Spatial patterns of ecosystem services and chemical risk assessment in freshwater catchments.

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"The world is full of great and wonderful things for those who are ready for them" – ***Moominpappa at Sea*** **by Tove Jansson**

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Thesis abstract

The ecosystem services concept has useful applications in chemical risk assessment, but there is a need to better understand spatial patterns of ecosystem service delivery and chemical risk assessment. With recreational fishery as a case study, a cultural ecosystem service enjoyed by millions of anglers at risk from chemical pollution, this thesis aims to investigate how effects of chemical stressors on a recreational fishery at local scales relate to ecosystem service delivery and risk assessment at the catchment-scale.

This thesis has three main objectives. First, to use a rapid evidence assessment to establish what ecosystem services tools are available. Second, to adapt and use a spatially explicit individual-based model to predict the effects of chemical stressors on a recreational fishery in a river reach. Third, to use the same model to investigate how effects within-reach translate to effects on a catchment-level. Finally, the implications of these findings for both angler satisfaction and ecological risk assessment were considered.

The effects of chemical stress on a recreational fishery varied spatially at the reach, river network, and landscape-scale. Isolated reach populations were most sensitive to stress effects on survival and resistant to effects on fecundity. Fish populations could recover from chemical stress but angler satisfaction decreased under all chemical stress scenarios. Impacts on angler satisfaction were not as apparent on the landscape-scale and spatial patterns did not directly translate from the local reach to the river network. The impacts of chemical stress on angling stock were variable at the local scale and neighbouring reaches could mitigate impacts at the catchment-scale.

Chemical risk assessment would be best applied with a local scale assessment to highlight the impact of chemicals coupled with a wider, landscape-scale assessment to better include dynamics as spatial patterns are useful in understanding chemical effects.

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Declaration

I, the author, confirm that the Thesis is my own work. I am aware of the University's Guidance on the Use of Unfair Means (www.sheffield.ac.uk/ssid/unfair-means). This work has not previously been presented for an award at this, or any other, university.

Chapters 4 and 5 are presented in manuscript form with the aim to publish post-submission.

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Chapter 1

General Introduction

1.1 What are ecosystem services and how are they delivered?

Ecosystems make multiple contributions to human well-being known as ecosystem services (*sensu* Ehrlich & Ehrlich, 1981) and are sometimes referred to as 'nature's services', 'environmental services', and 'ecological goods and services' (Haines-Young & Potschin, 2010; Kadykalo et al., 2019). Some examples of ecosystem services include drinking water, wild food, flood protection, and recreation (Peng et al., 2023). To understand ecosystem services and its scope, it is important to know that the ecosystem services are delivered in part by 'natural capital' (Smith et al., 2017), or the stock of available natural resources (Schumacher, 1973). Whilst 'natural capital' focuses on the resources, ecosystem services expands from this to consider the use of 'natural capital' through their flows, interactions, and contributions to human well-being (Costanza et al., 1997; Costanza, 2020; O'Higgins et al., 2020). As the ecosystem services concept includes these interactions as well as beneficiaries, it bridges natural science with social science by aiding decision-making, policy, and socioeconomics (Zhang et al., 2022).

These contributions to human well-being from ecosystem services provision and production are determined through ecosystem service delivery (Brauman et al., 2007). Ecosystem service delivery is dependent on organisms and their interactions with the environment (Birkhofer et al., 2015). Groups of organisms which are necessary for ecosystem service delivery are known as service providing units (SPUs, *sensu* Luck et al., 2009), for example, pollinators are the SPU for pollination. SPUs can be at the species, populations, functional groups, communities or ecosystems level (Luck et al., 2009; Birkhofer et al., 2015; Brown et al., 2017). However, SPUs are not exclusive as multiple species can contribute to one service or a single species can contribute to multiple services. For example, a stable fish population can contribute to many ecosystem services from food provision, recreation, and sediment retention (Holmlund & Hammer, 1999; Peng et al., 2023). SPUs rely on their habitats and biodiversity which facilitate and determine spatial variation of delivery and in turn are dependent on the spatial scale of where interactions can occur (Maltby et al., 2018). Therefore, ecosystem service delivery is not only dependent on SPUs but also on the service providing area or landscape (Maltby et al., 2021).

Ecosystem service delivery is also dependent on underlying ecosystem functions made up of ecosystem processes and structures (European Union, 2013; Petter et al., 2013). Ecosystem functions are the intermediary between ecosystem processes and services or the capacity for ecosystems to deliver ecosystem services, for example, primary production (Haines-Young & Potschin, 2010; Kumar, 2012). Ecosystem processes are the interactions between the biological, chemical, or physical components, and ecosystem structures are the biophysical attributes of ecosystems (Crossman et al., 2013). It is important to ensure ecosystem processes and ecosystem functions are well-defined as they may or may not all contribute to ecosystem service delivery (Pesce et al., 2023). Understanding the links between SPUs, ecosystem processes, ecosystem functions, and ecosystem services helps to determine ecosystem service delivery.

However, the ecosystem services concept is not without its own challenges in conceptualisation, especially as ecosystems and socioecological systems are inherently complex (IPBES, 2016; O'Higgins et al., 2020). Linking and measuring biodiversity loss to human benefits has proved challenging especially as biodiversity only focuses on services from living processes (Markandya, 2016; Potschin & Haines-Young, 2016; Devos et al., 2019). But the interactions of both biotic (living) and abiotic (non-living) processes contribute to ecosystem service delivery (European Union, 2013). The ecosystem services concept addresses these interactions by considering both the biotic and abiotic components of ecosystems (Maltby et al., 2022). Another challenge to conceptualisation is defining the final link to human well-being (van Wensem et al., 2017). To address this ecosystem services are often represented as either intermediate or final services (Heink & Jax, 2019). Intermediate services, or supporting services, are ecosystem functions which underlie other ecosystem services (Birkhofer et al., 2015), whilst final ecosystem services are those directly used or enjoyed by humans and do not include ecological processes but instead identify the beneficiaries (O'Higgins et al., 2010). Although intermediate services are important for ecosystem service delivery, focusing on final services can avoid double counting as intermediate services can contribute to multiple final ecosystem services (Birkhofer et al., 2015). For example, pollination is an intermediate service which underpins food provision in agricultural ecosystems and valuing both instead of only food provision would lead to double counting (Markandya, 2016). Ultimately, ecosystem services provide benefits to the end user (Petter et al., 2013). For example, in the case of a cultural ecosystem service, anglers benefit from recreational angling in a recreational fishery, (Haines-Young & Potschin, 2010). These benefits can be categorised as either use, related to services such as food provision and direct interaction with ecosystems through angling or hunting, or non-use such as aesthetics and existence (Hayes et al., 2018). If they do not relate to any beneficiaries then these are not services but instead considered ecosystem functions, or the potential that ecosystems have to deliver ecosystem services, which exist whether people benefit from them or not (European Union, 2013).

As ecosystems are complex, so is ecosystem service provision, especially as many ecosystem services are provided by similar components of ecosystems. Therefore, understanding how ecosystem services interact is important (Rodríguez et al., 2005, 2006). For example, utilising one ecosystem service can result in the reduction of other ecosystem services, known as trade-offs, such as catching fish for food will reduce the fish available for recreation (Rodríguez et al., 2006; Cord et al., 2017). Trade-offs can also occur when protection of one service negatively impacts the provision of another service, such as increasing timber production reduces soil and water conservation of forests (Chiang & Chuang, 2016). In contrast, ecosystem services can interact with each other to enhance their provision, often a win-win situation known as synergies, where using one service increases benefits supplied by another or managing and protecting one service will inadvertently protect or enhance another (Rodríguez et al., 2005; Cord et al., 2017). An example of synergies is improving pollination by planting wildflower meadows will also increase erosion control (Lindborg et al., 2017; Y. Zhang et al., 2021). Trade-offs and synergies are usually determined by weighing the costs and benefits, but this can lead to some services being valued over others (Costanza, 2020). Understanding trade-offs and synergies between ecosystem services and different decision-making outcomes is important to understand how ecosystem services interact (Lindborg et al., 2017; Ellis et al., 2019).

Different framings exist to conceptualise ecosystem services as well as how they might interact. The cascade model, for instance, is one of the first models to accompany ecosystem service assessment. It separates services into intermediate and final services as well as into its environmental and socioeconomic parts (De Groot et al., 2002; Haines-Young & Potschin, 2010; Potschin & Haines-Young, 2016; Czúcz et al., 2020). Cascading frameworks implement chains or pathways to connect from ecosystem structures and processes to functions and services all the way to social benefits (Potschin & Haines-Young, 2016; Zhang et al., 2022). However, unclear methods and guidance have led to conflicting interpretations of definitions hindering applications of the framework (Czúcz et al., 2020). Another conceptual framework is The Intergovernmental Panel on Biodiversity and Ecosystem Services (IPBES), that centres around decision-making as a driver and links various disciplines, instead of a chain-link pathway, whilst also considering the spatial and temporal scale (Díaz et al., 2015). Other framings include similar cyclical-based frameworks which map the interconnectedness of beneficiaries and ecosystems and their feedback loops (Daily et al., 2009) or underlie ecosystem service classifications such as The Economics of Ecosystems and Biodiversity (TEEB, Kumar, 2012). These framings all serve to better understand and communicate the ecosystem service concept. However, using the various framings with varying structures and definitions can influence the way human well-being and beneficiaries are recognised. The presence of feedback loops in cyclical frameworks are an essential addition to understanding provision especially in regard to potential trade-offs and synergies

(Heink & Jax, 2019). These framings highlight interactions between ecosystem services but are best understood with ecosystem service classifications which help to categorise ecosystem services.

Many typologies exist for different ecosystem services such as the Millennium Ecosystem Assessment (MEA, Millennium Ecosystem Assessment, 2005), TEEB (Kumar, 2012), and the Common International Classification of Ecosystem Services (CICES, Haines-Young & Potschin, 2023). Ecosystem services have broadly been grouped into four overarching types: i) provisioning (extractable goods); ii) supporting (intermediate); iii) regulatory (underlying processes); and iv) cultural (non-extractive interactions with ecosystems) (European Union, 2013). Both MEA and TEEB adopt these four broad groups, however, supporting services vary from the other groups as they do not directly benefit human well-being and are often considered intermediate services (Haines-Young & Potschin, 2010). A focus on final ecosystem services has led to supporting services no longer categorised in later typologies such as CICES which adopts only three overarching types as well as separates biotic and abiotic services. For these reasons, this thesis adopts CICES typology (Haines-Young & Potschin, 2010; European Union, 2013). CICES, with its hierarchical structure sub-divided into classes and class-type, allows detailed and measurable categories (Kosenius et al., 2013). This method of classifying ecosystem services allows better identification to select services for ecosystem service assessment through mapping and valuation and ultimately to better understand their delivery (Czúcz et al., 2018). As many classification systems exist and the ecosystem services concept is still evolving, it is important to specify which classification is applied within research and why.

1.2 How is ecosystem services delivery assessed?

Ecosystem service assessment serves to assess the baseline, monitor changes, and evaluate decision-making and policy outcomes (Neugarten et al., 2018). Broadly speaking ecosystem service assessment can be through bottom-up or top-down approaches (van Wensem & Maltby, 2013; IPBES, 2016; Faber et al., 2019). Bottom-up approaches typically take advantage of existing data to connect to ecosystem service delivery. Top-down approaches focus on framing and defining focal ecosystem services (usually in relation to spatial and temporal scales) to conceptualise the question at hand (Sieben et al., 2018). The application of top-down or bottom-up approaches depends on the context of assessment and type of service, such as specific services (e.g. nutrient cycling) can be assessed with bottom-up assessment with detailed biophysical methods whilst higher-level and qualitative analyses might be more appropriate for broader ecosystem services such as recreational fisheries (Apitz, 2013). However, applying only bottom-up or top-down approaches can restrict decision-making to the scale of assessment, as they are not mutually exclusive. Instead combining a bottom-up with a top-down approach can be a powerful tool and can capture both the conceptual (top-down) and data-driven (bottom-up) assessments of ecosystem service delivery.

There are multiple methods to assess ecosystem service assessment including mapping, valuation, modelling or quantification (Maes et al., 2012; European Union, 2013; Birkhofer et al., 2015; Cheng et al., 2019). However, determining which method is most appropriate for assessment is difficult as many databases and reviews exist but all vary, different methods have their own strengths and weaknesses, top-down and bottom-up approaches apply different methods, and the literature is dominated by a few tools (Crossman et al., 2013; IPBES, 2016; Sieben et al., 2018; Cheng et al., 2019). For example, mapping is effective at visualising where ecosystem services are produced, sourced, and used, as well as the location of their beneficiaries. However mapping is data dependent, based on land use and land cover maps, depends on the ecosystem service indicators used which can be either qualitative or quantitative, and there is uncertainty around scaling mapping (Maes et al., 2012; Birkhofer et al., 2015; Tardieu, 2017; Czúcz et al., 2020). Qualitative and quantitative methods also vary. Qualitative assessments involve identifying stakeholders and prioritising sites, whilst quantitative assessments measure or assign a specific value (Neugarten et al., 2018). For example, modelling and quantifying services are quantitative methods whilst valuing can be either qualitative or quantitative. Applying a combination of quantitative and qualitative assessments is beneficial. For example, qualitative assessments have the benefit of scoping and identifying services and beneficiaries whilst quantitative can better identify trade-offs and synergies as well as measure or model services in biophysical or economic terms, both important for ecosystem services assessment. In addition to this, no one method can capture all details of ecosystem service delivery so often an integrated approach is applied with many methods (IPBES, 2016). This is why ecosystem service tools tend to apply more than one method of assessment such as valuation and mapping to help span multiple biological, ecological, and spatial scales (Section 1.3).

Assessing service delivery by predicting how changes in SPUs affect ecosystem functions and services is hampered by a paucity of quantitative links from SPUs to final ecosystem service provision and tools which can assess these links (Faber et al., 2021; Maltby et al., 2021). These linkages are better defined as ecological production functions (EPFs) which predict the impact on ecosystem service delivery as a result of changes in SPUs (Daily et al., 2009; Wainger & Boyd, 2009; Faber et al., 2021). EPFs provide a means of quantitatively linking SPUs to ecosystem service delivery, but if this is not possible, a qualitative approach with evidence-based logic chains can be used instead (Hayes et al., 2018; Maltby et al., 2021). EPFs can also be useful to determine trade-offs and synergies, as well as supply and delivery of ecosystem services (Mandle et al., 2021). EPFs can quantify impacts based on ecosystem functions, indices, proxies, or indicators, where a number have been formulated for ecosystem services such as for freshwater provision (Logsdon & Chaubey, 2013; Birkhofer et al., 2015; Czúcz et al., 2020). These indices and indicators allow modelling outputs to be translated into ecosystem service delivery. However, they can only link to final ecosystem services and require translation to assess the value to beneficiaries (Faber et al., 2021).

Beneficiaries must be identified in order to assess the value of ecosystem services. Identifying beneficiaries is dependent on the context related to characteristics such as spatial scale, culture, and accessibility, as well as deciding the most appropriate approach of valuation (Crossman et al., 2013; IPBES, 2016). Value goes beyond just monetary and can be related to intrinsic value for beneficiaries or assigned a social context, whether that is holistic, well-being, economic, cultural, moral, aesthetic, or spiritual (Potschin & Haines-Young, 2016; Pascual et al., 2017; Cheng et al., 2019). This applies to many cultural ecosystem services such as parks and recreation which have an aesthetic and scenic value (Cheng et al., 2019). Although, most ecosystem services which have been valued are those which can be easily valued economically (i.e. monetary value) (Ellis et al., 2019). Socioeconomic valuation is usually in the form of cost-benefit analyses, willingness-to-pay, market price, benefit transfer, or with other economic methods (Costanza et al., 2017). However, this has led to an underrepresentation of services which do not fit into economic framing (Ellis et al., 2019). In addition, purely focusing on economic valuation only translates to economic incentives in policy and can cause conflicts between stakeholders, as the meaning of a monetary value is vague, it diminishes the value of nature, and it does not necessarily reflect people’s true perception of nature’s value. (Dallimer et al., 2014; Pascual et al., 2017). Classifications such as nature’s contributions to people (NCP, from IPBES) (Pascual et al., 2017), have approached this problem by centring value as more inclusive by reframing cultural services into its three main groups of regulating, material, and nonmaterial to boost their assessment (Kadykalo et al., 2019). There is clearly an added benefit in investigating the process up to (monetary) valuation where the utilisation of the service by beneficiaries is also understood and allows for a more focused (non-monetary) valuation of the service. A growing number of socioecological methods have been applied in combination with economic methods in ecosystem service assessment with non-monetary methods such as social media, observation, interviews, visitor numbers, and participation (Cheng et al., 2019; Kadykalo et al., 2019).

Methods for assessing ecosystem services are unclear, constantly evolving, and most of the time depend on economic valuation to determine value. Deciding which method to apply is dependent on previous reviews and databases and often each have their own considerations and aims (Neugarten et al., 2018). As a result, there is debate on how best to carry out an ecosystem services assessment, especially in cases where it is difficult to translate ecosystem service delivery into value for beneficiaries, as is the case for cultural ecosystem services (Birkhofer et al., 2015). Determining beneficiaries and impacts on them, beyond economic value, is also important. Therefore, research is needed to map out which tools are applicable to which type of assessment. This is especially important for determining the impact of spatial scales (Section 1.3) or threats and pressures on ecosystem service delivery (Section 1.4).

1.3 What are the importance of spatial patterns and scales in ecosystem service delivery and assessment?

Spatial scale is a fundamental component of ecological theory, underpinning the provision of ecosystem functions and processes (Wiens, 1989), and hence ecosystem service delivery. Observation of ecosystems, their structures, and processes is dependent on spatial scale as ecological patterns may be more or less detailed than their underlying structures and functions (Estes et al., 2018). For example, smaller spatial scales are effective for assessing biotic ecosystem service delivery by SPUs of communities such as forests and canopy cover within a region, whilst larger scales are appropriate for abiotic ecosystem services such as atmospheric reduction of carbon dioxide (Feld et al., 2009; Lindborg et al., 2017). Therefore, spatial scale is important for the delivery, assessment, and management of ecosystem services.

Spatial scale refers to the geographical area relevant for ecosystem service measurement (Feld et al., 2009). The spatial units used in ecosystem service assessment are typically ‘local’ (small scale e.g. < 100 m^2), ‘landscape’ (e.g. 100 m^2 – 1 km^2), ‘regional’ (e.g. 1 – 100 km^2), ‘national’ (defined by country boundaries) and ‘global’ (worldwide) (Lindborg et al., 2017; Deeksha & Shukla, 2022). Ecosystem service delivery relies on the SPUs and ecosystem processes which can span multiple biological scales, from individuals to populations to communities, and which are generated at a range of ecological scales (Tardieu, 2017). Tardieu (2017) classified the most relevant ecological scales for ecosystem service assessment as: ‘ecosystem’ (e.g. < 1km^2); ‘catchment’ (e.g. 1 – 10,000 km^2); ‘landscape’ (e.g. 10,000 – 1,000,000 km^2); and ‘global’ (e.g. > 1,000,000 km^2). Tardieu’s (2017) definition of ‘ecosystem’ can be better understood as ‘habitat’, the environment where an organism or population lives, which will be the term used in this thesis (Hein et al., 2006; EFSA Scientific Committee, 2016b; Tardieu, 2017). The biological, ecological, and spatial scales relevant to ecosystem service assessment and delivery are summarised in Figure 1.1.

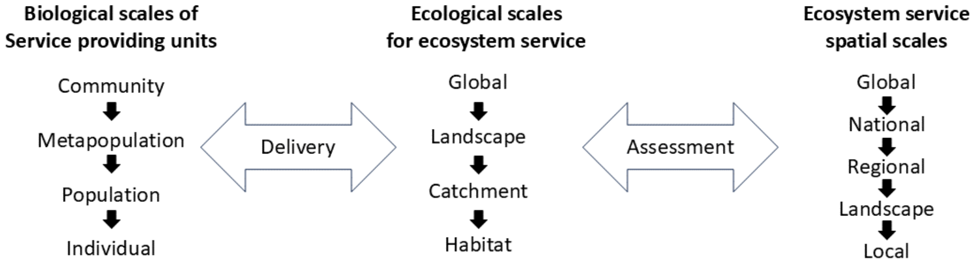


Figure 1.1 A summary of relevant biological, ecological, and ecosystem service spatial scales for ecosystem service delivery and ecosystem service assessment.

Bridging disciplines between biological, ecological, and ecosystem services scales is a challenge (Hein et al., 2006; Birkhofer et al., 2015). For example, ecologists need to scale site research to scales relevant for management of ecosystem services, however, outcomes might not translate across scales (Birkhofer et al., 2015). Taking water quality, ecologists may find that ecological processes indicate that there is low intensity water purification by surrounding riparian zones into a stream site, but this may not translate to improved water quality at a regional-level due to the heterogeneity across a landscape and, therefore, be insufficient to meet water quality standards set by environmental management (Hein et al., 2006; Lindborg et al., 2017; Tardieu, 2017). To add to this, ecosystem services are not heterogeneous across space as the ecosystems which provide them differ in properties such as species composition (Fisher et al., 2009; Maltby et al., 2018). The species composition of assemblages and the spatial structure of populations is a function of spatial population dynamics and the movement of individuals across landscapes as described in theories on metacommunities (Leibold et al., 2004) and meta-ecosystems (Loreau et al., 2003). Also, ecosystem service flows can range from finer, local scales, with a lack of accurate data, to larger scales (e.g. national and global) which is often the focus for most studies (Holt et al., 2015). In management practises the spatial scale is defined or framed with respect to the management focus (e.g. protecting ecosystem services or ecosystems) but this may disregard the supporting structures, functions, or other factors which all operate at their own scales, as well as any feedback beyond the management focus. For example, no-till farming can increase carbon sequestration and agricultural yield on a local scale, but on a larger scale can have negative effects and lead to increased populations of pests such as mice (Lindborg et al., 2017). Instead management focus can be defined by service providing areas or the spatial units or areas producing ecosystem services (Fisher et al., 2009; Syrbe & Walz, 2012). Ultimately, there is a clear challenge in translating between spatial scales and disciplines.

One way to address the challenge in translating spatial scales is to assess multiple scales to fully understand ecological patterns as they might extend beyond one defined scale (Birkhofer et al., 2015; Estes et al., 2018). This develops a more detailed picture which can be used to identify trade-offs or other interactions such as supply and demand between scales (Tardieu, 2017). For example, populations themselves are spatially structured and not uniform within their environment. The behaviour of these populations influences interactions across space, in other words, whether the organisms which make up the populations are mostly stationary or mobile will influence their ability to interact with their environment (EFSA Scientific Committee, 2016a). The spatial scale of measurement may reflect the interacting populations in a landscape but the SPU could be one specific population in its local habitat (Feld et al., 2009). This is a result of interactions between populations and their environment, where populations can respond in different ways such as through migrations, recolonisation, and extinctions or possible source-sink populations (Ares, 2003). One method to address multiple spatial scales is by using spatially explicit models of EPFs to assess delivery in response to spatial scales (Birkhofer et al., 2015).

An identified challenge for the ecosystem services concept is deciding which spatial scales are appropriate. Investigations can often be conducted at finer and more detailed scales in order to extrapolate to a wider scale assessment (Turner et al., 1989). For example, ecological focus is often done on short-term and local scales whilst ecosystem service protection is based on large spatial scales requiring extrapolation between the two different spatial scales (Birkhofer et al., 2015). This, however, relies on not losing information during this extrapolation. Care must be taken to include relevant information across scales and often validation of the transferability verified by established models, otherwise this can lead to a mismatch between the spatial focus of the two endeavours (Ochoa & Urbina-Cardona, 2017; Meng et al., 2020). Furthermore, ecosystem services are not only managed at varying spatial scales such as from local to global but there is a mismatch between scales at which services can be used and where they are made (Lindborg et al., 2017). “Hot spots” and “cold spots” can be determined to better understand where there are trade-offs and synergies, species contributing to multiple ecosystem services, or supply and demand overlaps. “Hot spots” and “cold spots” can influence spatial prioritisation of decision-making, for example, ecosystem function is usually more meaningful at the landscape or catchment-scale (Tardieu, 2017; Comín et al., 2018). Overall, considering appropriate spatial scales and scaling within assessments is a significant challenge in understanding the provision of ecosystem services (Birkhofer et al., 2015). Understanding how ecosystem service delivery can vary between scales or be translated across scales is especially important in ecosystem service assessment.

