (2015) have shown that there are substantial geographical variations in chloride

Table 1
Summary statistics from model calibration results. MAE = mean absolute error, MSE = mean square error, RMSE = root-mean square error. See Sterman (1984) for a detailed description of the summary statistics components. See appendix B for the underlying input and observation data used for the calibration.

						Error decomposition		
Payoff variable	MAE	MSE	RMSE	Correlation	R^2	Bias	Variation	Covariation
Public water use (Cubic meters/month)	354	343 k	585	0.859	0.737	0.184	0.155	0.661
Water transports (Cubic meters/month)	110	85.2 k	292	0.592	0.351	0.004	0.009	0.987
Beds (beds)	29.4	1.48 k	38.5	0.976	0.953	0.221	0.194	0.585
Groundwater level in the municipal aquifer (meters below ground)	0.118	0.020	0.143	0.647	0.419	0.0004	0.022	0.977
Groundwater level in private aquifers (meters below ground)	0.483	0.315	0.561	0.665	0.443	0.434	0.003	0.563
House price index (dimensionless)	0.301	0.192	0.438	0.903	0.815	0.015	0.013	0.972

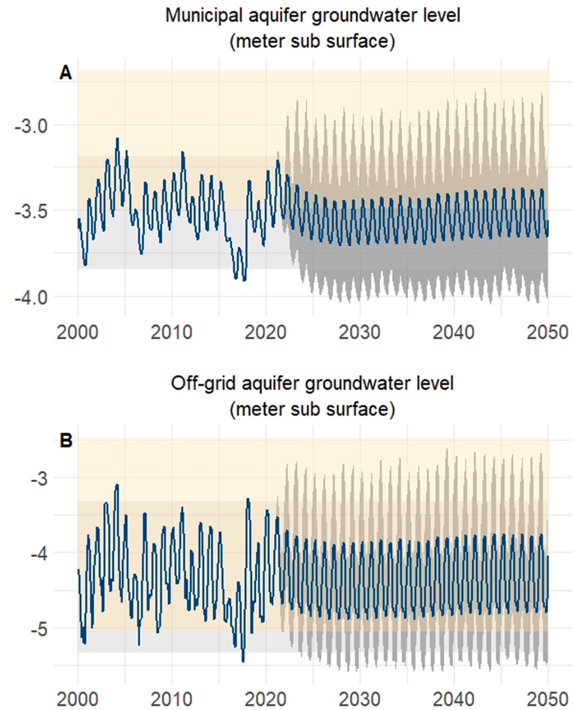


Fig. 11. Simulated groundwater level in the municipal and private aquifers on Fårö (panel A and B respectively). Blue lines are mean groundwater levels of the simulated ensemble, shaded areas represent the 95% confidence intervals, and the yellow and grey bands indicate the normal groundwater range (mean level +/- two standard deviations) for reference period P1 (1961–1990) and P2 (2000–2020), respectively.

Table 2
Frequencies and probability of extreme groundwater levels in the MC ensemble. The frequency columns represent how many times the groundwater table reaches a level equally or more extreme than the lowest level since 1961, or the 2.5th percentile in reference period P1 (1960–1990) and P2 (2000–2020). The probability column shows the probability of a new extreme low occurring at least once between 2020 and 2050.

	Frequency of a new extreme low	Probability of a new extreme low	Frequency of a < 2.5th percentile scenario (P1)	Frequency of a < 2.5th percentile scenario (P2)
Public aquifer	14.7	73.0%	211	31.6
Private aquifers	4.5	59.3%	36.1	10.0

base levels across the island. Accounting for both the spatial and temporal variability in groundwater chloride concentration when issuing new building permits is important to avoid an accumulation of houses in risk zones during periods when chloride levels are low. To mitigate this risk, further studies exploring spatial variation in chloride responsiveness to groundwater fluctuations are needed, so that locations with acceptable and stable groundwater quality can be identified for future building projects.

Table 3
Housing stock, tourist bed capacity and yearly water transports in 2020, and their estimated values in 2050 for the lower bound, mean, and upper bound of the simulated outcome space.

	2020	2050		
	Observed	Lower bound	MC mean	Upper bound
Houses	1300	2300	2650	3100
Tourist beds	800	1050	1220	1340
Water transports (m^3/year)	< 3000	2200	4000	5400

4.3. Socioeconomic impacts

4.3.1. Impacts on the housing sector

Our MC ensemble mean suggests that by 2050, the total number of households on Fårö will be between 2300 and 3100, compared to about 1300 in 2020 (Table 3). Most of the variation arises from uncertainty about future housing demand and about the strength of influence that groundwater chloride levels have on housing construction rates. On average, between 40% and 50% of well sites will have chloride levels exceeding 100 mg/L in the coming decades, but at the extreme of the simulated outcome space, that is during periods of severely low groundwater levels (as described in Section 4.1), the proportion can be as high as 75% for parts of the island. Detailed assessment of the impacts this would have on the housing sector requires further investigation of what areas that are attractive for housing development and how these areas correlate with risk zones for high chloride levels. In lack of this type of detailed spatial information, we make the simplifying assumption that housing development projects are homogenously distributed across the island. If this holds true, elevated chloride levels will pose a constraint for future housing development, increase housing prices, and reduce the number of households in 2050 by 4–11% compared with the unconstrained scenario where chloride levels have no effect on housing development.

For a region like Fårö, a 4–11% reduction in housing development is significant. For many years, RG has been striving towards increasing the number of permanent residents on the island through initiatives to enhance the availability of affordable housing. Despite these initiatives, reports by RG suggest that the high demand for summer houses, primarily by financially strong consumers from other regions of Sweden, have contributed to driving up house prices beyond what is affordable for the majority of the local community (Brunner, 2014). This effect has been confirmed by previous studies, showing how tourism intensification can lead to increase in local house prices (Paramati and Roca, 2019) and limit the availability of affordable housing for the local community (Mikulić et al., 2021). Our results suggest that future constraints in water availability could enhance these effects as the decline in housing availability that this would cause could contribute to further escalation of house prices.

4.3.2. Impacts on the tourist sector

The tourist sector is expected to grow from about 800 beds in 2020 to between 1000 and 1300 beds by 2050 (Table 3). The rate of growth is constrained by sustained low water self-sufficiency on the public grid, causing current restrictions on new connections to be maintained (see section 4.2.3.). This restraint leads to the establishment of a growing number of tourist accommodation sites relying on water from private wells instead of the municipal grid. The growth rate of these off-grid facilities experiences the same water quality constraints as the housing sector (described in section 4.2.1). Controlling for other factors, water supply limitations cause a 10–30% reduction in tourism growth compared with the unconstrained scenario.

As a whole, the proportion of tourist facilities relying on the municipal water system declines but, counterintuitively, in absolute terms the tourist sector demand for water from the municipal grid continues to increase. This is due to a significant share of tourism water consumption resulting from activities not associated with accommodation. For example, tourism water use arising in restaurant kitchens, spas and laundry facilities accumulates to on average 10–30 liters per guest night according to studies by Gossling et al. (2012). On Fårö, these facilities typically are connected to the public water grid and therefore continues to tax the public water system despite the accommodation facilities having their own wells. These spillover effects will cause an increase in the absolute municipal water use by the tourist sector, despite a growing number of tourist facilities having their own water supply. We argue that this is a challenge that is not unique to our case study. Introducing alternative water supply solutions (e.g. private wells) on top of an already existing centralized water supply system (e.g. the public grid) is likely to increase water use in the centralized system in the long run if it leads to an increase in the total number of consumers, and the water use of the new consumers is not confined to their private taps.

4.3.3. Impacts on the municipal water system

Persistence of the low groundwater regime experienced in the past 20 years will continue to limit groundwater extraction from the

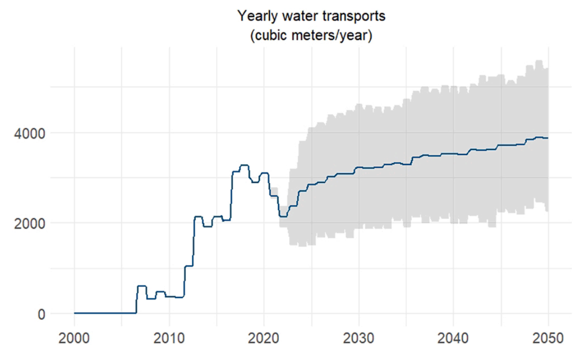


Fig. 12. Yearly municipal water transports. The blue line is the simulated mean and the shaded area represents the 95% confidence interval of the MC ensemble.

municipal aquifer and result in continued dependence on summertime supplementary water transports. As described in section 4.2.2, growing water use in the tourist sector, combined with higher summer water use due to warmer temperatures, causes water transports to increase steadily throughout the simulation (Fig. 12). By 2050 yearly water transports on average reach close to 4000 cubic meters per year (compared with the hitherto highest observed value of 3000 cubic meters), with a confidence bound ranging from 2200 to 5400 cubic meters (Table 3). A remarkable aspect of these results is that the local municipal water supply is insufficient to meet demand for the entire outcome space. This suggests that, even in the most optimistic climate scenario from a water supply perspective, maintaining the current trajectory of socioeconomic development will cause sustained reliance on supplementary water transports. Additionally, the high proportion of households with permanent or periodically elevated chloride levels is likely to result in increased pressure on the municipality to expand the borders of the municipal water management area and provide water services to more communities on the island. This would require substantial investments in infrastructure and further increase the reliance on supplementary water transports.

For Fårö to become water self-sufficient, a fundamental change in water supply solutions, growth strategy, and water use efficiency is needed. For instance, the current water supply system is completely reliant on groundwater, making it vulnerable to declines in both groundwater levels and groundwater quality (Schramm and Felmeden, 2012). Diversifying the portfolio of local water sources can reduce this vulnerability by making the system more resilient to unexpected climate events, and the large fluctuations in groundwater availability that our simulations project (Daigger and Crawford, 2007; Leigh and Lee, 2019). Rainwater, stormwater, and graywater are all potential sources of usable water that are not leveraged in most municipalities across the Nordic region. Utilizing these as alternatives for non-potable purposes can reduce water demand from conventional sources by an estimated 30–60% (Biggs et al., 2009; Zadeh et al., 2013). These solutions can improve overall resource efficiency, and increase redundancy by not wasting drinking quality water on uses with lower quality requirements (e.g. irrigation and toilet flushing). Reducing groundwater extraction also serves to maintain environmental flows that are critical for the health of freshwater dependent ecosystems (Leigh and Lee, 2019) and it can significantly reduce energy demand for water treatment and transfer (Xue et al., 2016). Several studies have concluded that because of their low energy costs, short construction times, and low capital intensity, decentralized solutions making use of alternative water sources are compatible, and often economically superior to conventional centralized alternatives (Brown et al., 2011; Leigh and Lee, 2019). On the other hand, a cost-benefit analysis conducted by Sjöstrand et al. (2019), comparing different water scarcity abatement measures in the Gotland region, concluded increased centralized groundwater extraction to be the most cost effective solution for the region. However, the analysis by Sjöstrand et al. (2019), like most conventional approaches for both economic and sustainability policy assessment, are based on a static view of the system (Lindqvist et al., 2019). That is, the system is assumed not to evolve or change over time and factors such as resilience to climate variability, effects of synergies and interactions between interventions, and socioeconomic feedbacks, are not accounted for in the assessment. Our simulation results clearly show that such a static assumption is misleading and, based on previous studies, can compromise the sustainability and resilience of future water systems (Leigh and Lee, 2019; Lindqvist et al., 2021). We call for further studies, on Fårö and elsewhere, to reassess alternative water supply solutions, some of which we have briefly mentioned, utilizing the type of feedback rich, dynamic, socio-hydrological system models we have developed in this study to identify sustainable and resilient pathways to mitigate future water scarcity.

5. Conclusions

We present a combined social and hydrological model using multivariate MC simulations to explore the effects of future climate and socioeconomic mechanisms on local supply and demand for drinking water on the Swedish island of Fårö. Our results suggest, given the available projections of future climate for the region, that the period with historically low groundwater levels experienced in the last decades will be sustained, and the probability of recurring periods with the groundwater table reaching lower levels than hitherto ever experienced is high. The low groundwater levels will limit water availability and increase the risk of saltwater contamination of drinking water wells. This will constrain growth in the housing sector (by 4–11%) and the tourist sector (by 10–30%), and maintain municipal reliance on supplementary water transports in summer months. The tourist sector will become increasingly reliant on private wells to support growth, but spillover effects will continue to increase consumption of municipal water and yearly municipal water transports.

To our knowledge, this is the first study to explore local impacts of future climate using an integrated social and hydrological model in the Scandinavian region. As in many other studies (Rusli et al., 2021; Tegegne et al., 2017), poor availability of local hydrological and water use data poses a challenge to model development for the region. For instance, limited data on historical groundwater levels and lack of spatially referenced water quality samples make spatially disaggregated modeling of future groundwater levels impossible. This necessitates a more exploratory modeling approach, investigating broad parameter ranges and presenting results in terms of outcome spaces rather than narrow predictions (Bankes, 1993). These limitations aside, our results provide important insights about the range of plausible futures that should be accounted for in local to regional water resource management and planning. Ensuring water self-sufficiency across the full outcome space will require investments targeting resilience in the water supply system. This can be achieved by leveraging alternative water sources, improving water use efficiency, and by accounting for socio-hydrological dynamics in the planning and management of future water system. We believe that the work presented here can support this necessary transition on Fårö and serve as a steppingstone for further climate impact and adaptation research in the Nordic region.

Funding

This research did not receive any specific grant from funding agencies in the public, commercial, or not-for-profit sectors.

CRediT authorship contribution statement

Andreas Nicolaidis Lindqvist: Conceptualization, Methodology, Formal analysis, Data curation, Writing – original draft. **Rickard Fornell:** Conceptualization, Methodology, Validation, Writing – review & editing. **Thomas Prade:** Writing – review & editing. **Sammar Khalil**: Supervision, Discussion. **Linda Tufvesson:** Supervision, Writing – review & editing. **Birgit Kopainsky:** Conceptualization, Methodology, Validation, Writing – review & editing, Supervision.

Declaration of Competing Interest

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

Acknowledgements

Thanks to participants from the Department of Water Management at Region Gotland for their contributions in data collection, model development and validation. Also thanks to Jake Jacobson and Len Malczynski at MindsEye Computing, Idaho Falls USA, for reviewing and providing invaluable feedback throughout the model development process.

Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.ejrh.2022.101066.

References

Ahopelto, L., et al., 2019. Can there be water scarcity with abundance of water? Analyzing water stress during a severe drought in Finland. Sustainability 11 (6), 1548.
Alexandersson, H., 2006. Vindstatistik för Sverige 1961–2004 [Wind statistics for Sweden 1961–2004].
Asp, M. et al., 2015. Framtidsklimat i Gotlands län - enligt RCP-scenarier [Future climate in Gotland county - according to the RCP scenarios], SMHI.
Bankes, S., 1993. Exploratory modeling for policy analysis. Oper. Res. 41, 435–449. https://doi.org/10.1287/opre.41.3.435.
Bessah, E., et al., 2020. Hydrological responses to climate and land use changes: the paradox of regional and local climate effect in the Pra River Basin of Ghana. J. Hydrol.: Reg. Stud. 27, 100654 https://doi.org/10.1016/j.ejrh.2019.100654.
Biggs, C., Ryan, C., Wiseman, J., Larsen, K., 2009. Distributed Water Systems: A Networked and Localised Approach for Sustainable Water Services. VEIL.
Brown, R., Ashley, R., Farrelly, M., 2011. Political and professional agency entrapment: an agenda for urban water research. Water Resour. Manag.: Int. J. Publ. Eur. Water Resour. Assoc. 25 (15), 4037–4050.
Brunner, F., 2014. FÖP Fårö - Fördjupad översiktsplan 2025 [Regional development plan for Fårö 2025]. In: Environment, D.o.B. (Ed.). Region Gotland, Visby, Gotland.
Dahlqvist, P. et al., 2015. SkyTEM-undersökningar på Gotland [SkyTEM analysis of Gotland].
Daigger, G., Crawford, G., 2007. Enhancing water system security and sustainability by incorporating centralized and decentralized water reclamation and reuse into urban water management systems. J. Environ. Eng. Manag. 17, 1–10.
Darvini, G., Memmola, F., 2020. Assessment of the impact of climate variability and human activities on the runoff in five catchments of the Adriatic Coast of south-central Italy. J. Hydrol.: Reg. Stud. 31, 100712 https://doi.org/10.1016/j.ejrh.2020.100712.
Di Baldassarre, G., et al., 2019. Sociohydrology: scientific challenges in addressing the sustainable development goals. Water Resour. Res. 55 (8), 6327–6355. https://doi.org/10.1029/2018wr023901.
Dimkić, D., 2020. Temperature impact on drinking water consumption. Environ. Sci. Proc. 2, 31. https://doi.org/10.3390/environsciproc2020002031.
Englund, P., 2011. Swedish House Prices in an International Perspective. Central Bank of Sweden, Stockholm.
Fabre, J., Ruelland, D., Dezetter, A., Grouillet, B., 2015. Simulating past changes in the balance between water demand and availability and assessing their main drivers at the river basin scale. Hydrol. Earth Syst. Sci. 19 (3), 1263–1285. https://doi.org/10.5194/hess-19-1263-2015.
Fernald, A., et al., 2012. Modeling sustainability of water, environment, livelihood, and culture in traditional irrigation communities and their linked watersheds. Sustainability 4 (11), 2998–3022.
Fraser, E.D.G., et al., 2011. Assessing vulnerability to climate change in dryland livelihood systems - conceptual challenges and interdisciplinary solutions. Ecol. Soc. 16 (3).
Gohari, A., et al., 2013. Water transfer as a solution to water shortage: a fix that can Backfire. J. Hydrol. 491, 23–39. https://doi.org/10.1016/j.jhydrol.2013.03.021.
Gossling, S., et al., 2012. Tourism and water use: Supply, demand, and security. An international review. Tour. Manag. 33 (1), 1–15. https://doi.org/10.1016/j.tourman.2011.03.015.
Gotlands Kommun, 2008. Riktlinjer för hantering av enskild vattenförsörjning vid nyetablering [Water supply guidelines for new establishments] Gotlands Kommun, Visby, Gotland, 8.
Hillel, D., 2004. Introduction to Environmental Soil Physics. Elsevier Science.
Höglund, L., 1999. Household demand for water in sweden with implications of a potential tax on water use. Water Resour. Res. 35 (12), 3853–3863. https://doi.org/10.1029/1999WR900219.
Homer, J.B., 2012. Partial-model testing as a validation tool for system dynamics (1983). Syst. Dyn. Rev. 28 (3), 281–294. https://doi.org/10.1002/sdr.1478.
International Monetary Fund, European Dept IMF Staff Country Reports 2015 Selected Issues Sweden A002 doi: 10.5089/9781513524368.002.A002.
IPCC, 2013. Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. IPCC, Cambridge, United Kingdom and New York, NY, USA.
Lantmäteriet, 2021. Offentlighetsuttag Fastighetsregistret [Extract from the real estate register]. In: Lantmäteriet (Ed.).
Leigh, N.G., Lee, H., 2019. Sustainable and resilient urban water systems: the role of decentralization and planning. Sustainability 11 (3), 918.
Lindqvist, A., Broberg, S., Tufvesson, L., Khalil, S., Prade, T., 2019. Bio-based production systems: why environmental assessment needs to include supporting systems. Sustainability 11 (17). https://doi.org/10.3390/su11174678.
Lindqvist, A.N., et al., 2021. Human-water dynamics and their role for seasonal water scarcity – a case study. Water Resour. Manag. 35 (10), 3043–3061. https://doi.org/10.1007/s11269-021-02819-1.

McCollum, D.L., Gambhir, A., Rogelj, J., Wilson, C., 2020. Energy modellers should explore extremes more systematically in scenarios. Nat. Energy 5 (2), 104–107. https://doi.org/10.1038/s41560-020-0555-3.
McLeod, A.I., 2011. Kendall: Kendall rank correlation and Mann-Kendall trend test.
Mei, W., Yu, G., 2020. basicTrendline: Add Trendline and Confidence Interval of Basic Regression Models to Plot.
Mikulić, J., et al., 2021. The effect of tourism activity on housing affordability. Ann. Tour. Res. 90, 103264 https://doi.org/10.1016/j.annals.2021.103264.
Paramati, S.R., Roca, E., 2019. Does tourism drive house prices in the OECD economies? Evidence from augmented mean group estimator. Tour. Manag. 74, 392–395. https://doi.org/10.1016/j.tourman.2019.04.023.
Penman, H.L., Keen, B.A., 1948. Natural evaporation from open water, bare soil and grass. Proc. R. Soc. Lond. Ser. A. Math. Phys. Sci. 193 (1032), 120–145. https://doi.org/10.1098/rspa.1948.0037.
Powell, M., 2009. The BOBYQA Algorithm for Bound Constrained Optimization without Derivatives. Technical Report, Department of Applied Mathematics and Theoretical Physics.
Pruyt, E., 2006. What is system dynamics? A paradigmatic inquiry.
R Development Core Team, 2019. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.
Region Gotland, 2021a. Building permits Fårö 2000–2019.
Region Gotland, 2021b. Pumping and delivery data Fårö water plant 2000–2019.
Rijsberman, F.R., 2006. Water scarcity: Fact or fiction? Agric. Water Manag. 80 (1), 5–22. https://doi.org/10.1016/j.agwat.2005.07.001.
Rivera, P., Ridderstolpe, Djurberg, 2011. VA-utredning för Fårö inför fördjupad översiktsplanering [Water and sewer assessment on Fårö for regional planning and development]. 2010–0342-B.
Rusli, S.R., Weerts, A.H., Taufiq, A., Bense, V.F., 2021. Estimating water balance components and their uncertainty bounds in highly groundwater-dependent and data-scarce area: an example for the Upper Citarum basin. J. Hydrol.: Reg. Stud. 37, 100911 https://doi.org/10.1016/j.ejrh.2021.100911.
SCB, 2021. SCB Statistical database. Statistics Sweden.
Schramm, E., Felmeden, J., 2012. Towards more resilient water infrastructures. Resilient Cities 2. Springer, Netherlands, Dordrecht, pp. 177–186.
SGU, 2021a. Kartvisare och diagram för beräknade nivåer [Map and diagrams of calculated groundwater levels]. Geological Survey of Sweden.
SGU, 2020. SGU dataset grundvattennivåer Sandhammaren 1 [SGU dataset groundwater levels Sandhammaren 1]. In: Geological Survey of Sweden (Ed.).
SGU, 2021b. SGU Kartvisare och diagram för mätstationer [SGU map and diagrams of monitoring stations]. SGU Geological Survey of Sweden.
Sivapalan, M., Savenije, H.H.G., Blöschl, G., 2012. Socio-hydrology: a new science of people and water. Hydrol. Process. 26 (8), 1270–1276. https://doi.org/10.1002/hyp.8426.
Sjökvist, E. et al., 2015. Klimatscenarier för Sverige - Bearbetning av RCP-scenarier för meteorologiska och hydrologiska effektstudier [Climate scenarios for Sweden - Processing of the RCP scenarios for metrological and hydrological effect studies], SMHI.
Sjöstrand et al., 2014. Metod för val av vattenförsörjning i områden med vattenbrist – en handbok [Methods for selecting water supply systems in areas with water scarcity - a manual]. 2014:54, SP Technical Research Institute of Sweden, Borås, Sweden.
Sjöstrand, K., Lindhe, A., Söderqvist, T., Dahlqvist, P., Rosén, L., 2019. Marginal abatement cost curves for water scarcity mitigation under uncertainty. Water Resour. Manag. 33 (12), 4335–4349. https://doi.org/10.1007/s11269-019-02376-8.
SMHI, 2021. SMHI Meteorologiska observationer [SMHI Metrological observations]. SMHI.
Stensen, K., Krunegård, A., Rasmusson, K., Matti, B., Hjerdt, N., 2019. Sveriges vattentillgång utifrån perspektivet vattenbrist och torka: – Delrapport 1 i regeringsuppdrag om åtgärder för att motverka vattenbrist i ytvattentäkter [Swedish water resources from a water scarcity and drought perspective: - report 1]. 02837722 (ISSN).
Sterman, J., 2000. Business Dynamics: Systems Thinking and Modeling for a Complex World. McGraw-Hill.
Sterman, J.D., 1984. Appropriate summary statistics for evaluating the historical fit of system dynamics models. Dynamica 10 (2), 51–66.
Strandberg, G. et al., 2014. CORDEX scenarios for Europe from the Rossby Centre regional climate model RCA4.
Swedish Environment and Energy department, 2007. Prop. (2006:412) Lagen om allmänna vattentjänster [2006:412 The law on public water services]. Miljö och energidepartementet.
Swedish Food Agency Livsmedelsverkets föreskrifter om ändring i Livsmedelsverkets föreskrifter (SLVFS 2001:30) om dricksvatten S.F. Agency [The Swedish Food Agency recommendations for changes in drinking water quality regulations (SLVFS 2001:30)] 2017 ISSN 1651-3533 Elanders Sverige AB 48.⟨https://www.livsmedelsverket.se/globalassets/om-oss/lagstiftning/dricksvatten—naturl-mineralv—kallv/livsfs-2017-2_web.pdf⟩.
Tegegne, G., Park, D.K., Kim, Y.-O., 2017. Comparison of hydrological models for the assessment of water resources in a data-scarce region, the Upper Blue Nile River Basin. J. Hydrol.: Reg. Stud. 14, 49–66. https://doi.org/10.1016/j.ejrh.2017.10.002.
United Nations, 2018. Sustainable Development Goal 6: Synthesis Report 2018 on Water and Sanitation. United Nations, New York, NY, USA. DOI:https://doi.org/10.18356/e8fc060b-en.
van Emmerik, T.H.M., et al., 2014. Socio-hydrologic modeling to understand and mediate the competition for water between agriculture development and environmental health: Murrumbidgee River basin, Australia. Hydrol. Earth Syst. Sci. 18 (10), 4239–4259. https://doi.org/10.5194/hess-18-4239-2014.
Swedish Water, 2020. Dricksvattenfakta [Drinking water facts].
Wiedmann, T., Lenzen, M., Keyßer, L.T., Steinberger, J.K., 2020. Scientists' warning on affluence. Nat. Commun. 11 (1), 3107. https://doi.org/10.1038/s41467-020-16941-y.
Wimmer, F., et al., 2015. Modelling the effects of cross-sectoral water allocation schemes in Europe. Clim. Change 128 (3), 229–244. https://doi.org/10.1007/s10584-014-1161-9.
Wu, W.-Y., et al., 2020. Divergent effects of climate change on future groundwater availability in key mid-latitude aquifers. Nat. Commun. 11 (1), 3710. https://doi.org/10.1038/s41467-020-17581-y.
Xu, L., Gober, P., Wheater, H.S., Kajikawa, Y., 2018. Reframing socio-hydrological research to include a social science perspective. J. Hydrol. 563, 76–83. https://doi.org/10.1016/j.jhydrol.2018.05.061.
Xue, X., Hawkins, T.R., Schoen, M.E., Garland, J., Ashbolt, N.J., 2016. Comparing the life cycle energy consumption, global warming and eutrophication potentials of several water and waste service options. Water 8 (4), 154.
Yeh, P.J.-F., Famiglietti, J.S., 2009. Regional groundwater evapotranspiration in Illinois. J. Hydrometeorol. 10 (2), 464–478. https://doi.org/10.1175/2008jhm1018.1.
Zadeh, S.M., Hunt, D.V.L., Lombardi, D.R., Rogers, C.D.F., 2013. Shared urban greywater recycling systems: water resource savings and economic investment. Sustainability 5 (7), 2887–2912.
Zhang, L., Potter, N., Hickel, K., Zhang, Y., Shao, Q., 2008. Water balance modeling over variable time scales based on the Budyko framework – model development and testing. J. Hydrol. 360 (1), 117–131. https://doi.org/10.1016/j.jhydrol.2008.07.021.