1.4 What are the threats to ecosystem service delivery?

Ecosystems services are under constant threat from multiple anthropogenic pressures such as land use, climate change, and toxic chemical pollution, causing environmental stress in the form of chemical, physical, or biological changes to ecosystems (Maltby et al., 2016; Rau et al., 2019; J. Wang et al., 2021). The triple threat of 'climate change, unsustainable land use and the sixth mass species extinction' all call for an urgency to protect ecosystems (O'Higgins et al., 2020). As a result nature has been significantly altered leading to a rapid decline in ecosystems, biodiversity, and the benefits humans receive from nature (IPBES, 2019; Z. Wang et al., 2021). This accelerated loss in biodiversity and ecosystem function has prompted concerns of reduced delivery of ecosystem services, which have been assessed at the global level by IPBES (IPBES, 2019a). This decline in ecosystem service delivery leads to pressures of unmatched supply and demand (Faber et al., 2021) which leads to overexploitation, degradation, destruction, and unsustainable use of ecosystem services (Haines-Young & Potschin, 2010).

Threats on ecosystem services are either a result of direct drivers or pressures which directly impact ecosystem service delivery or indirect drivers such as underlying causes of direct drivers (IPBES, 2016). Both direct and indirect drivers of change have accelerated in the last 50 years with the largest impact from direct drivers of land use, exploitation of organisms, climate change, pollution, and invasion of alien species and indirect drivers of human population dynamics, consumption, technology, and global governance (IPBES, 2019b). For decades, the importance of protecting ecosystem services from chemical exposure has been identified (Brown et al., 2017). Pollutants from waste, industry, mining, agriculture, toxic dumping, and oil spills all negatively impact ecosystems (IPBES, 2019b). Therefore, this thesis narrows its focus primarily to the examination of chemical stress on ecosystem service delivery.

Methods for quantifying impact depend on the direct driver. For example, land use or land management pressures can be suitably assessed with mapping to understand ecosystem service delivery changes, whilst effects of climate change are best assessed with process-based modelling (e.g. hydrological cycles, vegetative cover, soil erosion) which underlie ecosystem service tools, however, these methods are not mutually exclusive (Crossman et al., 2013). In this case of pollution or toxic chemicals, selecting a method is more nuanced as impacts are seen on lower levels of biological organisation, such as in populations and communities, making it necessary to be able to predict the effects of their changes on ecosystem functions. To add to this complexity, chemical stressors themselves can vary spatiotemporally from acute and local events such as point-source oil spills (Kaikkonen et al., 2021) to long-term chronic contamination (Seiler & Brinkmann, 2022). SPUs themselves can also respond to chemical stress where on one hand a local population impact might be mitigated through interacting populations with migrations from healthy populations (i.e. recolonisation) to impacted populations (Ares, 2003). On the other hand, dispersal of individuals from a stressed population can also spread chemical stress impacts to unstressed habitats. For example, lowering birth rate in an interacting population as a result of reduced spawning potential in stressed individuals (Schneeweiss et al., 2023). Chemical stress can not only disrupt spatial structures, such as source-sink dynamics within these populations, but can also produce dynamics where they might not have otherwise existed, for example, earthworms changed their population dynamics in response to soil with copper contamination (Ares, 2003). Furthermore, chemical exposure can have indirect effects on SPUs' population structures through impacts on lower trophic levels (e.g. food availability) altering ecosystem service provision as well as interact with other effects to either increase or decrease the impact (Pesce et al., 2023). Understanding these spatial scale overlaps or "hot spots" between chemical exposure and ecosystem delivery can help to determine where and what is needed to protect (Johnston et al., 2017).

A significant challenge to assessing the impacts of chemicals on ecosystem service delivery is linking threats by extrapolating impacts on SPUs or ecosystem functions to changes in ecosystem service delivery (Devos et al., 2019). In the case of chemical stressors this involves linking ecological impacts to human well-being by identifying the final ecosystem service impacted, although intermediate services can also be directly impacted (van Wensem et al., 2017). A method for protecting ecosystem services against the adverse effects of chemicals, both direct and indirect, is through ecological risk assessment. This enables a better understanding of how pressures influence changes across a spatial and temporal scale but also highlights the need to prioritise spatial scaling in environmental management or new approaches, such as the ecosystem services concept in risk assessment.

1.5 How is the ecosystem services concept used in chemical risk assessment?

Risk assessment is organised into hazard identification, effect assessment, exposure assessment, and risk characterisation (Ragas, 2011), where ecological risk assessment evaluates the environmental effects of stressors (Chen et al., 2013) and presents scientific advice to aid regulatory decision-making or management of risk. In the European Union (EU), there are different regulations for chemicals (REACH, Regulation (EC) No 1907/2006), biocides (Regulation (EU) No 528/2012), and plant protection products (Regulation (EC) No 1107/2009), and assessments are performed either by the European Chemicals Agency (ECHA, chemicals and biocides) or by the European Food Safety Authority (EFSA, plant protection products). For the most part, risk assessment follows three main steps; i) problem formulation; ii) hazard and exposure assessment; and iii) risk characterisation with a tiered approach depending on complexity, data availability, or risk mitigation (Benfenati, 2022). For hazard and exposure assessment, current risk assessment adopts a proxy-based approach, by either carrying out standardised testing for one chemical at a time on a test species and applying safety factors (Martin et al., 2014) or assessment factors to extrapolate effects on the individual to population-level (Grimm & Martin, 2013). Standardised tests make up much of what is required under EU regulations. However, this practice relies on the assumption that individual and higher levels such as population are intrinsically linked where responses at the individual-level directly translate to higher levels (Maltby et al., 2018; Forbes et al., 2019). In reality, effects across biological organisation are more nuanced between individuals and higher levels as factors can mitigate or exacerbate effects such as population dynamics, density dependence, and environment variability (Forbes et al., 2019). Additionally, standardised test species are not always relevant SPUs for ecosystem services delivery. This is only the case in specialised populations such as bees which are directly involved in ecosystem service delivery of pollination and even in these cases extrapolation is inferred (Maltby, 2013). There is a need for increasing ecological relevance of risk assessment and expanding beyond the one species – one test approach.

To address this need for ecological relevance, the ecosystem services concept has gained traction in its application in chemical risk assessment (Forbes & Calow, 2012; Nienstedt et al., 2012; Munns et al., 2016; Maltby et al., 2018; Faber et al., 2019). Ecosystem services not only improves ecological and societal relevance, but can help to better define specific protection goals (SPGs), which are more specific than current general protection goals (Maltby et al., 2018; Devos et al., 2019; Faber et al., 2019). General protection goals help develop problem formation and broadly define which environmental components should be protected such as protecting biodiversity (EFSA Scientific Committee, 2016b; Devos et al., 2019). However, general protection goals have been considered too generic and vaguely defined in legislations and hence not easily measurable from a scientific perspective (Forbes & Calow, 2012). Instead SPGs, formulated in terms of ecosystem services, can specify what environmental components to protect, maximum impacts that can be tolerated, when to protect, and where (Faber et al., 2021). SPGs define the ecological entity to protect which refers to the level of biological organisation of the relevant SPU, attributes such as abundance, magnitude of relevant effects, and the spatial and temporal scale of tolerable impacts (Devos et al., 2016; EFSA Scientific Committee, 2016b; Faber et al., 2021). Guidance for defining SPGs and implementing ecosystem services to plant protection products (Regulation (EC) No 1107/2009), and food additives under the regulation of the European Food Safety Authority (EFSA) have been developed (EFSA Scientific Committee, 2016b) with guidance on problem formation (European Commission, 2024) which have potential applications to other chemical groups (Maltby et al., 2017). However, there is a clear need to better define and align the concept amongst ecotoxicology researchers (Pesce et al., 2023). In order to frame SPGs in terms of ecosystem services, ecosystem services assessment needs to be transferable and understood by ecological risk assessment.

One such method for implementing the ecosystem services concept into risk assessment is with EPFs. Several EPFs relevant for chemical risk assessment have been identified from the literature (Faber et al., 2021). Faber et al. (2021) found EPFs covering a range of provisioning and regulatory & maintenance services, according to CICES classification, with the most prominent related to pollination. However, they concluded that other than the honeybee and pollination, no EPF was found which covered the entire range from standard testing species to ecosystem service valuation. Without the availability of standardised toxicity tests for SPUs, direct extrapolation from chemical effects on a standardised toxicity test individual all the way to ecosystem service delivery cannot be done (Faber et al., 2021). Furthermore, EPFs often focus more on ecosystem functions that drive ecosystem service delivery, ecological processes, or supporting services and still require translation to final ecosystem service delivery and value (Faber et al., 2019).

An alternative approach to EPFs involves *in silico* modelling (European Chemicals Agency, 2016) where models can utilise available EPFs (Faber et al., 2021). Currently, *in silico* methods are mainly applied in regulation to support and fill data gaps in the absence of ecotoxicological data (i.e. data poor scenarios) or for data intensive scenarios or landscape assessment (Benfenati, 2022). These methods are most often either conceptual, such as adverse outcome pathways, or related to read-across and dynamic energy budget models (DEB). However, they are not capable of modelling more complex ecologically relevant features including behavioural and habitat-level influences that can either remediate or exacerbate chemical impacts for an individual. Instead there has been extensive application and potential for modelling approaches which are capable of converting information from standardised tests on vital rates to population-level parameters in the form of individual-based models (Galic et al., 2010; Forbes & Calow, 2012; Grimm & Martin, 2013; Martin et al., 2014; Forbes et al., 2019). Individual-based models are population (or community-based) models which simulate how individuals behave in response to their abiotic and biotic environment and their own characteristics (Grimm & Martin, 2013).

Chemical stressors can cause long-term subtle changes to ecosystems across spatial scales (Cairns & Niederlehner, 1994) where spatial scaling has been identified as a priority challenge for applying the ecosystem services concept in chemical risk assessment (Maltby et al., 2018). For this reason, there is a need to identify the effects of stressors on ecosystem service delivery across spatial scales. However, assessing across spatial scales in risk assessment involves extrapolating between biological scales such as from effects on an individual to the ecosystem, a major challenge in ecotoxicology (Schneeweiss et al., 2023). In addition to this, current ecosystem service methods such as mapping are not capable of assessing individual-level impacts and are more often associated with biological data on functional traits, ecosystem structure, or habitat data (Maes et al., 2012). As methods in ecosystem service assessment are not directly transferable to risk assessment, and methods in risk assessment require extrapolation across scales, there is a need to identify appropriate methods. Models capable of extrapolating across biological hierarchy such as individual-based models do have the capability to be spatially explicit and model across spatial scales but this is dependent on their development and data availability. Spatially explicit and landscape assessments has typically been approached in risk assessment through field studies or modelling, mostly applied to retrospective assessments (Benfenati, 2022). As ecological risk assessment relies on both retrospective and prospective risk assessment, previous studies have found that the possible methods and key factors for both of these scenarios vary (e.g. prospective can model existing data whilst retrospective highlights the resilience of a system) (Maltby et al., 2021). Simulation with ecological models has been applied as an important method to tackle extrapolating chemical stress on lower levels of biological organisation to higher and more ecologically relevant levels (Roast et al., 2007). Ecological modelling, such as individual-based models, spans many forms and is capable of producing multiple endpoints including abundance and spatial distribution of individuals, populations, and communities, as well as trophic structure, species richness, and food webs (Mintram et al., 2018b; Larras & Charles, 2022). As models do exist which are capable of modelling populations, metapopulations, ecosystems, and landscapes, however, whether they are capable of implementing the effects of toxic chemicals is still unclear. Studies suggest applying ecological models in ecological risk assessment which could be useful in ecosystem service assessment for population-based or community-based ecosystem services (i.e. where the SPU is dependent on a population and can be directly linked to the service) (Galic et al., 2010).

A challenge for ecosystem service based chemical risk assessment is understanding how stressors impact ecosystem service delivery across spatial scales (Devos et al., 2019). The scale at which regulation and exposure of chemical stressors occurs does not always match ecosystem service delivery or assessment. For this reason, SPGs need to be spatially defined whilst also considering larger scales such as the landscape (van Wensem & Maltby, 2013). Spatially explicit models are needed to assess ecosystem service provision but models that can link ecosystem services to risk assessment as well as impact to beneficiaries across spatial scales needs to be identified.

1.6 What ecosystem services are delivered by freshwater catchments?

Freshwater ecosystems include surface flowing (e.g. lotic, rivers), standing (e.g. lentic, lakes), and sub-surface (e.g. groundwater) waters as well as wetlands, and riparian ecosystems (Geist, 2011). Although only 3% of the world's water is freshwater, they are the most impacted ecosystem (Geist, 2011; Palmer & Richardson, 2014) with anthropogenic pressures specifically on rivers expected to escalate in the coming years (Guerrero et al., 2022). In ecosystem service assessment, freshwater ecosystems are typically underrepresented compared to terrestrial ecosystems even though freshwaters such as rivers are influenced by their surrounding terrestrial catchments (IPBES, 2016). Catchments are bound areas of a landscape that make up the area surrounding surface waters particularly important in assessments (Vermaat et al., 2020). The greater catchment can influence freshwaters and their hydrological processes, for example, urban, agricultural, and terrestrial ecosystems surrounding freshwaters influence water flow and run-off into freshwaters (Qiu & Turner, 2013). Terrestrial processes such as soil erosion in the landscape can also influence sediment retention in freshwaters. Increases in urban land use and agricultural practices in a surrounding catchment also negatively influence freshwater water quality. Managing freshwater ecosystems is, therefore, based on the entire catchment to better capture external effects and the whole water cycle including its terrestrial component (Moss, 2012). For this reason freshwater catchments are the primary focal ecosystem of this thesis.

Freshwaters themselves provide multiple ecosystem services either related to hydrological cycling or interaction with freshwaters (Grizzetti et al., 2016). Ecosystem services dependent on the hydrological cycle are often referred to as hydrologic ecosystem services (Grizzetti et al., 2016). Hydrologic ecosystem services include water provision, water quality, climate regulation, and water retention or water-based services. Although hydrological ecosystems services do consider both the terrestrial and aquatic elements in the greater catchment, they only describe a subset of services provided by freshwaters (Lautenbach et al., 2012; Ochoa & Urbina-Cardona, 2017). Freshwater ecosystem services instead are focused on those related directly to freshwater ecosystems such as rivers and lakes. Some examples of freshwater ecosystem services include drinking water, food provision (e.g. fish or aquatic crops), hydropower, and water recreation activities such as swimming or angling (Brauman et al., 2007; Pan et al., 2020; Katharine et al., 2022). A summary of freshwater ecosystem services categorised by CICES (V5.1) and by abiotic and biotic services is provided in Table 1.1.

*Table 1.1 Table of freshwater ecosystem services from CICES (version 5.1) typology (Haines-Young & Potschin, 2018) divided into sections (provisioning, regulatory & maintenance, and cultural), and biotic and abiotic services. The CICES (version 5.1) division is in **bold** and the group listed underneath.*

Section	Biotic	Abiotic
Provisioning	**Biomass** - Cultivated aquatic plants - Reared aquatic animals - Wild aquatic plants - Wild aquatic animals	**Water** - Nutrition (e.g. drinking water) - Materials (e.g. water for non-drinking purposes) - Energy (e.g. hydropower)
	Genetic material from biota - Cultivated aquatic plants - Reared aquatic animals - Wild aquatic plants - Wild aquatic animals	
Regulatory & Maintenance	**Transformation of biochemical inputs to ecosystems**	
	- Mediation of wastes or toxic substances - Nuisances of anthropogenic origin (e.g. smell, noise, visual)	- Mediation of waste, toxics and other nuisances by non-living processes (e.g. dilution) - Mediation by other chemical or physical means
	Regulation of physical, chemical, biological conditions	
	- Regulation of baseline flows (e.g. sediment retention) - Lifecycle maintenance - Pest and disease control - Regulation of soil quality - Water conditions - Atmospheric conditions	- Regulation of baseline flows and extreme events (e.g. flood protection)
Cultural	**Direct, in-situ, and outdoor interactions with living systems that depend on presence in the environmental setting** - Physical and experiential interactions with natural environment (e.g. recreation) - Intellectual and representative interactions with natural environment (e.g. tourism)	
	Indirect, remote, often indoor interactions with living systems that do not require presence in the environmental setting - Spiritual, symbolic, and other interactions with natural environment (e.g. mental well-being) - Other abiotic and biotic characteristics that have a non-use value (e.g. identify, cultural meaning)	

Freshwater ecosystem services are impacted by the landscape and wider catchment (Lautenbach et al., 2012), highlighting the importance of spatial scale for freshwater ecosystem service assessment. Although studies often focus on freshwater ecosystem services as mainly regional services (Brauman et al., 2007), it is equally important to consider the water bodies catchment area either for their delivery or their beneficiaries. For example, there can be spatial disconnect between ecosystem service production and use, such as a clean water in rivers can be produced in upstream reaches because of protected areas but used downstream by people who benefit from clean drinking water (Neugarten et al., 2018). In the same way, ecosystem service producers and beneficiaries can be on difference scales, for example, a local habitat-scale supports fish for food provision which is utilised by beneficiaries not only elsewhere in the catchment but from different catchments (Maltby et al., 2017). Also, ecosystem service interactions such as trade-offs between supply and demand are often better understood at the catchment-scale (Kuemmerlen et al., 2019).

In particular, rivers are influenced by their catchment due to chemistry, hydrology, and sediment delivery with spatial scales for rivers usually classified as habitat – reach – segment – sub-catchment – catchment (Allan & Johnson, 1997). This added layer of spatial complexity on top of those relevant for biological, ecosystem services, and ecological scales highlights the disconnect between spatial scales across disciplines especially ecology and ecosystem service assessment (Figure 1.1). These spatial scales need to be matched in order to assess the influence of spatial patterns. Often local or habitat scales of the site or reach, respectively, are applied for ecological assessments (Kuemmerlen et al., 2019). However, finer spatial scales are complicated in rivers due to the dynamic, connected, and hierarchical nature of rivers and interactions occur across spatial scales (e.g. reach-scale hydraulics and catchment-scale hydrological patterns) (Baldan et al., 2020). While important, reach-scale alone does not consider river network-level mechanisms that can alleviate or exacerbate stressor impacts, such as habitat connectivity, and may not be appropriate to use to assess catchment-level attributes (Flotemersch et al., 2015). Therefore, complementing a reach-scale assessment with a catchment-wide assessment can help to spatially prioritise management efforts (Kuemmerlen et al., 2019). In freshwaters, the catchment-scale is often regarded as a fundamental unit, whilst spatially explicit assessments considering the arrangement of reaches are necessary for long-term management (Kuemmerlen et al., 2019). Catchments are also socioecological systems and beneficiaries to freshwater ecosystems services are influenced not only by the freshwater habitat but the greater landscape and catchment which can add in factors such as travel distance, accessibility, and human population density (Flotemersch et al., 2015). For example, a discrepancy between ecosystem services and humans can occur with increasing population density into urban areas causing a spatial mismatch in supply and demand as use is centralised in cities but ecosystem services production remains in rural areas e.g. wild crops and food consumption (Brauman et al., 2007).

In addition to spatial scales influencing freshwater catchments and their ecosystem service delivery, chemical, biological, and physical pressures pose as the greatest risk to aquatic environments (Borgwardt et al., 2019). These threats involve human activity such as land use, climate change, anthropogenic stress, and chemical pollution (Geist, 2011; Hering et al., 2015). The presence of multiple pressures on freshwater ecosystems can severely compromise their capacity to provide ecosystem services (Grizzetti et al., 2016). In relation to chemical pollution, freshwaters are the most impacted (Palmer & Richardson, 2014). Legislative efforts have been made to improve the quality of freshwaters, for example, in Europe the European Water Framework Directive (Directive 2000/60/EC) aimed to achieve good ecological status of water bodies based on the "quality of structure and functioning of aquatic ecosystems associated with surface waters" and ecosystem service profiles have been suggested as a way of assessing the ecological quality and status of ecosystems (Paetzold et al., 2010). Ecosystem service profiles present the overlap between provision of services (i.e. supply) and societal expectations (i.e. demand) for services which are important to understand in order to assess the risk to service provision under stress.

Recreation has been identified as the most studied ecosystem service in rivers and catchments (Kaval, 2019). An example of a recreational service is a recreational fishery. Fisheries can either be commercial, considered a provisioning service for food, or recreational, for enjoyment as a cultural ecosystem service (Ivanauskas et al., 2022) and these are usually considered as separate services in most framings such as CICES. Fisheries are a good example of where the establishment of a viable fish population (i.e. SPU) provides multiple services as it supports both types of fisheries (Kaiser et al., 2020) as well as the flow of supporting and regulatory services (Ivanauskas et al., 2022).

1.6.1 Recreational fisheries

Fisheries are one of the most valued ecosystem services, both socially and economically, provided by river ecosystems, whilst recreational fishing is an important cultural service (Katharine et al., 2022). Even more specific, freshwater recreational fisheries are a cultural ecosystem service supplying millions of people with recreation around the world (Birdsong et al., 2021). As freshwaters are one of the most impacted ecosystems, freshwater recreational fisheries, therefore, provide a good example where an ecosystem service approach can be applied to chemical risk assessment.

Recreational fisheries are made up of populations or communities of fish stock dependent on the fisheries management in place, such as stocking, or natural stock available within a fishery. The populations or communities of fish within the fisheries are otherwise known as the SPUs as without them the recreational fishery would not exist (Maltby et al., 2021). However, spatial structures of fish populations exist in a recreational fishery due to fish migration, resilience, and fragmentation (Kuemmerlen et al., 2019). Fish are mobile and link several spatial scales due to seasonal migration or can be disconnected in isolated populations due to habitat fragmentation (Holmlund & Hammer, 1999; Harvey & Railsback, 2011). These spatial structures and interactions between fish populations and their environment are important to determine where productivity is in a river for a fishery.

A recreational fishery itself can also exist across various spatial scales both in relation to the user (local, reach, and site-specific) and in relation to fisheries management (river network and catchment). The finer habitat scale is important to provide sufficient habitat for viable fish populations or the SPU (O'Higgins et al., 2010). Whilst the surrounding landscape will influence the benefits derived from fisheries such as competition for fishing sites and access to the fishery. To translate these scales to ecological and ecosystem scales the habitat scale can be considered as the local and reach area, and catchment to include the river network and its surrounding landscape. Spatial scales are also important factors which influence the value of an ecosystem, where rural residents interact and value a local river more than urban residents (Pan et al., 2020). Socioeconomic assessments often incorporate human preference, site choice, or willingness-to-pay practices based on the angler behaviour (Birdsong et al., 2021). Understanding recreational fisheries from the perspective of their benefits to the recreational angler is a useful tool for environmental decision-making and management. This way ecosystems and their benefits can be better safeguarded.

However, as recreational fisheries are a cultural service, they are inherently difficult to monetise (Kosenius et al., 2013). Instead, the benefit to the end user, specifically the recreational angler, can be understood in other units of utility such as the satisfaction gained from a successful fishing trip (Kosenius et al., 2013; Birdsong et al., 2021). A systematic review identified satisfaction as the most common measure of angler welfare applied independently in studies (Solomon et al., 2020). This is an important measure as recreational anglers are diverse through their fishing practices. For example, anglers' target species are two main groups of fish in recreational fisheries: game or coarse fish. Game fish includes trout and salmon whilst coarse fish includes any other species of fish (Winfield, 2016). In the UK, most anglers practise coarse fishing (Environment Agency, 2018) but game fish are often considered the more challenging and satisfying catch (Oliveira et al., 2009). Anglers are also diverse in which equipment they use (e.g. fly-fishing, netting), aesthetics they enjoy from their fishing trip (e.g. being in the water, lots of vegetation, scenery), their aspirations for the trip, and what they consider valuable from a fishing trip (e.g. catch-related or non-catch-related) (Pitman et al., 2019; Arostegui et al., 2021; Birdsong et al., 2021; Lomnicky et al., 2021). These differences are important as they can all influence angler satisfaction. In addition to angler diversity, recreational fisheries can either be managed or unmanaged. For example, in managed fisheries, there can be specific restrictions on catch limits or fish stocking to keep abundances high, which will also influence the anglers' experience (Palmer & Richardson, 2014; Ivanauskas et al., 2022). Therefore, benefits to anglers need to be framed from the perspective of the type of angler or type of recreational fishery. Assessing ecosystem service delivery based on the satisfaction it provides to the recreational angler, can better capture how benefits change due to impacts, such as those from chemical pollution, on the fishery.

A significant challenge with evaluating ecosystem services such as a recreational fishery in risk assessment is valuing the service without potentially undervaluing cultural services with monetary methods (Maltby et al., 2018). Part of this valuation is dependent on clearly identifiable human beneficiaries (van Wensem et al., 2017). However, as already mentioned, anglers are a diverse group of individuals with varying goals. Framing goals or identifying components of recreational fisheries which are important to anglers in general, can be a way forward. In addition, recreational fisheries exist across spatial scales and fish exhibit spatial structures in river networks. It is, therefore, crucial to investigate how spatial scales can influence the delivery of a stable fish population for recreational anglers.

1.7 Thesis aims and objectives

Despite efforts to apply the ecosystem services concept into ecological risk assessment approaches, understanding spatial scales is a significant challenge in assessing the provision of ecosystem services. However, methods which can apply spatial patterns as well as assess from chemical stress to impact on the beneficiary are largely unknown. Methods for applying ecosystem services in a chemical risk assessment are lacking in guidance and often focused on economic valuation for beneficiaries which is not the most appropriate valuation in the case of cultural ecosystem services. Current ecosystem service methods are not directly transferable to risk assessment, and methods in risk assessment require extrapolation across scales. Therefore, research is needed to map out which methods are applicable to risk assessment, capable of considering spatial scales, address chemical threats, and determine impacts to beneficiaries. Recreational fisheries are a good example of a freshwater ecosystem at risk to chemical stressors, where spatial scales are important, and where economic valuation is not the most appropriate method to address impacts on the value for beneficiaries, or the anglers. Understanding how various spatial scales such as the local reach-scale and catchment-scale in a case study recreational fishery is also important.

The overall aim of this thesis is to investigate how the effects of chemical stressors on ecosystem service providers at local scales relate to the delivery and risk assessment of ecosystem services at the catchment-scale, using a recreational fishery as a case study. Understanding spatial scales for recreational fisheries is important as freshwater fisheries are often managed as larger river sections across the network whilst populations of game fish are not spatially independent in rivers. Additionally, anglers are influenced by their landscape in the greater river catchment. Therefore, an integrated assessment is important to identify how chemical stressors can impact recreational fisheries.

Overall this thesis approaches this research question structured around the following four objectives:

Objective 1: To establish what ecosystem services tools and methods are currently available for freshwater catchments which could be used to apply an ecosystem services concept in ecological risk assessment (Chapter 2).

Guidance of applicable methods and ecosystem service tools to apply chemical stress in an ecological risk assessment is lacking. Although databases and reviews have identified and collated some ecological tools for extrapolating to higher levels of biological organisation these are not often developed to quantify the effects to ecosystem service delivery. In addition to this, ecosystem service tools have not been analysed for their applicability for ecological risk assessment. By conducting a systematic review, in the form of a rapid evidence assessment, available tools can be identified to determine what pressures and at which scales they perform. This review identifies which ecosystem services are currently assessed as well as information gaps in the tools. The review gives an indication of the breadth of current literature and how to address the need for finding tools which are fit for purpose.

Objective 2: To investigate how the effects of chemical stressors on fish life history parameters (growth, reproduction, survival) and fish prey abundance affect angling stock and subsequent angler satisfaction. (Chapter 3 and 4)

Objective 3: To investigate how chemical stress on a recreational fishery and angling stock spatial patterns influence angler satisfaction in a river network (Chapter 3 and 5)

To answer Objectives 2 and 3 an individual-based model was identified from the review in Chapter 2 and subsequently adapted to be fit for purpose to include the effects of toxic chemicals on individual life history parameters (in this case game fish) commonly used in ecotoxicology standardised testing (Chapter 3). The model is then applied on a river reach-level (Objective 2) to determine population responses and patterns on a local spatial scale (Chapter 4). The same model is then applied to a river network (Objective 3) and the wider landscape in the catchment to compare results between the local and landscape scale (Chapter 5) and how these influence the subsequent benefit to the end user, in this case, the recreational angler (Chapters 4 and 5).

Objective 4: Consider the implications of these findings for ecological risk assessment (Chapter 6)

Chapter 6, the final chapter, will address the implications and findings from previous Chapters (2 – 5) and their context in ecological risk assessment especially concerning the risks associated with interpreting results across an array of spatial scales. This chapter discusses how spatial patterns can be implemented into ecological risk assessment as well as in decision-making and then concludes with the findings of the thesis and future directions.

Chapter 2

Ecosystem service delivery in freshwater catchments: A review of current tools and their potential application in risk assessment

2.1 Introduction

Quantifying and understanding how ecosystem services contribute to human well-being are important for decision-making (Fenichel et al., 2010; Vigerstol & Aukema, 2011; IPBES, 2016). As the importance of assessing ecosystem services and identifying trade-offs has become more apparent, so has the need for methods and tools for this purpose. Many approaches to mapping and quantifying ecosystem services exist including valuing land cover with economic models (Daigneault et al., 2021), mapping with Geographical Information Systems (GIS) (Martínez-Harms & Balvanera, 2012; Reichel & Klug, 2018), and applying artificial intelligence (Villa et al., 2014b).

Ecosystem services are both generated and utilised at varying spatial scales (IPBES, 2016; Rieb et al., 2017) (Chapter 1). This can lead to mismatches as spatial scales might not overlap, for example, benefits such as food from crop production contribute not only to the local rural community (i.e. lifestyle for farmers and food) but also to urban areas that rely on rural food provision (Winkler et al., 2021). Furthermore, localised ecosystem service delivery may rely on ecological processes or components (i.e. service providing units or SPUs) occurring in the wider landscape. For example, instream water quality depends on catchment-level sediment delivery, water regulation, and nutrient cycling (J. Wang et al., 2014; Nabout et al., 2023). Understanding the relationship between ecosystem service delivery and their spatial scale is important as it not only influences the services available but also the tools that can be applied to quantify and map them (IPBES, 2016) (Chapter 1).

As many as eighty models and tools may be available for ecosystem service assessment (Bullock & Ding, 2018) with estimates for water regulation ecosystem service models at over 200 (Nedkov et al., 2022). Previous reviews of ecosystem service tool availability and usability have generally been limited to comparisons among a selected set of tools and their application to decision-making (Bagstad et al., 2013a; Bullock & Ding, 2018; Meraj et al., 2022; Nedkov et al., 2022). Ecosystem service tools have been applied in land use planning and to assess responses to pressures including climate change, human activity, and land use (IPBES, 2016; Torres et al., 2021). However, their use in other applications, such as ecological risk assessment of ecosystem services of toxic chemicals, is limited. Whilst the threat of toxic chemicals to freshwater ecosystem services has been recognised (Meraj et al., 2022; Nabout et al., 2023), assessments of pollution on ecosystem service delivery have tended to address general water quality or nutrient loading as opposed to toxic chemical pollution (IPBES, 2016; Meraj et al., 2022; Nedkov et al., 2022). There has been an increase in the number of studies addressing the ecological risk assessment of chemical pollution within an ecosystem service framing (Munns et al., 2016; Faber et al., 2019; Maltby et al., 2021; Peng et al., 2023) and it is timely to revisit the availability of tools that may be suitable for such an assessment. In applying the ecosystem services concept in ecological risk assessment knowing what pressures (e.g. both direct and indirect drivers, see Chapter 1) tools are capable of modelling (and whether this includes chemical stress), the spatial scales they can address, and the services that can be assessed are needed.

Systematic evidence reviews are an important and valuable research tool to map knowledge (Nabout et al., 2023) by collating and synthesising studies (Page et al., 2021). Although they have already been utilised to identify tools available for ecosystem service assessment (Wolff et al., 2015; Ochoa & Urbina-Cardona, 2017; Torres et al., 2021; Booi et al., 2022), systematic reviews can be time-consuming with most reviews requiring over a year to complete (Collins et al., 2015). An alternative, less exhaustive, but equally critical assessment of evidence, are rapid evidence assessments (Collins et al., 2015). Here I conduct a rapid evidence assessment as outlined by the Joint Water Evidence Group, JWEG (Collins et al., 2015), to identify methods and tools that are currently available to assess ecosystem services as well as highlight where there are gaps in ecosystem service tools. Preferred Reporting Items for Systematic reviews and Meta-Analyses (PRISMA, Moher et al., 2009) was developed to facilitate the transparent reporting of systematic reviews. An update of this guidance, PRISMA 2020 (Page et al., 2021), was used to augment the JWEG rapid evidence assessment guidelines.

Several methods and tools to assess ecosystem services across spatial scales do exist but understanding their application in regard to freshwater catchments is still unclear. Previous reviews have identified that most freshwater ecosystem service literature is at the catchment-scale (Hanna et al., 2018; Nabout et al., 2023). However, these reviews did not explicitly consider assessment tools and models nor the pressures applied to freshwater ecosystem services. These are both important as freshwaters are among the most impacted ecosystems (Palmer & Richardson, 2014).

Tools for assessing freshwater ecosystem services tend to focus on underlying hydrological processes rather than ecosystem service delivery. Vigerstol and Aukema (2011) reviewed four models which can be applied to freshwater ecosystem services, two of which were based on hydrological processes and require “post-processing” of results to determine effects on ecosystem services and two of which directly quantified ecosystem service outputs for a landscape. In the context of understanding spatial variation in services, and how they might be affected by stressors, it is likely to be important to focus on models which are based on underlying processes. These serve as a means of translating from stressors such as chemicals to ecosystem service delivery. For example, hydrological, ecological, and water quality models are more specific methods compared to generic ecosystem service models (Hallouin et al., 2018). However, a problem with post-processing is the need to ensure outputs are useful for ecosystem service assessment or that translation links exist (Hallouin et al., 2018) which require high data requirements and training (Lüke & Hack, 2018). On the other hand, equating to landscape changes can oversimplify underlying processes (Vigerstol & Aukema, 2011). This highlights a key difference in the approaches used in ecosystem service assessment especially regarding freshwater ecosystem services. Ultimately, in freshwater catchments, these two methods exist; 1) addressing underlying hydrological processes and translating this to delivery or 2) directly relating landscape changes to service delivery (e.g. value of surface water for drinking based on water storage abundance). Identifying the current state of literature will help to identify if these tools, which are now being incorporated into ecosystem service assessments, are capable of addressing all the way from chemical stressors impact to ecosystem service delivery.

A rapid evidence assessment can help identify the suite of tools and their applications in freshwater catchments as well as highlight gaps in current methods. The tools and methods can be specified according to spatial scale (habitat-type as well as local, regional, and global assessments) and pressure assessed, highlighting possible studies focusing on chemical stressors. This review will primarily answer the question what ecosystem services tools and methods are currently available for freshwater catchments. This could be used to apply an ecosystem services concept in ecological risk assessment by investigating:

1. What tools are available and which freshwater ecosystem services are currently assessed?
2. At what spatial scale are tools capable of modelling or assessing ecosystem services?
3. Which tools can be used to assess the effects of chemical stressors on ecosystem service delivery or could serve as translation links?

2.2 Method

This review followed the Joint Water Evidence Group (JWEG) guide on quick scoping reviews and rapid evidence assessments (Collins et al., 2015) and the PRISMA 2020 systematic review guidelines (Page et al., 2021). The only exception to these protocols is that an appraisal was not included as the purpose of this review was to identify approaches and not to assess the studies themselves.

2.2.1 Review scope

This review was limited to studies applying models or tools to assess freshwater ecosystem services in catchments. Freshwater catchments here include all inland water bodies including lakes (and ponds), rivers (and streams), wetlands, as well as the wider catchment or related ecosystems where freshwater ecosystem services have been assessed. There were no geographical limits to this review. The review considered all spatial scales for ecosystem services from local to global. In relation to the models and tools used, this included any means of assessing, quantifying, valuing, mapping, analysing, or modelling ecosystem services in response to any stressor or pressure. No limitations on the publication date of literature were applied. However, as "ecosystem services" is a relatively new concept (formally introduced 2005) (Millennium Ecosystem Assessment, 2005) most results were post-2000.

Eligibility criteria

Inclusion and exclusion criteria were applied during both initial (title, abstract, keyword) and final (full-text) screening (Table 2.1). Five main criteria were applied: 1) Is the paper in a potentially relevant field?; 2) Does the paper contain an analysis, review, or application with a model, tool, or some sort of qualitative or quantitative approach used to assess ecosystems services or their underlying processes?; 3) Is the approach(es) applied to freshwater catchments or has the possibility of being applied to a freshwater catchment?; 4) Does the approach consider assessment of freshwater ecosystem services or other similar terms?; 5) Is the full-text available?. Exclusion criteria also included socioeconomic methods such as surveys, human choice experiments, and human perception studies as socioeconomic valuation and assessment was beyond the scope of this review. By excluding socioeconomic valuation methods, the review should identify research aimed at quantifying ecosystem services, which is needed to counteract the slow uptake of applying the ecosystem services concept in management (Logsdon & Chaubey, 2013) and to better integrate into ecological risk assessment. However, tools applicable for socioecological analysis such as Social Values for Ecosystem Services (SolVES) were included as they are capable of assessing cultural services where survey data is not available and can help to assess these services based on social values (H. Zhang et al., 2019) and other socioecological methods were within the scope of this review.

Table 2.1 Inclusion and exclusion criteria for the rapid evidence assessment literature screening. Five decisions were made for each study screened: 1) relevant field; 2) contains a model or tool; 3) freshwater catchments; 4) assessed freshwater ecosystem services; and 5) full-text is available. Explanation of the inclusion and exclusion criteria are explained in depth here.

Decision	Inclusion	Exclusion
1) Is the paper in a potentially relevant field?	Yes, the paper is in a relevant field including ecology, ecotoxicology, environmental sciences, environmental chemistry, geography, chemistry, ecological economics, hydrological engineering, socioecology.	No, the paper is not in a relevant field including medicine and health sciences. Where topics in policy, management, governance, or regulatory analysis, ecological security patterns, zoning, landscape engineering are not included.
2) Does the paper contain an analysis, review, or application with a model, tool, or some sort of qualitative or quantitative approach used to assess ecosystems services or their underlying processes?	Yes, the paper describes or applies a model, tool, analysis, assessment, framework, decision support, mapping, spatial analysis, or GIS, used to assess ecosystem services or underlying hydrological processes.	No, the paper does not contain any method of assessing ecosystem services or the assessment is based on a human perception study, survey, choice experiment, monitoring study, experimental study, or willingness to pay study. Willingness to pay (and payment for ecosystem services) are included if they are applied alongside a tool assessing ecosystem services.
3) Is the approach(es) applied to freshwater catchments or has the possibility to be applied to a freshwater catchment?	Yes, assessment of ecosystem services are in freshwater catchments either in-channel (lakes, rivers, streams, pond, river network, and reservoir) or in the wider area (catchments, wetlands) with effects considered in the freshwater environments.	No, the ecosystem assessed is terrestrial, atmosphere (closed), marine, coastal, groundwater, estuary, groundwater, stormwater, salt marshes, irrigation, wastewater, glacial, snowmelt, or other terrestrial water. Also, if the study is in rangelands, grasslands, oases, deserts, dryland or floodplains or other seasonal freshwaters and connected wetlands.
4) Does the approach consider assessment of freshwater ecosystem services or other similar terms?	Yes, the paper assesses ecosystem services according to CICES freshwater ecosystems or other similar typologies such as MEA and TEEB or uses other similar terms such as natural capital and ecological production functions. Note: urban and agricultural ecosystems are assessed by if they consider freshwater ecosystem services (e.g. surface water body).	No, the study only assesses terrestrial ecosystem services in the catchment.
5) Is the full-text available?	Yes	No

2.2.2 Search strategy

Evidence for the review was obtained from peer review databases SCOPUS (accessed 05.05.2021) and Web of Science (accessed 21.05.2021). The PICO/PECO (population, impact/exposure, control, and outcome) approach (Aslam & Emmanuel, 2010; Morgan et al., 2018) was used to frame the review research question and develop search strings. PICO/PECO keywords were used to identify possible combinations relevant to literature searches based on the objectives of the review (Table 2.2).

Table 2.2 PICO/PECO was used to identify a list of potential keywords. These possible keywords for the search strings are categorised by freshwater catchments (population), ecosystem services (impact/exposure), risk (control), and assessment (outcome).

Population Freshwater catchments	**Impact/Exposure** Ecosystem services	**Control** Risk, pressures, stressors	**Outcome** Assessment (i.e. tools and methods applied)
Freshwater Standing waters Running waters Lakes Rivers Streams Wetlands Freshwater ecosystems Freshwater environments EUNIS habitats Hydrological basin Aquatic Fluvial and lotic Catchment Watershed Drainage basin	Ecosystem services Natural Capital Pollination Water erosion Water quality Water quantity Resources Ecological Indicators Water supply Nutrient retention Carbon sequestration Provisioning services Regulating services Food production Environmental benefits Environmental good Environmental service Fisheries Habitat quality Flood regulation	Risk assessment Ecological risk assessment Prospective Retrospective Predictive Resilience Adaptability Damage Vulnerability Exploitation Productivity Land use Stressors Chemicals Pressures	Tools Methods Model Toolkit Geographic information systems (GIS) Spatial mapping/analysis Valuation Resource management Comparative tools assessment Decision support Ecosystem modelling Ecosystem-based management Ecosystem management Land use planning Ecological Value Ecological Restoration Ecosystem assessment Production functions

Final search strings were compiled from a list of potential keywords using PICO/PECO (Table 2.2), a naïve search, and with the LitSearchR package in R (Grames et al., 2019) to predict search strings (Appendix Figure A2.1). LitSearchR is a quasi-automated search tool which applied text-mining and keyword co-occurrence networks to reduce bias in keyword selection.

A simple, naïve search of (freshwater) AND (model) AND (“ecosystem service*”) (327 results, Web of Science, accessed 28.04.2021), based on the PICO/PECO table (Table 2.2), input into LitSearchR (Grames et al., 2019) identified the most common words and phrases. The search words were ranked to show those which appeared most in literature and then output code was used to generate an automated search term (Appendix Figure A2.1). Comparison with the PICO/PECO table (Table 2.2) was used to remove irrelevant terms such as “coastal”, which was not within the scope of the review. Then, a combination of manual and automated methods was used to construct a final search string as the automated search returned alternative phrases for ecosystem services that were missing from the PICO/PECO table such as “food”, “fish”, and “habitat”. Therefore, the search was updated to include more specific freshwater ecosystem services. The final search string (see below) was used to screen titles, abstracts, and keywords (TITLE-ABS-KEY) for the four PICO/PECO categories (AND) and the possible keywords (“exact phrases” and their alternative spellings*) within those categories (OR).

TITLE-ABS-KEY(water* OR freshwater* OR river* OR lake*) **AND TITLE-ABS-KEY** (model* OR tool* OR "decision* support" OR "comparative tools assessment" OR "production function") **AND TITLE-ABS-KEY** ("water qua*" OR "water purification" OR "water provision" OR "water supply" OR fish* OR habitat OR "carbon sequestration" OR "carbon storage" OR flood* OR erosion* OR recreation OR pest* OR genetic* OR biodiversity OR "drinking water" OR "nutrient retention" OR food* OR "climate regulation" OR "aesthetic value" OR tourism OR "provisioning service*" OR "regulat* service*" OR "cultural service*" OR "hydrological service*" OR "water service*" OR benefit* OR "ecosystem function*") **AND TITLE-ABS-KEY** ("ecosystem service*")

Screening literature and evidence extraction

All literature was screened by a single reviewer (SMM). Initial screening of literature categorised studies as included or excluded and was carried out with exported citations from SCOPUS. Then literature from Web of Science was screened for duplicates against SCOPUS results after which non-duplicates were screened for inclusion.

Rayyan.ai was used for initial screening (Ouzzani et al., 2016). Rayyan.ai is a web-based systematic review screening tool which uses artificial intelligence and machine learning to aid screening by automatically annotating keywords and phrases identified by the user. Exported citations (.csv) screened for duplicates were imported into Rayyan.ai. After screening in Rayyan.ai, data was extracted (.csv) for retained citations and compiled into Microsoft Excel. Full-text screening with a reference manager was used for evidence extraction for the variables listed in Table 2.3. Any information which was not applicable to the study (NA) or not found within the study (NF) were marked accordingly.

Table 2.3 Variables for evidence extraction. ES = ecosystem service(s).

Variable	**Description**
Name and acronym	Name of tool or model
Type of study	e.g. case study or descriptive
Type of approach	How ecosystem services were assessed (e.g. valuation, mapping, decision support, hydrological process-based, hydroeconomic model, social media tool, empirical relationships, model results, calibrated simulation model) (Lautenbach et al., 2012)
Country or region of study	Name of study location
Habitat	Habitat or spatial scale e.g. stream, pond, river, lake, reservoir, wetland, catchment and other synonyms as classified by the study
If wider catchment, considers freshwater?	Yes, list
Total number	Number of freshwater ecosystem services evaluated
ES Section	CICES V5.1 sections e.g. provisioning, regulatory & maintenance, and cultural
Type of Freshwater ES	As stated in the study
CICES V5.1 equivalent coding	Translated services according to CICES V5.1 freshwater ecosystem services listed in Appendix Table A2.1
Timescale*	Study period e.g. days, months, years, seasons
Timeline*	Years of study (if applicable)
Number of years*	Total number of years studied (if applicable)
Temporal classification*	Assessment focus e.g. retrospective or predictive
General pressures assessed	Pressure assessed on ecosystem service delivery e.g. land use, pollution, human activity, climate change, and supply and demand
Pressure sub-type	Specific type of pressure e.g. chemical, nutrient loading, wastewater, etc.
Model input*	Data needs or input e.g. DEM, hydrological data, etc.
Model output	Output of the model e.g. map, quantification, value, yield, index value
Platform*	If hosted on a platform such as GIS, GIS toolkit, web-based, R package, desktop software, Excel

* Collected from studies but not included in subsequent analyses

Relevant freshwater ecosystem services, according to their CICES V5.1 classification (Haines-Young & Potschin, 2018) are listed in Appendix Table A2.1. Review studies which applied another ecosystem service typology, such as Millennium Ecosystem Assessment (MEA) (Millennium Ecosystem Assessment, 2005), were translated into their CICES equivalent.

Tool database selection

All tools and models were extracted from the reviewed studies and assessed for suitability. Tools were excluded if they were only found in one study; were deprecated, no longer updated, or not publicly available (e.g. LUCI, HSPF, LUONNIKAS, Estimap, BASS, ACRU); were related to frameworks (MAES) or economic methods (e.g. catch per unit effort, cost-benefit analysis, ecosystem service valuation, payment for ecosystem services) or were mathematical or spatial models which form the basis for newer tools (e.g. CASA, USLE, CA-Markov, power law models or multi criterion analyses). Suitable tools were compiled in the final tool database.

Data Analysis

Evidence from studies was analysed by grouping together summary characteristics such as number of ecosystem services assessed and location of study with the number of studies for each characteristic. Tools from the studies were then extracted and collated from studies by number of studies. Based on database selection criteria those with more than one study were then further analysed by number of studies for type of tool, freshwater ecosystem services, habitat and spatial scale, and pressure. Many studies applied more than one tool for their assessment, and therefore, freshwater ecosystem service, pressure, and habitat analyses only utilised studies where the tool was simulated alone.

All data analysis was done with R version 4.0.3 (R Core Team, 2024) in R Studio (RStudio Team, 2020) using packages dplyr (Wickham et al., 2023) and tidyverse (Wickham et al., 2019) for data manipulation and ggplot2 (Wickham, 2016), maps (Beckerand & Wilks, 2023), ggrepel (Slowikowski et al., 2024), scales (Wickham, Pedersen, & Seidel, 2023), and hrbrthemes (Rudis, 2017) for data visualisation.

2.3 Results

2.3.1 Search results

The results of the screening process are visualised as a PRISMA 2020 flow diagram (Figure 2.1). From the initial 5167 studies (after removing 2636 duplicates), 4389 studies were excluded based on inclusion criteria (Table 2.1), 778 studies (15%) were full-text screened and 332 studies (6.4%) were analysed for tools.