Appendix 3

Supplementary material

Impacts of future climate on local water supply and demand – A socio-hydrological case study in the Nordic region

Andreas Nicolaidis Lindqvist [a,b,*], Rickard Fornell [a], Thomas Prade [b], Sammar Khalil [b], Linda Tufvesson [b], Birgit Kopainsky [c]

[a] RISE Research Institutes of Sweden, Ideon Beta5, Scheelev¨agen 17, 22370 Lund, Sweden

[b] Department of Biosystems and Technology, Swedish University of Agricultural Sciences, POO Box 103, SE-230 53 Alnarp, Sweden

[c] Department of Geography, System Dynamics Group, University of Bergen, POO Box 7800, 5020 Bergen, Norway

[*] Correspondence: andreas.nicolaidis@ri.se

Content

Appendix A - Model documentation

This document contains the mathematical structure of the simulation model presented "Impacts of future climate on local water supply and demand - a socio-hydrological case study in the Nordic region" by Nicolaidis Lindqvist et al. First, a overview of the model is provided, followed by a detailed description of the individual submodules. Each description starts with an overview of the module and a summary of the key operations performed within it. The most important calculations performed in the module are described and a graphical representation of the module structure is provided in the form of a Stock-and-Flow Diagram (SFD). At the end of each description is a complete list of raw equations for the submodule.

Model overview

Table A.1. Model Information

Model Information	Number[1]
Total Number of Variables	389 (549)
Total number of stocks	20 (22)
Total number of flows	39 (46)
Converters	330 (481)
Constants	142 (243)
Equations	227 (284)
Graphical functions/Lookup tables	11 (12)
Time unit	Months
Initial time	0 (January 2000)
Final time	600 (January 2050)
Time step	0.0625

Modular structure

Figure A.1 illustrates the causal structure of the full model and the six major sub-modules *Climate*, *Public Water Supply*, *Private Water Supply*, *Household Water Use*, *Tourist Water Use* and *Public Water Supply Demand Balance.* Grey arrow links in Figure A.1 indicate information flow between modules.

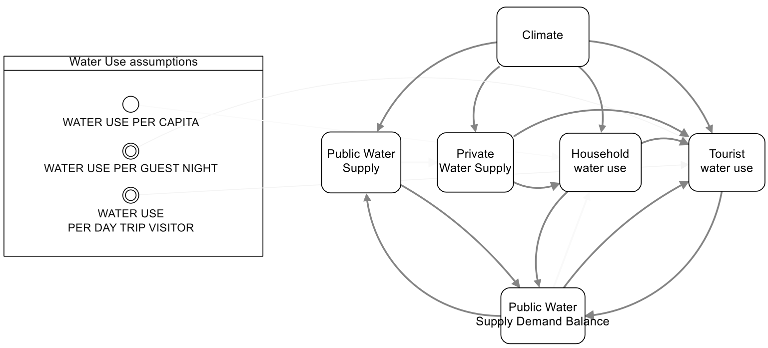


Figure A.1. Top level view of the model

Submodule descriptions

The Climate module

Key operations performed in the module:

[1] Array expansions in brackets.

1. Import of historic climate input variables (monthly precipitation, temperature, windspeed and solar radiation).
2. Import and calculation of future climate variables.
3. Calculation of monthly evapotranspiration using the Penman-Monteith method (Penman and Keen, 1948).
4. Calculation of the effect of temperature on per capita water use according to Dimkić (2020).

Historic climate variables

For the historic simulation period (2000 – 2020) the climate module imports monthly regional data on precipitation (mm/month), temperature (monthly daily max and monthly daily min temperature in Celsius), solar radiation (MJ/month*m2) and windspeed (m/sec). Precipitation and temperature data are based on observations from weather stations on Fårö (SMHI). Recorded data on solar radiation are used for periods where such data is available (2008 – 2020). Outside this time period, monthly mean solar radiation values from the 2008-2020 period are used. Monthly windspeeds are based on historic values provided by Alexandersson (2006).

Future climate variables

Mean future precipitation and temperature

Mean monthly future (2021 - 2050) precipitation and temperature inputs are based on county specific climate projections provided the Swedish Metrological and Hydrological Institute, SMHI (Asp et al., 2015). The climate projections are produced by the regional climate model RCA4 (Strandberg et al., 2014), by downscaling climate scenarios from an ensemble consisting of nine global climate models (CanESM2, CNRM-CM5, GFDL-ESM2M, EC-EARTH, IPSL-CM5A-MR, MIROC5, MPI-EMS-LR, NorESMI1-M and HadGEM2-ES) (Sjökvist et al., 2015). Two emission scenarios (Representative Concentration Pathways, RCP), published by the Intergovernmental Panel on Climate Change (IPCC, 2013), are considered in the regional climate projections: RCP4.5 and RCP8.5.

The county specific climate projections by SMHI provide climate inputs in the form of relative changes compared to the reference period 1961 – 1990. To estimate the local climate inputs in absolute terms we calculated the mean monthly precipitation and temperature variables for the reference period and multiplied or added the projected relative change:

$$P_{it} = P_{ir} + \alpha_{it} P_{ir} \qquad \text{Equation A.1}$$

$$T_{it} = T_{ir} + \beta_{it} \qquad \text{Equation A.2}$$

Where P_{it} and T_{it} are the projected absolute monthly precipitation (mm/month) and temperature (°C) variables for month *i* and time *t* respectively. P_{ir} and T_{ir} represent the mean precipitation and temperature for month *i* during the reference period. α_{it} is the relative change in precipitation (%) and β_{it} is the change in temperature (°C) for month *i* at time *t* in the future.

In the climate module the user can define which climate scenario (RCP4.5 or RCP8.5) he/she wants to activate using a *climate switch*. This allows the user to switch between the two climate scenarios in consecutive simulations.

Mean future solar radiation and windspeed

The county specific climate projections by SMHI do not include solar radiation or windspeed. Therefore, we assume historic observations of solar radiation and windspeed to be representative for future climates and use the monthly mean values from the historic simulation period in our future projections.

Future climate variability

The regional climate scenarios produced by SMHI show the projected mean values for the climate variables, but they do not include between year variation. To add between year variability to our simulations we fitted probability distribution functions (PDF) to the historic precipitation, temperature and solar radiation observations for each month of the year. Precipitation was approximated by a two parameter Weibull distribution and temperature and solar radiation were approximated by the Normal distribution. Future variation in climate inputs were then simulated by keeping the shape parameter of the Weibull distribution, and the relative standard deviation of the Normal distribution, constant whilst adjusting the scale and mean parameters according to the SMHI scenarios. In other words, each timestep the software was allowed to randomly sample from a PDF with a mean provided by the SMHI climate scenario and a shape estimated from the historical data for that specific month.

Monthly potential evapotranspiration

Monthly potential evapotranspiration (PET) is calculated using the Penman-Monteith method (Penman and Keen, 1948) using precipitation, temperature, windspeed and solar radiation as inputs. Due to missing data on relative humidity we used the simplifying assumption, as suggested by Zotarelli et al. (2015), that dewpoint temperature can be approximated by the minimum day temperature.

Effect of temperature on water use

Higher temperatures tend to increase per capita water use. To account for this we used the model developed by Dimkić (2020) where weekly per capita water use is estimated to increase by 1-5% for each degree Celsius that the hottest day of the week exceeds a threshold temperature of 15 °C. We assume a 2% increase in per capita water use when using monthly temperature values. The effect of temperature on per capita water use, $\varepsilon_{T_{MAX}}$, is calculated according to Equation A.3.

$$\varepsilon_{T_{MAX}} = q_H * 0.02 * MAX(0, T_{MAX} - T_{lim}) \qquad \text{Equation A.3}$$

Where q_H is the baseline water use per capita and month for Sweden, T_{MAX} is the temperature of the hottest day of the month and T_{lim} is the threshold temperature for increased water consumption suggested by Dimkić (2020).

Graphical representation of the Climate Module

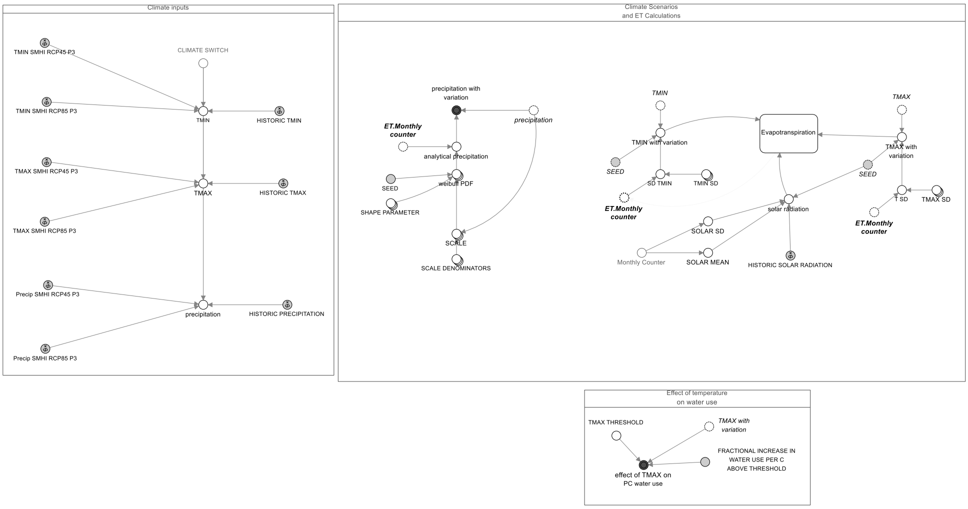


Figure A.2. Climate Module

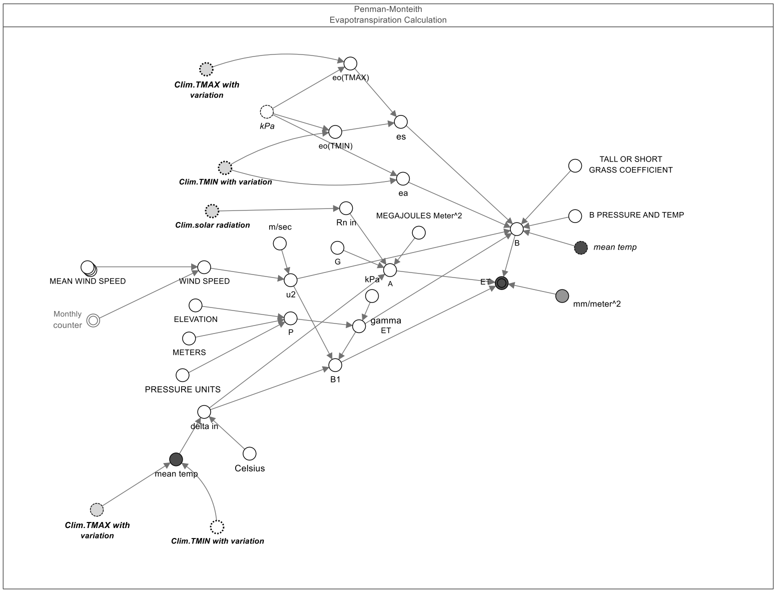


Figure A.3. Climate Module – Evapotranspiration

Complete list of equations for the Climate Module

Abbreviations
S = Stock
F = Flow
C = Constant
A = Auxiliary
D = Data input
I = Initial value
T = Graphical/Lookup Table

Table A.2. Climate module – variable constants

Climate module				
Type	**Variable name**	**Value**	**Unit**	**Description**
C	SCALE_DENOMINATORS[January;December]	0.887;0.886;0.886;0.892;0.892;0.900;0.891;0.887;0.887;0.888;0.886;0.886	dmnl	This is the denominator of the equation for calculating the scale parameter of the Weibull distribution for future monthly precipitation. The parameter values are calculated from historical precipitation data 1882-1995. Equation for calculating the scale parameter of the Weibull distribution: *scale = mean/GAMMA((1/shape)+1)* Where *mean* is the mean precipitation of the month, *GAMMA* is the gamma function as described by Abramowitz and Stegun (1988) and *shape* is the shape parameter of the Weibull distribution.
C	SHAPE_PARAMETER[January:December]	2.478;2.039;2.038;1.702;1.716;1.539;1.728;1.905;1.916;1.875;2.271;2.161	mm/meter ^2/month	This is the shape parameter of the Weibull distribution function for future precipitation. Calculated based on historic precipitation values 1882-1995.
C	CLIMATE_SWITCH	1;2	dmnl	This is a control parameter for choosing which future climate scenario to be used for the period 2021-2050. If CLIMATE_SWITCH = 1 then the SMHI RCP4.5 scenario will be used. If CLIMATE_SWITCH = 2 then the SMHI RCP8.5 scenario will be used.

C	FRACTIONAL INCERASE IN WATER USE PER C ABOVE THRESHOLD	0.02	dmnl/C	For each degree C above TMAX threshold (15 C) the average PC water use increases by 1-5 % on a weekly basis according to Dimkić (2020). We assume an effect of 2 % for Fårö when using monthly TMAX values.
C	TMAX_SD[January:December]	2.25;2.85;2.67;2.46;2.5;1.93; 2.25;1.9;1.93;1.78;1.67;2.49;	C	This is the standard deviation of the temperature of the observed hottest day per month in 2000-2020. Source: SMHI
C	TMIN_SD[January:December]	2.65;4.14; 3.57;1.37;1.69;1.47;1.29;1.35;1.85;1.49;1.77;3.45	C	This is the standard deviation of the temperature of the observed coldest day per month in 2000-2020. Source: SMHI
C	TMAX_THRESHOLD	15	C	Below this threshold temperature, water use is insensitive to temperature (Dimkić, 2020).
Climate Module - Evapotranspiration[2]				
C	B_PRESSURE_AND_TEMP	kPa/C	1	
C	Celsius	C	1	
C	ELEVATION	m	7	This is the elevation above sea level.
C	G	MJ/(month*meter^2)	0	Solar constant
C	kPa	kPa	1	
C	m/sec	m/sec	1	
C	MEAN_WIND_SPEED[January:December]	m/sec	7.7; 7.3; 6.7; 5.8; 5.5; 5.3; 5; 5.2; 6; 6.3; 6.6; 7.4	Mean wind speed per month 1991-2004 Source: SMHI
C	MEGAJOULES_m2	MJ/(month*meter^2)	1	
C	METERS	meter	1	
C	mm/meter^2	mm/meter^2	1	
C	PRESSURE_UNITS	kPa	1	
C	TALL_OR_SHORT_GRASS_COEFFICIENT	dmnl	900	This is the coefficient used for calculating evapotranspiration.

Table A.3. Climate module – auxiliary calculations

Type	Variable name	Equation	Units	Description
A	analytical_precipitation =	IF Evapotranspiration.Monthly_counter <= 1 THEN weibull_PDF[January] ELSE IF Evapotranspiration.Monthly_counter > 1 AND Evapotranspiration.Monthly_counter<=2 THEN weibull_PDF[February] ELSE IF Evapotranspiration.Monthly_counter > 2 AND Evapotranspiration.Monthly_counter<=3 THEN weibull_PDF[March] ELSE IF Evapotranspiration.Monthly_counter > 3 AND Evapotranspiration.Monthly_counter<=4 THEN weibull_PDF[April] ELSE IF Evapotranspiration.Monthly_counter > 4 AND Evapotranspiration.Monthly_counter<=5 THEN weibull_PDF[May] ELSE IF Evapotranspiration.Monthly_counter > 5 AND Evapotranspiration.Monthly_counter<=6 THEN weibull_PDF[June] ELSE IF Evapotranspiration.Monthly_counter > 6 AND Evapotranspiration.Monthly_counter<=7 THEN weibull_PDF[July] ELSE IF Evapotranspiration.Monthly_counter > 7 AND Evapotranspiration.Monthly_counter<=8 THEN weibull_PDF[August] ELSE IF Evapotranspiration.Monthly_counter > 8 AND Evapotranspiration.Monthly_counter<=9 THEN weibull_PDF[September] ELSE IF Evapotranspiration.Monthly_counter > 9 AND Evapotranspiration.Monthly_counter<=10 THEN weibull_PDF[October]	mm/meter^2 /month	Selects which precipitation distribution function (January:December) to use in the calculations depending on the month of the year.

[2] Unless otherwise stated, parameter values and calculations for modeling evapotranspiration are based on the FAO-56 Penman-Monteith method as outlined by Zotarelli et al., (2015).

		ELSE IF Evapotranspiration.Monthly_counter > 10 AND Evapotranspiration.Monthly_counter<=11 THEN weibull_PDF[November] ELSE weibull_PDF[December]		
A	effect_of_TMAX_on_PC_water_use =	IF TMAX_with_variation < TMAX_threshold THEN 1 ELSE 1 + (TMAX_with_variation-TMAX_threshold)*fractional_increase_in_water_use_per_C_above_threshold	dmnl	Per capita water use increases when temperatures reach values above a threshold temperature (15 °C). The effect_of_TMAX_on_PC-water_use is takes on values greater than 1 when the monthly TMAX exceeds the threshold temperature suggested by (Dimkić, 2020).
A	Monthly_Counter =	COUNTER(0, 12)	month	The Monthly_Counter is used as a multiplier in other calculations to keep track of the month of the year. COUNTER Value = Month 0≤1 = January 1≤2 = February 2≤3 = March 3≤4 = April 4≤5 = May 5≤6 = June 6≤7 = July 7≤8 = August 8≤9 = September 9≤10 = October 10≤11 = November 11≤12 = December
A	precipitation =	IF TIME < 0 THEN HISTORIC_PRECIPITATION ELSE IF CLIMATE_SWITCH = 1 THEN Precip_SMHI_RCP45_P3 ELSE Precip_SMHI_RCP85_P3	mm/meter^2/month	For simulations starting before January 2000 (TIME < 0) a historic precipitation dataset is used. When simulating January 2000 and forward either the SMHI RCP 4.5 or SMHI RCP 8.5 datasets are used, depending on the settings of the CLIMATE SWITCH.
A	precipitation_with_variation =	IF TIME <253 THEN precipitation ELSE analytical_precipitation	mm/meter^2/month	Before time 253 (January 2021) historic precipitation data are used. After time 253 simulated precipitation data, with variation between years, is used.
A	SD_TMIN =	IF Evapotranspiration.Monthly_counter <= 1 THEN TMIN_SD[January] ELSE IF Evapotranspiration.Monthly_counter > 1 AND Evapotranspiration.Monthly_counter<=2 THEN TMIN_SD[February] ELSE IF Evapotranspiration.Monthly_counter > 2 AND Evapotranspiration.Monthly_counter<=3 THEN TMIN_SD[March] ELSE IF Evapotranspiration.Monthly_counter > 3 AND Evapotranspiration.Monthly_counter<=4 THEN TMIN_SD[April] ELSE IF Evapotranspiration.Monthly_counter > 4 AND Evapotranspiration.Monthly_counter<=5 THEN TMIN_SD[May] ELSE IF Evapotranspiration.Monthly_counter > 5 AND Evapotranspiration.Monthly_counter<=6 THEN TMIN_SD[June] ELSE IF Evapotranspiration.Monthly_counter > 6 AND Evapotranspiration.Monthly_counter<=7 THEN TMIN_SD[July] ELSE IF Evapotranspiration.Monthly_counter > 7 AND Evapotranspiration.Monthly_counter<=8 THEN TMIN_SD[August] ELSE IF Evapotranspiration.Monthly_counter > 8 AND Evapotranspiration.Monthly_counter<=9 THEN TMIN_SD[September] ELSE IF Evapotranspiration.Monthly_counter > 9 AND Evapotranspiration.Monthly_counter<=10 THEN TMIN_SD[October]	C	SD_TMIN is the standard deviation of TMIN. Each month has a different value for SD_TMIN and which value to be used in the calculations is decided by the monthly counter. If Monthly_Counter has a value between 0 and 1 then SD_TMIN[January] will be used, etc.

		ELSE IF Evapotranspiration.Monthly_counter > 10 AND Evapotranspiration.Monthly_counter<=11 THEN TMIN_SD[November] ELSE TMIN_SD[December]		
A	SOLAR_MEAN =	IF Monthly_Counter <=1 THEN 41.92 ELSE IF Monthly_Counter > 1 AND Monthly_Counter <=2 THEN 102.85 ELSE IF Monthly_Counter > 2 AND Monthly_Counter <=3 THEN 273.62 ELSE IF Monthly_Counter > 3 AND Monthly_Counter <=4 THEN 478.73 ELSE IF Monthly_Counter > 4 AND Monthly_Counter <=5 THEN 673.26 ELSE IF Monthly_Counter > 5 AND Monthly_Counter <=6 THEN 715.10 ELSE IF Monthly_Counter > 6 AND Monthly_Counter <=7 THEN 665.17 ELSE IF Monthly_Counter > 7 AND Monthly_Counter <=8 THEN 515.32 ELSE IF Monthly_Counter > 8 AND Monthly_Counter <=9 THEN 326.40 ELSE IF Monthly_Counter > 9 AND Monthly_Counter <=10 THEN 156.49 ELSE IF Monthly_Counter > 10 AND Monthly_Counter <=11 THEN 50.42 ELSE 27.51	MJ/(month* m^2)	Sets the mean solar radiation per month by using historic mean values from 2008-2020 (SMHI).
A	solar_radiation =	IF TIME < 252 THEN HISTORIC_SOLAR_RADIATION ELSE NORMAL(SOLAR_MEAN, SOLAR_SD, SEED, 0, 800, 1)	MJ/(month* m^2)	Before 2020 historic values on solar radiation are used. After 2020 simulated values with a normal distribution are used.
A	SOLAR_SD =	IF Monthly_Counter <=1 THEN 3.44 ELSE IF Monthly_Counter > 1 AND Monthly_Counter <=2 THEN 15.77 ELSE IF Monthly_Counter > 2 AND Monthly_Counter <=3 THEN 30.43 ELSE IF Monthly_Counter > 3 AND Monthly_Counter <=4 THEN 40.02 ELSE IF Monthly_Counter > 4 AND Monthly_Counter <=5 THEN 53.21 ELSE IF Monthly_Counter > 5 AND Monthly_Counter <=6 THEN 33.63 ELSE IF Monthly_Counter > 6 AND Monthly_Counter <=7 THEN 40.13 ELSE IF Monthly_Counter > 7 AND Monthly_Counter <=8 THEN 46.78 ELSE IF Monthly_Counter > 8 AND Monthly_Counter <=9 THEN 30.78 ELSE IF Monthly_Counter > 9 AND Monthly_Counter <=10 THEN 25.57 ELSE IF Monthly_Counter > 10 AND Monthly_Counter <=11 THEN 8.39 ELSE 3.43	MJ/(month* m^2)	This is the standard deviation of monthly solar radiation values (SMHI).
A	T_SD =	IF Evapotranspiration.Monthly_counter <= 1 THEN TMAX_SD[January] ELSE IF Evapotranspiration.Monthly_counter > 1 AND Evapotranspiration.Monthly_counter<=2 THEN TMAX_SD[February] ELSE IF Evapotranspiration.Monthly_counter > 2 AND Evapotranspiration.Monthly_counter<=3 THEN TMAX_SD[March] ELSE IF Evapotranspiration.Monthly_counter > 3 AND Evapotranspiration.Monthly_counter<=4 THEN TMAX_SD[April] ELSE IF Evapotranspiration.Monthly_counter > 4 AND Evapotranspiration.Monthly_counter<=5 THEN TMAX_SD[May] ELSE IF Evapotranspiration.Monthly_counter > 5 AND Evapotranspiration.Monthly_counter<=6 THEN TMAX_SD[June]	C	Sets the standard deviation of TMAX to be used when simulating future TMAX_with_variation.