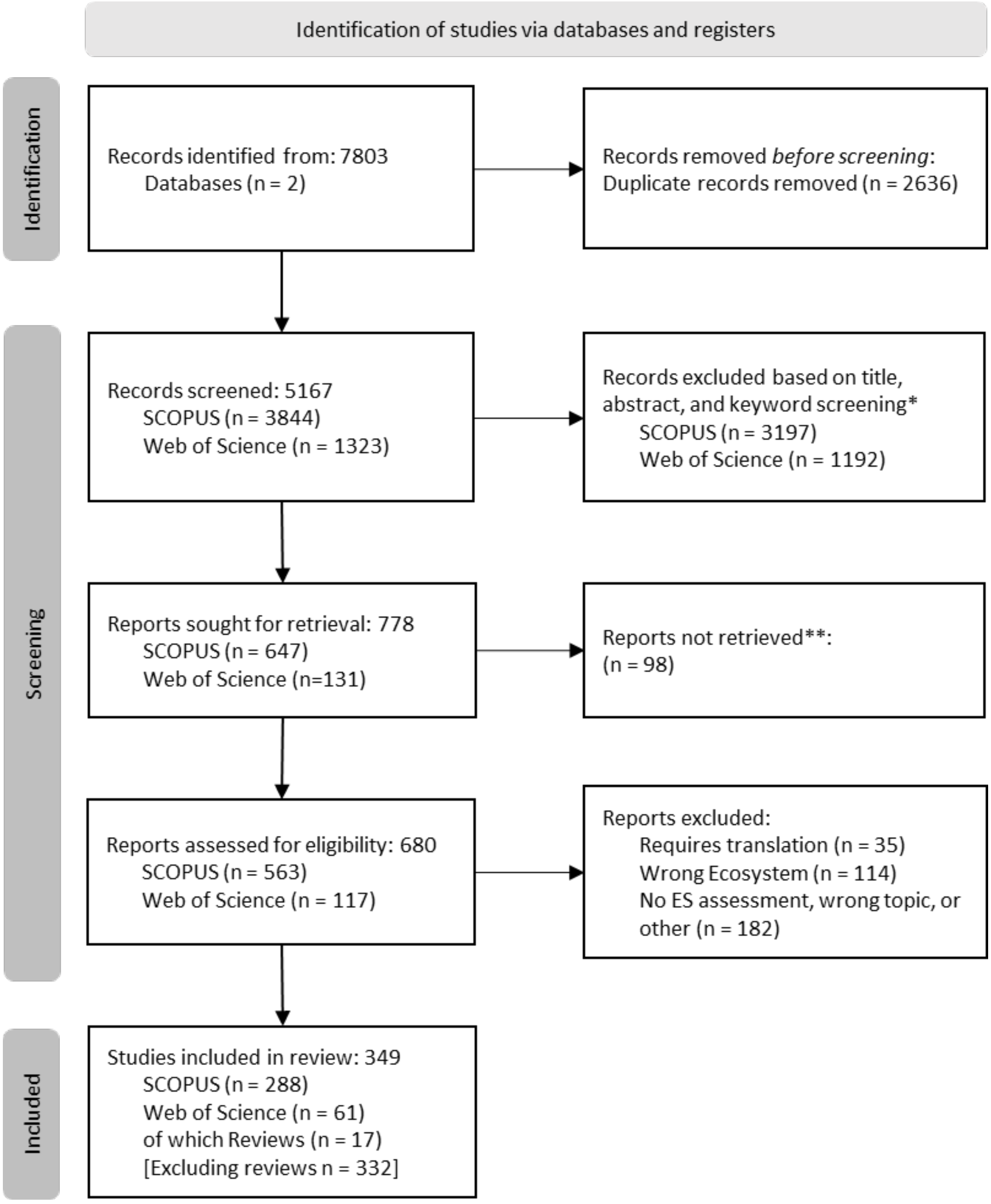


*Figure 2.1 PRISMA 2020 flow diagram for new systematic reviews which included searches of databases and registers only (Page et al., 2021). *based on exclusion criteria in Table 2. 1 **no full-text available. ES = Ecosystem service.*

There has been a clear upward trend in review studies published each year with the first studies appearing in 2004 (Figure 2.2). This reflects similar patterns seen in studies related to "ecosystem services'' in top journals (Costanza et al., 2017) and in wider literature (Fisher et al., 2009).

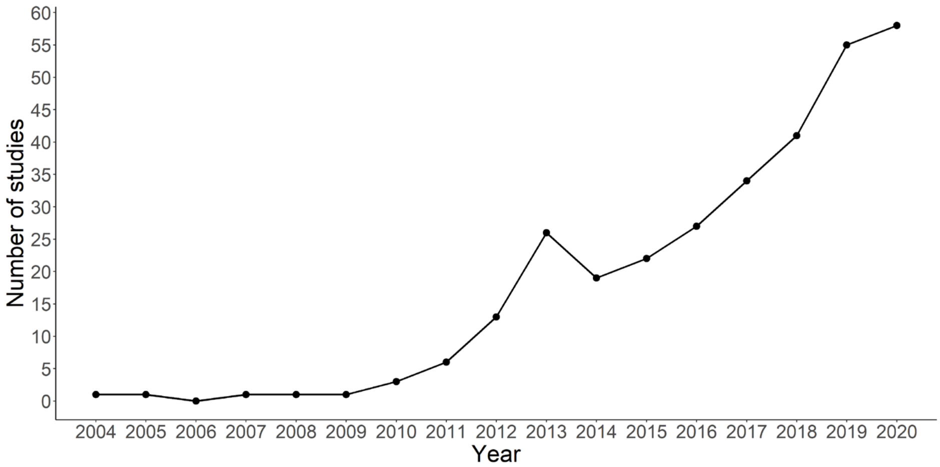


Figure 2.2 Studies included within the rapid evidence assessment by year of publication from 2004 to 2020. 23 studies were published in 2021 but as studies for the review were accessed mid-2021, these results were omitted from the graph.

The review identified studies in all continents except for Antarctica (Figure 2.3). China had the most studies with 23.5% (78) followed by the USA with 16.6% (55) and then Spain with 4.5% (15). All other countries had less than 10 studies per country. Studies performed at a regional-level or in more than one country within a certain region are not shown on the map, but included studies across Europe (12), Africa (3), Asia (2), South America (1), Mediterranean (1), and Global (1).

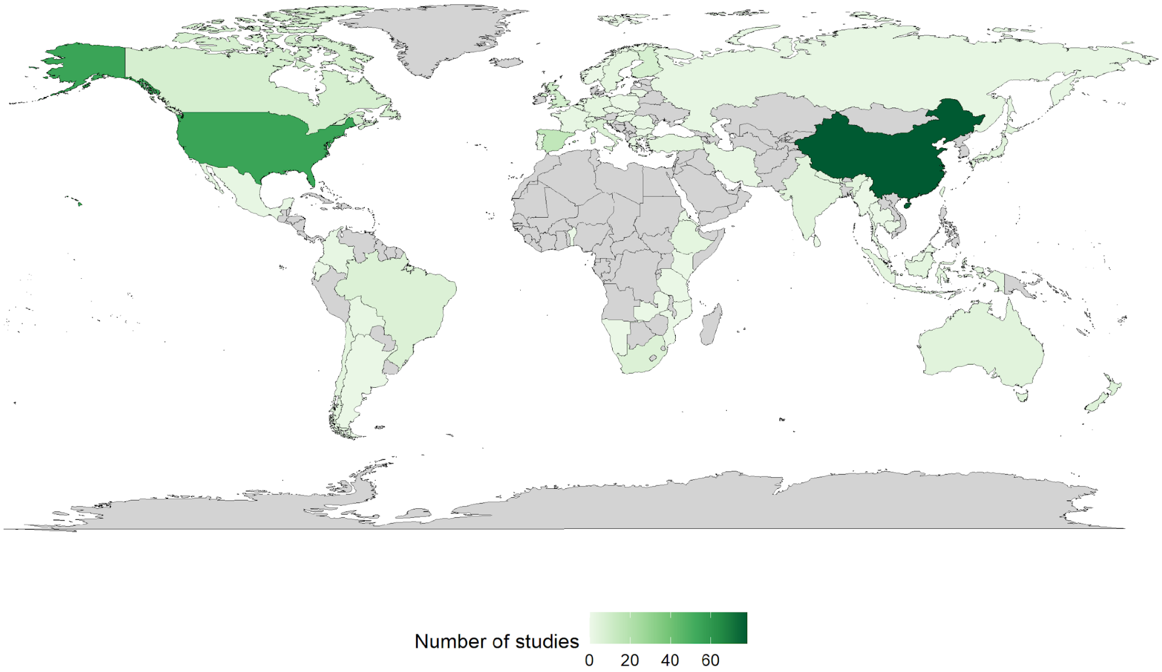


Figure 2.3 Geographical distribution of studies included within this review. The level of shading (green) represents the number of studies increasing with intensity of green. Countries in grey had no included studies. 14 studies were not associated with a specified region and are not included here.

The majority of studies assessed two to five ecosystem services (45%), focused mostly on regulatory & maintenance services (55%) and in the catchment (62%) (Table 2. 4). Studies mostly assessed pressure from land use (58%) with pollution as the third most common pressure (19%). Spatial modelling was the most common approach in studies with hydrological approaches as a close second.

Eight studies focused on chemical pollutants including sodium hypochlorite or bleach (Pioch et al., 2017a), 17a-ethinyl estradiol (EE2) an endocrine disruptor (Forbes et al., 2017; Forbes et al., 2019) and mercury (Harris et al., 2017; Johnston et al., 2017), as well as the cumulative stress from 32 stressors including chemical (polychlorinated biphenyls, copper, mercury) and non-chemical stressors (e.g. nutrient loading, invasive species, coastal development, climate change, and dams) (Allan et al., 2012). Five studies considered risk assessment explicitly (Harris et al., 2017; Johnston et al., 2017; Pioch et al., 2017; Forbes at al., 2019; J. Wang et al., 2021) and the approaches were applying indicators (J. Wang et al., 2021), habitat equivalency analysis (Pioch et al., 2017), probabilistic (Harris et al., 2017), individual-based modelling (Forbes et al., 2019), and integrated modelling (Johnston et al., 2017). No single tool directly linked chemical pressure to final ecosystem service provision. Studies that addressed chemical impacts on ecosystem service delivery modified or integrated an array of methods mostly involving ecological and socioeconomic approaches. These suggest promising ways forward include adapting existing ecological models (Forbes et al., 2019) or integrating models together in a stepwise assessment (Johnston et al., 2017).

Table 2.4 A summary of study findings and their characteristics. Some studies (*) applied a combination of approaches and so the total is greater than 332 studies or a subset of the studies.

Characteristic	**Number of studies**
Number of ecosystem services assessed	
1	136 (41%)
2-5	150 (45%)
6-10	23 (7%)
>10	10 (3%)
NA	13 (4%)
Ecosystem service section (CICES V5.1)	
Regulatory & Maintenance	182 (55%)
Provisioning	21 (6%)
Cultural	17 (5%)
Provisioning + Regulatory & Maintenance	43 (13%)
Cultural + Provisioning + Regulatory & Maintenance	37 (11%)
Cultural + Regulatory & Maintenance	15 (4.5%)
Cultural + Provisioning	6 (2%)
NA	11 (3%)
Habitat-type	
Lotic	46 (14%)
Lentic	22 (7%)
Catchment	205 (62%)

Characteristic (continued)	Number of studies (continued)
Wetland	15 (4.5%)
Other (regional, urban)	27 (8%)
Combined or many habitats	13 (4%)
NA	4 (1.2%)
Pressures assessed	
Land use	192 (58%)
Climate change	63 (19%)
Pollution	61 (19%)
Combination of pressures	53 (16%)
Supply and demand	45 (13.5%)
Human activity	17 (5%)
Invasive species	4 (1%)
NA	13 (4%)
Specific studies by pollution-type*	
Nutrient	36
Wastewater	11
Chemical	8
Mining	2
Microbial	2
General water quality	3
Tool type used in study*	
Spatial	111
Hydrological	101
Economic	56
Combination of tools	51
Spatial and temporal	37
Ecological	20
Other	13
Biophysical	12
Decision support	10
Index/indicator	7
Ecosystem	7
Bayesian network	6
Social media tool	5
Physical	4
Social value	3
Agent/individual-based model	3
Artificial neural network	2
Cost-benefit analysis	1
Bioeconomical	1
Total number of tools across all studies: 214 Total number of tools included in analyses: 30	

2.3.2 Tools available

A total of 214 individual models or tools were extracted from the reviewed studies and similar approaches were grouped together (e.g. GIS methods, social media methods). A total of 30 tools met the inclusion criteria, of which six were only applied in combination with other tools and therefore not included in subsequent analyses (Table 2.5).

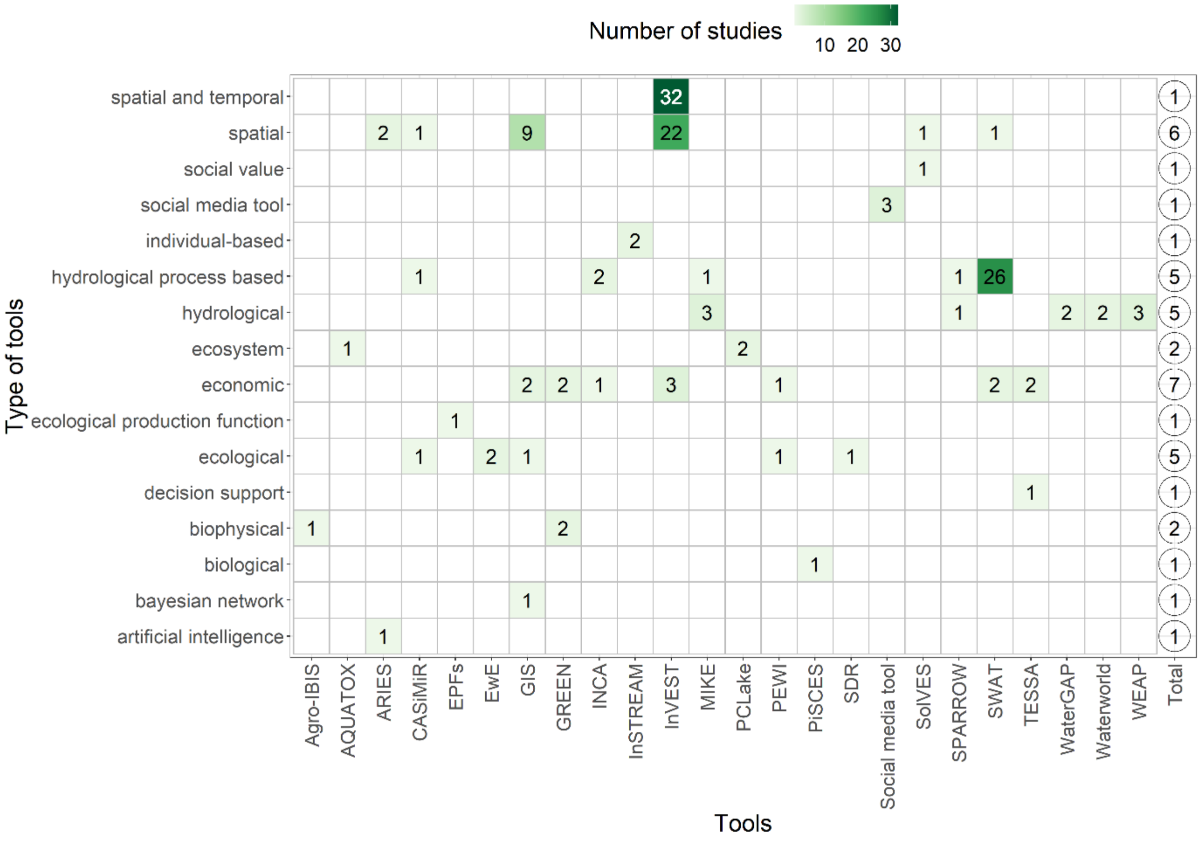
*Table 2.5 Summary of ecosystem service models and tools for freshwater ecosystem services which are still actively updated and managed. Tools which were found in more than one study have been described here. Some have been omitted as they are deprecated or no longer updated (BASS, HSPF, LUCI), not publicly available (LUONNIKAS, Estimap, ACRU) or are related to frameworks (MAES) or economic methods (CPUE or catch per unit effort, Cost-benefit analysis, ecosystem service valuation, payment for ecosystem services) or mathematical or spatial models which form the basis for tools (e.g. CASA, USLE, CA-Markov, Power law models, analysis (MCDA, MCA)). *Tools applied only in combination with other tools and therefore not included in subsequent analyses.*

Name	**Number of studies**	**Description**	**Output**	**Documentation available**	**Reference**
InVEST	76	Ecosystem service tool for spatial assessment across many ecosystem service models	Map	Yes	(Natural Capital Project, 2024)
SWAT	41	Hydrological process-based model	Quantification	Yes	(USDA Agricultural Research Service, 2024)
GIS	20	A variety of GIS toolkits	Map	Yes	GIS platforms (e.g. ArcGIS, QGIS, WebGIS)
MIKE	9	Several (e.g. MIKE SHE, MIKE 21, MIKE+) hydrological models for water management and engineering	Map, quantification	Yes	(MIKE Powered by DHI, 2022)
ARIES	6	Artificial intelligent model applying ecological process-based models	Map	Yes	(Bagstad et al., 2011; ARIES, 2024)
HEC-RAS*	6	Hydrological model for rivers	Map, quantification	Yes	(Brunner et al., 2024)
INCA	5	Various models for nutrients, sediments, and pathogens	Quantification	Yes	(Wade et al., 2002; Water Resource Associates LLP, 2024)
WaterGAP	5	Hydrological model	Quantification	No	(Schmied et al., 2023)
WEAP	5	Hydrological model	Quantification	Yes	(Stockholm Environment Institute, 2015)
CASiMiR	4	Various models for instream assessment (e.g. fish habitat, water flow)	Map, quantification	Yes	(Noack et al., 2013)
PCLake	4	Model for eutrophication in lakes and ponds	Quantification	Yes	(Janssen et al., 2019)

Name	Number of studies	Description	Output	Documentation available	Reference
SolVES	4	Decision support tool for social values	Map	Yes	(Sherrouse & Semmens, 2020)
TESSA	4	Decision support tool for ecosystem service assessment	Quantification	Yes	(Peh et al., 2013)
EPFs	4	Ecological production functions including lake water quality and Variable Infiltration Capacity Macroscale Hydrologic Model (VIC)	Quantification	Yes	Lake water quality (Rhodes et al., 2017; Moore et al., 2020) VIC (Wong et al., 2017; C. Zhang et al., 2021)
AQUATOX	3	Ecosystem model on water quality effects in aquatic life	Quantification	Yes	(U.S. Environmental Protection Agency, 2018)
EwE	3	Ecopath with Esim a food web model for marine ecosystems	Quantification	Yes	(Christensen & Walters, 2004; Ecopath International Initiative, 2022)
OVERSEER*	3	Nutrient management tool in New Zealand	Quantification	Yes	(AgResearch Ltd., 2015)
Social media tool	3	Various tools applying geotagging data from social media	Quantification	No	(Tieskens et al., 2018; Hale et al., 2019; Sinclair et al., 2019)
WASP*	3	Water quality analysis	Quantification	Yes	(U.S. Environmental Protection Agency, 2019)
WaterWorld	3	Process-based model for water quality and quantity	Quantification	Yes	(Mulligan, 2013) available on request
Agro-IBIS	2	Agricultural management model for Mid-West United States	Quantification	Yes	(Amuti et al., 2018)
CLUE*	2	Conversion of Land Use and its Effects dynamic, spatially explicit, land use and land cover change model (CLUE-S found within this review)	Map	Yes	(Verburg & Overmars, 2009)
GEOTRANSF*	2	Hydrological model hosted in FORTRAN used for water resource availability	Quantification	Yes	(Bellin et al., 2016) available on request
GREEN	2	Statistical model for nutrient impacts on the environment	Quantification	Yes	(Grizzetti et al., 2021)
PEWI	2	Game based learning tool for land uses	NA	Yes	(Chennault et al., 2020)

Name	Number of studies	Description	Output	Documentation available	Reference
PiSCES	2	Online tool for stream communities in the United States	Map	Yes	(Cyterski, 2014)
VEMALA*	2	Water quality and nutrient assessment model for catchments in Finland	Map	Yes	(Huttunen et al., 2016)
SDR	2	Species-discharge relationship model for aquatic species based on species-area relationships in terrestrial environments	Quantification	No	(Spooner et al., 2011; Xu et al., 2016)
SPARROW	2	Nutrient model for catchments	Quantification	Yes	(Schwarz et al., 2006)
inSTREAM	2	Individual-based model for trout species in stream environments	Quantification	Yes	(Railsback et al., 2022)

The 24 tools analysed varied in type, as defined in Table 2.4 (Figure 2. 4). The most common tools were InVEST and SWAT and the most common categories for type of tool were spatial and temporal, hydrological process-based, and spatial. Categories for types of tools were not exclusive. For example, InVEST was classified as spatial, spatial and temporal, and economic. Seven tools had an economic component and were capable of assessing economic valuation based on the integrated output from other types of approaches (i.e. ecological, decision support, Bayesian, hydrological process-based, individual-based, spatial, and spatial and temporal). For example, InVEST is a tool capable of mapping service provision and presenting results in economic terms (Natural Capital Project, 2024). In addition, other tools produced outputs that can be valued including five hydrological tools (hydrologic cycle) and five hydrological process-based tools (prediction based on physical processes) that assessed water quality and quantity within the catchment or water body. The outputs from these tools require post-processing to assess ecosystem service delivery. Some tools were the only one of their type for artificial intelligence (ARIES), biological (PiSCES), social value (SolVES), and individual-based model (inSTREAM).



	Agro-IBIS	AQUATOX	ARIES	CASiMiR	EPFs	EwE	GIS	GREEN	INCA	InSTREAM	InVEST	MIKE	PCLake	PEWI	PiSCES	SDR	Social media tool	SolVES	SPARROW	SWAT	TESSA	WaterGAP	Waterworld	WEAP	Total
spatial and temporal											32														1
spatial			2	1			9				22							1		1					6
social value																		1							1
social media tool																	3								1
individual-based										2															1
hydrological process based				1					2			1							1	26					5
hydrological												3							1			2	2	3	5
ecosystem		1											2												2
economic							2	2	1		3			1						2	2				7
ecological production function					1																				1
ecological				1		2	1							1		1									5
decision support																					1				1
biophysical	1							2																	2
biological															1										1
bayesian network							1																		1
artificial intelligence			1																						1

Figure 2.4 Heatmap of tools and type of tools across the 24 tools found in more than one study from the review. The number of studies where each tool was classified or applied as each type of tool are in green with increasing numbers leading to greater intensity of colour.

2.3.3 Freshwater ecosystem services assessed by tools

The final selection of 24 tools were analysed for their assessment of freshwater ecosystem service as classified by CICES (version 5.1) (Haines-Young & Potschin, 2018) (Figure 2.5). Collectively the tools could be used to assess 26 of the 52 ecosystem services classes across provisioning (3 classes), regulatory & maintenance (8 classes), cultural (11 classes), and abiotic services (4 classes). Freshwater ecosystem services not assessed by these tools are listed in Appendix Table A2.2.

Most tools (21) could assess regulatory & maintenance services, twelve tools could assess cultural services, ten could assess biotic provisioning services, and eight could assess abiotic services. The ecosystem services assessed by the most tools were 'regulation of chemical condition of freshwater' (13), 'hydrological cycle and water flow regulation' (12), and 'active or immersive interactions' (12). Eight tools could assess 'maintaining nursery populations and habitats', seven tools could assess 'surface water for drinking purposes' and 'control of erosion rates', and six tools could assess 'wild animals for nutrition' and 'reared aquatic animals for nutrition'.

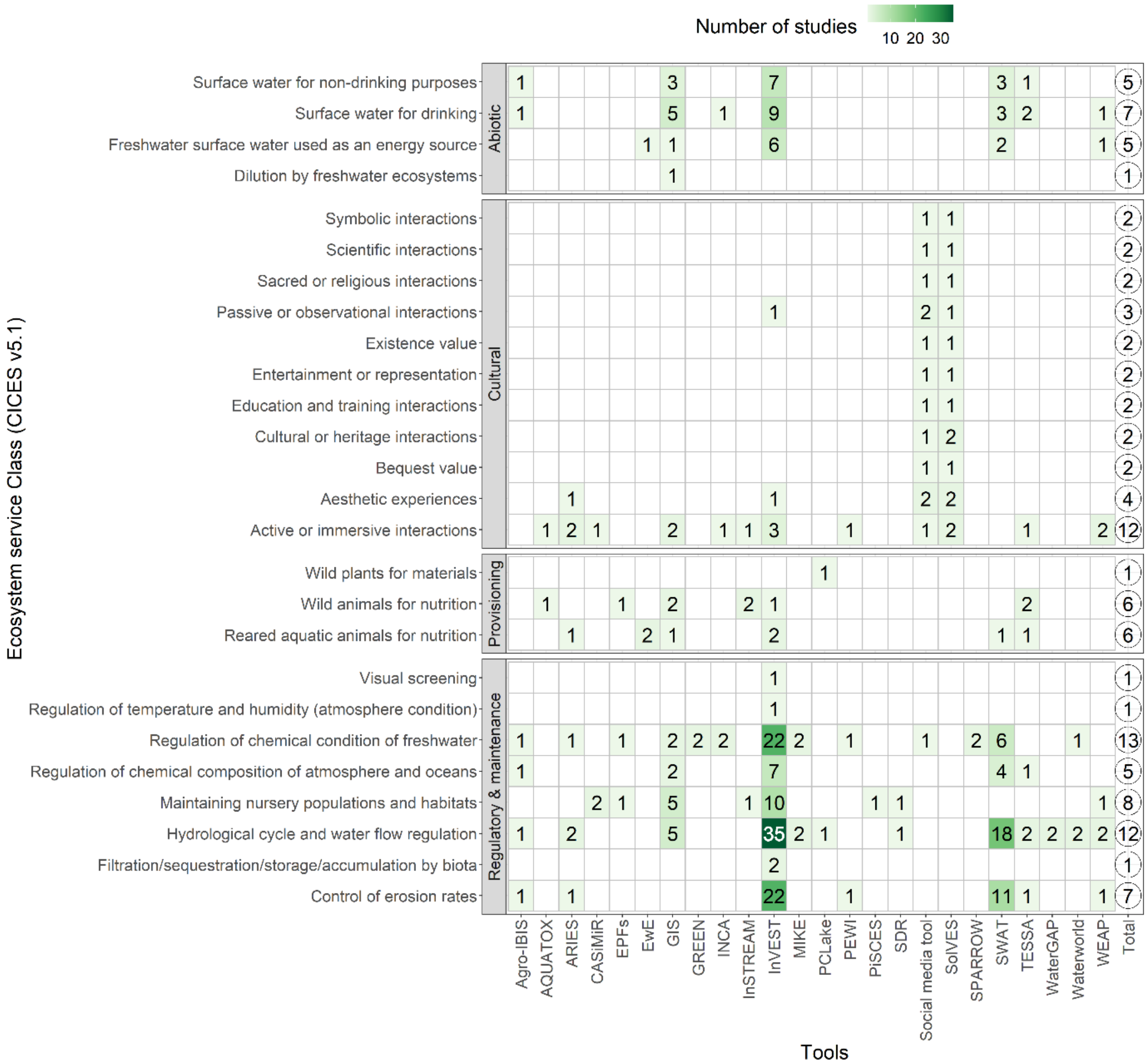


Figure 2.5 Freshwater ecosystem services (CICES V5.1) class assessed by tools available in more than one study where the tool was applied independently. Numbers denote the number of studies where a tool assessed the service and increasing numbers are shown in greater intensity of green.

Twenty tools were capable of assessing a suite of services and InVEST could assess 16 of the 26 ecosystem services included in Figure 2.5. Some combinations of assessed ecosystem services were more common than others. For example, seven tools assessed both 'regulation of chemical condition of freshwater' and 'hydrological cycle and water flow regulation', six tools assessed 'hydrological cycle and water flow regulation' and 'control of erosion rates', and six tools assessed 'hydrological cycle and water flow regulation' and 'surface water for drinking'.

2.3.4 Spatial scale of tools

The 24 selected tools have been used to assess ecosystem services across different habitats and spatial scales, from a single river to an entire region (Figure 2.6). The majority of tools (13) (Figure 2.6) and studies (Table 2.4) focused on assessing ecosystem services at the catchment and incorporated components of both freshwater and terrestrial ecosystems. Flowing surface water or lotic ecosystems had the second most tools (9) followed by other (e.g. urban, wetlands, and regions) (8), and standing surface water or lotic (7). InVEST was the only tool that has been used to assess ecosystem services at spatial scales from an individual water body to a catchment and across different habitat types (lentic and lotic). The spatial scales were also analysed in more detail in Appendix Figure A2.2.

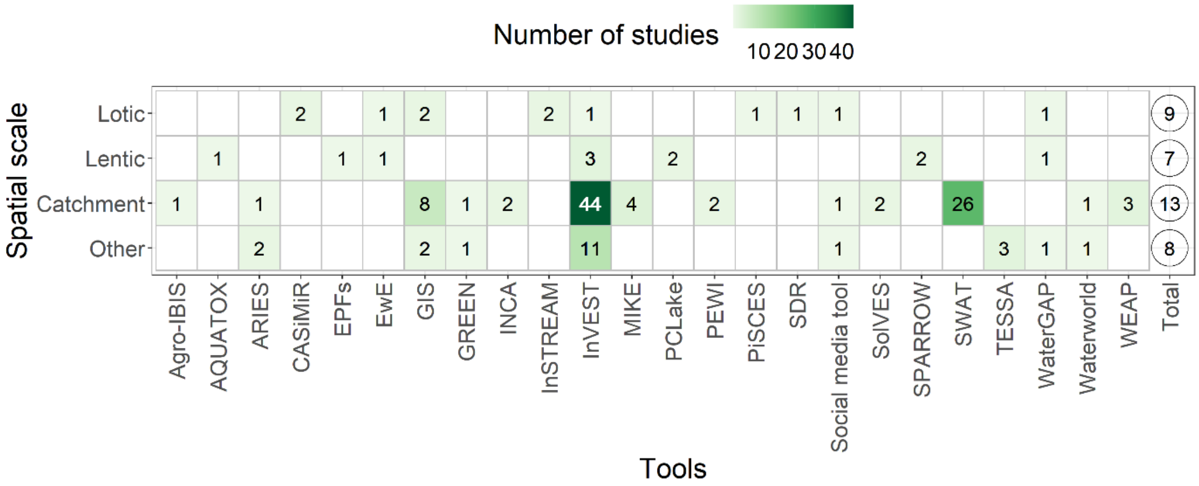
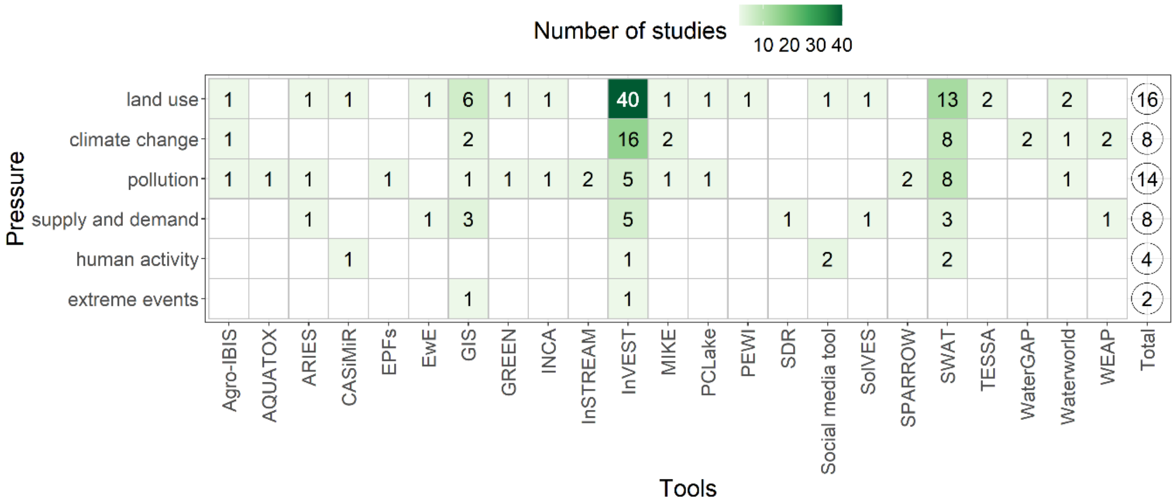


Figure 2.6. Spatial scale assessed by twenty-four tools which fulfilled inclusion criteria. Spatial scales are categorised as lotic or flowing surface waters, lentic or standing surface waters, catchment, and other (including wetlands, urban ecosystems and regions such as nations). Green signifies the number of studies where the tool was applied to assess the spatial scale, where increasing number is shown in greater intensity of green.

2.3.5 Pressures and chemical stressors assessed by tools

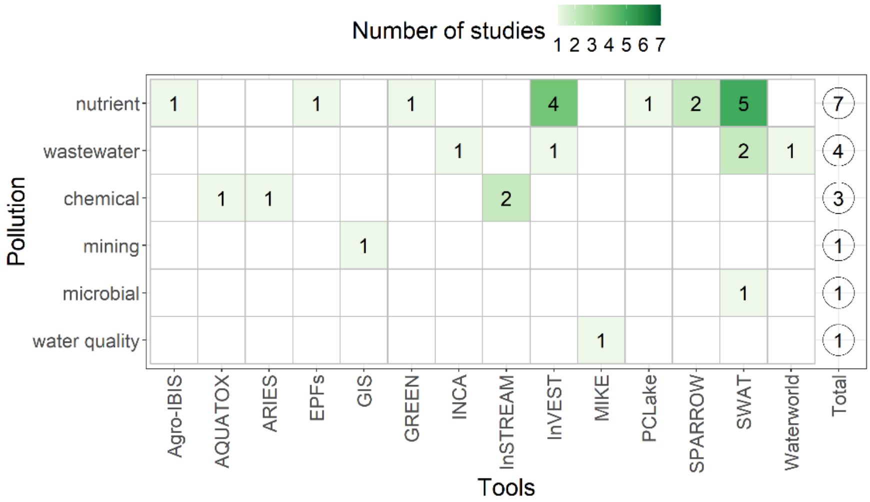
The twenty four selected tools have been used to assess the effect of climate change (8), land use/land cover change (16), pollution (14), human activity (e.g. including tourism and recreational activities) (4), supply and demand (i.e. availability and use of the service including trade-offs, mismatches, exploitation e.g. water usage) (8), and extreme events (e.g. landslides) (2) (Figure 2.7). InVEST was the only tool capable of addressing all the pressures assessed, with SWAT and GIS toolkits, as the next most broad tools.



	Agro-IBIS	AQUATOX	ARIES	CASiMiR	EPFs	EwE	GIS	GREEN	INCA	InSTREAM	InVEST	MIKE	PCLake	PEWI	SDR	Social media tool	SolVES	SPARROW	SWAT	TESSA	WaterGAP	Waterworld	WEAP	Total
land use	1		1	1		1	6	1	1		40	1	1	1		1	1		13	2		2		16
climate change	1						2				16	2							8		2	1	2	8
pollution	1	1	1		1		1	1	1	2	5	1	1					2	8			1		14
supply and demand			1			1	3				5				1		1		3				1	8
human activity				1							1					2			2					4
extreme events							1				1													2

Figure 2.7 Pressures assessed by tools within more than one study. Results are only analysed for those which are used independently. Green indicates the number of review studies where this connection appeared, where increasing number is shown in greater intensity of green.

Of the 14 tools used to assess pollution, seven assessed nutrient loading, four assessed wastewater, three (AQUATOX, inSTREAM, ARIES) assessed chemicals (Figure 2.8). AQUATOX is a food web model developed specifically for assessing the ecological risk of toxic chemicals (Park et al., 2008), inSTREAM is an individual-based model for trout species in stream environments (Railsback et al., 2022), and ARIES is an artificially intelligent integrated modelling platform (Bagstad et al., 2011). Whereas none of these models directly assess the impact of chemical stressors on ecosystem service delivery the studies show they have the potential to be adapted for this purpose.



	Agro-IBIS	AQUATOX	ARIES	EPFs	GIS	GREEN	INCA	InSTREAM	InVEST	MIKE	PCLake	SPARROW	SWAT	Waterworld	Total
nutrient	1			1		1			4		1	2	5		7
wastewater							1		1				2	1	4
chemical		1	1					2							3
mining					1										1
microbial													1		1
water quality										1					1

Figure 2.8 Specific tools which focused on pollution by the type of pollution including chemical, nutrient, wastewater, pesticide, chemical, mining, microbial, and general water quality (where the specific pollutant was unspecified in the study). Number of studies assessing pollution-type are in green, where increasing number is shown in greater intensity of green.

2.4 Discussion

A rapid evidence review identified 214 tools used for assessing freshwater ecosystem services, which is in line with a recent review by Nedkov et al. (2022) that identified over 200 models and approaches for assessing water-related ecosystem services. Twenty four tools met the criteria for inclusion as applied in more than one study and independently within the available studies. These tools were analysed to determine which freshwater ecosystem services have been assessed, the spatial scale of the assessment, and the environmental pressures they have assessed. Three tools capable of assessing chemical stressors to freshwater ecosystem services at varying spatial scales were identified; ARIES, AQUATOX, and inSTREAM.

2.4.1 Available tools to assess freshwater ecosystem services

The review studies assessed mostly regulatory & maintenance, then provisioning, and then cultural freshwater ecosystem services. This is a similar finding found from other reviews across freshwater ecosystem services catchments (Nabout et al., 2023), quantification of river ecosystem services (Hanna et al., 2018), or for mapping freshwaters in general (Martínez-Harms & Balvanera, 2012). However, previous reviews have focused on identifying tools and the services and summarising ecosystem services occurrence across studies but not highlighting the capabilities of tools across the studies, only focusing on one form of assessment (e.g. mapping), or summarising tools application for selection (e.g. training and data requirements) (Trabucchi et al., 2012; Ochoa & Urbina-Cardona, 2017; Hallouin et al., 2018; Deeksha & Shukla, 2022; Meraj et al., 2022; Nabout et al., 2023). As this review addresses the tools and not studies, it is clear that what is most frequently addressed in studies does not necessarily reflect the capability of the tools applied. Previous reviews identified cultural services as the least assessed ecosystem services across studies as cultural services are usually considered less tangible and difficult to quantify (Hale et al., 2019; Deeksha & Shukla, 2022). In contrast to previous reviews, the current review identified cultural services as the second most assessed ecosystem services by tools available. However, most of the cultural services had been assessed by less than four tools, with the exception of ‘active or immersive interactions’ with the second most tools across all freshwater ecosystem services. This explicit water-human interaction is present for some cultural services, such as ‘active and immersive interactions’, and can be quantified in assessments compared to less tangible cultural ecosystem services which do not have these interactions (Apostolaki et al., 2019).

This review identified two main types of tools for freshwater ecosystems service assessment; generic tools capable of modelling across a suite of ecosystem services and specialised tools which are more focused and capable of modelling a few services. On the one hand, generic tools include ecosystem service tools which have been simplified missing certain levels of complexity (Rieb et al., 2017). For example, InVEST is a generic ecosystem service tool made up of suite of models which can map and value ecosystem services. Within this review, InVEST was the most applied tool across studies capable of assessing regulatory & maintenance services. This aligns with a previous study which analysed InVEST application logs and found use most often associated with assessing regulatory & maintenance services (Posner et al., 2016). But InVEST it is not the only generic tool found in this review. ARIES, MIKE, SWAT, and INCA are all capable of assessing an array of freshwater ecosystem services, however, unlike InVEST they are not all ecosystem service tools. For example, SWAT is a regional catchment tool, with a suite of models that can address catchment management questions related to water and soil systems, albeit not originally developed as an ecosystem service tool (Arnold et al., 2012; USDA Agricultural Research Service, 2024). SWAT, as a hydrological tool, addresses freshwater ecosystem services related to the water cycle such as 'hydrological cycle and water flow regulation' and 'surface water for drinking/non-drinking purposes' both found within this review and others (Nedkov et al., 2022). However, it can also model services which rely on water transport and flow such as 'control of erosion rates' and 'regulation of chemical condition of water'. Tools such as InVEST and SWAT, are process-based tools and as they can assess multiple services, they are capable of addressing trade-offs and dynamics between services and the socioecological system, for example where the supply of one service might reduce the supply of another (Haase et al., 2014; Wolff et al., 2015). However, as some generic tools are not ecosystem service tools, such as SWAT, they would still require post-processing as their original development was not for an ecosystem services assessment (Vigerstol & Aukema, 2011; Lüke & Hack, 2018)

On the other hand, some tools are more specialised applying methods such as hydrological and ecological tools being more bespoke (Hallouin et al., 2018). For example, AQUATOX (Park et al., 2008), an ecosystem model for closed systems (e.g. lakes), simulates the fate of pollutants and was only capable of simulating the impact on 'wild animals for nutrition' and the influence this has on 'active or immersive interactions' (e.g. recreation such as fishing or swimming) and 'aesthetic experiences' (Galic et al., 2019). Tools with specificity are not originally intended for ecosystem service assessment so, therefore, have specialised outputs but can be useful to, for example, quantify ecosystem processes and structures (Harrison et al., 2018). However, specialised tools require post-processing to quantify ecosystem service delivery and ensure they are useful for ecosystem service assessment. Furthermore, as these tools do not model a range of services, they are not capable of determining interactions between ecosystem services such as trade-offs.

This sets up another dichotomy for modelling freshwater ecosystem services as either using hydrological tools or not (Vigerstol & Aukema, 2011). Hydrological tools are those which were not originally intended to be applied in ecosystem services assessment, such as the hydrological models SWAT and MIKE, and require detailed data input, modelling expertise, and post-processing of outputs to ecosystem services assessment (Vigerstol & Aukema, 2011). A majority of the tools in this review were hydrological or hydrological process-based. Hydrological tools mainly addressed regulatory & maintenance services associated with the water cycle or hydrological ecosystem services such as 'hydrological cycle and water flow regulation' seen both in this review and previous reviews (Nedkov et al., 2022). Hydrological and biogeochemical tools are widely accepted as appropriate methods of assessment of water-related (or hydrological) ecosystem services in catchments (Grizzetti et al., 2016). This is logical as earlier ecosystem service assessments focused on ecosystem service provision by assigning one value to one habitat type. In the case of freshwater ecosystem services this was based on their underlying hydrological processes (Rieb et al., 2017). Common combinations of freshwater ecosystem services assessed included 'regulation of chemical condition' with 'hydrological cycle and water flow regulation', 'hydrological cycle and water flow regulation' with 'control of erosion rates', and 'hydrological cycle and water flow regulation' with 'surface water for drinking', services all related to the underlying hydrological processes. This indicates that the modelling mechanisms behind these assessments are similar. For example, water quality can be assessed from water flow models such as VEMALA and SPARROW to determine influx of e.g. nutrients from the wider catchment (Milstead et al., 2013; Huttunen et al., 2016). Tools which assessed water supply from water yield or flow quantification would then either value or post-process the results from hydrologic or hydrological process-based models to identify the effect on the water provisioning service. In the case of VEMALA and SPARROW (Milstead et al., 2013; Huttunen et al., 2016), the resulting output is the nutrient loading or concentration which still needs to be translated to ecosystem services such as aesthetics or water quality.

Only direct valuation methods considering 'surface water for drinking' were found, otherwise these water-related studies, which were the majority, assessed the underlying hydrological process. This indicates that freshwater ecosystem service assessment is biased towards measuring the underlying hydrological processes or in other words the ecosystem structures and processes and not considering the impact to beneficiaries beyond monetary valuation (Ochoa & Urbina-Cardona, 2017). For example, WaterWorld produces hydrological outputs which can be equated to economic values (Mulligan, 2013; Bullock & Ding, 2018). However, some ecosystem service tools have more recently expanded to include hydrological models (e.g. InVEST) and would, therefore, not require post-processing but instead require a level of pre-processing of data (Lüke & Hack, 2018).

Fish are considered the most investigated group of organisms in freshwater ecosystem services research as they provide provisioning as food, regulatory & maintenance in nutrient regulation, and cultural services such as recreational fishing (Nabout et al., 2023). From this review 'wild animals for nutrition', 'reared aquatic animals for nutrition' and 'maintaining nursery populations and habitats' often considered fish as the focal aquatic organism from freshwaters with several models focused on fish population or communities (PiSCES and inSTREAM), the food web (AQUATOX and EwE), or fisheries stock for food or their habitat availability (ARIES and InVEST). Furthermore some 'active or immersive interactions' involved recreation such as recreational fishing. Considering that all regulatory & maintenance other than 'maintaining nursery populations and habitat' and abiotic services consider the hydrological cycle, aquatic organisms made up most of the tools available for provisioning services and fish can be considered an important organism in freshwater ecosystems

2.4.2 Spatial scales of freshwater ecosystem service tools

The spatial scale for assessing freshwater ecosystem services depends on the capability of the tools as they operate at particular spatial scales. Most tools were seen to assess the catchment-level scale. This aligns with the findings from other reviews, although the focus was on studies across freshwater ecosystem services (Hallouin et al., 2018; Hanna et al., 2018; Nabout et al., 2023). As catchment was the most assessed spatial scale, this highlights that tools are more likely to consider both the aquatic and terrestrial elements to assess ecosystem service delivery. Literature has found that there is a strong bias towards terrestrial ecosystems in the methods and models for ecosystem service assessment (IPBES, 2016). As tools found in this review were dominated by the catchment-scale this indicates that this terrestrial bias is still apparent in freshwater services assessment. The second most assessed spatial scale was lotic ecosystems. This was also reflected with the findings from other reviews (Hallouin et al., 2018; Hanna et al., 2018; Nabout et al., 2023). One in particular, Hanna et al. (2018), reviewed the spatial scale of rivers found across studies, and found that the most found spatial scale was multiple river reaches where studies did not usually consider the whole river or the one reach (i.e. habitat) scale. However, this current review did not apply the same level of detail for describing the lotic ecosystem (with the exception of some classifications of the river network in Appendix Figure A2.2). Understanding which scale is appropriate or useful to assess particular ecosystem services is important especially as in ecosystem service assessment, spatial scales are often predetermined (Paetzold et al., 2010; Wu et al., 2022). Tools can then be selected based on the framing of spatial scales.

Tools were generally not able to model across spatial scales. Only three tools were found which were able to assess both the catchment and lotic scale: InVEST, social media tools, and GIS tools. InVEST was the only tool capable of assessing across all spatial scales including the lentic, lotic, and catchment-scales. Given that GIS tools and social media were a conglomeration of similar methods, this shows that tools might not be directly transferable between the different spatial scales. For example, spatial tools applying mapping assessments are not capable of assessing small-scale local habitats such as rivers and lakes where biophysical assessment is more important for ecosystem service delivery (Urbina-Cardona et al., 2023). This separation between scales was also reflected in the type of tool. Some tools, which were not originally intended for ecosystem service assessment, usually were found to assess local scale (lentic and lotic) ecosystems. For example, one of the tools found in this review, AQUATOX was applied on the local scale for a lake exposed to chemical stress (Forbes et al., 2017). However, further research into AQUATOX shows that the tool is capable of modelling six types of sites including ponds, lakes, rivers, reservoirs, enclosures, and estuaries (Park et al., 2008; U.S. Environmental Protection Agency, 2018), but examples of these were not found within this review. Furthermore, another tool found in this review, inSTREAM is spatially explicit for rivers (Railsback et al., 2022), but has the capability to assess several spatial scales from the reach-scale to the entire river network.

In contrast, tools found in this review which are originally intended for ecosystem service assessment were usually on the catchment-scale such as InVEST, ARIES, SolVES, and TESSA. This highlights that the spatial scale of tools are usually dependent on their intended use (Vigerstol & Aukema, 2011). For example, at the ecological scale, habitat-level, local, reach, and small scales are usually the scale of interest (Kuemmerlen et al., 2019). Whilst the smallest scale for ecosystem services are usually on the local scale in relation to the beneficiaries (human well-being) and not ecosystem service delivery (e.g. SPU) (Tardieu, 2017). However, applications beyond the intended scale of the tools were also seen. For example, SPARROW is originally a catchment-scale tool but was found in this review assessing a lake (Milstead et al., 2013) and pond (W. Zhang et al., 2019) ecosystems. This is dependent on what is of interest and if the catchment has an influence on other spatial scales, as is the case with such ecosystem services related to the water cycle (Lüke & Hack, 2018). Some tools were found to be confined to a particular location as opposed to a spatial scale such as PiSCES (USA), VEMALA (Finland), OVERSEER (New Zealand), Agro-IBIS (Mid-West USA), but it is unclear if they have the potential to be applied elsewhere with the same methods. There is a clear gap for tools which are capable of addressing multiple spatial scales, which has been identified as a need for develop better methods capable of quantifying and managing ecosystem services across spatial scales (Costanza et al., 2017).

2.4.3 Tools available to assess the risk of chemicals to ecosystem service delivery in freshwater catchments

The majority of tools within this review focused mostly on land use, second as pollution, and third as climate change. In literature, ecosystem service assessment methods are biased towards assessing climate change as a driver with significant gaps and lack of attention paid to pollution (IPBES, 2016; Kosnik et al., 2024). However, in ecological risk assessment, a previous knowledge mapping identified methods mostly assessed land use and climate change as pressures (Qian et al., 2023). As land use was the frontrunner in this current review, this highlights that there was a similarity of tools found more with ecological risk assessment methods as opposed to ecosystem service methods. InVEST was the most widely applied tool across pressures, aligning with the findings from another review (Deeksha & Shukla, 2022), and is capable of assessing the impacts of climate change and land use. However, at the time of writing, when considering pollution, InVEST only has the capability to model nutrient loading, not chemical pollutants (Natural Capital Project, 2024). InVEST applies hydrological modelling to calculate run-off and estimated nutrient loading to surface waters from, for example, agricultural practices (Cong et al., 2020). Another tool found in this review, SWAT, also applied the same method to model pesticide transport in the catchment and subsequent loading to freshwaters (Arnold et al., 2012). However, this approach cannot determine the subsequent effects on biological organisms in surface waters, such as on the SPUs, and are not useful for chemical effects assessment. These approaches could instead be used as an exposure assessment tool for chemicals, although an example of this was not found in this current review.

Chemical stressors was considered in 2.4% of the review studies. This aligns with a previous review which identified risk assessment as the least studied theme for ecosystem service assessment found in less than 2.2% of studies (Urbina-Cardona et al., 2023). All of the studies which assessed chemical pollution introduced an approach and were, therefore, assumed the first of their type, indicating that literature and tool application is limited and still being developed. However, from the perspective of the tools, three out of the 24 tools analysed considered chemical stress (12.5%). Although there is still a clear application gap for tools to assess the risk of chemicals to freshwater ecosystem services, it was more than would have been expected had the focus been on studies alone.