		ELSE IF Evapotranspiration.Monthly_counter > 6 AND Evapotranspiration.Monthly_counter<=7 THEN TMAX_SD[July] ELSE IF Evapotranspiration.Monthly_counter > 7 AND Evapotranspiration.Monthly_counter<=8 THEN TMAX_SD[August] ELSE IF Evapotranspiration.Monthly_counter > 8 AND Evapotranspiration.Monthly_counter<=9 THEN TMAX_SD[September] ELSE IF Evapotranspiration.Monthly_counter > 9 AND Evapotranspiration.Monthly_counter<=10 THEN TMAX_SD[October] ELSE IF Evapotranspiration.Monthly_counter > 10 AND Evapotranspiration.Monthly_counter<=11 THEN TMAX_SD[November] ELSE TMAX_SD[December]		
A	TMAX =	IF TIME < 0 THEN HISTORIC_TMAX ELSE IF CLIMATE_SWITCH = 1 THEN TMAX_SMHI_RCP45_P3 ELSE TMAX_SMHI_RCP85_P3	C	Selects what time series of mean monthly TMAX to use depending on the TIME of the simulation.
A	TMAX_with_variation =	IF TIME <253 THEN TMAX ELSE NORMAL(TMAX, T_SD, SEED)	C	Introduces between year variation in future TMAX. Before time 253 historical TMAX data are used. After time 253 TMAX is normally distributed around the mean TMAX provided by the SMHI scenarios.
A	TMIN =	IF TIME < 0 THEN HISTORIC_TMIN ELSE IF CLIMATE_SWITCH = 1 THEN TMIN_SMHI_RCP45_P3 ELSE TMIN_SMHI_RCP85_P3	C	Selects what time series of mean monthly TMIN to use depending on the TIME of the simulation.
A	TMIN_with_variation =	IF TIME <253 THEN TMIN ELSE NORMAL(TMIN, SD_TMIN, SEED)	C	Introduces between year variation in future TMIN. Before time 253 historical TMIN data are used. After time 253 TMIN is normally distributed around the mean TMIN provided by the SMHI scenarios
A	SCALE[January:December] =	precipitation/SCALE_DENOMINATORS[January:December]	mm/meter^2 /month	This is the equation for the scale parameter of the Weibull probability distribution used for simulation between year variation in future precipitation.
A	weibull_PDF[January] =	WEIBULL(SHAPE_PARAMETER, SCALE[January], SEED, 0, 104.7, 1)	mm/meter^2 /month	To simulate between year variation in future monthly precipitation we assume precipitation to follow a Weibull distribution. The shape of the Weibull PDF is set by the SHAPE and SCALE parameters, estimated from historic precipitation data (1882-1995) and the projected monthly precipitation in the SMHI RCP4.5 and 8.5 scenarios. The distribution is bounded by the historic maximum and minimum observed precipitation for each month. See documentation for Stella Architect for further details about the Weibull function: ISEE Systems
A	weibull_PDF[February] =	WEIBULL(SHAPE_PARAMETER, SCALE[February], SEED, 0, 74, 1)	mm/meter^2 /month	
A	weibull_PDF[March] =	WEIBULL(SHAPE_PARAMETER, SCALE[March], SEED, 0, 69.7, 1)	mm/meter^2 /month	
A	weibull_PDF[April] =	WEIBULL(SHAPE_PARAMETER, SCALE[April], SEED, 0, 91, 1)	mm/meter^2 /month	
A	weibull_PDF[May] =	WEIBULL(SHAPE_PARAMETER, SCALE[May], SEED, 0, 93, 1)	mm/meter^2 /month	
A	weibull_PDF[June] =	WEIBULL(SHAPE_PARAMETER, SCALE[June], SEED, 0, 122, 1)	mm/meter^2 /month	
A	weibull_PDF[July] =	WEIBULL(SHAPE_PARAMETER, SCALE[July], SEED, 0, 161, 1)	mm/meter^2 /month	
A	weibull_PDF[August] =	WEIBULL(SHAPE_PARAMETER, SCALE[August], SEED, 0, 164, 1)	mm/meter^2 /month	
A	weibull_PDF[September] =	WEIBULL(SHAPE_PARAMETER, SCALE[September], SEED, 0, 146,1)	mm/meter^2 /month	
A	weibull_PDF[October] =	WEIBULL(SHAPE_PARAMETER, SCALE[October], SEED, 0, 170.2, 1)	mm/meter^2 /month	
A	weibull_PDF[November] =	WEIBULL(SHAPE_PARAMETER, SCALE[November], SEED, 0, 134, 1)	mm/meter^2 /month	
A	weibull_PDF[December] =	WEIBULL(SHAPE_PARAMETER, SCALE[December], SEED, 0, 112.9, 1)	mm/meter^2 /month	
		Climate Module – Evapotranspiration[2]		
A	A =	(0.408*delta_in*(Rn_in-G))/megajoules_m2	dmnl	See Zotarelli et al., (2015).
A	B =	((gamma_ET*tall_or_short_grass_coefficient*u2*(es-ea))/(mean_temp+273))/B_pressure_and_temp	dmnl	
A	B1 =	delta_in+(gamma_ET*(1+(0.38*u2)))	dmnl	
A	delta_in =	(2503*EXP((17.27*(mean_temp/Celsius))/((mean_temp/Celsius) +237.3)))/(((mean_temp/Celsius)+237.3)^2)	dmnl	

A	ea =	0.6108*EXP((17.27*Clim.TMIN_with_variation)/(Clim.TMIN_with_variation+237.3)) *kPa	kPa	
A	eo(TMAX) =	0.6108*EXP((17.27*Clim.TMAX_with_variation)/(Clim.TMAX_with_variation+237.3))*kPa	kPa	
A	eo(TMIN) =	0.6108*EXP((17.27*Clim.TMIN_with_variation)/(Clim.TMIN_with_variation+237.3)) *kPa	kPa	
A	es =	(eo(TMIN)+eo(TMAX))/2	kPa	
A	ET =	((A+B)/B1) *mm/meter^2	mm/meter^2	
A	gamma_ET =	(0.000665*P)/kPa	dmnl	
A	mean_temp =	(Clim.TMAX_with_variation+Clim.TMIN_with_variation)/2	C	
A	P =	101.3*((293-(0.0065*(ELEVATION/meters)))/293)^5.26 *PRESSURE_UNITS	kPa	
A	Rn_in =	Clim.solar_radiation	MJ/(month*m2)	
A	u2 =	WIND_SPEED*4.87/(LN((67.8*10)-5.42))/(m/sec)	dmnl	
A	WIND_SPEED =	IF Monthly_counter <=1 THEN MEAN_WIND_SPEED[January] ELSE IF Monthly_counter >1 AND Monthly_counter <=2 THEN MEAN_WIND_SPEED[February] ELSE IF Monthly_counter >2 AND Monthly_counter <=3 THEN MEAN_WIND_SPEED[March] ELSE IF Monthly_counter >3 AND Monthly_counter <=4 THEN MEAN_WIND_SPEED[April] ELSE IF Monthly_counter >4 AND Monthly_counter <=5 THEN MEAN_WIND_SPEED[May] ELSE IF Monthly_counter >5 AND Monthly_counter <=6 THEN MEAN_WIND_SPEED[June] ELSE IF Monthly_counter >6 AND Monthly_counter <=7 THEN MEAN_WIND_SPEED[July] ELSE IF Monthly_counter >7 AND Monthly_counter <=8 THEN MEAN_WIND_SPEED[August] ELSE IF Monthly_counter >8 AND Monthly_counter <=9 THEN MEAN_WIND_SPEED[September] ELSE IF Monthly_counter >9 AND Monthly_counter <=10 THEN MEAN_WIND_SPEED[October] ELSE IF Monthly_counter >10 AND Monthly_counter <=11 THEN MEAN_WIND_SPEED[November] ELSE MEAN_WIND_SPEED[December]	meter/sec	This is the monthly mean wind speed as 10 meter above ground per month.

Table A.4. Climate module – Data inputs

Type	Variable name	Units	Description
D	HISTORIC_PRECIPITATION	mm/meter^2/month	This is the historic precipitation per month 1961 to 2000. Used for generating the groundwater range for reference period P1 (1961-1990). Source: SMHI. See appendix C for full dataset.
D	HISTORIC_SOLAR_RADIATION	MJ/(month*m2)	This is the historic solar radiation values per month in 2000-2020. Source: SMHI. See appendix C for full dataset.
D	HISTORIC_TMAX	C	This is the temperature of the hottest observed day per month in the historic period 1961 to 2000. Used for generating the groundwater range for reference period P1 (1961-1990). Source: SMHI. See appendix C for full dataset.
D	HISTORIC_TMIN	C	This is the temperature of the coldest observed day in the historic period 1961 to 2000. Used for generating the groundwater range for reference period P1 (1961-1990). Source: SMHI. See appendix C for full dataset.
D	Precip_SMHI_RCP45_P3	mm/meter^2/month	This is the projected mean monthly precipitation according to the SMHI RCP4.5 scenario for the Gotland region up to 2050. Based on Asp (2015). See appendix C for full dataset.
D	Precip_SMHI_RCP85_P3	mm/meter^2/month	This is the projected mean monthly precipitation according to the SMHI RCP8.5 scenario for the Gotland region up to 2050. Based on Asp (2015). See appendix C for full dataset.
D	TMAX_SMHI_RCP45_P3	C	This is the projected temperature of the hottest day per monthly according to the SMHI RCP4.5 scenario for the Gotland region up to 2050. Based on Asp (2015). See appendix C for full dataset.
D	TMIN_SMHI_RCP85_P3	C	This is the projected temperature of the hottest day per monthly according to the SMHI RCP8.5 scenario for the Gotland region up to 2050. Based on Asp (2015). See appendix C for full dataset.
D	TMIN_SMHI_RCP45_P3	C	This is the projected temperature of the coldest day per monthly according to the SMHI RCP4.5 scenario for the Gotland region up to 2050. Based on Asp (2015). See appendix C for full dataset.

D	TMIN_SMHI_RCP85_P3	C	This is the projected temperature of the coldest day per monthly according to the SMHI RCP8.5 scenario for the Gotland region up to 2050. Based on Asp (2015). See appendix C for full dataset.

The Public Water Supply module

The Public Water Supply module represents the municipal aquifer utilized for public water supply. The aquifer is modeled as a two-stock structure with a soil storage stock, S(t), and a groundwater storage stock, GW(t).

Key operations performed in the module:

1. Imports data on precipitation and evapotranspiration from the Climate module, and data on desired groundwater pumping from the Public Water Supply Demand Balance module.
2. Partitions precipitation into rain and snow based on temperature.
3. Calculates the water balance of the aquifer using a Budyko-based approach as described by Zhang et al. (2008).
    a. Available rain and snowmelt are partitioned into runoff and infiltration based on the soil dryness index and the soil retention efficiency.
    b. Evapotranspiration from the soil layer is calculated as a function of the soil storage, the PET and the evapotranspiration efficiency of the ground cover.
    c. Groundwater recharge, from the soil layer to deep groundwater storage, is calculated as a function of the soil saturation level, the PET and the evapotranspiration efficiency of the ground cover.
    d. Groundwater discharge is calculated as a function of the groundwater saturation level, and a discharge factor (the groundwater baseflow).
    e. Vertical groundwater flow is estimated using Darcy's flow equation (Hillel, 2004). This represents the groundwater flow between the municipal aquifer and the surroundings if pumping would lower the water table of the municipal aquifer below that of the surroundings
4. Estimates the groundwater level in the aquifer as a function of the depth of the aquifer, the physical properties of the medium and the volume of water stored in the groundwater stock.
5. Limits groundwater pumping by the municipality as the groundwater level approaches the depth of the municipal wells.

Simulating snow and snowmelt

Precipitation is partitioned into rain, $P_r(t)$, and snowfall, $P_s(t)$, using a temperature-based allocation method (Federer, 1995). If the mean monthly temperature is above the threshold temperature for rain, or below the threshold temperature for snow, all precipitation will be allocated to rain or snow respectively. Between the two threshold temperatures the proportion of rain increases linearly with temperature. Snow accumulates as a stock of snow storage, Snow(t), and melts at a rate, S_m, determined by the same temperature thresholds used snow formation. The level of the snow storage stock is calculated by Equation A.4

$$Snow(t) = Snow(t-dt) + (P_s - S_m)dt \qquad \text{Equation A.4.}$$

where dt is the time step used in the simulation.

Water balance calculations

The dynamics of the S(t) and GW(t) are governed by the following two equations:

$$S(t) = S(t-dt) + (I + ET_d - ET - R)dt \qquad \text{Equation A.5.}$$

$$GW(t) = GW(t-dt) + (R + F_v - ET_d - D - E)dt \qquad \text{Equation A.6.}$$

I, ET_d, ET and R are infiltration, deep evapotranspiration (representing evapotranspiration occurring directly from the groundwater table at times with high groundwater levels), evapotranspiration and recharge respectively. F_v, D and E are vertical groundwater flow (representing exchange between the aquifer and surrounding aquifers), groundwater discharge and groundwater extraction respectively.

Partitioning rain and snowmelt into runoff and infiltration

The sum of rain and snowmelt, P(t), is partitioned into infiltration, I(t), and direct runoff, $Q_d(t)$ (Equation A.7). The amount that infiltrates is calculated according to Equation A.8.

$$P(t) = I(t) + Q_d(t) \qquad \text{Equation A.7}$$

$$I(t) = P(t)f\left(\frac{X_0(t)}{P(t)}, \alpha_1\right) = P(t)\left[1 + i_s - \left(1 + i_s^{\alpha_1}\right)^{\frac{1}{\alpha_1}}\right] \qquad \text{Equation A.8}$$

$X_0(t)/P(t)$ is an index of dryness, i_s, with $X_0(t)$ being the sum of the available soil storage capacity in the soil layer and PET. α_1 is a constant representing the rainfall retention efficiency of the soil. The partitioning of rain and snowmelt into runoff and infiltration is schematically illustrated in Figure A.4.

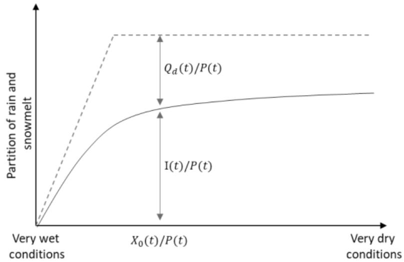


Figure A.4. Ratio of the mean monthly rain and snowmelt, P(t), partitioned into direct runoff, $Q_d(t)$, and soil infiltration, I(t), as a function of the index of dryness, $X_0(t)/P(t)$. The shape of the partitioning curve is determined by the retention efficiency of the soil, α_1. Areas with a higher value for α_1 will have a larger proportion of P(t) partitioned to infiltration and less to direct runoff. The demand and supply limits are shown as dashed lines

$$I(t) \,/\, P(t) \rightarrow 1 \text{ as } X_0(t)/P(t) \rightarrow \infty \text{ (very dry conditions)} \qquad \text{Equation A.9}$$

$$I(t) \rightarrow X_0(t) \text{ as } X_0(t)/P(t) \rightarrow 0 \text{ (very wet conditions)} \qquad \text{Equation A.10}$$

Calculating evapotranspiration and groundwater recharge

Evapotranspiration, ET(t) is a function of the level of the soil storage stock, S(t), PET(t), and the evapotranspiration efficiency of the soil cover α_2. PET(t) and S(t) can be considered the demand and supply limits for evapotranspiration, and the ratio PET(t)/S(t) is the demand/supply index for evapotranspiration, i_{ET} (Equation A.11).

$$ET(t) = S(t)f\left(\frac{PET(t)}{S(t)}, \alpha_2\right) = S(t)\left[1 + i_{ET} - \left(1 + i_{ET}^{\alpha_2}\right)^{\frac{1}{\alpha_2}}\right] \qquad \text{Equation A.11}$$

Recharge, R(t), equals S(t) minus the evapotranspiration opportunity, Y(t), (Equation A.11). Y(t) is calculated as a function of PET, S(t), soil storage capacity, S_{max}, and α_2 according to Equation A.13

$$R(t) = S(t) - Y(t) \qquad \text{Equation A.12}$$

$$Y(t) = S(t)f\left(\frac{PET(t)+S_{max}}{S(t)}, \alpha_2\right) = S(t)\left[1 + i_{Y(t)} - \left(1 + i_{Y(t)}^{\alpha_2}\right)^{\frac{1}{\alpha_2}}\right] \qquad \text{Equation A.13}$$

where $\frac{PET(t)+S_{max}}{S(t)}$ represents the demand/supply index for evapotranspiration opportunity $i_{Y(t)}$.

The partitioning of soil storage into evapotranspiration, recharge and soil retention to the next time step, S(t+1), is schematically illustrated in Figure A.5.

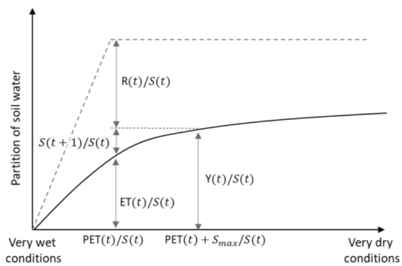


Figure A.5. Ratio of soil storage, S(t), partitioned into recharge, R(t), evapotranspiration, ET(t), and soil storage retention to the next time step, S(t+1), as a function of the demand/supply index for evapotranspiration, PET(t)/S(t), and the demand/supply index for evapotranspiration opportunity, [PET(t)+S_{max}]/S(t)]. The shape of the partitioning curve is determined by evapotranspiration efficiency of the soil cover α2. The demand and supply limits are shown as dashed lines.

$$Y(t) \,/\, S(t) \;\rightarrow 1 \;\text{ as }\; \frac{PET(t)+S_{max}}{S(t)} \rightarrow \;\infty \text{ (very dry conditions)} \qquad \text{Equation A.14}$$

$$Y(t) \rightarrow \;[PET(t) + S_{max}] \;\text{ as }\; \frac{PET(t)+S_{max}}{S(t)} \;\rightarrow\; 0 \text{ (very wet conditions)} \qquad \text{Equation A.15}$$

Calculating groundwater discharge

Groundwater discharge, D(t), is calculated as a function of the saturation level of the aquifer and a discharge factor, α_3.

$$D(t) = \alpha_3 \left(GW(t) * \frac{GW(t)}{GW_{max}}\right) \qquad \text{Equation A.16}$$

Where GW_{max} is the estimated maximum storage capacity of the aquifer, estimated from the size of the aquifer and the physical properties of the medium.

Modeling groundwater extraction and horizontal groundwater flow

Groundwater extraction, E(t), is set equal to the desired pumping, $E^*(t)$ (calculated in the *Water Supply Demand Balance module*), as long as the groundwater level in the aquifer, $GWL_A(t)$, remains higher than the depth of the municipal wells, WD. If the distance between $GWL_A(t)$ and WD is less than one meter, $E(t)/E^*(t)$ will linearly approach zero as $GWL_A(t)$ approaches WD.

$$\text{If } GWL_A(t) - WD < 1\,m \;\text{ then }\; E(t)/E^*(t) \;\rightarrow 0 \;\text{ as }\; GWL_A(t) - WD \rightarrow \;0 \qquad \text{Equation A.17}$$

Vertical groundwater flow, $F_v(t)$, is estimated using Darcy's flow equation (Hillel, 2004). The groundwater level of the municipal aquifer is compared to a baseline groundwater level of the surrounding aquifers, $GWL_B(t)$, where we assume no groundwater extraction to occur. This is a realistic assumption as the municipal aquifer on Fårö is located in a nature reserve with few other wells in the near proximity. $F_v(t)$ is assumed to be positive if E(t) causes $GWL_A(t)$ to fall below $GWL_B(t)$. The rate of the flow is calculated by Equation A.18 where Δh(t) is the difference between $GWL_A(t)$ and $GWL_B(t)$, $A_{GW}(t)$ is the cross-sectional area of the flow, K is the hydraulic conductivity of the medium and L_{GW} is the vertical flow distance.

$$F_v(t) = K * A_{GW}(t) \left(\frac{\Delta h}{L_{GW}}\right) \qquad \text{Equation A.18}$$

Estimating the groundwater level

GWL_A and GWL_B are both calculated by equation A.19.

$$GWL(t) = d + \frac{\left(\frac{GW(t)}{\Phi}\right)}{A} \qquad \text{Equation A.19}$$

Where d is the depth of the aquifer, Φ is the water holding capacity of the medium and A is the aquifer area.

Graphical representation of the Public Water Supply Module

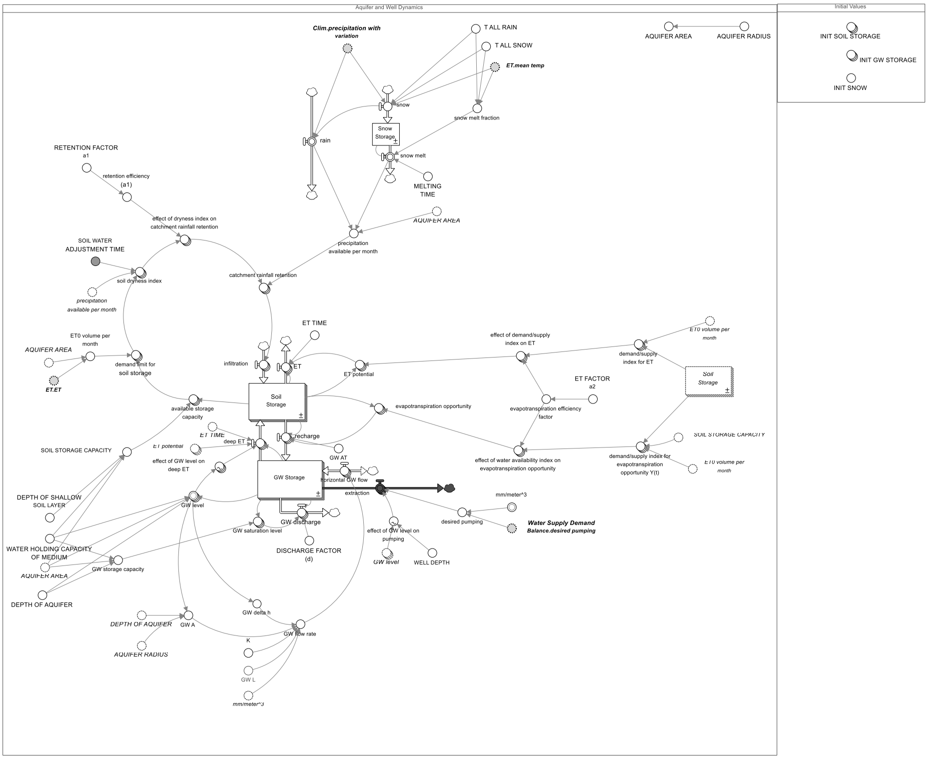


Figure A.6. Public Water Supply Module

Complete list of equations for the Public Water Supply Module

Abbreviations
S = Stock
F = Flow
C = Constant
A = Auxiliary
D = Data input
I = Initial value
T = Graphical/Lookup Table

Table A.5. Assumptions & constants

Public Water Supply Module				
Type	**Variable name**	**Unit**	**Value**	**Description**
C	SOIL WATER ADJUSTMENT TIME	month	1	This is a time constant representing the time interval over which incoming precipitation is distributed (one month).
C	AQUIFER RADIUS	Meter^2	21.7	This is the radius of the municipal aquifer.
C	MELTING TIME	month	0.376	This is a constant representing the time for surface runoff from snowmelt to reach the soil surface. Calibrated value.
C	DEPTH_OF_AQUIFER	meter	-10	This is the depth of the municipal aquifer.
C	DEPTH_OF_SHALLOW_SOIL_LAYER	meter	1.16	This is the depth of the soil storage layer of the municipal aquifer. Calibrated value.
C	DISCHARGE FACTOR	Dmnl/month	0.01	This is a parameter governing the groundwater baseflow from the municipal aquifer. Calibrated value.
C	ET_FACTOR_a2	dmnl	0.6	This is a model parameter representing the evapotranspiration efficiency of the ground cover of the municipal aquifer. Based on studies by Zhang et al. (2008). Calibrated value.

C	RETENTION_FACTOR_a1	dmnl	0.65	This is a model parameter representing the capacity of the soil surface to retain water. Based on studies by Zhang et al. (2008). Calibrated value.
C	ET_TIME	month	1	This is the evapotranspiration time.
C	GW_AT	month	0.773	This is the time for water to flow from the soil storage layer to the groundwater storage layer of the municipal aquifer. Calibrated value.
C	GW_L	meter	50.94	This is the average horizontal distance the groundwater flows from the surroundings to reach the municipal aquifer. Calibrated value.
C	INFILTRATION TIME	month	1	This is the time for precipitation to infiltrate to the soil storage stock.
C	INIT_GW_STORAGE[Aquifer_GW:Baseline_GW]	mm/meter^2	1150:1265	This is the volume of water stored per square meter of aquifer (the municipal aquifer and the surrounding aquifers) at the start of the simulation (January 2000). Calibrated values.
C	INIT_SNOW	mm/meter^2	0	This is the volume of water stored as snow at the start of the simulation (January 2000).
C	INIT_SOIL_STORAGE	mm/meter^2	100	This is the volume of water stored in the shallow soil layer per square meter of aquifer area at the start of the simulation (January 2000).
C	K	meter/month	549.8	This is the hydraulic conductivity (K) of the municipal aquifer. This corresponds to 3*10^-5 meter/second, the hydraulic conductivity of fine-grained sand. Soil samples from Region Gotland suggest a conductivity in the range: 2*10^-4 to 3*10^-4 meter/s or 518 -777 meter/month.
C	mm/meter^3	mm/meter^3	1000	This is the number of mm precipitation per cubic meter of water. 1 mm precipitation = 1 liter/meter^2.
C	T_ALL_RAIN	C	3.3	This is the threshold mean monthly temperature above which all precipitation falls as rain.
C	T_ALL_SNOW	C	-10	This is the threshold mean monthly temperature below which all precipitation falls as snow.
C	WATER_HOLDING_CAPACITY_OF_MEDIUM	mm/meter^3	180	This is the water holding capacity of the soil medium. Based on porosity estimates provided by Region Gotland and then calibrated to optimize fit to historical data.
C	WELL_DEPTH[3]	meter	---	This is the depth of the municipal wells. Classified information. For inquiries about the data consult Region Gotland.