This current review has highlighted the availability of tools which can address chemical pollution from an ecosystem service perspective. Examining if there are overlaps with ecological risk assessment methods can better help to determine the suitability of tools identified in this review for risk assessment. Previous reviews have examined the application of models for chemicals pollution and contamination in ecological risk assessment (Roast et al., 2007; Chen et al., 2013). The main groups of models in ecological risk assessment are population, metapopulation, ecosystem, landscape, and toxicity models (Roast et al., 2007). These include mechanistic, individual-based, and food-web models to assess these varying levels of organisation. Approaches for implementing an ecosystem service assessment to ecological risk assessment recommend the use of mechanistic models or ecological models, especially to translate from chemical effect data to more relevant biological levels for SPU assessment (Galic et al., 2010; Forbes & Calow, 2012; Grimm & Martin, 2013; Martin et al., 2014; Forbes et al., 2019) (Chapter 1). Although when considering freshwater ecosystem services tools in this review, hydrological and spatial tools were the most common, but there were examples of ecological, ecosystem, and individual-based tools, suggesting these methods are applicable across both ecosystem service and ecological risk assessment. Two tools found in this review of this type, inSTREAM and AQUATOX, were used to assess chemical pressure both by adapting to include chemical effects (inSTREAM) or applying a model developed for risk assessment (AQUATOX) (Forbes et al., 2017; Forbes et al., 2019). It is clear that there might be overlaps between the types of approaches and methods suggested in ecological risk assessment and ecosystem service assessment. Integrating ecosystem and socioecological tools into risk assessment has been suggested as a method to consider the wider systems (Chen et al., 2013), therefore, integrating ecosystem service-related tools which consider both the ecosystem as well as socioecology can help to achieve this.

One method which has been suggested for implementing ecosystems service assessment in risk assessment is using ecological production functions (EPFs) to estimate service provision from characteristics within the site (Andrew et al., 2015). EPFs have previously been screened for their applicability in risk assessment but none were found capable of addressing from SPU to ecosystem service valuation (Faber et al., 2021). From the EPF database (Faber et al., 2021), 22 EPFs for eight of the CICES (V5.1) classes of freshwater ecosystem services were identified (Appendix Table A2.3) with applications in chemical risk assessment and ecosystem service delivery. The two EPFs found in this current review for lake water quality and VIC were not found in this database. However, they are also not capable of addressing ecosystem service valuation alone and were coupled or applied in integrated methods to translate to ecosystem service valuation. Some of the tools found use the Revised Universal Soil Loss Equation (RUSLE) to estimate erosion control which is itself considered an EPF (Andrew et al., 2015) and incorporated into tools, such as InVEST (Estrada-Carmona et al., 2017). RUSLE was not found in the EPF database but also requires post-processing if applied alone as opposed to within InVEST which will then translate outputs to ecosystem service valuation. Although few EPFs were found in this review, they might be incorporated into tools so focusing on tools readily available enable applications of established EPFs coupled with other methods to assess ecosystem services.

2.4.4 Strengths and limitations

Most tools were found from studies published post-2004. This aligned with the history of the "ecosystem services" concept which was first introduced in 1981 (Ehrlich & Ehrlich, 1981) but first formally classified in 2005 (Millennium Ecosystem Assessment, 2005). Similar concepts have been used and still exist such as "nature's services'' (Costanza et al., 2017) and "ecological services" (Fisher et al., 2009) and terminology is still changing with the introduction of newer concepts such as "nature's contributions to people" (NCP) (Pascual et al., 2017). By including "ecosystem service" as one of the categories for the literature search string, these other terms might not have been picked up with this review. However, freshwater assessments existed before the introduction of the concept of "ecosystem services" which investigated ecosystem functions and freshwater health (Norris & Thoms, 1999). Furthermore, newer tools and models are built from the concepts of past models and in some cases have the capabilities to address several ecosystem services provisions at once (Cong et al., 2020). So, although the review only found literature post-2004, this search was unlikely to miss key models as later models tend to incorporate the concepts of earlier ones.

As the ecosystem services concept in risk assessment is of interest, it is important to know whether there are tools currently applied in risk assessment in known databases. To address the utility of methods found in this review, an additional database was screened, the United States Environmental Protection Agency (EPA) Science Models and Research Tools (SMaRT) search (U.S. Environmental Protection Agency, 2024). SMaRT is a collection of US EPA developed tools and was screened (accessed 28.05.2024) by searching for 'ecosystem services' and selecting Environmental Topic Areas as 'water'. As the database is for tools the third inclusion criteria was assumed satisfied. This database search returned 15 results from the 151 entries (10%). (Appendix Table A2.4). The only tool found both in this review and the database search is AQUATOX. The database also includes the tool PiSCES (Cyterski, 2014), which was also found within this review, however, it was not considered for ecosystem services in the database and did not show up with the search. This implies that the database formulated within this review is unique and fit for the purpose for the research in this thesis. This highlights that linking research to practice is important but that practice is not always reflective of research. Therefore, reviews are needed to determine the current status of tools as databases or other reviews might not highlight exact tools suitable for research needs.

The original search for this review was conducted in May 2021 and therefore, an afterword has been added to discuss any updates or changes to the literature. An updated search with the same string was done in both SCOPUS and Web of Science (accessed 15.04.2024). The search string returned 2455 studies in SCOPUS and 2554 in Web of Science totalling 5009 additional studies since the original search. Using Rayyan.ai to screen the additional literature, 3333 duplicates were found between the two databases and resolved in Rayyan.ai for above 75% similarity. This resulted in 33% of the studies removed as duplicates (1666 articles). This ended with a total of 3343 new articles were found since the original search was conducted. To investigate if the literature has significantly changed since the original screening of this study, 10% of new literature (334 articles) was randomly sub-sampled using Rayyan.ai's create sample function. This sample was screened with the initial screening process of title, abstract, and keywords. The results of this screening are in Table 2.6.

Table 2.6 Results of the sampled screening of updated literature.

10% Sample	Articles	Percentage
Included	53	~16%
Excluded	281	~84%

The afterword screening resulted in 16% of sampled literature included for the next stage (full-text) of screening (Table 2.6). The original review search had 15% remaining after initial screening (Figure 2.1) and therefore, from the new search, a similar number (6.7%) could be estimated as possibly included from new literature (~223 new studies). It is clear that the increasing trend in studies (Figure 2.2) has continued since the original search literature. The tools or type of tool found in the abstracts of studies in the afterword are summarised in Table 2.7. Not enough information in the abstracts was found to specify the tool in 31 studies. Tools which were already familiar to this review were InVEST, GIS, WaterGAP, and SWAT. New approaches found were CADDIES, INSEAT, SCIMAP, INEGI, PlaceMarker Survey, River Park Assessment, and Geographical Detector model, however, these were only found once and would need to be found in other studies to be included in the assessment carried out within this chapter.

Table 2.7 A summary of the tools from afterword screening of abstracts. Tools which been applied with other tools are denoted (*).

Tool	Number of studies
InVEST	10
GIS	3
WaterGAP*	1
SWAT	1
CADDIES	1
INSEAT	1
SCIMAP	1
INEGI	1
PlaceMarker Survey	1
RIVER PARK ASSESSMENT	1
Geographical Detector model	1
Not enough information found in abstract	31
Total	53

2.4.5 Implications for risk assessment

Reviews are an effective way to compile information on tools capable of addressing the problem of assessing chemical pressures on ecosystem services. Focusing on tools that have been documented in scientific literature also highlights tools which are applied as a method of ecosystem service assessment with supporting documentation that is available for new users (Ochoa & Urbina-Cardona, 2017). Model documentation is especially important for regulatory ecological risk assessment (European Food Safety Authority, 2014). However, a database of tools still only presents possible options and guidance is needed to select the most appropriate method for assessing chemical risk to ecosystem services. Decision trees can aid in selecting between options. Harrison et al. (2018) developed a decision-tree for ecosystem service assessment based on biophysical methods, appropriate for ecological risk assessment, and highlights tools that are suitable for specific purposes: i) ecosystem processes and functions can be assessed with ecological and hydrological models; ii) ecosystem services with ecosystem service tools; iii) interactions between nature and society can be assess with either agent-based models (otherwise known as individual-based models) or integrated modelling methods. These approaches all reflect the types of tools identified within this current review. Therefore, Figure 2.4 – 2.8 can be used to select a method based on the framing of spatial scale, ecosystem service, and pressure for ecological risk assessment. However, applying ecosystem service tools individually may be insufficient to meet the needs of assessment and often combining tools will be the best approach (Dunford et al., 2018). Evidence-based logic chains can help identify the plausible links between chemical exposure, species-level effects, and ecosystem service(s) impacts (Maltby et al., 2017, 2021) to, therefore, help to identify which modelling approaches to use for each step of the logic chain.

2.5 Conclusion

This review aimed to answer what ecosystem services tools and methods are currently available for freshwater catchments. This was done by addressing the freshwater ecosystem services which can be assessed and found that most freshwater ecosystem services were assessed. The review also addressed the spatial scale of tools found, where most were related to the catchment-scale with some tools capable of local habitat-scale assessments in lentic and lotic ecosystems. In terms of tools which could assess the effects of chemical stressors on ecosystem service delivery, only three tools were found in more than one study capable of linking chemical stressors to ecosystem service delivery usually with integrated modelling or post-processing the underlying ecological processes, mostly serving as translation links. When it comes to the availability of ecosystem service tools and methods and their usefulness in applying an ecosystem services concept in ecological risk assessment it is clear that methods are limited. Ecosystem service tools in particular are not developed to assess chemical pollution and require adaptation to be fit for purpose whilst other methods found require post-processing. There is clearly an interest in assessing ecosystem service delivery with respect to pollution, however, no universal method was found from literature. Future research is needed to identify ways to adapt tools and combine tools to be fit for purpose and guidance to ensure that adaptations are acceptable for ecological risk assessment.

Chapter 3

Method development: Using ecological models to determine the impact of toxic chemicals on a recreational fishery across spatial scales

3.1 Introduction

Ecological modelling is a method of representing and exploring a system of interest under different scenarios (Galic et al., 2018) and provides a means of addressing ecological complexity. Ecological modelling has been suggested as a solution to overcome current limitations of risk assessment and reduce animal testing (Thorbek et al., 2009; Galic et al., 2012). Ecological models can address many levels of biological organisation from the individual to the ecosystem and landscape (Galic et al., 2012) and have increasing applications in ecosystem management (Frank et al., 2011). Their capacity to model different levels of biological organisation means that, in the context of ecosystem services, they can focus on a variety of possible service providing units (SPUs) whether individual, populations, or metapopulations. In this way, ecological models also help to develop ecological production functions (EPFs) (Tallis & Polasky, 2009). Individuals can be modelled in detail with individual-based models and populations can be modelled with differential equation models (Roast et al., 2007; Cornell et al., 2019). More complex assessments of communities or ecosystem interactions can be modelled with food web or ecosystem models (Forbes et al., 2017).

To determine the impact of toxic chemicals on a freshwater recreational fishery across spatial scales, both the population at the reach-scale and interacting populations at the river network-scale need to be modelled. This means that the model needs to be both spatially explicit and capable of modelling varying elements, such as the population, interacting populations, and ecosystem interactions. All model types can theoretically be spatially explicit (Galic et al., 2012) but this is not always the case (Wilensky, 1999; Pal et al., 2020) as models are often restricted by computing power (Galic et al., 2012) and host platforms (Pal et al., 2020). Recent advancements to computational power have now made spatial scaling more possible and host platforms such as NetLogo can support spatial assessments (Wilensky, 1999; Pal et al., 2020).

Ecological risk assessment relies on standardised toxicity testing at lower levels of biological organisation and often considers the direct effects of chemicals on life history parameters such as growth rate, reproduction, and survival in individuals (Rudén et al., 2017). Individuals can also undergo combined stress indirectly through trophic interactions such as chemical toxicity to prey and subsequent reduced food availability (Vaugeois et al., 2020; Accolla et al., 2022). A model, therefore, needs to be able to extrapolate chemical effects at the individual-level to population-level consequences (e.g. population abundance, growth rate, or biomass, and habitat quality and distribution (Frank et al., 2011) and produce outputs that are relevant to address ecosystem service delivery to the anglers (e.g. likelihood of catching large fish).

Individual-based models are increasingly applied to investigate ecological questions (Dominguez Almela et al., 2020) and can extrapolate effects at the individual-level to the population (Forbes et al., 2008), metapopulation, community, or ecosystem. Individual-based models have previously been used in risk assessment (Mintram et al., 2018b; Seiler & Brinkmann, 2022) as they provide a tool to extrapolate from known individual-level effects from standard toxicity tests on test organisms to population-level effects within the ecosystem and are more flexible to adaptations than other models (Accolla et al., 2021). In this way, not only are the individuals considered but so are their interactions with one another (e.g. competition) and with their environment (e.g. river flow). Previous individual-based models assessing chemical impacts to fish populations are readily calibrated to chemical impacts such as those for zebrafish (Hazlerigg et al., 2014) and three-spined stickleback (Mintram et al., 2018a). There are also examples of individual-based models being adapted to include risk assessment endpoints (Forbes et al., 2019; Hazlerigg et al., 2023). For example, the salmonid model, inSTREAM (Railsback et al., 2022), has been adapted by incorporating the impact of endocrine disrupters on reproduction through the number of viable eggs or fecundity (Forbes et al., 2019).

InSTREAM has the advantage of being spatially specific, has the potential to be adapted to incorporate toxicity test endpoints, and can provide critical population-level parameters including population abundance, size variation, and spatial distribution. However, these outputs are insufficient for assessing the use value of a fishery, which includes other aspects such as accessibility and competition for the angler (Maltby et al., 2021). Therefore, the model needs to not only be adapted to include chemical stressor endpoints but relevant outputs identified for ecosystem service delivery assessment.

Depending on the goals and angling methods of anglers using a fishery, the game fish population can be better described by a subset of the population of interest, referred to as the angling stock. In the case of, for example, trophy hunters, the angling stock would be the largest and oldest fish within the game fish population. This is different from catchable stock which has been used to refer to mature adult fish (Forbes et al., 2019) or the legal restrictions on minimum or maximum catchable size (Oliveira et al., 2009; Schmutz & Sendzimir, 2018). Other measures of angler benefit include catch, harvest, and size of fish caught (Birdsong et al., 2021). However, the net benefit to the main beneficiaries (i.e. anglers) is dependent on more than just the fish caught (Hutt & Jackson, 2008). Benefit to anglers also includes non-catch-related factors such as access to the site, length of fishing season, and crowding or competition for fishing sites. Ecological models such as individual-based models can assess the impact of chemicals to the angling stock but post-processing will be needed to equate changes in angling stock to impact on the recreational angler. This can be addressed through socioecological methods such as for angler satisfaction (Birdsong et al., 2021)

This chapter is focused on adapting inSTREAM, an individual-based/ecological model in trout, to: 1) incorporate common toxicity test endpoints (i.e. individual reproduction, growth, and survival) and indirect effects on fish prey availability to predict the effects of toxicants on fish populations across spatial scales; 2) coupling the outputs from inSTREAM with an ecosystem service delivery assessment in a recreational fishery.

3.2 Model description and rationale: Spatially explicit individual-based model inSTREAM

InSTREAM (version 7.3) is an individual-based model developed by Steve Railsback, Bret Harvey and Daniel Ayllón to model population-level effects in trout species dependent on individual behaviour (Railsback et al., 2022). InSTREAM (2022) is publicly available and free to download from https://ecomodel.humboldt.edu/instream-7-and-insalmo-. The Overview, Design concepts and Detail (ODD) protocol is the standard for describing individual-based models (Grimm, et al., 2020a). The full ODD for inSTREAM is available for download with the model (Railsback et al., 2022).

InSTREAM has been in use for over 25 years since 1999, and included in over 25 publications (Railsback et al., 2021). It was primarily built for investigating different river management scenarios such as for hydropower (Railsback et al., 2021). InSTREAM is regularly updated and a major update involved changing the host software from SWARM to NetLogo. The current version is now hosted in NetLogo version 6.2+, (free to download from https://ccl.northwestern.edu/netlogo/download.shtml) (Wilensky, 1999). NetLogo allows GIS integration through its GIS extension and as a result can be used to simulate individuals based on site data and confined spatial units, which makes the model spatially explicit. InSTREAM coupled with NetLogo, is therefore a powerful tool to determine the effects across spatial scales at the single reach-level to the catchment or river network-level. As a note, during the completion of this thesis, two minor updates of inSTREAM version 7.3 were released, the first (8 August 2022) with minor changes to the BehaviorSpace output and file outputs and the second (7 July 2023) again with minor improvements. Each update has an accompanying user manual which lists all changes but no changes to the underlying mechanisms of inSTREAM were made in either of these updates. The simulations within this thesis have been done prior to the most recent update (7 July 2023) but the results would not change as no major updates have been published. Other versions or sister models of inSTREAM exist including inSTREAM-Gen which considers evolutionary influences (Ayllón et al., 2016) and inSALMO which models migratory salmon (Railsback, 2021).

InSTREAM is capable of simulating salmonid species population responses to river management actions and environmental changes within streams and rivers (Railsback et al., 2022). Any salmonid species can be simulated depending on the input parameters used for inSTREAM. InSTREAM also has the capacity to model either a single trout species or multiple trout species. But for the purpose of this research, the focus is on one trout species. As this thesis aims to investigate the chemical impacts on trout within a recreational fishery, rainbow trout (*Oncorhynchus mykiss)*, a common species both used in ecological risk assessment and a common recreational fishery game fish, was selected as the focal species (Serchuk et al., 1980; European Commission, 2024).

InSTREAM, as an ecological model, has the capability to simulate effects from stressors in a river ecosystem. Previous uses are mainly focused on temperature and water flow assessments and only two studies have adapted the model for use with chemical stress (Forbes et al., 2019; Hazlerigg et al., 2023). However, these adaptations were in previous versions of inSTREAM (hosted in SWARM) or are not publicly available and the model itself has not been fully adapted or analysed regarding the choice of endpoints and implementation of chemical effects. Therefore, use of inSTREAM to explore the effects of chemical stressors on ecosystem service delivery, required i) adaptation of the model to allow simulation of effects of chemical stressors (Section 3.4), ii) decisions about the model running conditions and set-up, such as the number of replicates required and appropriate time periods for simulations (Section 3.5) and iii) post-processing model outputs to assess angler satisfaction (Section 3.6).

3.3 Data

3.3.1 Study sites

Two spatial scales will be used to address spatial scaling, the river reach (Chapter 4) at the local scale and the river network (Chapter 5) at the catchment-scale. To consider variation between reaches, four reaches were selected for simulation at the reach-scale. Three of these reaches of varying stream order (first, second, and third) were then compiled into a hypothetical river network.

3.3.2 River reaches

Figure 3.1 shows the four reach shapefiles which were used for simulations which are available in GIS Shapefile format from ArcGIS Pro (ESRI, 2021) (see Chapter 4 for further details on the simulations).

Reach A was used in developing the adaptation of the model (Figure 3.1A). Reach A is based on a stream which supports steelhead trout (*Oncorhynchus mykiss irideus*), an anadromous form of coastal rainbow trout, but was calibrated for rainbow trout (Railsback et al., 2022). The other three reaches (Figure 3.1B – D) were originally calibrated with cutthroat trout (*Oncorhynchus clarkii*) and not rainbow trout (Harvey & Railsback, 2011). Model parameters were changed to those for rainbow trout with the exception of initial input of fish length parameters which is not species-specific, as this size distribution reflects the capacity that fish can grow within their specific habitat, or reach (Railsback et al., 2022).

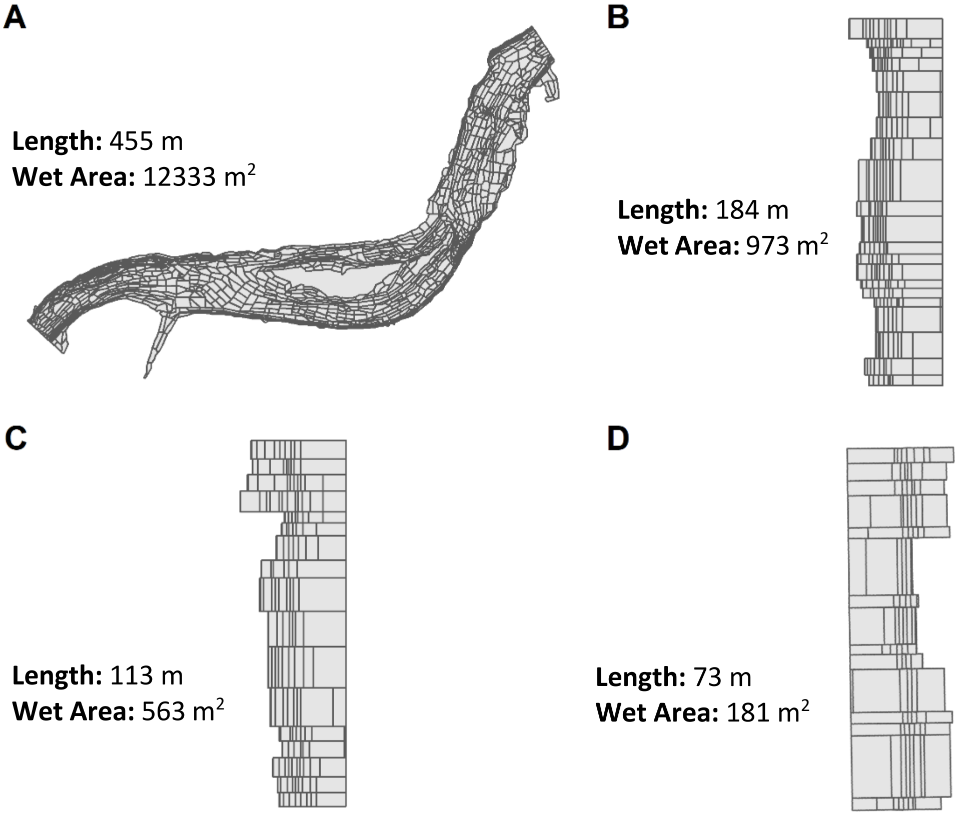


Figure 3.1 GIS Shapefiles with habitat cells; A) Reach A (Railsback et al., 2013, 2015)*, downloadable with inSTREAM version 7.3, delineated habitat cells based on a hydrodynamic model mesh and habitat observations, B) Reach B, C) Reach C, and D) Reach D* (Harvey & Railsback, 2011) *are all from the same river network with habitat cells depicted as rectangular cells. All reaches are based on rivers in California, USA. Not to the same size scale.*

3.3.3 River network

A hypothetical river network (Figure 3.2), available in GIS Shapefile format from ArcGIS Pro (ESRI, 2021), was constructed from river reaches of different stream orders: first order (Figure 3.1B), second order (Figure 3.1C), and third order (Figure 3.1D) (see Chapter 5 for further details on the simulations). Reaches were connected to make up a network of six tributary branches, each with two copies of a first order reach, three forks each with two copies of a second order reach, and a mainstem segment of two copies of a third order reach. The hypothetical river network was constructed by Jason White (Pacific Southwest Research Station, U.S. Department of Agriculture) and provided by Steve Railsback (Humboldt Department of Mathematics and Lang, Railsback and Associates) and Bret Harvey (Pacific Southwest Research Station, U.S. Department of Agriculture) based on a network previously applied in an older version of inSTREAM (Harvey & Railsback, 2011).

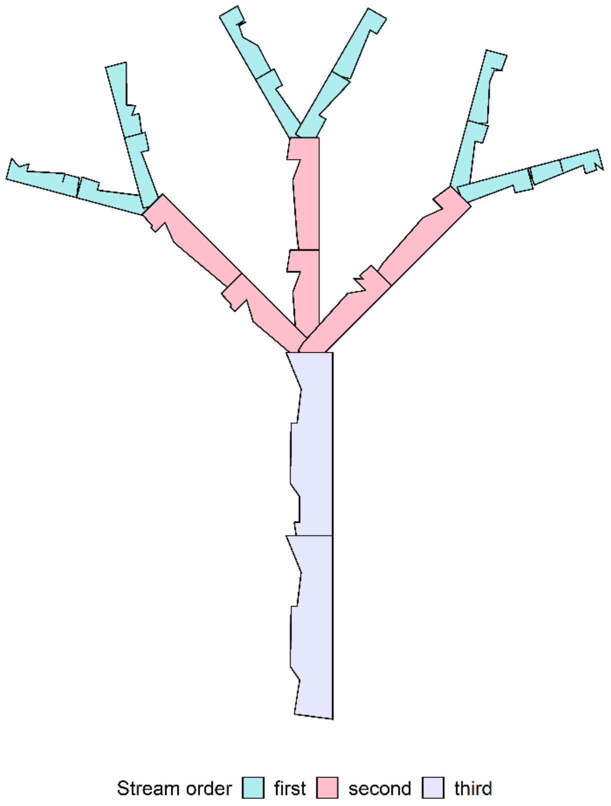


Figure 3.2 A hypothetical river network for simulations in inSTREAM (version 7.3). The river network is made up of 20 river reach sites composed of replicates of first order (12) (blue), second order (6) (pink), and third order (2) (purple) reach sites. Reach copies for each stream order are identical environmentally and in terms of initial trout populations.

3.3.4 Time series data

Reach sites

In order to run simulations in inSTREAM, environmental time series data was needed for daily measurements of flow, temperature, and turbidity (Railsback et al., 2022). Historical monitoring data for the study reach sites were used for simulations and the time series was selected to exclude extreme environmental events (reaches B – D) or restoration activities (reach A (Railsback et al., 2013)). Time series data are presented in this Chapter as actual years (Figure 3.3 and Figure 3.4), but in subsequent analyses (i.e. Chapter 4 and 5) it is presented as time since the start of the simulation or exposure period. In order to have a long enough time period (i.e. 19 years), reach site A time series data was extended by duplicating a stable period in the historical data set.

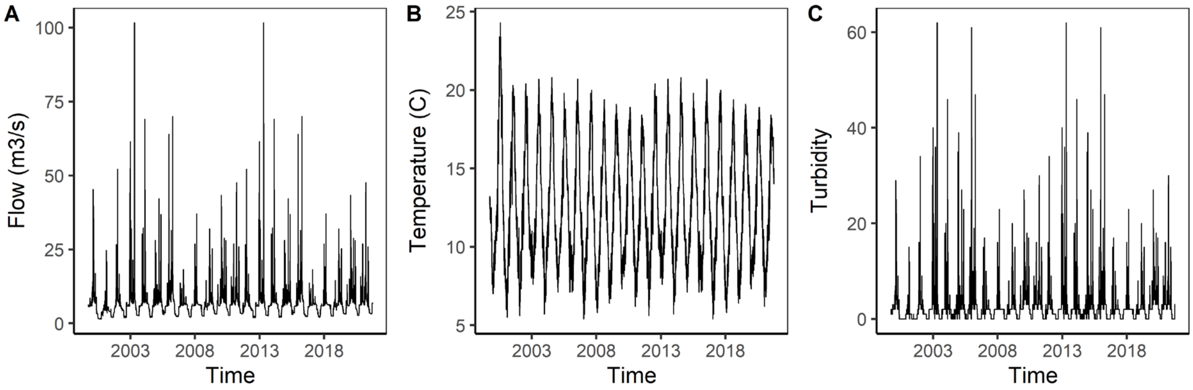


Figure 3.3 Historical daily monitoring data on A) flow (m^3/s), B) temperature (°C), and C) turbidity for Reach site A. The data was extended (by duplicating data) in order to have a long enough time period to conduct the simulations.

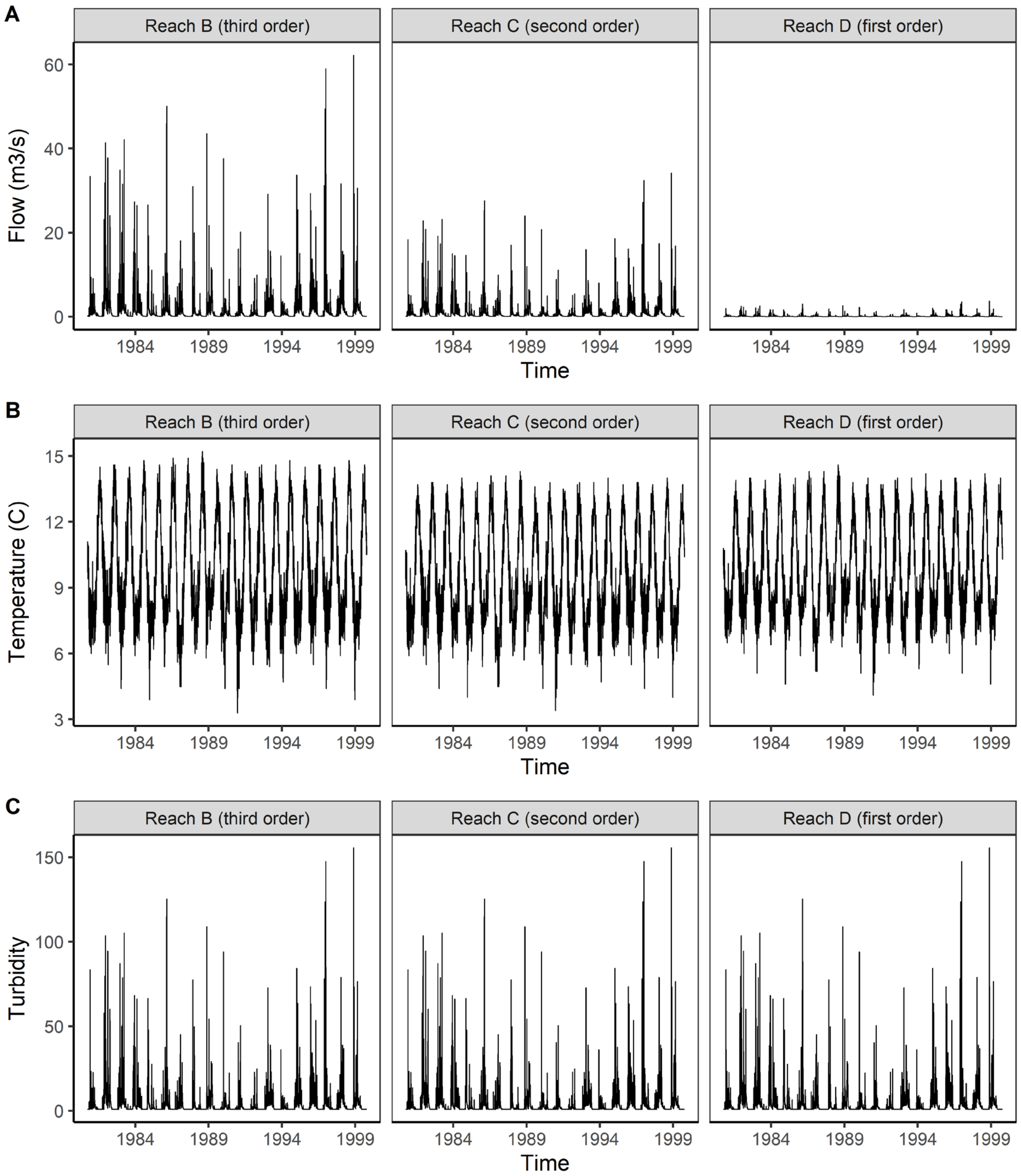


Figure 3.4 Historical daily monitoring data on A) flow (m^3/s), B) temperature (°C), and C) turbidity for reaches B, C, and D.

River network

For the river network, the same time series data was used for each reach based on their historical monitoring data. To verify that the measured flow data was indicative of network flows, cumulative flows were estimated and compared to measured (historical data) for both second and third order streams or branches (Figure 3.5). Cumulative flows for third and second order streams were calculated by summing the flow from upstream reaches. For both second and third order stream flow, measured values exceed cumulative values. This indicates that measured reach flows reflect ecological realism within the simulated river network as water flow is not only from upstream reaches but also from the surrounding landscape.

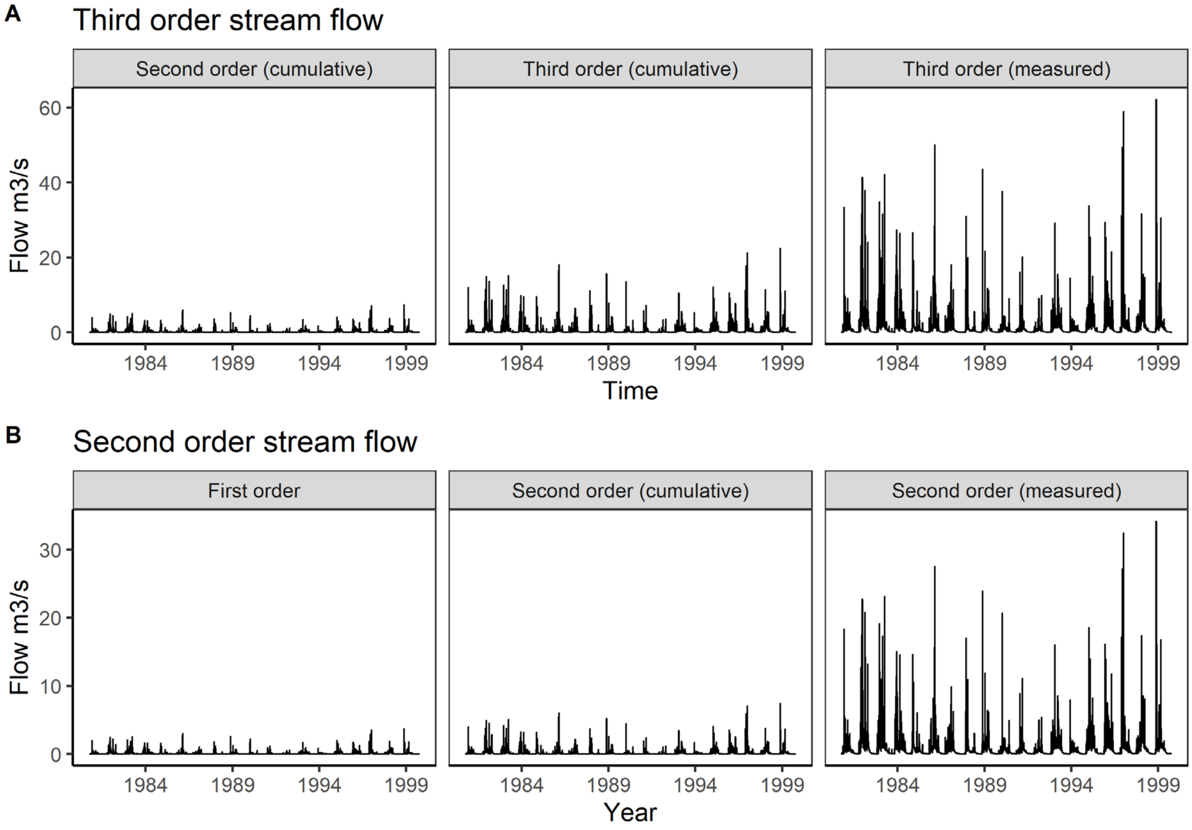


Figure 3. 5. Historical daily flow (m^3/s) data for the simulation period of a river network. A) Third order stream flow from cumulative flows of second order to find the cumulative flow in third order were compared to measured flows. B) Second order stream flow from first order to find the cumulative flow in second order were compared to measured flows.

3.3.5 Initial populations

Initial rainbow trout populations used in the initialisation of inSTREAM are shown for each reach in Table 3.1. The same initial populations were used for the corresponding reach in the river network. These population numbers are based on previous applications of reaches A (Railsback et al., 2013), B, C, and D (Harvey & Railsback, 2011) and are estimated on typical numbers found in the reaches.

Table 3.1 Initial populations for each reach

Reach	**Age 0**	**Age 1**	**Age 2+**
A	1500	350	100
B	60	30	30
C	71	14	12
D	28	6	2

3.3.6 Data analysis

All data analysis for Chapters 3, 4, and 5 was done with R version 4.0.3 (R Core Team, 2024) in R Studio (RStudio Team, 2020) using R packages: dplyr (Wickham et al., 2023), tidyverse (Wickham et al., 2019), tidyr (Wickham, Vaughan, & Girlich, 2023), reshape2 (Wickham, 2022), lubridate (Grolemund & Wickham, 2011), and stringr (Wickham, 2023) for data manipulation; ggplot2 (Wickham, 2016), RColorBrewer (Neuwirth, 2022), patchwork (Pedersen, 2023), ggnewscale (Campitelli, 2024), ggrepel (Slowikowski et al., 2024), ggpubr (Kassambara, 2023) and ggprism (Dawson, 2024) for data visualisation; and sf (Pebesma, 2018) for spatial analysis. Chapter 4 also used R package ggforce (Pedersen, 2024) for data visualisation.

3.4 Model adaptation and testing: tailoring inSTREAM for simulations

Chemical stressors may have many different proximate mechanisms but ultimately these impact whole organism performance characteristics such as growth, mortality, and reproductive output, common chemical effect endpoints used in risk assessment (Ashauer & Brown, 2013; Vaugeois et al., 2020; Vlaeminck et al., 2022). Therefore, growth rate, survival (more relevant in ecological modelling) and reproduction (or fecundity in inSTREAM) were selected as suitable targets to decide the closest corresponding parameters in inSTREAM. Additional changes to inSTREAM included combined stress with prey availability, analysing stress in a river network, and angler satisfaction for ecosystem service delivery. All code changes made to inSTREAM are documented in Appendix 3.10.

3.4.1 Individual trout life history parameters and sensitivity analyses

Inhibition of the life history parameters fecundity, growth rate, and survival were added to inSTREAM to mimic chemical stress. This was done by first selecting the most appropriate target in inSTREAM (e.g. parameter or sub-model) and then adding an inhibition parameter to this target. A subsequent sensitivity analysis was performed for each added inhibition parameter by simulating the effects of none to complete inhibition.

Fecundity

Spawning of trout in inSTREAM occurs between 1 April to 30 June and during this period the model determines whether female trout are ready to spawn every day based on their age, length, condition, mate selection, and season (Railsback et al., 2022). A number of sub-models exist for reproduction in inSTREAM: spawning readiness, spawning, spawning site selection, and spawning mate selection (Railsback et al., 2022). As spawning site and mate selection both represent reproductive behaviours of the trout, and spawning readiness is related to maturity, the spawning sub-model was selected as the most appropriate target for chemical stress. Fecundity (the number of viable eggs produced per spawning day in a spawning season) is a common endpoint of individual toxicity tests in risk assessment (European Food Safety Authority, 2010; Larras & Charles, 2022) and is often used to extrapolate to population-level effects (Ford et al., 2021). Within the spawning sub-model, the number of viable eggs (n), is a function of spawner length (L, cm), two species-dependent constants (m and e) and the fraction of eggs which become viable in redds (v) (Equation 3.1).

$$n = (mL^e \times v)$$ Equation 3.1 (Railsback et al., 2022)

The sub-model was adapted to include chemical inhibition on the parameter v by multiplying this with a new parameter f (see Appendix 8.2.1 for detailed code changes) as shown in Equation 3.2

$$n = (mL^E \times (v \times f))$$ Equation 3.2

where f indicates inhibition of fecundity in individual trout as a fraction of potential impact and ranges from 0 – 1.0, where 0 is 100% inhibition and 1.0 is no inhibition present in the simulation (i.e. baseline). The effect of the 10 years inhibition of fecundity on mean abundance and mean fish length is shown in Figure 3.6 for three trout age groups (Age 0, Age 1, Age 2+).

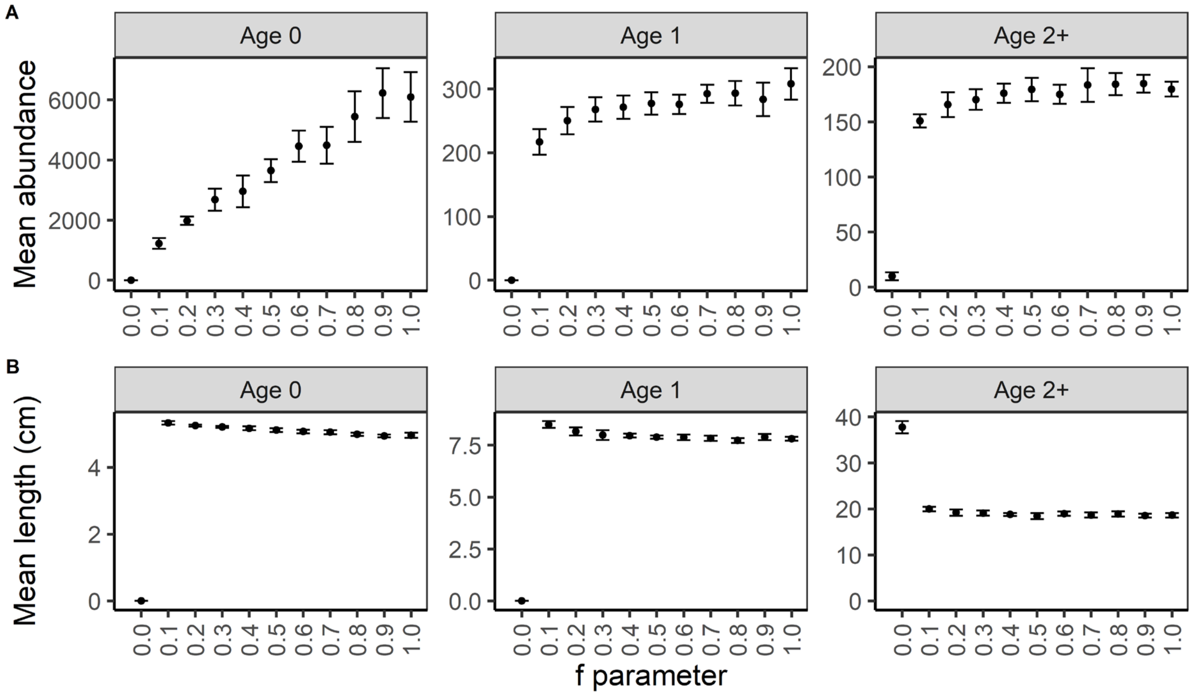


Figure 3.6 The A) mean abundance and B) mean length (cm) for Age 0, Age 1, and Age 2+ trout after 10 years of fecundity inhibition (1 = no inhibition, 0 = 100% inhibition) from ten replicate runs. Error bars denote +/- one standard deviation.

The percentage change in output from a sensitivity analysis of f was calculated with Equation 3.3

$$\Delta Obs = \frac{\overline{Obs} - \overline{Obs_{1.0}}}{Obs_{1.0}} \times 100$$ Equation 3.3 (Vaugeois et al., 2020)

where Obs is the observed value with modified parameters and $Obs_{1.0}$ is the observed value at baseline (set at 1.0 or 0% inhibition). Equation 3.3 was used in subsequent sensitivity analyses for all parameters added to inSTREAM. The resulting sensitivity analyses for abundance and length are shown in Table 3.2.

Table 3.2 Sensitivity analysis of abundance and length to inhibition of f parameter

Trout age group	**f parameter** *(inhibition of fecundity)* (% inhibition)	**Mean abundance percentage change** (±%Δ)	**Mean length (cm) percentage change** (±%Δ)
Age 2+	1.0 (0%)	0.00	0.00
	0.9 (10%)	2.78	-0.36
	0.8 (20%)	2.50	1.40
	0.7 (30%)	2.06	0.25
	0.6 (40%)	-2.62	1.76
	0.5 (50%)	-0.22	-0.96
	0.4 (60%)	-2.06	0.90
	0.3 (70%)	-5.29	2.46
	0.2 (80%)	-7.85	3.09
	0.1 (90%)	-16.03	7.19
	0.0 (100%)	-94.55	102.54
Age 1	1.0 (0%)	0.00	0.00
	0.9 (10%)	-7.92	0.96
	0.8 (20%)	-4.74	-1.13
	0.7 (30%)	-5.03	0.23
	0.6 (40%)	-10.42	0.84
	0.5 (50%)	-10.00	0.98
	0.4 (60%)	-11.85	1.89
	0.3 (70%)	-13.02	2.25
	0.2 (80%)	-18.70	4.49
	0.1 (90%)	-29.57	8.75
	0.0 (100%)	-100.00	-100.00
Age 0	1.0 (0%)	0.00	0.00
	0.9 (10%)	2.18	-0.48
	0.8 (20%)	-10.69	0.55
	0.7 (30%)	-26.26	1.79
	0.6 (40%)	-26.83	2.22
	0.5 (50%)	-40.14	3.12
	0.4 (60%)	-51.44	4.18
	0.3 (70%)	-55.98	4.96
	0.2 (80%)	-67.53	5.80
	0.1 (90%)	-79.90	7.43
	0.0 (100%)	-100.00	-100.00

The sensitivity analysis shows that for Age 1 and Age 2+ trout abundance and mean length were both less sensitive to the f parameter than for Age 0 (hatchlings) as abundance and mean length were between ± 30% until 100% inhibition (0.0) where both were 100% impacted. Age 0 trout, however, surpassed -30% at 50% (0.5) inhibition with the f parameter. Although the adult population is of direct importance to recreational anglers, they are not as sensitive as hatchlings which were increasingly affected. This is important for anglers as this shows that the population might not be sustainable as new trout are less likely to be born into the stream.

Growth rate

Growth rate is calculated by the inSTREAM growth rate sub-model for each trout depending on their activity at each time step. Growth rate is calculated sub-daily, four times a day when trout activity is feeding. Feeding activity is either drift or search feeding and non-feeding behaviour is hiding. Hiding does not provide any energy intake as the fish is not actively feeding (Railsback et al., 2022). Both drift feeding and search feeding were selected as targets for inhibition by adding the parameter g into Equation 3.4

$$G = \frac{I}{D} \qquad \text{Equation 3.4 (Railsback et al., 2022)}$$

where G is the growth rate (g/day), I is the energy intake from drift or search feeding activity (J/day), and D is the energy density of the trout (J/g) from net energy intake and weight change (Railsback et al., 2022). The updated sub-model is shown in Equation 3.5

$$G = \left(\frac{I}{D}\right) \times g \qquad \text{Equation 3.5}$$

where g indicates inhibition of growth rate in individual trout as a fraction of potential impact and ranges from 0 – 1.0 where 0 is 100% inhibition and 1.0 is no inhibition present in the simulation (see Appendix 8.2.2 for detailed code changes). The mean abundance and length of fish under different levels of growth rate is shown in Figure 3.7 and the results of a sensitivity analysis of the impact of inhibition on abundance and length are summarised in Table 3.3.

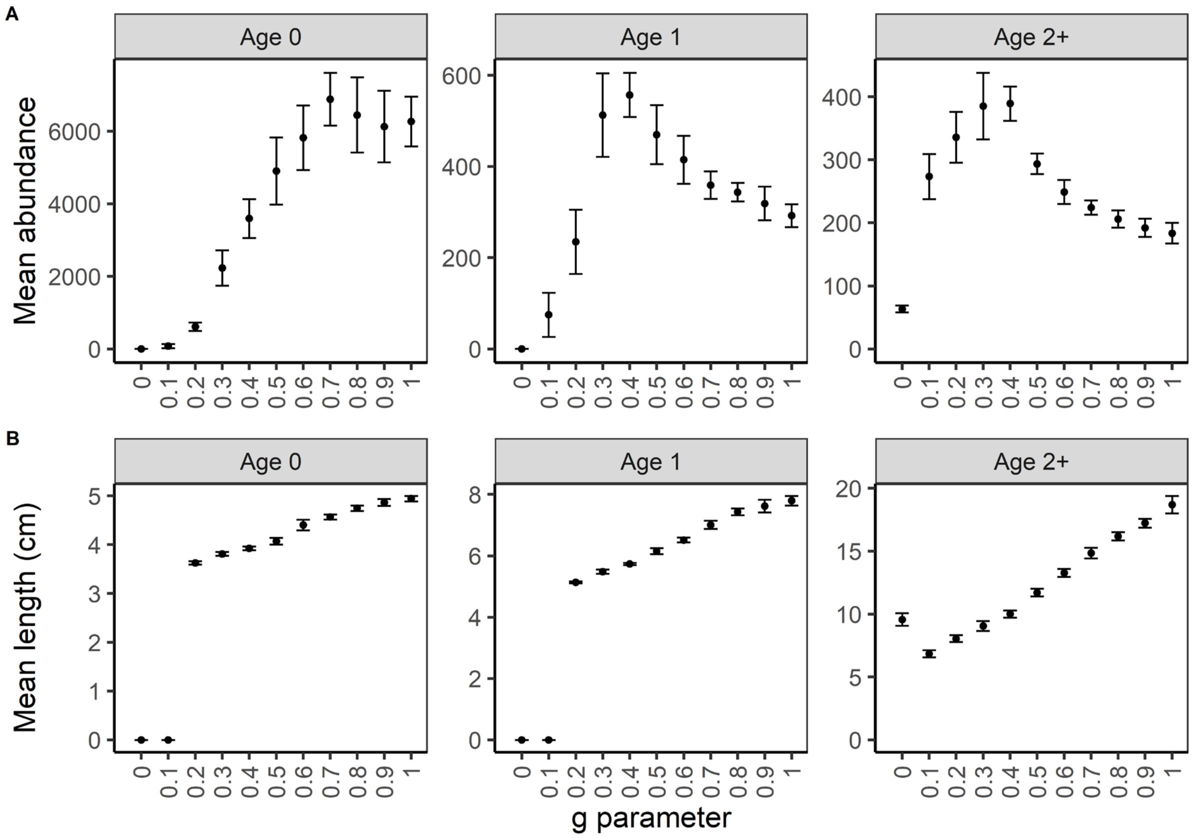


Figure 3.7 The A) mean abundance and B) mean length (cm) for Age 0, Age 1, and Age 2+ trout after 10 years of growth rate inhibition (1 = no inhibition, 0 = 100% inhibition) from ten replicate runs. Error bars denote +/- one standard deviation.