Table A.6. Stocks and Flows

Public Water Supply Module				
Type	**Variable name**	**Equation**	**Units**	**Description**
S	GW_Storage(t)=	GW_Storage(t - dt) + (recharge + horizontal_GW_flow - GW_discharge - deep_ET - extraction) * dt	mm	This is the groundwater stock of the municipal aquifer. It increases by recharge and horizontal groundwater flow, and it is drained by deep evapotranspiration and groundwater extraction.
F	Recharge =	(Soil_Storage-evapotranspiration_opportunity)/GW_AT	mm/month	Recharge is the flow of water from Soil Storage to Groundwater Storage. It is calculated as the difference between Soil Storage and the evapotranspiration opportunity any given month.
F	horizontal_GW_flow[Aquifer_GW] =	horizontal_GW_flow[Aquifer_GW] = GW_flow_rate = K*GW_A*(GW_delta_h/GW_L) *"mm/meter^3	mm/month	The *horizontal groundwater flow* allows *groundwater storage* to adjust to the groundwater level in surrounding catchments. If extraction causes the groundwater level in the municipal aquifer to drop below that of the surroundings, this causes a horizontal flow into the aquifer until the hydraulic gradient is reduced to zero. The flow is calculated using Darcy's flow equation (Hillel, 2004).

[3] Details about the municipal wells and water supply system is classified and can therefore not be published. For inquiries about the data contact Region Gotland.

F	horizontal_GW_flow[Baseline_GW] =	0	mm/month	No horizontal groundwater flow is assumed for the surroundings of the municipal aquifer.
F	GW_discharge =	GW_Storage*GW_saturation_level*DISCHARGE_FACTOR	mm/month	GW_discharge is the flow of water from the aquifer to the ocean. It is calculated as the product of *groundwater storage,* the *groundwater saturation level,* and the *discharge factor* of the aquifer.
F	deep_ET =	MIN(GW_Storage, ET_potential*effect_of_GW_level_on_deep_ET)/ET_TIME	mm/month	deep_ET represents evapotranspiration reaching deeper than the upper soil layer and is primarily caused by trees and plants with deep root systems. It is calculated as the minimum of the available GW_storage and the ET_potential multiplied by the effect of GW level on deep ET. This structure makes sure that evapotranspiration stops if the groundwater level falls beyond the reach of the plant root system.
F	extraction[Aquifer_GW] =	desired_pumping*effect_of_GW_level_on_pumping	mm/month	Extraction from the municipal aquifer is the product of *desired pumping* and the *effect of GW level on pumping*. The effect function makes sure that as the groundwater level approaches the depth of the municipal wells, extraction declines linearly.
F	extraction[Baseline_GW] =	0	mm/month	We assume no extraction occurs in aquifers surrounding the municipal aquifer.
S	Snow_Storage(t) =	Snow_Storage(t - dt) + (snow - snow_melt) * dt	mm/meter ^2	Snow storage is a stock variable that increases by the flow *snow* and is drained by *snow melt*.
F	snow =	MAX(0, Clim.precipitation_with_variation* ((T_ALL_RAIN-ET.mean_temp)/ (T_ALL_RAIN-T_ALL_SNOW)))	mm/meter ^2/month	Precipitation is partitioned into rain and snowfall using a temperature-based allocation method (Federer, 1995). If the mean monthly temperature is above T_ALL_RAIN or below T_ALL_SNOW, all precipitation will be allocated to rain or snow respectively. Between the two threshold temperatures a fraction of the precipitation comes as snow.
F	Rain =	Clim.precipitation_with_variation-snow	mm/meter ^2/month	Rain is the total precipitation minus the snow fraction.
F	snow_melt =	Snow_Storage*snow_melt_fraction/MELTING_TIME	mm/meter ^2/month	Snow melt every month is a fraction of the Snow Storage stock.
S	Soil_Moisture_Storage(t) =	Soil_Storage(t - dt) + (infiltration + deep_ET - recharge - ET) * dt	mm	Soil Storage is a stock that increases by infiltration and capillary rise caused by deep ET, and is drained by ET.
F	infiltration =	catchment_rainfall_retention/INFILTRATION_TIME	mm/month	Precipitation retained by the soil surface infiltrates to the soil layer.
F	ET =	MIN(ET_potential, Soil_Moisture_Storage)/ET_TIME	mm/month	Evapotranspiration is the minimum of the ET_potential and the available water in the Soil_Moisture_Storage stock.

Table A.7. Auxiliary calculations

Public Water Supply Module				
Type	**Variable name**	**Equation**	**Units**	**Description**
I	INIT Soil_Moisture_Storage	INIT_SOIL_STORAGE*AQUIFER_AREA	mm	The starting value for the Soil_Moisture_Storage stock is calculated as the initial water storage per square meter of aquifer times the total area of the aquifer.
A	AQUIFER_AREA =	2*PI*AQUIFER_RADIUS^2	meter^2	We assume the area of the aquifer can be approximated by a circle.
A	available_storage_capacity =	MAX(0, "SOIL_STORAGE_CAPACITY_(Smax)"-"Soil_Moisture_Storage")	mm	Available storage capacity represents the difference between the current amount of water stored in the soil and the maximum amount of water the soil can hold when saturated.
A	catchment_rainfall_retention =	precipitation_available_per_month*effect_of_dryness_index_on_catchment_rainfall_retention	mm	Catchment rainfall retention is the precipitation withheld by the soil surface. It is influenced by the dryness index of the soil. When the soil is dry retention is higher and vice versa (Zhang et al., 2008).
A	demand_limit_for_soil_storage =	available_storage_capacity+ET0_volume_per_month	mm	The theoretical maximum amount of rain that can be retained by the soil each month. This is the sum of the available storage capacity and the potential evapotranspiration (Zhang et al., 2008).
A	demand/supply_index_for_ET =	ET0_volume_per_month/Soil_Moisture_Storage	dmnl	This index represents the relationship between the water demand from potential evapotranspiration and the supply/availability of water in the soil layer. It is used to calculate the evapotranspiration potential (Zhang et al., 2008).
A	demand/supply_index_for_evapotranspiration_opportunity =	(ET0_volume_per_month+SOIL_STORAGE_CAPACITY)/Soil_Moisture_Storage	dmnl	This index represents the relationship between the demand side for water in the aquifer system (the sum of potential evapotranspiration and the maximum soil storage capacity) and the supply side (the amount of water already stored in the soil). The index is used to calculate the evapotranspiration_opportunity (Zhang et al., 2008).
A	desired_pumping =	WSDB.desired_pumping*"mm/meter^3"	mm/month	desired_pumping is the amount of water the municipality want to extract from the aquifer. It is equal to the desired_pumping from the Water Supply Demand Balance (WSDB) module (in meter^3) converted to mm.
A	effect_of_demand/supply_index_on_ET =	1+"demand/supply_index_for_ET" -(1+"demand/supply_index_for_ET"^evapotranspiration_efficiency_factor)^(1/evapotranspiration_efficiency_factor)	dmnl	This is a non-linear function for calculating the proportion of water stored in the soil that is partitioned to evapotranspiration each month. If the demand/supply index is large, more water is allocated to evapotranspiration and vice versa (Zhang et al., 2008).
A	effect_of_dryness_index_on_catchment_rainfall_retention[GW_1] =	1+soil_dryness_index-(1+soil_dryness_index^"retention_efficiency_(a1)")^(1/"retention_efficiency_(a1)")	dmnl	This is a non-linear function for partitioning incoming precipitation between catchment rainfall retention and runoff. Rainfall retention increases with soil dryness (Zhang et al., 2008).
A	effect_of_water_availability_index_on_evapotranspiration_opportunitY =	1+demand/supply_index_for_evapotranspiration_opportunity -(1+demand/supply_index_for_evapotranspiration_opportunity^evapotranspiration_efficiency_factor)^(1/evapotranspiration_efficiency_factor)	dmnl	This is a non-linear function for calculating the proportion of water stored in the soil that is withheld in the soil layer (that is, it remains available for evapotranspiration) and what proportion that forms groundwater recharge (Zhang et al., 2008).
A	ET_potential =	Soil_Storage*effect_of_demand/supply_index_on_ET	mm	This is the proportion of water stored in the soil layer that is partitioned to evapotranspiration each month.
A	ET0_volume_per_month =	Evapotranspiration.ET*AQUIFER_AREA	mm	This is the potential evapotranspiration for the entire aquifer area each month.
A	evapotranspiration_efficiency_factor =	1/(1-ET_FACTOR_a2)	dmnl	This step converts the normalized ET_FACTOR_a2 to its proper scale as described by Zhang et al. (2008).

A	evapotranspiration_opportunity =	MAX(0, Soil_Moisture_Storage*effect_of_water_availability_index_on_evapotranspiration_opportunity)	mm	evapotranspiration_opportunity is the proportion of water stored in the soil that is withheld in the soil layer (that is, it is unavailable for recharge) each month.
A	GW_A =	AQUIFER_RADIUS*(GW_level - DEPTH_OF_AQUIFER)	meter^2	This is the cross-sectional area (A) of the Darcy's flow equation (Hillel, 2004) used to estimate the horizontal groundwater flow.
A	GW_delta_h =	GW_level[Baseline_GW]-GW_level[Aquifer_GW]	meter	This is the hydraulic gradient between the municipal aquifer [Aquifer_GW] and the surrounding aquifers [Baseline_GW]. Used in the Darcy's flow equation to calculate horizontal groundwater flow.
A	GW_level =	DEPTH_OF_AQUIFER+ GW_Storage/WATER_HOLDING_CAPACITY_OF_MEDIUM/AQUIFER_AREA	meter	The groundwater level (meters sub surface) is modeled by calculating the height of a water column sitting on top of the bottom of the aquifer. The height of the column is the volume of water stored divided by the water holding capacity of the medium and the total surface area of the aquifer.
A	GW_saturation_level =	GW_Storage/GW_STORAGE_CAPACITY	dmnl	Groundwater saturation takes on values between 0 and 1.
A	GW_STORAGE_CAPACITY =	WATER_HOLDING_CAPACITY_OF_MEDIUM*(AQUIFER_AREA*DEPTH_OF_AQUIFER*-1)	mm	This is the maximum volume of water the groundwater layer of the aquifer can hold. It is the product of the volume of the aquifer and the water holding capacity of the medium.
A	precipitation_available_per_month =	MAX((rain+snow_melt)*AQUIFER_AREA*Accumulation_TIME, 1)	mm	This is the water available for catchment rainfall retention each month. This is the sum of the rain and snow melt for the entire aquifer area.
A	retention_efficiency_(a1)" =	1/(1-RETENTION_FACTOR_a1)	dmnl	This step converts the normalized RETENTION_FACTOR_a1 to its proper scale as described by Zhang et al. (2008).
A	snow_melt_fraction =	(ET.mean_temp-T_ALL_SNOW)/(T_ALL_RAIN-T_ALL_SNOW)	dmnl	This fraction determines how much of the stored snow that melts each month as a result of the mean monthly temperature.
A	soil_dryness_index =	demand_limit_for_soil_storage/precipitation_available_per_month	dmnl	This is an index representing the soil saturation level each month when accounting for storage and incoming precipitation.
A	SOIL_STORAGE_CAPACITY =	WATER_HOLDING_CAPACITY_OF_MEDIUM *AQUIFER_AREA *DEPTH_OF_SHALLOW_SOIL_LAYER	mm	This is the maximum water storage capacity of the Soil_Storage stock. It is a product of the volume of the soil layer and its water holding capacity.

The Private Water Supply module

The Private Water Supply module represents the aquifers outside the municipal water system and it is used to model the water supply of private households not connected to the public water grid. The module is structurally very similar to the Public Water Supply module, with the following key differences:

- Groundwater extraction is not included in the computation of the water balance in the Private Water Supply module. The reasons are that reliable data on groundwater extraction at the household level are not available, and historic data on groundwater levels at the local scale is limited to a few locations. We therefore model only the "natural" groundwater dynamics for the region (that is, only including precipitation, evapotranspiration and groundwater discharge as driving processes) by calibrating the off-grid water balance model to the available groundwater data whilst assuming extraction patterns to remain constant.
- In contrast to the public water supply system, the primary factor limiting the water supply for off-grid households has historically been water quality, specifically chloride concentration, and not water quantity. Data shows that there is a linear effect of groundwater level on average chloride concentration in private wells ($p = 0.0035$).
  We use this relationship to estimate the mean groundwater chloride concentration on the island for different groundwater levels, and we use the historic frequency of households with above recommended chloride levels in their wells for different mean chloride concentrations to estimate the proportion of well sites with chloride concentrations exceeding the regulatory requirements for drinking water.

Key operations performed in the module:

1. Imports data on precipitation and evapotranspiration from the Climate module.
2. Calculates the water balance of the aquifer using a Budyko-based approach as described by Zhang et al. (2008).
3. Estimates the groundwater level in the aquifer as a function of the depth of the aquifer, the physical properties of the medium and the volume of water stored in the groundwater stock.
4. Estimates the mean chloride concentration of in private groundwater wells across the island as a function of the groundwater level.
5. Estimates the proportion of well sites with chloride levels exceeding the recommended limit values for drinking water and the maximum permissible chloride level for issuing of building permits.

The following section only covers structures related to the modeling of groundwater chloride concentration as the water balance structure of the Private Water Supply Module is largely identical to the structure already presented in the section describing the Public Water Supply module.

Estimating mean groundwater chloride concentration

Mean chloride concentration, Cl_μ, is calculated as a function of the groundwater level in the region, GWL_R, according to Equation A.20

$$Cl_\mu = MAX(0,\ \varepsilon_{Cl} * GWL_R + \beta_0) \qquad \text{Equation A.20.}$$

where ε_{Cl} is the effect of the groundwater level on the average chloride concentration (-121 mg/l/meter, SE = 39.6, p = 0.0035) and β_0 is the intercept of the linear function (-343 mg/l, SE = 169.6, p = 0,048). The MAX command ensures that chloride concentration does not go negative at extreme groundwater levels.

Estimating proportion of well-sites exceeding chloride limit values for drinking water and building permits

The proportion of private wells exceeding 100 mg/l Cl, P(Cl>100), and 300 mg/l Cl, P(Cl>300), for different values of Cl_μ is estimated by Equation A.21 and A.22, both calibrated to measurements of chloride concentration from 329 water samples taken between 2008 and 2020.

$$P(Cl > 100) = \varepsilon_{Cl100} * Cl_\mu + \beta_{100} \qquad \text{Equation A.21.}$$

$$P(Cl > 300) = \varepsilon_{Cl300} * \ln(Cl_\mu) + \beta_{300} \qquad \text{Equation A.22.}$$

ε_{Cl100} (9.45*10^-4 dmnl/mg/l, SE = 3.63*10^-4, p = 0.0283) and ε_{Cl300} (0.132 dmnl, SE = 0.00332, p = 0.00328) are the effect coefficients for the effect of Cl_μ on P(Cl>100) and P(Cl>300) respectively. B_{100} (0.291 dmnl/mg/l, SE = 0.0743, p = 0.00357) and β_{300} (-0.505 dmnl, SE = 0.174, p = 0.0174) are the intercept coefficients. Both proportions are bounded to values between zero and one in the simulation.

Graphical representation of the Private Water Supply Module

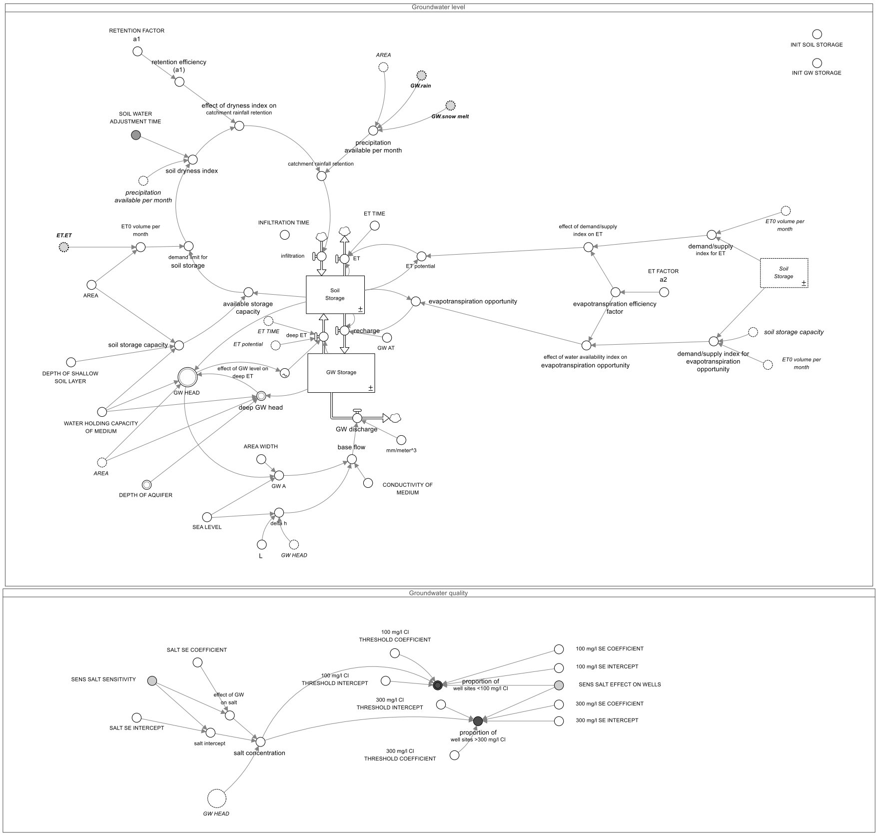


Figure A.7. Private Water Supply Module

Complete list of equations for the Public Water Supply Module

Abbreviations
S = Stock
F = Flow
C = Constant
A = Auxiliary
D = Data input
I = Initial value
T = Graphical/Lookup Table

Table A.8. Private Water Supply module - Constants

Private Water Supply Module				
Type	**Variable name**	**Unit**	**Value**	**Description**
C	AREA	meter^2	1	This is the area unit used for modeling groundwater storage in the off-grid aquifers.
C	100_mg/l_Cl_THRESHOLD_COEFFICIENT	dmnl/(mg/l)	0.000948	This is the effect coefficient in the linear function for calculating proportion_of_well_sites_<100_mg/l_Cl. It governs the slope of the linear function governing the effect of the groundwater salt concentration on the proportion of well-sites with chloride levels above the 100 mg/liter Cl threshold.
C	100_mg/l_Cl_THRESHOLD_INTERCEPT	dmnl/(mg/l)	0.291	This is the intercept coefficient in the linear function for calculating proportion_of_well_sites_<100_mg/l_Cl. It determines the intercept value of

				the linear function governing the effect of groundwater salt concentration on the proportion of well-sites with chloride levels above the 100 mg/liter Cl threshold.
C	100_mg/l_SE_COEFFICIENT	dmnl/(mg/l)	0.000363	This is the standard error of the “100 mg/l Cl THRESHOLD COEFFICIENT”.
C	100_mg/l_SE_INTERCEPT	dmnl/(mg/l)	0.0743	This is the standard error of the “100 mg/l Cl INTERCEPT COEFFICIENT”.
C	300_mg/l_Cl_THRESHOLD_COEFFICIENT	dmnl	0.132	This is the effect coefficient in the function for calculating the proportion_of_well_sites_>300_mg/l_Cl. It governs the slope of the function governing the effect of the groundwater salt concentration on the proportion of well-sites with chloride levels above the 300 mg/liter Cl threshold.
C	300_mg/l_Cl_THRESHOLD_INTERCEPT	dmnl	-0.505	This is the intercept coefficient in the function for calculating proportion_of_well_sites_>300_mg/l_Cl. It determines the intercept value of the linear function governing the effect of groundwater salt concentration on the proportion of well-sites with chloride levels above the 300 mg/liter Cl threshold.
C	300_mg/l_SE_COEFFICIENT	dmnl	0.0332	This is the standard error of the “300 mg/l Cl THRESHOLD COEFFICIENT”.
C	300_mg/l_SE_INTERCEPT	dmnl	0.174	This is the standard error of the “300 mg/l Cl INTERCEPT COEFFICIENT”.
C	SOIL_WATER_ADJUSTMENT_TIME	month	1	This is a time constant representing the time interval over which incoming precipitation is distributed (one month).
C	AREA_WIDTH	meter	1	This is the cross-section area of the groundwater flow used to calculate the groundwater baseflow by Darcy’s flow equation.
C	CONDUCTIVITY_OF_MEDIUM	m/month	0.13	This is the hydraulic conductivity of the bedrock medium. Source: SGU
C	DEPTH_OF_AQUIFER	meter	-56	This is the depth of the freshwater layer of the aquifer. Estimated based on studies by Dahlqvist et al. (2015).
C	DEPTH_OF_SHALLOW_SOIL_LAYER	meter	4	This is the estimated depth of the shallow soil layer, representing the depth that can be reached by plant roots (Schenk and Jackson, 2002).
C	ET_FACTOR_a2	dmnl	0.6	This is a model parameter representing the evapotranspiration efficiency of the ground cover of the municipal aquifer. Based on studies by Zhang et al. (2008). Calibrated value.
C	RETENTION_FACTOR_a2	dmnl	0.65	This is a model parameter representing the capacity of the soil surface to retain water. Based on studies by Zhang et al. (2008). Calibrated value.
C	ET_TIME	month	1	This is the evapotranspiration time.
C	GW_AT	month	1	This is the time for water to flow from the soil storage layer to the groundwater storage layer of the municipal aquifer.
C	INFILTRATION TIME	month	1	This is the time for precipitation to infiltrate to the soil storage stock.
I	INIT_GW_STORAGE	mm	5334	This is the volume of water stored per square meter of aquifer at the start of the simulation (January 2000). Calibrated values.
I	INIT_SOIL_STORAGE	mm	80	This is the volume of water stored per square meter in the soil layer at the start of the simulation (January 2000). Calibrated values.
C	L	meter	500	This is the distance to the sea from the wells used for calibrating the private aquifer module and it is also the distance to the sea for many of the households on Fårö. This is used to calculate the hydraulic gradient of off-grid aquifers, used to estimate the groundwater base flow using Darcy’s flow rate.
C	SALT_SE_COEFFICIENT	mg/liter/meter	39.6	This is the standard error of the effect_of_GW_on_salt variable used to determine how groundwater salt concentration varies with groundwater levels outside the public grid area.
C	SALT_SE_INTERCEPT	mg/liter	169.6	This is the standard error of the salt_intercept variable used to determine how groundwater salt concentration varies with groundwater levels outside the public grid area.
C	SENS_SALT_SENSITIVITY	dmnl	0	This is a parameter used for sensitivity analysis of the effect of groundwater levels on salt concentration. It takes values between -1.96 and +1.96 and is multiplied with the “SALT_SE_COEFFICIENT” and the “SALT_SE_INTERCEPT” in the equations for calculating effect_of_GW_on_salt and salt_intercept. This yields the 95 % confidence intervals groundwater salt concentration at any given groundwater level.
C	SENS_SALT_EFFECT_ON_WELLS	dmnl	0	This is a parameter used for sensitivity analysis of the effect of groundwater salt concentration on the proportion of well-sites above 100 or 300 mg/liter Cl. It takes values between -1.96 and +1.96 and is multiplied with the standard error of the intercept and coefficient variables in the equations for proportion_of_well-sites_<100_mg/l_Cl and proportion_of_well-sites_>300_mg/l_Cl. This gives the 95 % confidence intervals for the proportion of well-sites above or below the 100 and 300 mg/liter Cl threshold values at any given groundwater level.
C	SEA_LEVEL	meter	-9	This is the sea level in relation to the ground surface.
C	WATER_HOLDING_CAPACITY_OF_MEDIUM	mm/meter^3	100	This is the water holding capacity of the limestone making up the Fårö bedrock. This is in the lower range suggested by Wolff (1982), in line with previous

				studies on Fårö that have concluded that the groundwater storage capacity of the bedrock is generally low (Dahlqvist et al., 2015).

Table A.9. Private Water Supply module – Stocks and Flows

Private Water Supply Module				
Type	**Variable name**	**Equation**	**Unit**	**Description**
S	GW_Storage(t) =	GW_Storage(t - dt) + (recharge - GW_discharge - deep_ET) * dt	mm	The groundwater storage stock increases by recharge and is drained by GW_discharge and deep_ET.
F	Recharge =	(Soil_Storage-evapotranspiration_opportunity)/GW_AT	mm/month	Recharge is the flow of water from Soil Storage to Groundwater Storage. It is calculated as the difference between Soil Storage and the evapotranspiration opportunity any given month.
F	GW_discharge =	base_flow*"mm/meter^3"	mm/month	GW_discharge is equal to the baseflow.
F	deep_ET =	MIN(GW_Storage, ET_potential*effect_of_GW_level_on_deep_ET)/ET_TIME	mm/month	deep_ET represents evapotranspiration reaching deeper than the upper soil layer and is primarily caused by trees and plants with deep root systems. It is calculated as the minimum of the available GW_storage and the ET_potential multiplied by the effect of GW level on deep ET. This structure makes sure that evapotranspiration stops if the groundwater level falls beyond the reach of the plant root system.
S	Soil_Storage(t) =	Soil_Storage(t - dt) + (infiltration + deep_ET - recharge - ET) * dt	mm	Soil Storage is a stock that increases by infiltration and capillary rise caused by deep ET, and is drained by ET.
F	infiltration =	catchment_rainfall_retention/INFILTRATION_TIME	mm/month	Precipitation retained by the soil surface infiltrates to the soil layer.
F	ET =	MIN(ET_potential, Soil_Moisture_Storage)/ET_TIME	mm/month	Evapotranspiration is the minimum of the ET_potential and the available water in the Soil_Moisture_Storage stock.

Table A.10. Private Water Supply module – auxiliary calculations

Type	Variable name	Equation	Unit	Description
A	available_storage_capacity =	MAX(0, SOIL_STORAGE_CAPACITY - Soil_Storage)	mm	Available storage capacity represents the difference between the current amount of water stored in the soil and the maximum amount of water the soil can hold when saturated.
A	base_flow =	CONDUCTIVITY_OF_MEDIUM*GW_A*delta_h	meter^3/months	Base flow represents the continuous flow of grounwater towards the ocean. We assume this can be approximated using Darcy's flow equation (Hillel, 2004). Thus, the base flow is a function of the height of the groundwater column, the hydraulic gradient (delta_h) and the hydraulic CONDUCTIVITY_OF_THE MEDIUM.
A	catchment_rainfall_retention =	precipitation_available_per_month*effect_of_dryness_index_on_catchment_rainfall_retention	mm	Catchment rainfall retention is the precipitation withheld by the soil surface. It is influenced by the dryness index of the soil. When the soil is dry retention is higher and vice versa (Zhang et al., 2008).
A	DEEP_GW_HEAD =	DEPTH_OF_AQUIFER+ GW_Storage/WATER_HOLDING_CAPACITY_OF_MEDIUM/AREA	meter	This is the depth to the saturated groundwater zone.
A	delta_h =	(GW_HEAD-SEA_LEVEL)/L	dmnl	This is the hydraulic gradient between the groundwater head and sea level.
A	demand_limit_for_soil_storage =	available_storage_capacity+ET0_volume_per_month	mm	The theoretical maximum amount of rain that can be retained by the soil each month. This is the sum of the available storage capacity and the potential evapotranspiration (Zhang et al., 2008).
A	demand/supply_index_for_ET =	ET0_volume_per_month/Soil_Storage	dmnl	This index represents the relationship between the water demand from potential evapotranspiration and the supply/availability of water in the soil layer. It is used to calculate the evapotranspiration potential (Zhang et al., 2008).
A	demand/supply_index_for_evapotranspiration_opportunity =	(ET0_volume_per_month+SOIL_STORAGE_CAPACITY)/Soil_Storage	dmnl	This index represents the relationship between the demand side for water in the aquifer system (the sum of potential evapotranspiration and the maximum soil storage capacity) and the supply side (the amount of water already stored in the soil). The index is used to calculate the evapotranspiration_opportunity (Zhang et al., 2008).
A	effect_of_demand/supply_index_on_ET =	1+"demand/supply_index_for_ET" - (1+"demand/supply_index_for_ET"^evap	dmnl	This is a non-linear function for calculating the proportion of water stored in the soil that is partitioned to evapotranspiration each month. If the demand/supply index

		otranspiration_efficiency_factor)^(1/evapotranspiration_efficiency_factor)		is large, more water is allocated to evapotranspiration and vice versa (Zhang et al., 2008).
A	effect_of_dryness_index_on_catchment_rainfall_retention[GW_1] =	1+soil_dryness_index-(1+soil_dryness_index^"retention_efficiency_(a1)")^(1/"retention_efficiency_(a1)")	dmnl	This is a non-linear function for partitioning incoming precipitation between catchment rainfall retention and runoff. Rainfall retention increases with soil dryness (Zhang et al., 2008).
A	effect_of_GW_on_salt =	-121 + SALT_SE_COEFFICIENT*SALT_SENSITIVITY_SPECS	mg/liter/meter	This is the effect coefficient governing the how much a change in groundwater level impacts the average salt concentration in wells on the island. The estimate of the coefficient is -121, the SALT_SE_COEFFICIENT is the standard error of the coefficient, and the SALT_SENSITIVITY_SPECS is a parameter for adjusting the effect coefficient between -198 and -43 (+/- two sigmas).
A	effect_of_water_availability_index_on_evapotranspiration_opportunity =	1+demand/supply_index_for_evapotranspiration_opportunity-(1+demand/supply_index_for_evapotranspiration_opportunity^evapotranspiration_efficiency_factor)^(1/evapotranspiration_efficiency_factor)	dmnl	This is a non-linear function for calculating the proportion of water stored in the soil that is withheld in the soil layer (that is, it remains available for evapotranspiration) and what proportion that forms groundwater recharge (Zhang et al., 2008).
A	ET_potential =	Soil_Storage*effect_of_demand/supply_index_on_ET	mm	This is the proportion of water stored in the soil layer that is partitioned to evapotranspiration each month.
A	ET0_volume_per_month =	Evapotranspiration.ET*AREA	mm	This is the potential evapotranspiration per square meter each month.
A	evapotranspiration_efficiency_factor =	1/(1-ET_FACTOR_a2)	dmnl	This step converts the normalized ET_FACTOR_a2 to its proper scale as described by Zhang et al. (2008).
A	evapotranspiration_opportunity =	MAX(0, Soil_Moisture_Storage*effect_of_water_availability_index_on_evapotranspiration_opportunity)	mm	evapotranspiration_opportunity is the proportion of water stored in the soil that is withheld in the soil layer (that is, it is unavailable for recharge) each month.
A	GW_A =	(GW_HEAD-SEA_LEVEL)*AREA_WIDTH	meter^2	This is the cross-sectional area (A) of the Darcy's flow equation (Hillel, 2004) used to estimate the base_flow per square meter.
A	GW_HEAD =	DEEP_GW_HEAD+ Soil_Storage/WATER_HOLDING_CAPACITY_OF_MEDIUM/AREA)	meter	The groundwater head/level (meters sub surface) is modeled by calculating the height of a water column sitting on top of the bottom of the aquifer. The height of the column is the volume of water stored divided by the water holding capacity of the medium and the total surface area of the aquifer.
A	precipitation_available_per_month =	MAX((rain+snow_melt)*AQUIFER_AREA*Accumulation_TIME, 1)	mm	This is the water available for catchment rainfall retention each month. This is the sum of the rain and snow melt for the entire aquifer area.
A	proportion_of_well_sites_<100_mg/l_Cl" =	MAX(0, 1-(salt_concentration*(100_mg/l_Cl_THRESHOLD_COEFFICIENT + 100_mg/l_SE_COEFFICIENT*SENS_SALT_EFFECT_ON_WELLS) + (100_mg/l_Cl_THRESHOLD_INTERCEPT + 100_mg/l_SE_INTERCEPT*SENS_SALT_EFFECT_ON_WELLS)))	dmnl	This is the proportion of well-sites on Fårö that have a chloride level within the regulatory limit of 100 mg/liter Cl. It is calculated as a linear function of the average salt_concentration on the island. The model is based water samples from Fårö taken between 2012 and 2020. y = 1-(ax + b) Where y is the proportion of wells above 100 mg/l Cl, a is the Cl_threshold_coefficient, x is the salt_concentration and b is the 100_mg/l_Cl_threshold_intercept. The SE_Coefficient variables and the SENS_SALT_EFFECT_ON_WELLS variable are used for sensitivity analysis to explore how results react to alternative assumptions regarding the intercept and coefficient variables in the linear model.
A	retention_efficiency_(a1)" =	1/(1-RETENTION_FACTOR_a1)	dmnl	This step converts the normalized RETENTION_FACTOR_a1 to its proper scale as described by Zhang et al. (2008).
A	salt_concentration =	MAX(0.001, GW_HEAD*effect_of_GW_on_salt + salt_intercept)	mg/l	This is the average salt concentration of well-sites on Fårö. The salt concentration is modelled as a linear function of the groundwater level: y = ax + b Where y is the salt_concentration, a is the effect_of_GW_on_salt and b is the salt_intercept.

A	salt_intercept =	-343 + salt_sd_intercept*SALT_SENSITIVITY_SPECS	mg/l	This is the intercept coefficient of the salt_concentration function. The range of the salt_intercept is: Min: -674 Max: -11 Estimate: -343. The intercept value can be varied y adjusting the SALT_SENSITIVITY_SPECS parameter.
A	soil_dryness_index =	demand_limit_for_soil_storage/precipitation_available_per_month/SOIL_WATER_ADJUSTMENT_TIME	dmnl	This is an index representing the soil saturation level each month when accounting for storage and incoming precipitation.
A	SOIL_STORAGE_CAPACITY =	WATER_HOLDING_CAPACITY_OF_MEDIUM *AREA *DEPTH_OF_SHALLOW_SOIL_LAYER	mm	This is the maximum water storage capacity of the Soil_Storage stock. It is a product of the volume of the soil layer and its water holding capacity.

The Household Water Use module

The Household Water Use module consists of two subsectors: Housing and Water Demand. The former simulates the growth of the private housing sector on the island, driven by an expected future demand for new houses, the construction rate of new houses, and the sensitivity of housing demand and supply to housing prices.

In the Water Demand subsector, the total water demand for households with both private water and public water are calculated based on the average water use per capita in Sweden, the number of residents per household, the number of houses in use at any given time, the effect of temperature on water consumption, and a water use index that represents the effect of household affluence on water consumption.

Key operations performed in the Housing sector of the Household Water Use module:

1. Imports an estimate of historic and future housing demand based on historic data on summer house prices and summer houses sold on Gotland between 2000 and 2020 (SCB). The trend in demand growth seen in 2000 to 2020 is extrapolated to the period 2020 to 2050.
2. Estimates the number of potential buyers based on the historic housing demand, house construction rates and the house price index.
3. Calculates the house construction rate as a function of the number of potential buyers, house construction capacity, and the proportion of well sites with chloride levels exceeding the regulatory limits for building permits (P(Cl>100) from the Private Water Supply module).
4. Estimates the housing price index based on the sensitivity of house prices to the ratio between housing demand and supply.
5. Calculates the number of houses with private water and public water and estimates the number of these houses that are in use any given time based on the month of the year.

Key operations performed in the Water Demand sector of the Household Water Use module:

1. Calculates average water use per household for houses with private and public water. The calculation is based on the number of people per household, a baseline level of water use per capita, and the effect of temperature and the water use index.
2. Calculates the water use index as a function of household affluence, normalized to year 2000.
3. Summarizes the total household water use on the public grid and outside the public grid.

Housing dynamics

Houses on the island are categorized into two groups, I, houses with private water, $H_p(t)$, and houses connected to the public grid, $H_m(t)$. Both are represented as stock variables that increase by new constructions, $C_p(t)$ and $C_m(t)$, and decrease by demolitions, $D_p(t)$ and $D_m(t)$. In the base run settings, demolitions are compensated for by new constructions, assuming that the existing housing stock is continuously maintained. Any net addition of new houses is assumed to enter the $H_p(t)$ stock as there are strict restrictions on new connections to the public grid. The general formula for the two house stocks are

$$H_i(t) = H_i(t - dt) + (C_i - D_i)dt \qquad \text{Equation A.23}$$

$$D_i(t) = H_i(t) * \gamma_D \qquad \text{Equation A.24}$$

$$C_i(t) = D_i(t) + S_i(t) \qquad \text{Equation A.25}$$

where γ_D is a demolition fraction representing the fraction of the existing housing stock that is demolished each month, and $S_i(t)$ is the number of finished new houses sold per month of category i.

The rate of $S_i(t)$ is driven by a feedback loop with three stocks: potential buyers of new houses, B(t), houses under construction, H'(t), and house price index, $i_H(t)$. The feedback loop drives new house constructions according to Equations A.26 – A.37.

House construction rate

$$H'(t) = H'(t-dt) + (H_{st} - S_i)dt \qquad \text{Equation A.26}$$

$$S_i(t) = \frac{H\prime(t)}{\tau_C} \qquad \text{Equation A.27}$$

$$H_{st}(t) = \; H'_{st}(t) * \varepsilon_{Hst}(t) = MIN[B(t), H^*_{st}(t)] * [1 - P(Cl > 100)] \qquad \text{Equation A.28}$$

$$H^*_{st}(t) = \gamma_2 * i_H(t)\hat{}\eta_s \qquad \text{Equation A.29}$$

$H_{st}(t)$ is the number of new house construction projects started per month. T_C is the construction delay, representing the time it takes to build a new house, ε_{Hst} is the effect of the groundwater salt concentration on construction starts (with chloride levels above 100 mg/l building permits are not issued), $H^*_{st}(t)$ represents construction capacity per month given the size of the construction industry on the island, C_0 is the base construction capacity in year 2000, and η_s is the price elasticity of housing supply (assuming that higher house prices tend to increase supply by making house construction more profitable).

Potential buyers

$$B(t) = B(t-dt) + (B_a - B_s - B_l)dt \qquad \text{Equation A.30}$$

$$B_a(t) = \delta_{Hd} * \varepsilon_p \qquad \text{Equation A.31}$$

$$\varepsilon_p = i_H(t)\hat{}\eta_d \qquad \text{Equation A.32}$$

$$B_s(t) = MIN[h_s(t), B(t)]/\tau_d \qquad \text{Equation A.33}$$

$$B_l(t) = B(t)/\tau_3 \qquad \text{Equation A.34}$$

$B_a(t)$, $B_s(t)$, and $B_l(t)$ are the flows regulating the number of potential buyers, B(t), and represent new potential buyers added per month, the buyer's satisfaction rate per month and buyers leaving because they find a house somewhere else. $B_a(t)$ is the product of δ_{Hd}, which is a time dependent function representing the historic and projected housing demand, and ε_p, which is the effect of house prices on demand. ε_2 is calculated from the housing price index and the price elasticity of demand for houses, η_d. $B_s(t)$ is calculated as the minimum of $H_s(t)$ and B(t), divided by τ_d which is a delay time constant. Lastly, $B_l(t)$ is calculated as B(t) divided by τ_w which is the average waiting time a potential buyer is willing to wait for a house on Fårö. We assume that potential buyers will lose patients or find a house somewhere else after an average waiting time of 12 months.

Housing price index

The house price index is calculated by Equation A.35 and it is a measure of the relative house prices at any time t compared to the house prices in January 2000, adjusted for inflation by accounting for changes in consumer prices, CPI. The index is adjusted by the flow $\Delta i_H(t)$ according to Equation A.36 where i_H^* (t) is an indicated housing price index and τ_{rH} is the adjustment delay time for $i_H(t)$ to adjust to the indicated price level. The indicated price level is calculated by equation A.37 as a function of the ratio between housing demand and supply, r_H, and the price sensitivity to this ratio, η_{rH}.

$$i_H(t) = \frac{i_H(t-dt) + \Delta i_H * dt}{CPI} \qquad \text{Equation A.35}$$

$$\Delta i_H(t) = \frac{i^*_H(t) - i_H(t)}{\tau_{iH}} \qquad \text{Equation A.36}$$

$$i^*_H(t) = r_H{}^{\eta_{rH}} \qquad \text{Equation A.37}$$

Household water demand

Water demand per household

Average water demand per month is calculated on household level for both $H_p(t)$ and $H_m(t)$. Both categories consist of full-time and part-time households. The household water use for the types of is calculated according to Equation A.38.

$$\theta_{ij}(t) = q_H * r_{ij} * \varepsilon_{T_{MAX}} * i_\theta \qquad \text{Equation A.38}$$

θ_{ij} is the water demand per month for household category i (houses with private water or public water) and utilization type j (full-time or part-time), q_H is average water use per capita and month in Sweden, r_{ij} is the average number of residents per household of category i and type j, $\varepsilon_{T_{MAX}}$ is the effect of temperature on water use as calculated by Equation A.3, and i_θ is the water use index reflecting the positive correlation between affluence and water consumption (Höglund, 1999).

The water use index, i_θ, increases with household affluence. Due to lack of data on true household finances we approximate affluence by $i_H(t)$. As $i_H(t)$ increases, i_θ increases with a 12-month smoothed time delay. The delay represents the estimated time it takes to close the gap between the actual water use index and the new indicated water use index $i_\theta^*(t)$ after a change in affluence. This can be interpreted as the time for consumers to adjust their behaviors to their new level of affluence. The magnitude of change in the water use index for a given change in affluence is calculated according to Equation A.40 where $i_H(t=0)$ is the housing price index in year 2000, η_{iH} is the affluence elasticity of water use and $\tau_{\Delta i\theta}$ is the adjustment time of the water use index.

$$i_\theta(t) = i_\theta(t-1) * \Delta i_\theta * dt \qquad \text{Equation A.39}$$

$$\Delta i_\theta(t) = \frac{i_\theta^*(t) - i_\theta(t)}{\tau_{\Delta i\theta}} = \frac{\left(\frac{i_H(t)}{i_H(t=0)}\right)^{\eta_{iH}} - i_\theta(t)}{\tau_{\Delta i\theta}} \qquad \text{Equation A.40}$$

Total water use in the household sector

The total monthly water use by households on the public grid and households with private wells is the product of the number of houses in use and their average monthly water use (Equation A.41).

$$Q_i(t) = \sum[H_{ij}(t) * \delta_{u_{ij}} * \theta_{ij}(t)] \qquad \text{Equation A.41}$$

In Equation A.41, $Q_i(t)$ is the total water use of household category *i* (households with private water, p, or public water, m), $\delta_{u_{ij}}$ is a time-dependent function representing the fraction of houses of category *i* and type *j* (full-time or part-time houses) that are in use any given month of the year, and $\theta_{ij}(t)$ is the average water demand per household in category *i* and type *j* from Equation A.38.

Graphical representation of the Household Water Use Module

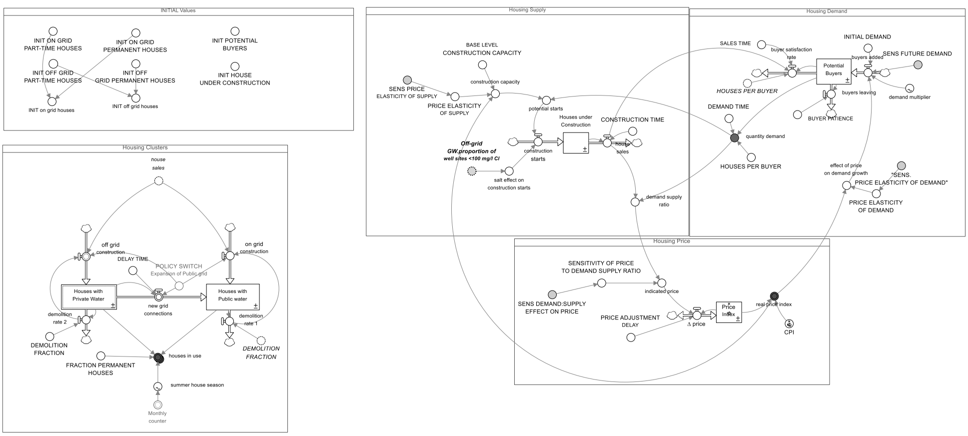


Figure A.8. Household Water Use Module - Housing

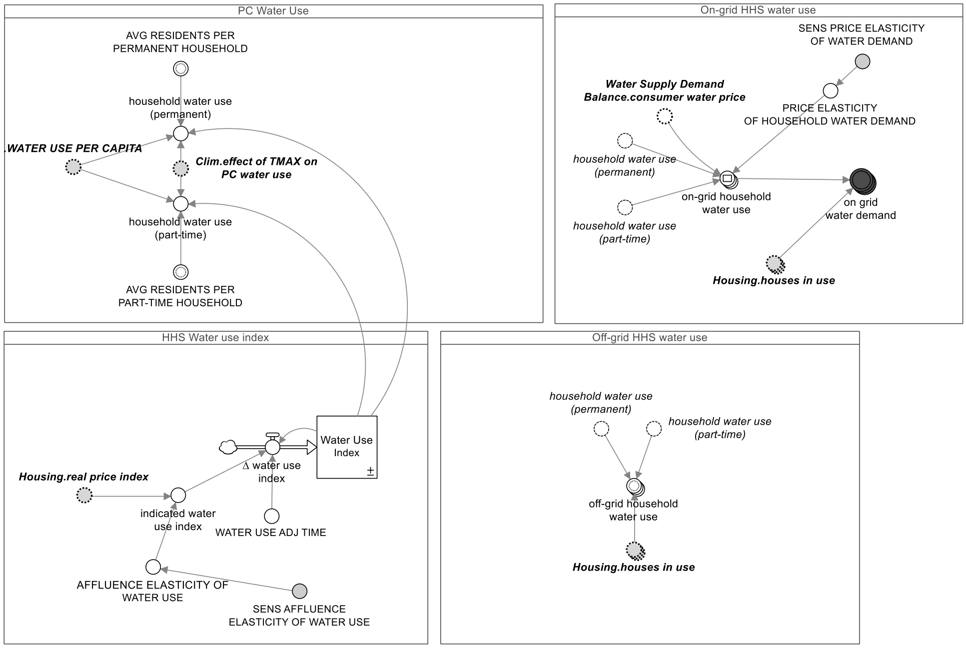


Figure A.9. Household Water Use Module – Water Demand

Complete list of equations for the Household Water Use Module

Abbreviations
S = Stock
F = Flow
C = Constant
A = Auxiliary
D = Data input
I = Initial value
T = Graphical/Lookup Table

Table A.11. Household Water Use module - Constants

Household Water Use Module – Housing				
Type	**Variable name**	**Unit**	**Value**	**Description**
C	SENS._PRICE_ELASTICITY_OF_DEMAND	dmnl	-0.5	This is a parameter for sensitivity analysis. Sets future price elasticity of housing demand.
C	BASE_LEVEL_CONSTRUCTION_CAPACITY	houses/month	3	This is the maximum number of new houses that can be built per month at the start of the simulation. Based on historic building permits issued on Fårö in the early 2000s.
C	BUYER_PATIENCE	month	12	This is the average time a potential house buyer will wait for a house on Fårö before leaving the Potential Buyers stock to find a house elsewhere.
C	CONSTRUCTION_TIME	month	12	This is the average time to build a house.
C	SALES_TIME	month	1	This is the time it takes to sell a newly constructed house.
C	DEMAND_ADJUSTMENT_TIME	month	1	
C	DEMOLITION_FRACTION	dmnl/month	1/100	This is the fraction of existing houses taken down each month.
C	FRACTION_PERMANENT_HOUSES	dmnl	0.4	This is the fraction of year round houses on Fårö.
C	HOUSES_PER_BUYER	houses/households	1	This is the number of houses owned per household.
I	INIT_HOUSE_UNDER_CONSTRUCTION	houses	40	This is the number of houses under construction at the start of the simulation (January 2000).
I	INIT_OFF_GRID_PART-TIME_HOUSES	houses	300	This is the number of off-grid part-time houses at the start of the simulation (January 2000).

I	INIT_OFF_GRID_PERMANENT_HOUSES	houses	300	This is the number of off-grid permanent houses at the start of the simulation (January 2000).
I	INIT_ON_GRID_PART-TIME_HOUSES	houses	43	This is the number of on-grid part-time houses at the start of the simulation (January 2000). In 2014 there were 74 on grid part-time houses (Sjöstrand et al., 2014) and between 2000 and 2014 the part-time house stock on Fårö increased by about 72 %. We assume the on-grid part time house stock follows the same trend as the total part time house stock, giving an initial on-grid part time house stock of 43 houses in 2000.
I	INIT_ON_GRID_PERMANENT_HOUSES	houses	6	This is the number of permanent houses on the public grid in January 2000 (Sjöstrand et al., 2014). Assumed to be constant between 2000 and 2019
I	INIT_POTENTIAL_BUYERS	households	3.35	This is the number of potential buyers at the start of the simulation (January 2000). Calibrated value.
I	INIT Price_Index =	dmnl	1	This is the house Price_Index at the start of the simulation (January 2000).
I	INITIAL_DEMAND	Household/month	4	This is the quantity demand for houses on Fårö at the start of the simulation (January 2000). Calibrated value.
C	PRICE_ADJUSTMENT_DELAY	month	36	House prices do not change instantaneously. This value determines how quickly price can change.
C	PRICE_ELASTICITY_OF_DEMAND =	dmnl	-0.5	This is the sensitivity of housing demand to changes in house prices. Estimate based on studies by Englund (2011). Range: -0.25 to -0.75.
C	PRICE_ELASTICITY_OF_SUPPLY =	dmnl	0.1	(International Monetary Fund. European Dept., 2015). Range: 0 to 0.5
C	SENS_DEMAND:SUPPLY_EFFECT_ON_PRICE	dmnl	1	This is a parameter for analyzing the sensitivity of simulation results to the effect of the housing demand:supply ratio on house prices.
C	SENS_FUTURE_DEMAND	dmnl	1	This is a parameter for analyzing the sensitivity of simulation results to the effect of assumptions about future housing demand.
C	SENS_PRICE_ELASTICITY_OF_SUPPLY	dmnl	0.1	This is a parameter for analyzing the sensitivity of simulation results to assumptions about housing price elasticity of supply.
C	DEMAND_TIME	month	1	This is the time unit of quantity demand.
C	AFFLUENCE_ELASTICITY_OF_WATER_USE =	dmnl	0.1	This is the sensitivity of household water use to change in household affluence. Range: 0.07 - 0.13 (Höglund, 1999)
C	AVG_RESIDENTS_PER_PART-TIME_HOUSEHOLD = 6 UNITS:	People/house	6	This is the average number residents per part-time household when these are in use. According to Region Gotland and SCB (2017) the average number is 6 people per household with a variation from 3 to 9 people.
C	AVG_RESIDENTS_PER_PERMANENT_HOUSEHOLD	People/house	1.83	This is the average number of people per permanent household on Fårö. Based on data from Brunner et al. (2014).
I	INIT Water_Use_Index	dmnl	1	This is an index representing the relative water use intensity of households compared to year 2000.
C	PRICE_ELASTICITY_OF_HOUSEHOLD_WATER_DEMAND =	dmnl	-0.43	The short-term price elasticity of water demand in is set to -0.43 (Arnaud, 2015).
Household Water Use Module – Water Demand				
C	SENS_AFFLUENCE_ELASTICITY_OF_WATER_USE	dmnl	0.1	This is a parameter used for assessing the sensitivity of simulation results to assumptions about how household affluence influence household water use in the future.
C	SENS_PRICE_ELASTICITY_OF_WATER_DEMAND	dmnl	-0.43	This is a parameter used for assessing the sensitivity of simulation results to assumptions about how water price influence household water use in the future.
C	WATER_USE_ADJ_TIME	month	12	This is the assumed time it takes for household water use to adjust to changes in water price and household affluence.

Table A.12. Household Water Use module – Stocks and Flows

Household Water Use Module - Housing				
Type	**Variable name**	**Equation**	**Units**	**Description**
S	Houses_under_Construction(t) =	Houses_under_Construction(t - dt) + (construction_starts - house_sales) * dt	houses	This is the number of houses under construction on Fårö. It increases by construction_starts and decreases by house_salses (assuming all produced houses are sold).
F	construction_starts =	potential_starts*salt_effect_on_construction_starts	houses/ month	The number of new house projects is equal to the number of potential_starts multiplied by the salt_effect_on_construction_starts. For our analysis we make the simplifying assumption that water quality is the only limiting factor to housing development on Fårö.
F	house_sales =	Houses_under_Construction/CONSTRUCTION_TIME	houses/ month	We assume all houses take the same time to build and thus house_sales each month is a fraction of the Houses_under_Construction.

S	Houses_with_Private_Water(t) =	Houses_with_Private_Water(t - dt) + (off_grid_construction - new_grid_connections - demolition_rate_2) * dt	houses	This stock represents the total number of houses on Fårö with private wells. It increases by off_grid_constructions and declines by either houses with private wells being connected to the public grid (=new_grid_connections) or by demolition_rate_2, which represents the aging and demolition of the house stock.
F	off_grid_construction =	IF POLICY_SWITCH_Expansion_of_Public_grid = 1 THEN 0 ELSE demolition_rate_2+house_sales	houses/ month	off_grid_construction is equal to the demolition_rate_2 (in our base case we assume continuous maintenance of the houses on Fårö so no houses are demolished due to aging) plus house_sales. If the POLICY_SWITCH in the equation is set to 1 we simulate a scenario where new constructions only occur in areas with public water supply.
F	new_grid_connections =	IF POLICY_SWITCH_Expansion_of_Public_grid = 1 THEN Houses_with_Private_Water/DELAY_TIME ELSE 0	houses/ month	In the base case scenario the POLICY SWITH in the equation is set to 0 so no new connections to the public grid occurs. If the POLICY SWITCH is set to 1 then an expansion of the public grid can be simulated by allowing houses to flow from Houses_with_Private_Water to Houses_with_Public_Water at a chosen rate.
F	demolition_rate_2 =	Houses_with_Private_Water*DEMOLITION_FRACTION	houses/ month	Houses_with_Private_Water degrades and are demolished at a set fractional rate.
S	Houses_with_Public_water(t) =	Houses_with_Public_water(t - dt) + (on_grid_construction + new_grid_connections - demolition_rate_1) * dt	houses	This is the stock of houses connected to the municipal grid. It increases by new constructions and new connections from the Houses_with_Private_Water stock. It declines by demolitions.
F	on_grid_construction =	IF POLICY_SWITCH_Expansion_of_Public_grid = 1 THEN demolition_rate_1+house_sales ELSE demolition_rate_1	houses/ month	on_grid_constructions is the addition of houses to the Houses_with_Public_Water stock by construction. If the POLICY_SWITCH is set to 1 then this means new houses are allowed to be built on the public grid and the flow will equal demolition_rate_1 (replacing old houses) and new house_sales. If the switch is set to 0 no new houses are added, only the degraded ones are replaced.
F	demolition_rate_1 =	Houses_with_Public_water*DEMOLITION_FRACTION	houses/ month	Houses_with_Public_Water degrades and are demolished at a set fractional rate.
S	Potential_Buyers(t) =	Potential_Buyers(t - dt) + (buyers_added - buyer_satisfaction_rate - buyers_leaving) * dt	households	Potential_Buyers represent households looking for a house or summerhouse on Fårö. The stock increases by the flow buyers_added, representing new households becoming interested in buying a house on Fårö. It decreases by buyers either finding and buying a house (=buyers_satisfaction_rate) or leaving because they find a house somewhere else (=buyers_leaving).
F	buyers_added =	IF TIME > 252 THEN INITIAL_DEMAND*demand_multiplier*effect_of_price_on_demand_growth *SENS_FUTURE_DEMAND ELSE INITIAL_DEMAND*demand_multiplier*effect_of_price_on_demand_growth	households/month	The number of buyers_added each month is a product of the INITIAL_DEMAND, the demand_multiplier and the effect_of_price_on_demand_growth. This allows the demand for housing to adjust to changes in house prices. Between January 2000 and December 2019 the demand_multiplier is estimated from historic data on building permits applications and the genera trend in building permit applications is extrapolated into the future to estimate future demand. The SENS_FUTURE_DEMAND parameter is used to test the sensitivity of simulation results to changes in our assumptions about future demand.
F	buyer_satisfaction_rate =	MIN(house_sales/HOUSES_PER_BUYER,Potential_Buyers/DEMAND_ADJUSTMENT_TIME)	households/month	Ponetial_Buyers leave the stock at the same rate as new houses are constructed.
F	buyers_leaving =	Potential_Buyers/BUYER_PATIENCE	households/month	We assume potential house buyers will leave the system after a set waiting time because they find a house somewhere else than on Fårö.

S	Price_Index(t) =	Price_Index(t - dt) + (Δ_price) * dt	dmnl	The Price_Index represents the relative house price on Fårö normalized to year 2000.
F	Δ_price =	(indicated_price-Price_Index)/PRICE_ADJUSTMENT_DELAY	dmnl/months	The Price_Index adjusts to indicated_price with a set time delay.
Household Water Use Module – Water Demand				
S	Water_Use_Index(t) =	Water_Use_Index(t - dt) + (Δ_water_use_index) * dt	dmnl	This is an index representing the relative water use intensity of households compared to year 2000. The index is influenced by consumer water price and the affluence level of the households. Higher prices tend to reduce water use and higher affluence tend to increase water use (Höglund, 1999).
F	Δ_water_use_index =	(indicated_water_use_index-Water_Use_Index)/WATER_USE_ADJ_TIME	dmnl/month	The change in Water_Use_Index is modelled by a gap closing function where the gap between indicated_water_use_index and the Water_Use_Index is closed over a set adjustment time.

Table A.13. Household Water Use module – Auxiliary Calculations

Household Water Demand Module - Housing				
Type	**Variable name**	**Equation**	**Unit**	**Description**
A	construction_capacity =	BASE_LEVEL_CONSTRUCTION_CAPACITY*(real_price_index/INIT(real_price_index))^ PRICE_ELASTICITY_OF_SUPPLY	house/month	This is the building capacity of the housing sector on the island. It adjusts to price changes. If prices go up it becomes more attractive for construction companies to build more so construction capacity goes up.
A	demand_supply_ratio =	quantity_demand/house_sales	dmnl	This is the ratio of quantity demand and quantity supply on the housing market.
A	effect_of_price_on_demand_growth =	(real_price_index/INIT(real_price_index))^PRICE_ELASTICITY_OF_DEMAND	dmnl	Demand for housing responds to changes in house prices according to a set price elasticity of demand.
A	houses_in_use[Permanent, Ongrid] =	Houses_with_Public_water*FRACTION_PERMANENT_HOUSES	houses	The number of permanent on-grid houses in use.
A	houses_in_use[Permanent, Offgrid] =	Houses_with_Private_Water*FRACTION_PERMANENT_HOUSES	houses	The number of permanent off-grid houses in use.
A	houses_in_use[Parttime, Ongrid] =	Houses_with_Public_water*(1-FRACTION_PERMANENT_HOUSES)*summer_house_season	houses	The number of part-time on-grid houses in use.
A	houses_in_use[Parttime, Offgrid] =	Houses_with_Private_Water*(1-FRACTION_PERMANENT_HOUSES)*summer_house_season	houses	The number of part-time off-grid houses in use.
A	indicated_price =	1*(demand_supply_ratio^SENSITIVITY_OF_PRICE_TO_DEMAND_SUPPLY_RATIO)	dmnl	This is the equilibrium price as set by the demand-supply ratio_ratio. The actual price will reach this value after a delay specified by the price_change_delay.
A	INIT_off_grid_houses =	INIT_OFF_GRID_PERMANENT_HOUSES+INIT_OFF_GRID_PART-TIME_HOUSES	houses	This is the total number of off-grid houses in 2000.
A	INIT_on_grid_houses =	INIT_ON_GRID_PART-TIME_HOUSES + INIT_ON_GRID_PERMANENT_HOUSES	houses	This is the total number of on-grid houses in 2000.
A	potential_starts =	MIN(quantity_demand/START_TIME, construction_capacity)	houses/month	The number of new house construction projects possible each month if water quality limitations are not accounted for.
A	quantity_demand =	Potential_Buyers*HOUSES_PER_BUYER	houses	The quantity demand for houses.
A	real_price_index =	Price_Index/CPI	dmnl	The house price index normalized to prices in year 2000.
A	salt_effect_on_construction_starts =	IF TIME < 84 THEN 1 ELSE SMTHN(Off-grid_GW.proportion_of_well_sites_<100_mg/l_Cl, 12, 1)/ INIT(SMTHN(Off-grid_GW.proportion_of_well_sites_<100_mg/l_Cl, 12, 1))	dmnl	Before time 84 (2008) groundwater chloride levels did not influence building permits. After time 84 a building permits are not issued for housing projects where the groundwater chloride concentration exceeds 100 mg/l Cl. However, the water sample can be taken any time from May to October so we use the smoothed yearly salt concentration to capture that potential house builders can take multiple water samples across the year and use the "best" one when they are applying for a building permit.

				On the aggregate level, we assume that the construction starts will correlate negatively with the number of well-sites exceeding the 100 mg/l Cl threshold and that the effect of salt levels on construction starts is normalized to the average salt concentration in year 2000.
		Household Water Demand Module – Water Demand		
A	household_water_use_(part-time) =	.WATER_USE_PER_CAPITA* Clim.effect_of_TMAX_on_PC_water_use* Water_Use_Index* "AVG_RESIDENTS_PER_PART-TIME_HOUSEHOLD"	meter^3/month/houses	The average water use per part-time household. Influenced by temperature, the water use index and the number of residents per household.
A	household_water_use_(permanent) =	.WATER_USE_PER_CAPITA* Clim.effect_of_TMAX_on_PC_water_use* Water_Use_Index* AVG_RESIDENTS_PER_PERMANENT_HOUSEHOLD	meter^3/month/houses	The average water use per permanent household. Influenced by temperature, the water use index and the number of residents per household.
A	indicated_water_use_index =	(Housing.real_price_index/INIT(Housing.real_price_index))^AFFLUENCE_ELASTICITY_OF_WATER_USE	dmnl	This is the indicated water use index when house prices change. The actual water use index will reach the indicated water use index after a time delay.
A	off-grid_household_water_use"[Permanent] =	Housing.houses_in_use[Permanent,Offgrid]*household_water_use_(permanent)	meter^3/month	The total water use of all off-grid permanent households on the island.
A	off-grid_household_water_use [Parttime] =	Housing.houses_in_use[Parttime,Offgrid]*household_water_use_(part-time)	meter^3/month	The total water use of all off-grid part-time households on the island.
A	on_grid_water_demand[Permanent] =	Housing.houses_in_use[Permanent,Ongrid]*on-grid_household_water_use[Permanent]	meter^3/month	The total water use of all on-grid permanent households on the island.
A	on_grid_water_demand[Parttime] =	Housing.houses_in_use[Parttime,Ongrid]*on-grid_household_water_use[Parttime]	meter^3/month	The total water use of all on-grid part-time households on the island.
A	on-grid_household_water_use[Permanent] =	household_water_use_(permanent)* SMTH1((Water_Imports.consumer_water_price/INIT(Water_Imports.consumer_water_price))^PRICE_ELASTICITY_OF_HOUSEHOLD_WATER_DEMAND, 12)	meter^3/month/house	The average water use of on-grid permanent households.
A	on-grid_household_water_use[Parttime] =	household_water_use_(part-time)* SMTH1((Water_Imports.consumer_water_price/INIT(Water_Imports.consumer_water_price))^PRICE_ELASTICITY_OF_HOUSEHOLD_WATER_DEMAND, 12)	meter^3/month/house	The average water use of on-grid part-time households.

The Tourist Water Use module

Water use in the tourist sector is modeled based on the number of tourists at the island any given month and the average water use per guest night. Growth in the tourist sector is driven by investments in new bed capacity by the hotel companies on the island. Investment decisions are made based on historic capacity utilization and water availability. If water availability on the public grid is high more facilities will be connected to the grid, but if water availability on the grid is low the increase in bed capacity will depend on the chloride concentration of the groundwater. If chloride concentrations are high, fewer building permits will be issued and fewer new tourist facilities will be built. The number of tourists any given month, and the fractions of facilities connected to the public water grid governs the public water use of the sector.

Key operations performed in the module:

- Modeling the growth in bed capacity using a three-stock ageing chain of installed beds, beds on order and beds planned. Beds can be added to the plan every three years and then flow through the chain in batches representing yearly investments in new capacity.
- Modeling the decision rules for adding new beds to the plan based on historic capacity utilization and water availability.
- Modeling decision rules for yearly investments that move beds from the stock of beds planned to beds ordered and later beds installed, and for selecting the water source the new beds will rely on.
- Calculating the number of tourists on the island as a function of the bed capacity installed, the duration of the tourist season and the destination attractiveness of the island.
- Estimating the number of day-trip visitors on the island.
- Calculating the total public water use by the tourist sector as the sum of the water use by tourists on the public grid and day-trip visitor water use.

Bed capacity dynamics

BP(t), BO(t) and BI(t) represent the stocks beds planned, beds on order and installed beds. They are connected by the flows $TB_p(t)$, $TB_o(t)$ and $TB_i(t)$, representing the processes of adding new tourist beds to the plan, ordering new tourist beds, and installing new tourist beds (Equation A.42 – A.44).

$$BP(t) = BP(t-dt) + (TB_p - TB_o)dt \qquad \text{Equation A.42.}$$

$$BO(t) = BO(t-dt) + (TB_o - TB_i)dt \qquad \text{Equation A.43.}$$

$$BI(t) = BI(t-dt) + TB_i * dt \qquad \text{Equation A.44.}$$

$TB_p(t)$ is calculated from Equation A.45, where $BI_{gap}(t)$ is the difference between the desired number of beds installed, $BI^*(t)$, and the total number of beds already in the ageing chain (Equation A.46). This gap is closed by a pulse function where, at a time interval of τ_{BI}, the gap is multiplied by ε_{Bp} if $\varepsilon_{Bp} > 0$, or multiplied by (1-P(Cl>100)) if $\varepsilon_{Bp} = 0$. ε_{Bp} is a nonlinear function that can take on values between zero and one, and it represents the effect water scarcity has on the willingness of the municipality to grant building permits for new tourist facilities. When water scarcity is severe ε_{Bp} will limit the number of new beds added to the plan and force the tourist sector to drill private wells instead. At this point, the limiting factor to expansion is the chloride concentration of the groundwater, which is captured by multiplying $BI_{gap}(t)$ by the proportion of well sites that have chloride concentrations within the regulatory limits for building permits, (1-P(Cl>100)).

$$TB_p(t) = \begin{cases} PULSE(BI_{gap}(t) * \varepsilon_{Bp}(t), \tau_{BI}) & , \ \varepsilon_{Bp}(t) > 0 \\ PULSE[BI_{gap}(t) * (1 - P(Cl > 100)), \tau_{BI}] & , \ \varepsilon_{Bp}(t) = 0 \end{cases} \qquad \text{Equation A.45.}$$

$$BI_{gap}(t) = BI^*(t) - \big(BP(t) + BO(t) + BI(t)\big) \qquad \text{Equation A.46.}$$

$BI^*(t)$ is determined by the level of capacity utilization, U(t), the tourist sector has experienced in the past seasons. U(t) is calculated as the number of tourists, T(t), divided by BI(t) (Equation A.47).

The number of tourists any given month is calculated by Equation A.48, where the net increase of tourists each month, $\Delta T(t)$, is a goal-gap function that works to close the gap between T(t) and the target number of tourists, $T^*(t)$, over a set adjustment time, τ_{Ta}. $T^*(t)$ is the product of the installed bed capacity and the time-dependent function δ_{season} which takes on values between zero and one depending on the month of the year (Equation A.49).

$$U(t) = T(t)/B(t) \qquad \text{Equation A.47.}$$

$$T(t) = T(t-dt) + \Delta T * dt \qquad \text{Equation A.48.}$$

$$\Delta T(t) = \frac{T(t) - T^*(t)}{\tau_{Ta}} = \frac{T(t) - (BI(t) * \delta_{season})}{\tau_{Ta}} \qquad \text{Equation A.49.}$$

U^*, is a fractional value representing the capacity utilization level required for the tourist sector to make investments in additional capacity. If U(t), exceeds U^*, the sector will update $BI^*(t)$ by increasing it by γ_{beds} bed units. The speed of the update is determined by the perception delay τ_{BI*}.

$$BI^*(t) = SMTHN\big((BI(t) + \gamma_{beds}) * u, \tau_{BI^*}, 24\big) \qquad \text{Equation A.50.}$$

In Equation A.50, SMTHN is an information delay that makes sure $BI^*(t)$ is gradually updated, and *u* is a variable that takes on a value of zero if $U(t) < U^*$ and a value of one if $U(t) > U^*$.

The flows $TB_o(t)$ and $TB_i(t)$ are calculated according to Equations A.51 and A.52. Ordering of new beds happen according to a yearly ordering cycle (τ_{BO}) as long as *u* equals 1. The MIN function in Equation A.51 ensures the size of the order never exceeds 75 beds per year. $TB_i(t)$ is a delayed version of $TB_o(t)$ with a time delay of $\tau_{B_{new}}$ months, representing the time it takes to build the new tourist facilities.

$$TB_o(t) = PULSE\big(MIN(75, BP(t)), \tau_{BO}\big) * u \qquad \text{Equation A.51.}$$

$$TB_i(t) = TB_o\big(t - \tau_{B_{new}}\big) \qquad \text{Equation A.52.}$$

Water source of new bed capacity

New tourist facilities can either be connected to the public water grid or have a private well. Which one of these the new facility gets depends on the level of water scarcity experienced by the municipal water supply system. If water scarcity is severe (ε_{Bp} is close to one) more of the new tourist facilities will have private wells and vice versa. The number of tourist beds relying on public vs. private water is calculated by equations A.53 and A.54 where $BI_m(t)$ and $BI_p(t)$ are the stocks of installed beds with public water and installed beds with private water respectively.

$BI_m(t) = BI_m(t - dt) + (TB_i * (1 - \varepsilon_{Bp}))dt$ Equation A.53.

$BI_p(t) = BI_p(t - dt) + (TB_i * \varepsilon_{Bp})dt$ Equation A.54.

Total public water use of the tourist sector

Total public water use in the tourist sector, $Q_T(t)$, is the sum of: the total water use from tourists staying in facilities with public water, $\theta_{Tm}(t)$, the activity water use from tourists staying in facilities with private wells (activity water use is water use not occurring as at the accommodation, e.g. in restaurants, beach showers, etc.), $\theta_{Tp}(t)$, and the activity water use of day trip visitors, $\theta_V(t)$.

$Q_T(t) = \theta_{Tm}(t) + \theta_{Tp}(t) + \theta_V(t)$ Equation A.55.

$\theta_{Tm}(t) = T(t) * \frac{BI_m(t)}{BI_m(t)+BI_p(t)} * q_T * \varepsilon_{T_{MAX}} * i_\theta$ Equation A.56.

$\theta_{Tp}(t) = T(t) * \frac{(1-BI_m(t))}{BI_m(t)+BI_p(t)} * q_T * \varepsilon_{T_{MAX}} * i_\theta$ Equation A.57.

$\theta_V(t) = V(t) * q_V * \varepsilon_{T_{MAX}}$ Equation A.58.

$V(t) = \frac{BI(t)}{BI(t=0)} * V(t = 0) * \delta_{season}$ Equation A.59.

In the above equations q_T is the baseline water use per guest night and q_V is the baseline water use per daytrip visitor. We assume the number of visitors, V(t), to grow in line with the tourist sector, BI(t), and to follow the same seasonal pattern as the tourists, δ_{season}. We also assume that temperature and affluence have the same effect on tourist water use as they do on household water use.

Graphical representation of the Tourist Water Use Module

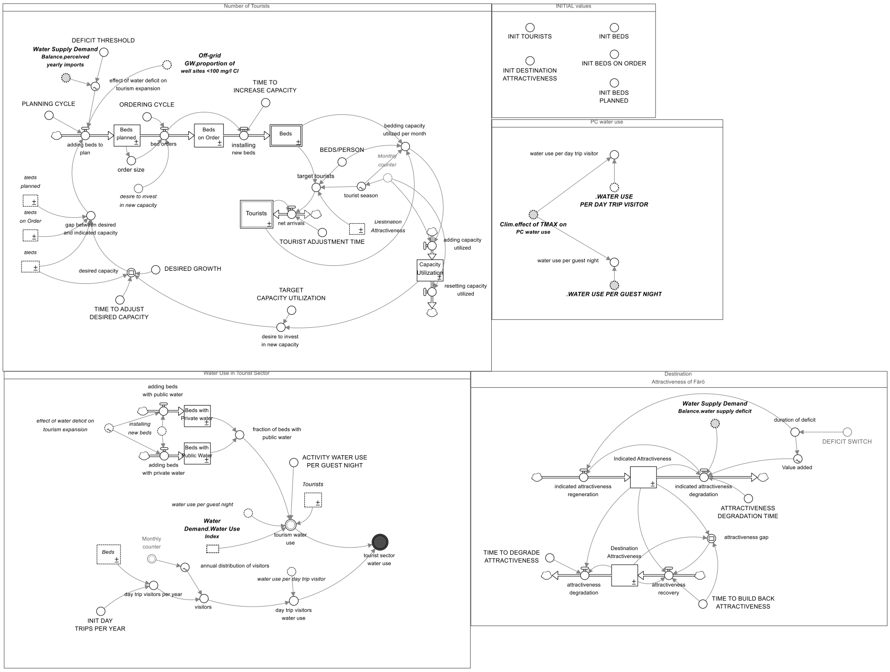


Figure A.10. Tourism Module

Complete list of equations for the Tourist Water Use Module

Abbreviations
S = Stock
F = Flow
C = Constant
A = Auxiliary
D = Data input
I = Initial value
T = Graphical/Lookup Table

Table A.14. Tourist Water Use module – Constants

Type	Variable name	Unit	Value	Description
C	ATTRACTIVENESS_DEGRADATION_TIME	month	1/30	This is the time it takes for the "indicated attractiveness" of Fårö as a destination to degrade when water supplies are insufficient.
I	INIT_DESTINATION_ATTRACTIVENESS	dmnl	1	This is the normal attractiveness of Fårö as a destination. Can tale values between 0 and 1.
C	BEDS/PERSON	bed/person	1	This is the number of installed beds utilized per tourist.
C	DESIRED_GROWTH	beds	235	This is how many beds the tourist sector plans for in each planning cycle. A planning cycle lasts for three years and each year the tourist sector can build about 8-12 new tourist houses. The number of houses that can be built per year is limited by the time it takes to build each house and the start of the tourist season. The new houses need to be finished by to the start of the tourist season and this means that on average 10 new houses can be added per year. Each house has on average 4-9 beds, giving a desired growth per planning cycle of 120 - 270 beds depending on what type of houses are built. Source: Rolf Lindvall, Sudersand Resort, personal communication 2020-10-15.
C	DESTINATION_ATTRACTIVENESS	dmnl	1	This is a measure representing how attractive Fårö is as a holiday destination. It can take values between 0 and 1, zero being very low and one being the high level of attractiveness experienced in the last 20 years. At a value of one bed capacity is the limiting factor to the number of tourists visiting Fårö every year. As our base case we

				assume destination attractiveness is constant throughout our simulation but other assumptions can be introduced by adjusting the value of the parameter.
I	INIT_BEDS	beds	290	This is the number of beds at the start of the simulation period (January 2000). Source: Rolf Lindvall, Sudersand Resort, personal communication 2020-10-15.
I	INIT_BEDS_OR_ORDER	beds	90	This is the number of beds on order at the start of the simulation (January 2000)
I	INIT_BEDS_PLANNED	beds	180	This is the number of beds planned at the start of the simulation (January 2000).
I	INIT BEDS_WITH_PRIVATE_WATER =	beds	0	This is the number of tourist beds with their own water wells at the start of the simulation (January 2000).
I	INIT BEDS_WITH_PUBLIC_WATER	beds	290	This is the number of tourist beds with public water at the start of the simulation (January 2000).
I	INIT_CAPACITY_UTILIZATION	dmnl	1	This is the capacity utilization at the start of the simulation (January 2000)
I	INIT_TOURISTS	people	0	This is the number of tourists on Fårö at the start of the simulation (January 2000).
I	INIT_DAYTRIPS_PER_YEAR	people	1000	This is the number of day trip visitors per year at the start of the simulation (January 2000).
C	ORDERING_CYCLE	month	12	This is how often new beds are ordered to the tourist sector (Rolf Lindvall, Sudersand Resort, 2020-10-15).
C	PLANNING_CYCLE	month	36	This is how frequently Beds planned is adjusted. According to Sudersand Resort (Rolf Lindvall, Sudersand Resort, 2020-10-15), they have a planning cycle of "a couple of years" years, for which they plan upcoming expansions and investments.
C	ACTIVITY_WATER_USE_PER_GUEST NIGHT	Meter^3/person/month	0.6	This is the estimated water use per guest night from activities not related to accommodation. Even tourist cottages and hotel rooms with their own wells will consume part of the water from their stay using municipal water sources (showers, food and drink, activities, etc.). (Gossling et al., 2012).
C	DEFICIT_THRESHOLD	Meter^3/year	650	This is a parameter representing the sensitivity of the municipality to water scarcity. It can be interpreted as the amount of water that can be transported each year before limitations are imposed on tourism expansion.
C	TARGET_CAPACITY_UTILIZATION	0.92	dmnl	This is the desired level of capacity utilization of the tourist sector. If the capacity utilization the previous year is above the TARGET_CAPACITY_UTILIZATION this will trigger a desire to invest in new beds. Calibration against observed capacity investments has been used to achieve an estimated value of 0.92.
C	TIME_TO_ADJUST_DESIRED_CAPACITY	month	20.4	This is the time for "desired capacity" to update to the level of “capacity utilization”. A lower value will give more vigorous reactions to changes in "capacity utilization” and a higher value will give a more dampened response.
C	TIME_TO_BUILD_BACK_ATTRACTIVENESS	month	12	This is the time delay to build back destination attractiveness following a degradation caused by water supply failure.
C	TIME_TO_DEGRADE_ATTRACTIVENESS	month	1	This is the time for destination attractiveness to degrade in case of a water supply failure.
C	TIME_TO_INCREASE_CAPACITY	month	12	This is the time to build and install new bed capacity (Rolf Lindvall, Sudersand Resort, 2020-10-15).
C	TOURIST_ADJUSTMENT_TIME	month	0.5	This is how long time it takes for tourists to arrive/depart from Fårö when the tourist season starts/ends.

Table A.15. Tourist Water Use module – Stocks and Flows

Type	Variable name	Equation	Unit	Description
S	Beds(t) =	Beds(t - dt) + (installing_new_beds) * dt	bed	This is the stock of installed beds available for tourists. The number of beds increase by the flow installing_new_beds
F	installing_new_beds =	DELAY(bed_orders, TIME_TO_INCREASE_CAPACITY)	bed/month	Installing_new_beds drain the stock of Beds_on_Order. The rate is equal to the flow bed_orders but with a fixed time delay (represented by TIME_TO_INCREASE_CAPACITY). The delay represents the construction time to build new facilities.
S	Beds_on_Order(t) =	Beds_on_Order(t - dt) + (bed_orders - installing_new_beds) * dt	bed	This stock represents beds under construction. It increases by the inflow of bed_orders and decreases by the outflow installing_new_beds.
F	bed_orders =	PULSE(ORDER_SIZE, 1, ORDERING_CYCLE) *desire_to_invest_in_new_capacity	bed/month	bed_orders represent the ordering of new beds by the tourist facility. It drains the stock of Beds_Planned and ads to the stock of Beds_on_Order. Orders are done according to a fixed ordering cycle (once per year). New orders only occur if there is a desire_to_invest_in_new_capacity, which depends on the level of capacity utilization in the last year.
S	Beds_planned(t) =	Beds_planned(t - dt) + (adding_beds_to_plan - bed_orders) * dt	bed	Beds_Planned is a stock variable representing the planned bed capacity growth for the coming three

				years by the tourist facilities. More beds are added to the plan by the flow adding_beds_to_plan and the stock is drained by bed_orders as new beds are being ordered.
F	adding_beds_to_plan =	IF effect_of_water_deficit_on_tourism_expansion > 0 THEN PULSE(gap_between_desired_and_indicated_capacity* effect_of_water_deficit_on_tourism_expansion, 1, PLANNING_CYCLE) ELSE PULSE(gap_between_desired_and_indicated_capacity* "Off-grid_GW"."proportion_of_well_sites_<100_mg/l_Cl", 1, PLANNING_CYCLE)	bed/month	The tourist facilities on Fårö plan for capacity expansion following a three year planning cycle. Every three years (= the duration of the PLANNING_CYCLE) the desired capacity is compared to the installed capacity. If there are no restrictions on tourism expansion imposed by the municipality then the number of new beds added to the stock of Beds_planned will equal the gap_between_desired_and_indicated_capacity. If water scarcity on the municipal grid increases, then the effect_of_water_deficit_on_tourism_expansion will cause only a fraction of the gap_between_desired_and_indicated_capacity to be installed. If water scarcity is severe, no new beds will be allowed by the municipality. In this situation we assume the tourist facilities will drill their own wells instead of relying on the public grid to supply water to their new facilities (Rolf Lindvall, Sudersand Resort, personal communication 2020-10-15). When this occurs, the number of new beds added to the Beds_planned stock each planning cycle will be limited by the proportion_of_well_sites < 100 mg/l Cl. In other words, groundwater chloride concentration becomes the limiting factor to tourism expansion.
S	Beds_with_Private_water(t) =	Beds_with_Private_water(t - dt) + (adding_beds_with_public_water) * dt	bed	Beds_with_Private_water represents the number of installed tourist beds that rely on private wells for their water supply. This is a subset of the total number of installed Beds stock.
F	adding_beds_with_private_water =	installing_new_beds*effect_of_water_deficit_on_tourism_expansion	bed/month	This flow adds new beds to the stock of Beds_with_Private_water and is a co-flow to the installing_new_beds flow that is active when municipal restrictions on grid expansion force tourist facilities to drill own wells.
S	Beds_with_Public_Water(t) =	Beds_with_Public_Water(t - dt) + (adding_beds_with_private_water) * dt	bed	Beds_with_Public_water represents the number of installed tourist beds that rely on the municipal grid for their water supply. This is a subset of the total number of installed Beds stock.
F	adding_beds_with_public_water =	installing_new_beds*(1-effect_of_water_deficit_on_tourism_expansion)	bed/month	This flow adds new beds to the stock of Beds_with_Public_water and is a co-flow of the installing_new_beds flow. As water deficit increases the fraction of new beds added that have public water decreases.
S	Capacity_Utilization(t) =	Capacity_Utilization(t - dt) + (adding_capacity_utilized - resetting_capacity_utilized) * dt	dmnl	The level of capacity utilization is modelled as a stock that can take values between 0 and 1. In month 7 each year the level of the stock is recalculated so it takes on a new value representing the fraction of tourist beds in use at the peak of the tourist season that year. Doing so the Capacity_Utilization stock is always updated based on the tourist occupancy rate from last year.
F	adding_capacity_utilized =	IF Monthly_counter = 7 THEN bedding_capacity_utilized_per_month/DT ELSE 0	dmnl/month	The adding_capacity_utilized flow updates the Capacity_Utilization stock with a new value month 7 of each year.
F	resetting_capacity_utilized =	IF Monthly_counter = 7 THEN capacity_utilization/DT ELSE 0	dmnl/month	This flow resets the Capacity_Utilization stock in month 7 each year by extracting the value from the previous year.
S	Destination_Attractiveness(t) =	Destination_Attractiveness(t - dt) + (attractiveness_recovery - attractiveness_degradation) * dt	dmnl	This is a stock representing how attractive Fårö is as a holiday destination to potential tourists.
F	attractiveness_recovery =	MIN(attractiveness_gap, Indicated_Attractiveness-Destination_Attractiveness)/TIME_TO_BUILD_BACK_ATTRACTIVENESS	dmnl/month	If Destination_Attractiveness is lower than the Indicated_Attractiveness the gap between the two will be closed after a set time delay.

F	attractiveness_degradation =	(Destination_Attractiveness-Indicated_Attractiveness)/TIME_TO_DEGRADE_ATTRACTIVENESS	dmnl/month	If Destination_Attractiveness is higher than the Indicated_Attractiveness the gap between the two will be closed after a set time delay.
S	Indicated_Attractiveness (t) =	Indicated_Attractiveness(t - dt) + (indicated_attractiveness_regeneration - indicated_attractiveness_degradation) * dt	dmnl	This is the target Destination_Attractiveness. It reacts quickly to water supply failures and sets the goal for the stock Destination_Attractiveness.
F	indicated_attractiveness_regeneration =	DELAY(indicated_attractiveness_degradation, DURATION_OF_DEFICIT)	dmnl/month	The Indicated_attractiveness is updated each time step.
F	Indicated_attractiveness_degradation =	IF Water_Supply_Demand_Balance.water_supply_deficit > 0 THEN MIN(1-Value_added, Indicated_Attractiveness)/ATTRACTIVENESS_DEGRADATION_TIME ELSE 0	dmnl/month	
S	Tourists(t) =	Tourists(t - dt) + (net_arrivals) * dt	people	The Tourists stock represents tourists staying at tourist facilities connected to the public grid on Fårö.
F	net_arrivals =	(target_tourists-Tourists)/tourist_adjustment_time	people/month	The net flow of tourists to and from the island is formulated as a gap closing function where the number of tourists is compared to a target number of tourists and the gap is closed over a set adjustment time.

Table A.16. Tourist Water Use module – Auxiliary Calculations

Type	Variable name	Equation	Unit	Description
A	attractiveness_gap =	SMTHN((Indicated_Attractiveness-Destination_Attractiveness), TIME_TO_BUILD_BACK_ATTRACTIVENESS, 3)	dmnl	This is the difference between the actual attractiveness of Fårö as a destination and the perceived attractiveness by potential future visitors. We assume that if attractiveness is degraded due to water shortages it will take some time before the perceived attractiveness responds to the change. Similarly, building back attractiveness will take time even after the water supply disruption has been resolved.
A	bedding_capacity_utilized_per_month =	IF tourist_season = 1 THEN Tourists/Beds * "BEDS/PERSON" ELSE 0	dmnl	During the tourist season the bed capacity utilization is the number of tourists divided by the number of beds installed. Outside the tourist season the bedding capacity utilization is not measured by the tourist facility because the occupancy rate is neglectable.
A	day_trip_visitors_per_year =	(Beds/INIT(Beds)) *INITIAL_DAY_TRIPS_PER_YEAR	people	We assume the number of day trip visitors (not staying over night on the island) scales with the overall size of the tourist sector (measured by the number of installed beds).
A	day_trip_visitors_water_use =	visitors*water_use_per_day_trip_visitor	meter^3/month	The water use by day trip visitors is the product of the number of visitors and their average water use per person and day.
A	desire_to_invest_in_new_capacity =	IF capacity_utilization<TARGET_CAPACITY_UTILIZATION THEN 0 ELSE 1	Dmnl	If 0 then there is no desire to make new investments. If 1 then there is a desire to invest in new capacity.
A	desired_capacity =	SMTHN(Beds+desire_to_invest_in_new_capacity*DESIRED_GROWTH,TIME_TO_ADJUST_DESIRED_CAPACITY, 24, 290)	bed	The desired bed capacity of the tourist facility is the installed beds plus the desired growth. Desired capacity does not react instantaneously to changes in demand for tourist facilities. Instead a two year smooth function is used to replicate the decision process of the hotel owners before they invest in more beds.
A	duration_of_deficit =	IF DEFICIT_SWITCH = 0 THEN 0 ELSE IF DEFICIT_SWITCH = 1 THEN 0.033 ELSE IF DEFICIT_SWITCH = 2 THEN 0.25 ELSE 1	month	If water demand exceeds supply on the public grid there will be a supply failure. This variable sets the duration of a potential supply failure on the public grid. The supply failure can las for one day, one week and one month.
A	fraction_of_beds_with_public_water =	Beds_with_Public_Water/(Beds_with_Private_water+Beds_with_Public_Water)	dmnl	This is the fraction of installed beds that rely on the municipal grid for their water supply.
A	gap_between_desired_and_indicated_capacity =	desired_capacity-(Beds_planned+Beds_on_Order+Beds)	bed	This is the difference between the desired and desired bed capacity.

A	order_size =	MIN(75, Beds_Planned)	bed	The number of new beds ordered each ordering cycle is 75 beds or the number of beds planned, whatever is the smallest number.
A	target_tourists =	Beds*tourist_season* Destination_Attractiveness/"BEDS/PERSON"	people	This is the target of the Tourists stock each month.
A	tourism_water_use =	Tourists*(1-fraction_of_beds_with_public_water)*public_water_use_per_private_water_guest_night*Water_Demand.Water_Use_Index + Tourists*fraction_of_beds_with_public_water* water_use_per_guest_night*Water_Demand.Water_Use_Index	meter^3/month	This is the total water use from tourists each month. It includes the activity water use of tourists staying in facilities with private wells and the full water use of tourists staying in facilities with public water. The water use adjusts with the water use index.
A	tourist_sector_water_use =	tourism_water_use+day_trip_visitors_water_use	meter^3/month	This is the total water use from the entire tourism sector.
A	visitors =	day_trip_visitors_per_year*annual_distribution_of_visitors	people	The total yearly number of visitors are distributed according to the seasonal distribution of the annual_distribution_of_visitors_function (the majority visiting in the peak summer).
A	water_use_per_day_trip_visitor =	.WATER_USE_PER_DAY_TRIP_VISITOR* Clim.effect_of_TMAX_on_PC_water_use	meter^3/person/month	Water use per visitor is a function of the base level water use multiplied by an effect function that cause per capita water use to increase when temperatures are high according to studies by (Dimkić, 2020).
A	water_use_per_guest_night =	.WATER_USE_PER_GUEST_NIGHT*Clim.effect_of_TMAX_on_PC_water_use	meter^3/person/month	Water use per tourist is a function of the base level water use multiplied by an effect function that cause per capita water use to increase when temperatures are high according to studies by (Dimkić, 2020).

The Public Water Supply Demand Balance module

The Public Water Supply Demand Balance (PSDB) module keeps track of the total public grid water demand, regulates desired water extraction from the municipal aquifer, monitors the need for supplementary water transports to meet demand, and calculates the net profits of the municipal water supply system on the island. When there is a risk that water demand on the public grid exceeds the local supply capacity, the municipality will supplement the local grid with water transports from outside the island.

Key operations performed in the module:

- Calculates the desired pumping from the municipal aquifer based on water demand from the household and tourist sectors.
- Calculates the required water transports as the difference between the total public water demand and the local water supply.
- Calculates municipal revenues from water tariffs.
- Calculates municipal costs from OPEX, CAPEX and costs of water transports.
- Calculates municipal net profits as the difference between revenues and costs.

Total public water demand and water transports

Total public water demand, $Q_{total}(t)$, is the sum of the water demand from households with public water, $Q_m(t)$, and the public water demand in the tourist sector $Q_T(t)$ (Equation A.60). The desired extraction from the municipal aquifer, $E^*(t)$, equals Q_{total} plus an additional margin to account for filtration losses, γ_{loss}, when the water is treated (Equation A.61).

$$Q_{total}(t) = Q_T(t) + Q_m(t) \quad \text{Equation A.60.}$$

$$E^*(t) = Q_{total}(t) * (1 + \gamma_{loss}) \quad \text{Equation A.61.}$$

Water transports, $Q_{transport}(t)$, is the difference between $Q_{total}(t)$ and the volume of clean water exiting the municipal water plant, $Q_{clean}(t)$ (Equation A.62).

$$Q_{transport}(t) = Q_{total}(t) - Q_{clean}(t) = Q_{total}(t) - \big(E(t) * (1 - \gamma_{loss})\big) \quad \text{Equation A.62.}$$

Municipal net profits

Net profits, $\chi_{net}(t)$ is the difference between municipal revenues, $\chi_{rev}(t)$, and municipal costs, $\chi_{cost}(t)$ (Equation A.63). $\chi_{rev}(t)$ and, $\chi_{cost}(t)$ are calculated according to Equations A.64 and A.65 respectively.

$$\chi_{net}(t) = \chi_{rev}(t) - \chi_{cost}(t) \qquad \text{Equation A.63}$$

$$\chi_{rev}(t) = \left(H_m(t) + \frac{BI_m(t)}{\gamma_{B_{eqv}}}\right) * \gamma_{fee_1} + Q_{total}(t) * \rho_w + s_m(t) * \gamma_{fee_2} \qquad \text{Equation A.64}$$

$$\chi_{cost}(t) = \chi_{opex}(t) + \chi_{capex}(t) + \chi_{transport}(t) \qquad \text{Equation A.65}$$

$$\chi_{capex}(t) = s_m(t) * \gamma_{cost_1} \qquad \text{Equation A.66}$$

$$\chi_{transport}(t) = Q_{transport}(t) * \gamma_{cost_2} \qquad \text{Equation A.67}$$

Municipal revenues are composed of: (1) a yearly fee, γ_{fee_1}, paid by every consumer unit connected to the grid. The total number of units is the on-grid households ($H_m(t)$) plus the installed beds divided by $\gamma_{B_{eqv}}$, which is a weighting factor used to calculate the yearly fee for tourist facilities; (2) a consumption-based fee, ρ_w, multiplied by the total water use, $Q_{total}(t)$; and (3) a connection fee γ_{fee_2} that is paid by all new connections to the public grid $S_m(t)$.

Municipal costs consist of: (1) operational costs, $\chi_{opex}(t)$, which are assumed to be constant, (2) capital investment costs, $\chi_{capex}(t)$, which is the cost associated with extending the grid to new units and is the product of new connections multiplied by a fixed connection fee, γ_{cost_1}, and (3) costs of water transports, $\chi_{transport}(t)$, which is the product of the volume of water transported per month and a fixed cost per cubic meter, γ_{cost_2}.

Graphical representation of the Public Water Supply Demand Balance Module

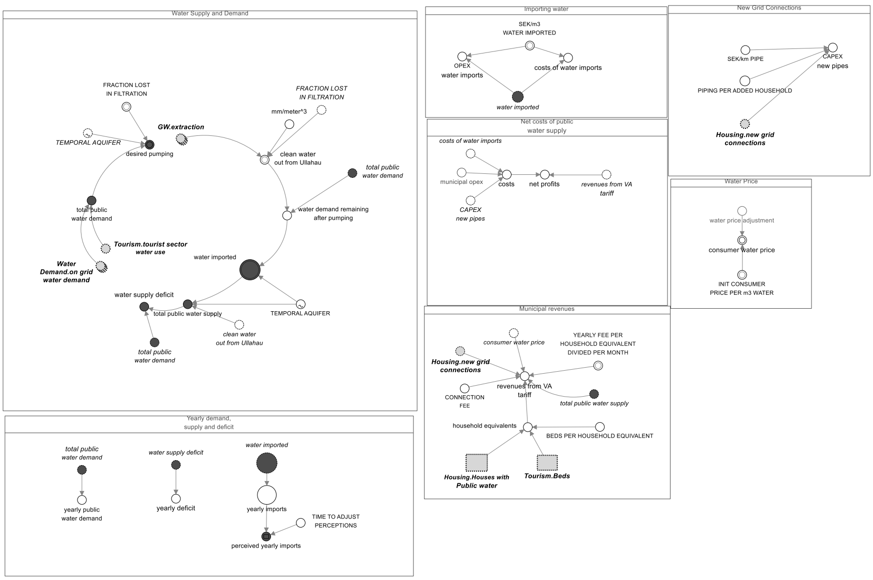


Figure A.11. Public Water Supply Demand Balance Module

Complete list of equations for the Public Water Supply Demand Balance Module

Abbreviations
S = Stock
F = Flow
C = Constant
A = Auxiliary
D = Data input
I = Initial value
T = Graphical/Lookup Table

Table A.17. Public Water Supply Demand Balance module – Constants

Type	Variable name	Unit	Value	Description
C	BEDS_PER_HOUSEHOLD_EQUIVALENT	beds/house	4	Local water tax is calculated using "household equivalents". We estimate that four tourist beds correspond to one household equivalent and is taxed accordingly. The possible range is wide since the number of beds per rental apartment/cottage/studio varies. We assume a plausible range of 2-8 beds/household equivalent.
C	CONNECTION_FEE	SEK/house	199 392	This is a fee paid by households when they get connected to the public water grid (Region Gotland, 2019). - 199 392 SEK/house inc. taxes for water and sewer connection. - 111 854 SEK/house inc. taxes for only water connection.
C	INIT_CONSUMER_WATER_PRICE	SEK/meter^3	26.72	This is the price per meter^3 of public water paid by consumers (Region Gotland, 2019).
C	PIPING_PER_ADDED_HOUSEHOLD	km/house	0.1	This is the average length of new water and sewer pipes needed for every new household added to the public grid in case of expansion. We assume priority will be given to houses near the existing grid.
C	SEK/km PIPE	SEK/km	1000 000	Installation of new water pipes cost 1 (+/- 0.3) MSEK/km (Sjöstrand et al., 2019).
C	SEK/m3_WATER_IMPORTED	SEK/meter^3	125	This is the average cost per cubic meter of water transported to Fårö. Estimated based on an hourly rate at 1043 SEK from the trucking company. During the summer period on average two trips are made per day = 8 hours. With 30 days per month this gives a monthly cost of about 250 000 SEK. Assuming an average of 2000 m3 of water transported per month during the tourist season this gives a cost of 125 SEK/m3. Source: Lars Westerlund, Region Gotland, personal communication, 2 Mars 2021.
C	TIME_TO_ADJUST_PERCEPTIONS	month	72	This is the time it takes for the perception about magnitude local water supply deficit to be update, and to trigger a reaction in the form of restrictions on water use and tourism expansion. It is an indicator of how fast the organization will react to deficit in local water supply capacity (e.g. by imposing restrictions or limiting growth in the tourist sector). The time delay is estimated to 72 months based on the observation that the local water supply deficit was first seen in 2006 (when water imports started) but it took to 2012 until the realization about the deficit in local water supply capacity resulted in actions to limit growth in the water-demanding tourism sector.
C	YEARLY_FEE_PER_HOUSEHOLD_EQUIVALENT_DIVIDED_PER_MONTH	SEK/month/house	1185/12	This is the fee every household equivalent connected to the public grid pays for his/her water services (Region Gotland, 2019). Yearly fee excluding tax = 1185 SEK/household/year equivalent Yearly fee including tax = 1481,25 SEK/household equivalent/year

Table A.18. Public Water Supply Demand Balance module – Auxiliary Calculations

Type	Variable name	Equation	Unit	Description
A	CAPEX_new_pipes =	Housing.new_grid_connections*"SEK/km_PIPE"*PIPING_PER_ADDED_HOUSEHOLD	SEK/month	The capital expenditures the municipality have to pay for expanding the public grid is the product of the number of new houses added to the grid, the distance of piping needed and the cost per kilometer of new pipe added.
A	clean_water_out_from_Ullahau =	GW.extraction*(1-FRACTION_LOST_IN_FILTRATION)/"mm/meter^3"	meter^3/month	This is the volume of water reaching the public grid after filtration and treatment.
A	consumer_water_price =	INIT_CONSUMER_PRICE_PER_m3_WATER*water_price_adjustment	SEK/meter^3	The consumer water price is the price per cubic meter of water consumed by consumers on the public grid. It can be adjusted by changing the water_price_adjustment policy parameter.
A	costs =	costs_of_water_imports+ municipal_opex+ CAPEX_new_pipes	SEK/month	This is the total cost the municipality pays to supply water to the public grid each month. Includes operational and capital investments and the additional costs for water imports.
A	costs_of_water_imports =	SEK/m3_WATER_IMPORTED*water_imported	SEK/month	The costs of water imports is the product of the volume imported each month and the cost per cubic meter.
A	desired_pumping =	MAX(0, (total_public_water_demand/(1-	meter^3/month	The volume of water the municipality would like to extract from the municipal aquifer each month. The TEMPORAL_AQUIFER was an aquifer supplying water

		FRACTION_LOST_IN_FILTRATION)) - TEMPORAL_AQUIFER)		to the municipal grid for a short period in 2015-2016. It is no longer in use.
A	FRACTION_LOST_IN_FILTRATION =	0 + STEP (0.15, 73) + STEP (0.15, 133)	dmnl	This is the fraction of extracted water from the public aquifer that is lost in the filtration/treatment process of the water. It has increased in two steps since 2000 due to stricter regulations on water quality.
A	household_equivalents =	Housing.Houses_with_Public_water + (Tourism.Beds/BEDS_PER_HOUSEHOLD_EQUIVALENT)	houses	The total number of household equivalents connected to the public grid.
A	municipal_opex =	IF TIME < 132 THEN 25000 ELSE 75000	SEK/month	In 2011 a new water treatment plant was introduced. Prior to this date only very basic water treatment had been used, with limited costs (estimated to 25 000 SEK/month). However, the new treatment plant was needed to ensure sufficient water quality. The new treatment facility is rented by the municipality with an estimated cost per month at 75 000 SEK. (Lars Westerlund, Region Gotland, personal communication 2 March 2021)
A	net_profits =	CumMonthlyToYearlySEK(revenues_from_VA_tariff-costs)	SEK/year	The yearly net profits of the municipal water system.
A	OPEX_water_imports =	"SEK/m3_WATER_IMPORTED"*water_imported	SEK/month	The operational expenditures of municipal water imports/transports.
A	perceived_yearly_imports =	SMTHN(yearly_imports, TIME_TO_ADJUST_PERCEPTIONS,12)	meter^3/year	It takes time for the municipality to perceive, plan and react to the growing water imports. An information delay is used to capture this inertia in the system.
A	revenues_from_VA_tariff =	(YEARLY_FEE_PER_HOUSEHOLD_EQUIVALENT_DIVIDED_PER_MONTH*household_equivalents)+ (total_public_water_supply*consumer_water_price)+ Housing.new_grid_connections*CONNECTION_FEE	SEK/month	This is the total revenues to the municipality from the water services they supply. It includes a yearly fee per household equivalent, a consumption-based tariff and a connection fee for all new consumers that are added to the gird.
A	total_public_water_demand =	Water_Demand.on_grid_water_demand[Permanent]+ Water_Demand.on_grid_water_demand[Parttime]+ Tourism.tourist_sector_water_use	meter^3/month	The total water demand by all consumers on the municipal grid.
A	total_public_water_supply =	clean_water_out_from_Ullahau + water_imported + TEMPORAL_AQUIFER	meter^3/month	The total supply of water to the public grid.
A	water_demand_remaining_after_pumping =	total_public_water_demand- clean_water_out_from_Ullahau	meter^3/month	The water demand remaining after what water can be produced from the municipal aquifer has been produced.
A	water_imported =	MAX(0, water_demand_remaining_after_pumping- TEMPORAL_AQUIFER	meter^3/month	The volume of water imported/transported each month is the water demand remaining after extraction from the municipal aquifer (and the TEMPORAL_AQUIFER in 2015-2016).
A	water_supply_deficit =	MAX(0, total_public_water_demand- total_public_water_supply)	meter^3/month	The water supply deficit. This occurs if total water demand is greater than the total water supply on the public grid.
A	yearly_deficit =	CumMonthlyToYearly(water_supply_deficit)	meter^3/year	This is the accumulated water deficit per year on the municipal grid.
A	yearly_imports =	CumMonthlyToYearly(water_imported)	meter^3/year	This is the accumulated water imports/transports per year.
A	yearly_public_water_demand =	CumMonthlyToYearly(total_public_water_demand)	meter^3/year	This is the accumulated water demand per year on the municipal grid.

General constants, time series and graphical functions

Abbreviations
S = Stock
F = Flow
C = Constant
A = Auxiliary
D = Data input
I = Initial value
T = Graphical/Lookup Table

Table A.19. Assumptions & constants

Top level				
Type	**Variable name**	**Value**	**Unit**	**Description**
C	WATER_USE_PER_CAPITA	3.6	Meter^3/person/month	This is the estimated average water use per person and month, estimated from the national average (Swedish Water, 2020)and down adjusted according to estimates by Region Gotland (Lars Westerlund, personal communication 2 March 2021).
C	WATER_USE_PER_DAY_TRIP_VISITOR	0.3	Meter^3/person/month	This is the estimated water use of tourists and visitors excluding water use associated with accommodation according to Gössling et al (2012). Corresponding to 10-30 liter/person/day.
C	WATER_USE_PER_GUEST_NIGHT	2.4	Meter^3/person/month	This is the average water use per guest night multiplied by 30 to give the water us per tourist month. This corresponds to 80 liter/person/day (Gossling et al., 2012).
C	SEED =	NAN	dmnl	Sets the seed used for the simulation.
C	DEFICIT_SWITCH =	0	dmnl	This parameter allows us to experiment with different durations of water supply failures on the public grid. If there is a water supply deficit in the municipal system and the switch is set to 0, the duration of the deficit is assumed to be too short to affect attractiveness. If the switch is set to 1 we assume the deficit lasts for one day. If the switch is set to 2 the deficit is assumed to last one week. If the switch is set to 3 the deficit is assumed to last one month.
C	water_price_adjustment =	0	dmnl	This is a policy variable to explore the effect of water price changes on the public grid. The value assigned represents the fractional change in consumer water price compared to the baseline water price used in the simulation.
C	POLICY_SWITCH_Expansion_of_Public_grid =	IF TIME < 265 THEN 0 ELSE 0	dmnl	This is a policy variable used to explore the effect of introducing a municipal policy of expanding the public water grid. Assigning the parameter a value of 1 introduces a policy where the municipal gird starts expanding after time 265.

Table A.20. Graphical functions and time-dependent variables

Type	Variable name	Value	Unit	Description
T	demand_multiplier =	GRAPH(TIME) (0,1.00) (12,0.92) (24,0.87) (36,1.01) (48, 0.86) (60,1.10) (72,1.06) (84,1.25) (96,1.03) (108,1.25) (120,1.18) (132,1.16) (144,0.91) (156,1.39) (168,1.38) (180,1.73) (192,1.75) (204,1.49) (216,1.44) (228,1.48) (240,1.78) (252,1.64) (600,2.79)	dmnl	This is the estimated quantity demand for houses on Fårö normalized to year 2000. Based on data on summer house prices and summer houses sold on Gotland between 2000 and 2020 (SCB). The trend in demand growth seen in 2000 to 2020 is extrapolated to the period 2020 to 2050.
T	summer_house_season =	GRAPH(Monthly_counter,summer) (1.00, 0.045), (1.50, 0.045), (2.00, 0.049), (2.50, 0.045), (3.00, 0.045), (3.50, 0.047), (4.00, 0.073), (4.50, 0.091), (5.00, 0.128), (5.50, 0.170), (6.00, 0.500), (6.50, 0.788), (7.00, 1.000), (7.50, 0.715), (8.00, 0.276), (8.50, 0.193), (9.00, 0.107), (9.50, 0.089), (10.00, 0.056), (10,50, 0.049), (11.00, 0.047), (11.50, 0.045), (12.00, 0.044), (12.50, 0.044), (13.00, 0.045)	dmnl	This is the fraction of part-time households that are in use any given month of the year. Based on estimates by Region Gotland (Lars Westerlund personal communication 11 May 2020).
T	effect_of_GW_level_on_deep_ET =	GRAPH(GW_level) (-4.0,0.0), (-3.6, 0.1), (-3.2, 0.2), (-2.8, 0.3), (-2.4, 0.4), (-2.0, 0.5), (-1.6, 0.6), (-1.2, 0.7), (-0.8, 0.8), (-0.4, 0.9), (0.0, 1.0)	dmnl	If the groundwater table reaches the root zone it becomes available for evapotranspiration. At depths deeper than 4 meter the groundwater is unavailable to plant roots but availability increases linearly as the water table rises into the root zone.

T	effect_of_GW_level_on_pumping =	GRAPH(GW_level[Aquifer_GW]-WELL_DEPTH) (0.0, 0.0), (0.1, 0.1), (0.2, 0.2), (0.3, 0.3), (0.4, 0.4), (0.5, 0.5), (0.6, 0.6), (0.7, 0.7), (0.8, 0.8), (0.9, 0.9), (1.0, 1.0)	dmnl	Pumping capacity declines as the groundwater table approaches the depth of the municipal wells.
T	annual_distribution_of_visitors =	GRAPH(Monthly_counter) (0, 0.005), (1, 0.010), (2, 0.010), (3, 0.010), (4, 0.010), (5, 0.012), (6, 0.009), (7, 0.400), (8, 0.014), (9, 0.014), (10, 0.012), (11, 0.010), (12, 0.005)	dmnl	This is the fractional distribution of yearly visitors per month. Based on data from Region Gotland (Region Gotland, 19 March 2020) and validated through personal communication, Lars Westerlund Region Gotland 11 May 2020).
T	effect_of_water_deficit_on_tourism_expansion =	GRAPH(Water_Imports.perceived_yearly_imports/DEFICIT_THRESHOLD) (0.0, 1.0), (0.1, 1.0), (0.2, 1.0), (0.3, 1.0), (0.4, 1.0), (0.5, 1.0), (0.6, 1.0), (0.7, 1.0), (0.8, 1.0), (0.9, 0.9), (1.0, 0.0)	dmnl	Water supply deficit on the public gird causes a non-linear effect on tourism expansion. Only when the perceived water transports are very high (90 % of the deficit threshold) do we see a response in the form of limitation on tourism expansion.
T	tourist_season =	GRAPH(Monthly_counter) (0.0, 0.0), (0.5, 0.0), (1.0, 0.0), (1.5, 0.0), (2.0, 0.0), (2.5, 0.0), (3.0, 0.0), (3.5, 0.0), (4.0, 1.0), (4.5, 1.0), (5.0, 1.0), (5.5, 1.0), (6.0, 1.0), (6.5, 1.0), (7.0, 1.0), (7.5, 1.0), (8.0, 0.0), (8.5, 0.0), (9.0, 0.0), (9.5, 0.0), (10.0, 0.0), (10.5, 0.0), (11.0, 0.0), (11.5, 0.0), (12.0, 0.0)	dmnl	The tourist season is centered around the summer months (June, July and August) with a peak in July.
T	Value_added =	GRAPH(DURATION_OF_DEFICIT) (0.0, 1.0), (0.033, 0.28), (0.25, 0.0), (1.0, 0.0)	dmnl	The duration of a water supply failure has nonlinear impacts on the value creation of the tourist sector. Water supply failures longer then one day would force many tourist facilities to close down. The effect of water disruption on value added is based on results from Sjöstrand et al. (2020).
T	TEMPORAL_AQUIFER =	TEMPORAL_AQUIFER = GRAPH(TIME) (0, 0), (194, 0), (195, 0), (196, 2100), (197, 2100), (198, 2100), (199, 2100), (200, 0), (201, 0), (202, 0), (203, 0), (204, 0), (205, 0), (206, 0), (207, 0), (208, 0), (209, 300), (210, 1800), (211, 600), (212, 0), (213, 0), (214, 0), (215, 0), (216, 0), (217, 0), (218, 0), (219, 0), (220, 0), (221, 1700), (222, 1700), (223, 1300), (224, 0)	meter^3/month	The temporal aquifer was used in the summer months between 2016 and 2018. Data from Region Gotland.

References

Abramowitz, M., Stegun, I.A., Romer, R.H., 1988. Handbook of Mathematical Functions with Formulas, Graphs, and Mathematical Tables. American Journal of Physics, 56(10): 958-958. DOI:10.1119/1.15378

Alexandersson, H., 2006. Vindstatistik för Sverige 1961-2004 [Wind statistics for Sweden 1961-2004].

Arnaud, R., 2015. Modelling Household Water Demand in Europe - Insights from a Cross-Country Econometric Analysis of EU-28 countries.

Asp, M. et al., 2015. Framtidsklimat i Gotlands län - enligt RCP-scenarier [Future climate in Gotland county - according to the RCP scenarios], SMHI.

Brunner, F., 2014. FÖP Fårö - Fördjupad översiktsplan 2025 [Regional development plan for Fårö 2025]. In: Environment, D.o.B. (Ed.). Region Gotland, Visby, Gotland.

Dahlqvist, P. et al., 2015. SkyTEM-undersökningar på Gotland [SkyTEM analysis of Gotland].

Dimkić, D., 2020. Temperature Impact on Drinking Water Consumption. Environmental Sciences Proceedings, 2: 31. DOI:10.3390/environsciproc2020002031

Englund, P., 2011. Swedish house prices in an international perspective, Central Bank of Sweden, Stockholm.

Federer, 1995. BROOK90: A simulation model for evaporation, soil water, and

streamflow, version 3.1. Computer freeware and documentation. U.S. Forest Service, Durham, USA.

Gossling, S. et al., 2012. Tourism and water use: Supply, demand, and security. An international review. Tourism Management, 33(1): 1-15. DOI:10.1016/j.tourman.2011.03.015

Hillel, D., 2004. Introduction to Environmental Soil Physics. Elsevier Science.

Höglund, L., 1999. Household demand for water in sweden with implications of a potential tax on water use. Water Resources Research, 35(12): 3853-3863. DOI:https://doi.org/10.1029/1999WR900219

International Monetary Fund. European Dept., 2015. Sweden: Selected Issues. IMF Staff Country Reports, 2015(330): A002. DOI:10.5089/9781513524368.002.A002

IPCC, 2013. Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change, IPCC, Cambridge, United Kingdom and New York, NY, USA.

Penman, H.L., Keen, B.A., 1948. Natural evaporation from open water, bare soil and grass. Proceedings of the Royal Society of London. Series A. Mathematical and Physical Sciences, 193(1032): 120-145. DOI:doi:10.1098/rspa.1948.0037

Region Gotland, 2019. Avgifter för Region Gotlands vatten- och avloppsverksamhet. In: Teknikförvaltningen VA-avdelningen (Ed.).

Schenk, H.J., Jackson, R.B., 2002. Rooting depths, lateral root spreads and below-ground/above-ground allometries of plants in water-limited ecosystems. Journal of Ecology, 90(3): 480-494. DOI:https://doi.org/10.1046/j.1365-2745.2002.00682.x

Sjökvist, E. et al., 2015. Klimatscenarier för Sverige - Bearbetning av RCP-scenarier för meteorologiska och hydrologiska effektstudier [Climate scenarios for Sweden - Processing of the RCP scenarios for metrological and hydrological effect studies], SMHI.

Sjöstrand et al., 2014. Metod för val av vattenförsörjning i områden med vattenbrist – en handbok [Methods for selecting water supply systems in areas with water scarcity - a manual]. 2014:54, SP Technical Research Institute of Sweden, Borås, Sweden.

Sjöstrand, K., Lindhe, A., Söderqvist, T., Dahlqvist, P., Rosén, L., 2019. Marginal Abatement Cost Curves for Water Scarcity Mitigation under Uncertainty. Water Resour. Manag., 33(12): 4335-4349. DOI:10.1007/s11269-019-02376-8

Sjöstrand, K., Lindhe, A., Söderqvist, T., Rosén, L., 2020. Water Supply Delivery Failures—A Scenario-Based Approach to Assess Economic Losses and Risk Reduction Options. Water, 12(6): 1746.

Strandberg, G. et al., 2014. CORDEX scenarios for Europe from the Rossby Centre regional climate model RCA4.

Swedish Water, 2020. Dricksvattenfakta [Drinking water facts].

Wolff, R.G., 1982. Physical properties of rocks; porosity, permeability, distribution coefficients, and dispersivity. 82-166. DOI:10.3133/ofr82166

Zhang, L., Potter, N., Hickel, K., Zhang, Y., Shao, Q., 2008. Water balance modeling over variable time scales based on the Budyko framework – Model development and testing. Journal of Hydrology, 360(1): 117-131. DOI:https://doi.org/10.1016/j.jhydrol.2008.07.021

Zotarelli, L., Dukes, M., Romero, C., Migliaccio, K., Kelly, T., 2015. Step by Step Calculation of the Penman-Monteith Evapotranspiration (FAO-56 Method).

Appendix B – Calibration inputs and assumptions

Table B.1. Complete list of inputs and outputs used in partial- and full model calibration. The first column, from left to right, indicates the module, the second column contains the payoff variables used in the calibration. Column number three contains names of the calibrated parameters, column four shows the calibration range explored for each parameter, and column five provides the estimated parameter value. The sixth column provides references to the source of the applied calibration range, and in column seven contains additional comments with regards to the estimated parameter values.

Module	Payoff variables[4]	Calibrated parameters[1]	Calibration range	Estimate	References and range assumptions	Comments
Public Water Supply	Groundwater level	Water holding capacity of medium	140-220 mm/meter^3	180	Range estimated from soil samples provided by Region Gotland.	The estimated water holding capacity is in line with estimates from available soil sample data.
		Discharge Factor	0.0-1.0 dmnl/month	0.01	Range taken from Zhang et al. (2008)	Studies by Zhang et al. (2008) show large variations in the value of the discharge factor parameter between aquifers. A more or less uniform probability distribution is suggested, ranging from 0.0 to 1.0. Our estimate is in the lower, but plausible, range of the distribution.
		Retention Factor a1	0.0-1.0 dmnl	0.65	Range taken from Zhang et al. (2008)	Studies by Zhang et al. (2008) show a majority of aquifers having values between 0.5 and 0.8. Our estimate fits well within this range.
		ET Factor a2	0.0-1.0 dmnl	0.6	Range taken from Zhang et al. (2008)	Studies by Zhang et al. (2008) show a majority of aquifers having values between 0.55 and 0.8. Our estimate fits well within this range.
		GW AT	0.5 – 3 months	0.773	Estimated by the modelers.	A percolation time of about three weeks is assumed reasonable for the studied area.
Private Water Supply	Groundwater level	Water holding capacity of medium	75 – 140 mm/meter^3	100	Estimated from Wolff (1982) and studies by Dahlqvist et al. (2015).	The estimated water holding capacity is in line with representative values for limestone provided by Wolff (1982).
		Hydraulic conductivity	0.1-5.8 m/month	0.13	Range estimated from the digital map of hydrological conductivity provided by SGU	The range for hydraulic conductivity is taken from a geological model developed by SLU. The value can vary significantly
		Retention Factor a1	0.0-1.0 dmnl	0.65	Range taken from Zhang et al. (2008)	Studies by Zhang et al. (2008) show a majority of aquifers having values between 0.5 and 0.8. Our estimate fits well within this range.
		ET Factor a2	0.0-1.0 dmnl	0.6	Range taken from Zhang et al. (2008)	Studies by Zhang et al. (2008) show a majority of aquifers having values between 0.55 and 0.8. Our estimate fits well within this range.
		GA AT	0.5 – 3 months	1	Range estimated by the modelers.	A percolation time of one month is assumed reasonable for the studied area.
Household Water Use	House Price Index	Price elasticity of demand	(-0.25) – (-0.75) dmnl	-0.5	Range taken from Englund (2011)	Our estimate suggests housing demand is quite inelastic to changes in price.
	Total Houses	Price elasticity of supply	0.5 – (-0.5)	0.1	Range taken from IMF (2015)	Housing Supply is inelastic to price changes. This suggests other factors are more important determinants of supply than price.
		Base level construction capacity	1-6 houses/month	3	Estimated by the modelers.	The estimate is in line with the number of building permits issued between 2000 – 2020.
Tourist Water Use	Installed Beds	Deficit Threshold	300 – 2000 meter^3/year	650	Range estimated by the modelers.	If the local water deficit grows large enough on the public grid this will lead to the municipality acting to limit further connections. Our estimate suggests that the municipality will not restrict expansion until a considerable deficit has been reached. This corresponds to the historical development on the island where it took several years of increased local water deficit until restrictions were introduced.
		Time to Adjust Desired Capacity	12 – 36 months	20.4	Range estimate based on personal communication with Rolf Lindvall, Sudersand Resort, 2020-10-15.	The estimated value suggests that the tourist sector will adjust its desired capacity with a considerable time delay. Hence, a single good or bad season is not enough to change plans. This corresponds well information provided by Sudersand resort.
		Target Capacity Utilization	0.75 – 1.0	0.92	Range estimate based on personal communication with Rolf Lindvall, Sudersand Resort, 2020-10-15.	A high level of capacity utilization is required for investments in additional capacity to occur.
		Desired Growth	120 – 270 beds	235	Range estimate based on personal communication with Rolf Lindvall, Sudersand Resort, 2020-10-15.	A desired growth of 235 beds corresponds to an average growth rate of about 10 to 15 new cottages per year, which corresponds well to the growth strategy applied by Sudersand Resort.
Public Water Supply Demand Balance	Total Public Water Demand	Water Use Per Guest night	80 – 200 liter/person/day	80	Range estimated from Gossling et al. (2012)	80 liter per person and day is on the low side of the range suggested by Gossling et al. (2012). However, many of the tourists visiting Fårö are staying at basic facilities and/or in facilities with water-efficient showers, WCs, etc. so the low estimate is considered plausible.

[4] See appendix A for variable documentation.

	Water Transports	Water Use Per Capita	100 – 140 liter/person/day	120	Range based on national estimates provided by Swedish Water (2020) and Region Gotland.	120 liter per person per day is in line with estimates provided by Region Gotland.
		Hydraulic conductivity	518-777 meter/month	550	Range based on soil samples from Region Gotland.	This estimate corresponds well to documented hydraulic conductivity of medium-grained sand.
		Effect of GW on pumping	Graphical function calibration	1	Graphical function. Shape estimated by the modelers.	This is a graphical function regulating the responsiveness of water extraction capacity to the level of the groundwater head. We assume that as the groundwater head approaches within one meter of the depth of the municipal wells, extraction capacity declines linearly.
		GW L	25-100 meter	51	Estimated by the modelers.	The estimate is assumed plausible based on discussions with hydrogeologists from SGU.
		Aquifer Radius	10-100 meter	21.7	Estimate based on the size of the aquifer.	The estimate is assumed plausible based on maps of the municipal aquifer provided by Region Gotland.

Dahlqvist, P. et al., 2015. SkyTEM-undersökningar på Gotland [SkyTEM analysis of Gotland].
Gossling, S. et al., 2012. Tourism and water use: Supply, demand, and security. An international review. Tourism Management, 33(1): 1-15. DOI:10.1016/j.tourman.2011.03.015
Swedish Water, 2020. Dricksvattenfakta [Drinking water facts].
Wolff, R.G., 1982. Physical properties of rocks; porosity, permeability, distribution coefficients, and dispersivity. 82-166. DOI:10.3133/ofr82166
Zhang, L., Potter, N., Hickel, K., Zhang, Y., Shao, Q., 2008. Water balance modeling over variable time scales based on the Budyko framework – Model development and testing. Journal of Hydrology, 360(1): 117-131. DOI:https://doi.org/10.1016/j.jhydrol.2008.07.021

Appendix C. Monte Carlo inputs

Multivariate Monte Carlo simulations are used to explore a wide ensemble of plausible futures. Each simulation of the Monte Carlo analysis the parameters presented in Table C.1 are provided a new value, sampled from the probability distributions and specified ranges described below. Shape and range of the distribution are based on empirical data or literature studies to the extent such information is available. Otherwise, a uniform distribution is used with range estimated by the modelers.

Table C.1. Parameter inputs and sampling ranges used in the multivariate Monte Carlo simulations.

Variable name	Distribution	Range	Description
Clim.SEED	Uniform	0 – 10000	A new seed value is generated for every simulation in the Monte Carlo analysis.
Future precipitation [January:December]	Weibull	0 – 105 [January] 0 – 74 [February] 0 – 69.7 [March] 0 – 91 [April] 0 – 93 [May] 0 – 122 [June] 0 – 161 [July] 0 – 164 [August] 0 – 146 [September] 0 – 170 [October] 0 – 134 [November] 0 – 113 [December]	Monthly precipitation in 2020 – 2050 is randomly sampled from a Weibull distribution that conforms to the specified shape and scale parameter values calculated for each month. The shape and scale parameters are estimated from historic precipitation data (1882 – 1995) (SMHI, 2021) and the projected monthly mean precipitation values provided by the SMHI climate scenarios for Gotland (Asp et al., 2015). Future precipitation is bounded by the maximum observed precipitation in the historic dataset.
Future TMIN [January:December]	Normal (mean, SD)	Mean, 2.65 [January] Mean, 4.14 [February] Mean, 3.57 [March] Mean, 1.37 [April] Mean, 1.69 [May] Mean, 1.47 [June] Mean, 1.29 [July] Mean, 1.35 [August] Mean, 1.85 [September] Mean, 1.49 [October] Mean, 1.77 [November] Mean, 3.45 [December]	Monthly TMIN (the coldest day of the month) in 2020 – 2050 is sampled from a Normal distribution with a mean set by the TMIN values estimated in the SMHI climate scenarios (Asp et al., 2015) and a standard deviation calculated from historic temperature observations on Fårö (SMHI, 2021).
Future TMAX [January:December]	Normal (mean, SD)	Mean, 2.25 [January] Mean, 2.85 [February] Mean, 2.67 [March] Mean, 2.46 [April] Mean, 2.50 [May] Mean, 1.93 [June] Mean, 2.25 [July] Mean, 1.90 [August] Mean, 1.93 [September] Mean, 1.78 [October] Mean, 1.67 [November] Mean, 2.49 [December]	Monthly TMAX (the hottest day of the month) in 2020 – 2050 is sampled from a Normal distribution with a mean set by the TMAX values estimated in the SMHI climate scenarios (Asp et al., 2015) and a standard deviation calculated from historic temperature observations on Fårö (SMHI, 2021).
SENS_DEMAND:SUPPLY_EFFECT_ON_PRICE	Uniform	0.75 – 1.25	The elasticity of housing prices to the quantity demand:quantity supply ratio is allowed to vary +/- 25 % compared to the base case.
SENS_FUTURE_DEMAND	Uniform	0.5 – 1.5	Assumptions about future housing demand is allowed to vary +/- 50 % compared to the base case.
SENS_PRICE_ELASTICITY_OF_SUPPLY	Uniform	0 – 0.5	This is the confidence range for the price elasticity of housing supply (International Monetary Fund. European Dept., 2015).
SENS_SALT_SENSITIVITY	Uniform	-1.96 – 1.96	The responsiveness of groundwater chloride levels to changes in groundwater levels is allowed to vary within its 95 % confidence intervals (+/- two standard deviations). This is based on the empirical relationship between groundwater level and chloride concentration described by equation A.20 in appendix A.

SENS_AFFLUENCE_ELASTICITY_OF_WATER_USE	Uniform	0.07 – 0.13	This is the confidence interval for the affluence elasticity of per capita water use (Höglund, 1999).
SENS_SALT_EFFECT_ON_WELLS	Uniform	-1.96 – 1.96	The effect of changes in groundwater chloride levels on the proportion of well-sites with chloride concentrations exceeding 100 mg/liter Cl is allowed to vary within its 95 % confidence intervals (+/- two standard deviations). This is based on the empirical relationship between groundwater level and chloride concentration described by equation A.21 in appendix A.

Asp, M. et al., 2015. Framtidsklimat i Gotlands län - enligt RCP-scenarier [Future climate in Gotland county - according to the RCP scenarios], SMHI.

Höglund, L., 1999. Household demand for water in sweden with implications of a potential tax on water use. Water Resources Research, 35(12): 3853-3863. DOI:https://doi.org/10.1029/1999WR900219

International Monetary Fund. European Dept., 2015. Sweden: Selected Issues. IMF Staff Country Reports, 2015(330): A002. DOI:10.5089/9781513524368.002.A002

SMHI, 2021. SMHI Meteorologiska observationer [SMHI Metrological observations]. SMHI.

Appendix D. Data inputs

Data inputs for the base run, climate data and calibration data are available in the electronic version of the paper (https://ars.els-cdn.com/content/image/1-s2.0-S2214581822000799-mmc5.xlsx) or from the corresponding author upon request.

Acta Universitatis agriculturae Sueciae

Doctoral Thesis No. 2022:70

Understanding the dynamics of coupled human and natural systems (CHANS) is key for addressing many sustainability challenges. In this thesis, I combine systems thinking, modelling and simulation, to build understanding of the mechanisms driving complex behaviour in CHANS, and to develop effective management support tools. The results revealed that many contemporary tools are not suited to complex systems, and this contributes to unsustainable management. Simulation-based approaches can complement conventional tools by allowing for more realistic representation of complex systems.

Andreas Nicolaidis Lindqvist completed his graduate education at the Department of Biosystems and Technology at the Swedish University of Agricultural Sciences (SLU), Alnarp, Sweden. He received his MSc in Environmental Science and BSc in Biology from Lund University.

Acta Universitatis agriculturae Sueciae presents doctoral theses from the Swedish University of Agricultural Sciences (SLU).

SLU generates knowledge for the sustainable use of biological natural resources. Research, education, extension, as well as environmental monitoring and assessment are used to achieve this goal.

ISSN 1652-6880

ISBN (print version) 978-91-8046-016-3

ISBN (electronic version) 978-91-8046-017-0