Table 3.3 Sensitivity analysis of abundance and length to inhibition of g parameter

Trout age group	**g parameter** *(inhibition of growth rate)* (% inhibited)	**Mean abundance percentage change** (±%Δ)	**Mean length (cm) percentage change** (±%Δ)
Age 2+	1.0 (0%)	0.00	0.00
	0.9 (10%)	4.74	-7.88
	0.8 (20%)	12.21	-13.46
	0.7 (30%)	22.19	-20.62
	0.6 (40%)	35.71	-29.02
	0.5 (50%)	60.03	-37.38
	0.4 (60%)	112.16	-46.50
	0.3 (70%)	109.98	-51.56
	0.2 (80%)	83.04	-56.96
	0.1 (90%)	49.07	-63.43
	0.0 (100%)	-65.54	-48.82
Age 1	1.0 (0%)	0.00	0.00
	0.9 (10%)	9.07	-2.23
	0.8 (20%)	17.59	-4.63
	0.7 (30%)	22.90	-10.10
	0.6 (40%)	41.92	-16.41
	0.5 (50%)	60.71	-21.08
	0.4 (60%)	90.59	-26.44
	0.3 (70%)	75.50	-29.66
	0.2 (80%)	-19.68	-34.11
	0.1 (90%)	-74.44	-100.00
	0.0 (100%)	-100.00	-100.00
Age 0	1.0 (0%)	0.00	0.00
	0.9 (10%)	-2.25	-1.60
	0.8 (20%)	2.84	-3.99
	0.7 (30%)	9.81	-7.63
	0.6 (40%)	-7.15	-10.96
	0.5 (50%)	-21.79	-17.69
	0.4 (60%)	-42.68	-20.63
	0.3 (70%)	-64.45	-22.90
	0.2 (80%)	-90.25	-26.70
	0.1 (90%)	-98.78	-100.00
	0.0 (100%)	-100.00	-100.00

The sensitivity analysis shows that for Age 0, Age 1, and Age 2+ abundance are sensitive to changes in the g parameter to a similar degree but Age 1 and Age 2 abundance peak at 0.4 (60% inhibition) whilst Age 0 peaks earlier at 0.7 (30%) so is more sensitive than the older trout. However for mean length, Age 0 and Age 1 are more similar, steadily decreasing from 1.0 – 0.2 (0% -– 80% inhibition), where mean length suddenly drops, whilst for Age 2+ it steadily decreases across the range 1.0 – 0.1 (0 – 90%) with no sudden drops in but overall a more marked decline in length. Therefore larger (and older) trout lengths are more sensitive to the g parameter then Age 0 and Age 1, but Age 0 abundance is more sensitive to the parameter then Age 1 and Age 2+ trout.

Survival

Growth rate and fecundity were already present as sub-models within inSTREAM, however, there was no existing sub-model that could be used to target the effects of chemical stress on survival. Survival within inSTREAM is determined by several sub-models for stressors such as high temperature, low temperature, stranding, and condition. Therefore, to simulate inhibition of survival from chemical exposure, a new sub-model was added to inSTREAM (see Appendix 8.2.3 for detailed code changes). This model was added following the format of existing survival sub-models, where trout survival is determined at each time step by first calculating the daily probability of surviving the source of mortality followed by a random Bernoulli trial to determine if the trout survives, but in this case the source is chemical exposure (Railsback et al., 2022). The probability of survival is calculated as an $Adjusted\ survival$, between 0 – 1.0 (100% – 0% inhibition), estimated from $Observed\ survival$ over the course of the year with the following Equation 3.6

$$Adjusted\ survival = Observed\ survival^{\left(\frac{t_s}{t_o}\right)}$$ Equation 3.6 (Krebs, 2014)

where t_s is the standardised test time interval and t_o is the observed (or known) time interval. Here the adjusted survival time period is 14 days, based on the length of the Organisation for Economic Co-operation and Development (OECD) Test No. 204: Fish, Prolonged Toxicity Test (OECD, 1984). The observed survival time period is the course of the entire year (365 days) identified as the new parameter $Observed\ survival$. These time points are one example, but ultimately serve to help identify the survival probability to chemical exposure where inSTREAM can still produce viable populations and evaluate the responses of rainbow trout populations to the addition of a new survival sub-model, which trout were sensitive to. The calculated values of adjusted survival with the new sub-model are shown in Table 3.4. The adjusted survival value is the probability of survival that inSTREAM will apply to each trout individual. The observed survival over the year was used to report the amount of inhibition (Table 3.4). The probability to survive is the same for each trout in the simulations regardless of age, size, or sex.

Table 3.4 Calculated probability to survive chemical exposure by the new sub-model added to inSTREAM based on extrapolating from an observed survival time of 365 days to an adjusted survival time of 14 days.

Adjusted survival (14 days)	**Observed survival (365 days)**
1.0 (0%)	1.0 (0%)
0.99961458195 (~4 x 10^{-4}%)	0.99 (1%)
0.99922540179 (~8 x 10^{-4}%)	0.98 (2%)
0.99883238382 (~ 1 x 10^{-4}%)	0.97 (3%)
0.99843545005 (~15 x 10^{-4}%)	0.96 (4%)
0.99803452005 (~ 20 x 10^{-4}%)	0.95 (5%)
0.9959669295 (~40 x 10^{-4}%)	0.90 (10%)
0.99378578581 (~62 x 10^{-4}%)	0.85 (15%)
0.99147759263 (~85 x 10^{-4}%)	0.80 (20%)
0 (100%)	0.0 (100%)

The mean abundance and length when inhibition of survival was between 0 – 100% is shown in Figure 3.8 for all trout age groups and a sensitivity analysis of the impact on outputs of abundance and length are summarised in Table 3.5.

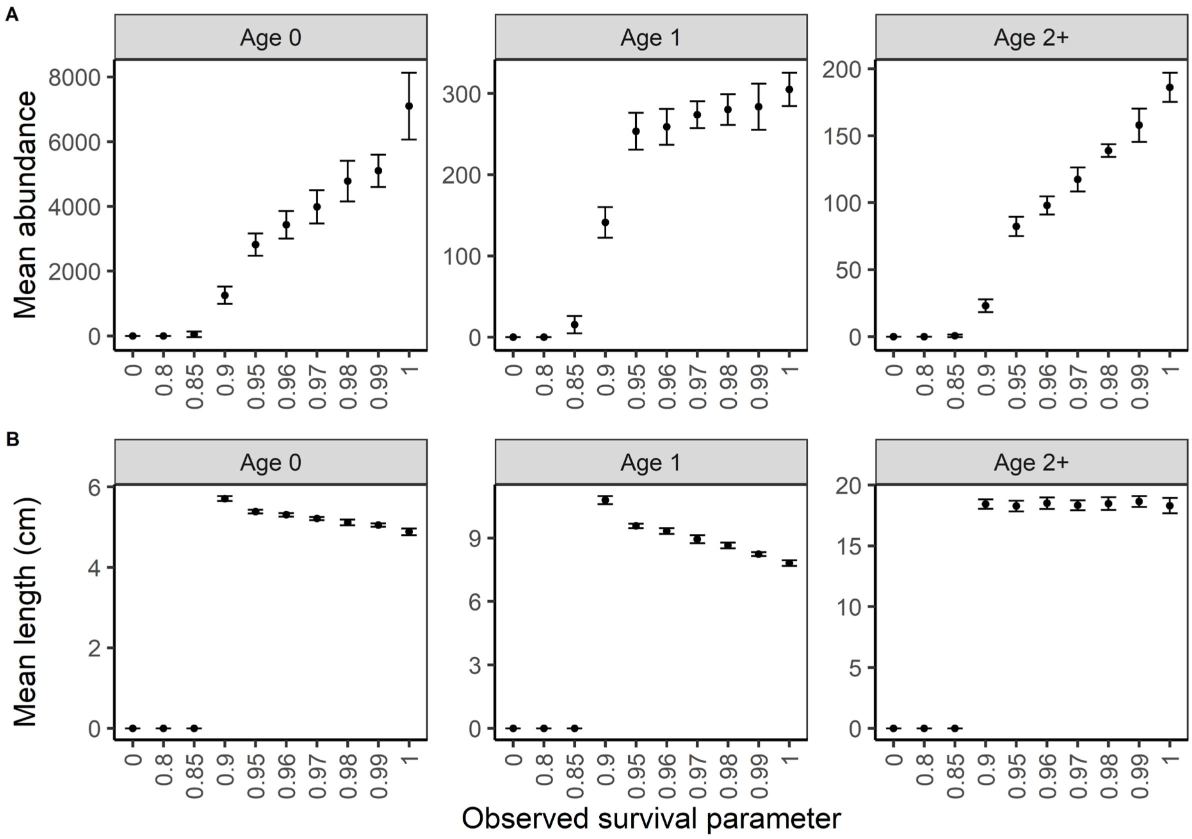


Figure 3.8 The A) mean abundance and B) mean length (cm) for Age 0, Age 1, and Age 2+ trout after 10 years of survival inhibition (1 = no inhibition, 0 = 100% inhibition) from ten replicate runs. Error bars denote +/- one standard deviation.

Table 3.5 Sensitivity analysis of abundance and length to inhibition of $Observed\ survival$ *parameter*

Trout age group	$\boldsymbol{Observed\ survival}$ **parameter** *(inhibition of survival)* (% inhibited)	**Mean abundance percentage change** (±%Δ)	**Mean length (cm) percentage change** (±%Δ)
Age 2+	1.0 (0%)	0.00	0.00
	0.99 (1%)	-15.21	1.89
	0.98 (2%)	-25.42	0.97
	0.97 (3%)	-36.97	0.21
	0.96 (4%)	-47.39	1.10
	0.95 (5%)	-55.83	-0.16
	0.90 (10%)	-87.64	0.74
	0.85 (15%)	-99.68	-100.00
	0.80 (20%)	-100.00	-100.00
	0.0 (100%)	-100.00	-100.00
Age 1	1.0 (0%)	0.00	0.00
	0.99 (1%)	-7.01	5.37
	0.98 (2%)	-8.13	10.67
	0.97 (3%)	-10.23	14.40
	0.96 (4%)	-15.14	19.42
	0.95 (5%)	-16.88	22.53
	0.90 (10%)	-53.75	38.15
	0.85 (15%)	-94.95	-100.00
	0.80 (20%)	-100.00	-100.00
	0.0 (100%)	-100.00	-100.00
Age 0	1.0 (0%)	0.00	0.00
	0.99 (1%)	-28.12	3.37
	0.98 (2%)	-32.64	4.78
	0.97 (3%)	-43.88	6.72
	0.96 (4%)	-51.64	8.64
	0.95 (5%)	-60.28	10.21
	0.90 (10%)	-82.32	16.90
	0.85 (15%)	-99.30	-100.00
	0.80 (20%)	-100.00	-100.00
	0.0 (100%)	-100.00	-100.00

The sensitivity analysis shows that Age 0, Age 1, Age 2+ were all similarly sensitive to the $Observed\ survival$ parameter for abundance and length where 0.95 – 0.96 results in 50% reduction in abundance across all age groups. The same applies to mean length which is ultimately unaffected in Age 2+ trout but slightly increases in Age 0 and Age 1 trout probably a result of less trout growing within the reach to compete for resources.

3.4.2 Indirect impact on prey availability and sensitivity analysis

Sometimes fish are indirectly impacted by chemicals through their food source or prey availability as a result of direct toxicity on prey. Prey density is a reach specific parameter in inSTREAM and the prey energy content (J/g) within a reach (i.e. *reach-prey-energy-density,* P) is used to convert intake from drift and search feeding (i) to net energy intake of drift or search feeding by individuals (E) as shown in Equation 3.7.

$$E = i \times P \qquad \text{Equation 3.7 (Railsback et al., 2022)}$$

Parameter P was inhibited to reflect chemical stress by multiplying it with the new parameter A as shown in Equation 3.8

$$E = i \times (P \times A) \qquad \text{Equation 3.8}$$

where A indicates the inhibition of prey availability and ranges from 0 – 1.0 (or 100% – 0% inhibition) where 1.0 is no inhibition present in the simulation (see Appendix 8.2.4 for detailed code changes). The effect of inhibition of prey availability on mean abundance and fish length is shown in Figure 3.9 and a sensitivity analysis is summarised in Table 3. 6. The sensitivity analysis shows that abundance is more sensitive to inhibition of prey availability (Table 3.6) than inhibition of growth rate (Table 3.3) even though both inhibition parameters target the same growth rate sub-model within inSTREAM. All age groups are equally as sensitive to changes in prey availability, with between 0.5 – 0.6 (40% – 50%) inhibition resulting in no trout persisting within the simulation after 10 years. Trout mean length is seen to decrease for all age groups, although most drastically for Age 2+ trout.

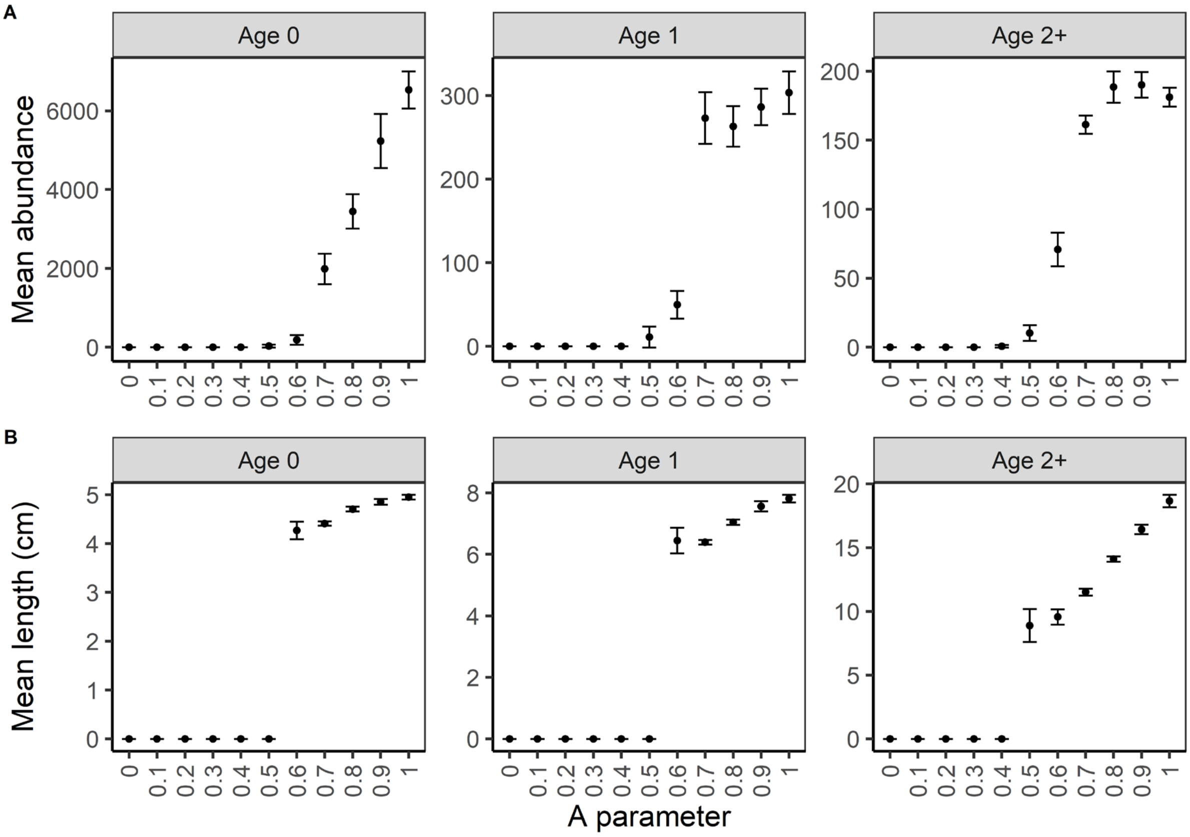


Figure 3.9 The mean abundance (top) and mean length (cm) (bottom) for Age 0, Age 1, and Age 2+ trout after 10 years of growth rate inhibition (1 = no inhibition, 0 = 100% inhibition) from ten replicate runs. Error bars denote +/- one standard deviation.

Table 3.6 Sensitivity analysis of abundance and length to inhibition of A parameter

Trout age group	**A parameter** *(inhibition of prey density)* (% inhibited)	**Mean abundance percentage change** (±%Δ)	**Mean length (cm) percentage change** (±%Δ)
Age 2+	1.0 (0%)	0.00	0.00
	0.9 (10%)	2.78	-0.36
	0.8 (20%)	2.50	1.40
	0.7 (30%)	2.06	0.25
	0.6 (40%)	-2.62	1.76
	0.5 (50%)	-0.22	-0.96
	0.4 (60%)	-2.06	0.90
	0.3 (70%)	-5.29	2.46
	0.2 (80%)	-7.85	3.09
	0.1 (90%)	-16.03	7.19
	0.0 (100%)	-94.55	102.54
Age 1	1.0 (0%)	0.00	0.00
	0.9 (10%)	-7.92	0.96
	0.8 (20%)	-4.74	-1.13
	0.7 (30%)	-5.03	0.23
	0.6 (40%)	-10.42	0.84
	0.5 (50%)	-10.00	0.98
	0.4 (60%)	-11.85	1.89
	0.3 (70%)	-13.02	2.25
	0.2 (80%)	-18.70	4.49
	0.1 (90%)	-29.57	8.75
	0.0 (100%)	-100.00	-100.00
Age 0	1.0 (0%)	0.00	0.00
	0.9 (10%)	2.18	-0.48
	0.8 (20%)	-10.69	0.55
	0.7 (30%)	-26.26	1.79
	0.6 (40%)	-26.83	2.22
	0.5 (50%)	-40.14	3.12
	0.4 (60%)	-51.44	4.18
	0.3 (70%)	-55.98	4.96
	0.2 (80%)	-67.53	5.80
	0.1 (90%)	-79.90	7.43
	0.0 (100%)	-100.00	-100.00

3.4.3 Chemical exposure scenarios

Chemical exposure was added to inSTREAM in order to simulate chemical stress on the trout. This was achieved by adding chemical exposure concentrations to the time series input file for daily environmental characteristics of flow, temperature, and turbidity in a reach. Although a time-variable chemical concentration profile could be used, for the purpose of this thesis a chemical is considered as either present or absent in the reach. InSTREAM was then adapted to read the subsequent time series data for chemical concentration in the same way as for flow, temperature, and turbidity. For each sub-model, where inhibition of a life history parameter was added to inSTREAM, the model first checks that chemical exposure is present (or greater than 0) before it can inhibit any of the life history parameters (see Appendix 8.2.5 for detailed code changes).

By adding chemical exposure into inSTREAM, this also enables modelling of the potential effects of chemical exposure as continuous or pulsed on life history parameters. As a minimum of 6 years post-exposure was determined suitable for simulating impacts (Section 3.5.2), a simulation period of 19 years is needed for investigating a pulse exposure (i.e. 2 years pre-exposure, 6 years exposure, 11 years post-exposure). The run time was expanded from 10 years to 19 years for all simulations to allow for comparisons. The subsequent continuous and pulse chemical exposure scenarios are shown in Figure 3.10 with reach A as an example.

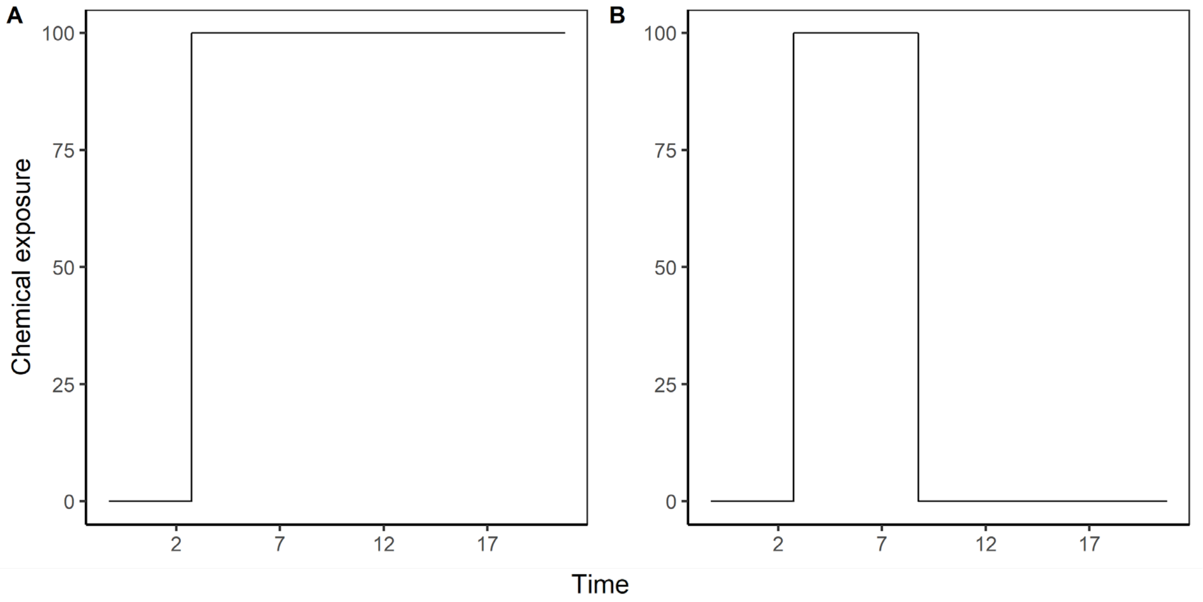


Figure 3.10 A) Continuous chemical exposure and B) pulse exposure for reach A over time (years of simulation). The same exposure profiles were applied to all reach sites. A chemical was either present (exposure set at 100) or absent (exposure set at 0). The model activates inhibition when chemical exposure > 0.

3.4.4 NetLogo interface and BehaviorSpace simulations

All above modifications of the original inSTREAM model (version 7.3) were added to the model code as well as in the model controls on the NetLogo interface, allowing the three direct (fecundity, survival, and growth rate) and one indirect (prey density) parameters to be selected as either on or off, and their magnitude controlled. This allowed simulations to be carried out both with the direct impacts on life history parameters independently and in combination with an indirect stress on prey density.

BehaviorSpace is the NetLogo environment to set-up and execute a large number of experiments (Banos et al., 2015). This is where the parameter values and ranges were specified (0 – 1.0), switches for effects turned on or off for virtual experiments, replicate numbers specified, output files selected, or additional exported data defined. All simulations were carried out using the BehaviorSpace functionality in NetLogo (see Appendix 8.2.6 for detailed code changes).

3.4.5 Angling stock

Angling stock was determined as the largest (> 20 cm length) and oldest (older than 1 year) trout based on fishery size restrictions for salmonids (UK Government, 2024). This is also based on a study which examined the probability of satisfaction reported based on rainbow trout length caught, which found high angler satisfaction was reported for 80% of cases when a rainbow trout larger than 20 cm was caught (McCormick & Porter, 2014). The size limit gives an indication of the abundance of large fish in the population, although trophy hunters would still favour larger fish. The age limit also ensures angling stock individuals are adult trout, promoting growth of trophy-sized stock (Arostegui et al., 2021).

InSTREAM has the built-in ability to count the abundance of specific age groups of trout, i.e. Age 0, Age 1, and Age 2+. The abundance of angling stock was determined by summing up the number of trout which are greater than 20 cm in length and older than 1 year old. The BehaviorSpace output file then summarises these values for each reach and replicate (see Appendix 8.2.7 for detailed code changes).

Wetted area was also added to BehaviorSpace outputs so that angling stock density could be calculated for each reach. InSTREAM determines the wetted area of reaches as cell areas which have a depth greater than zero (cm^2) (Railsback et al., 2022). This wetted area of reaches varies across time depending on the daily water flow. InSTREAM tracks this data in order to model subsequent habitat use by trout (see Appendix 8.2.7 for detailed code changes).

3.4.6 River network and angling stock

Targeting stress in a network reach

Chemical exposure was only added to the reach that was targeted for stress and it was assumed that the chemical degrades before it reaches neighbouring reaches. Consequently, all other reaches within the river network had no chemical exposure for the duration of the simulation. Chemical exposures (present/absent) for individual reaches in a river network were set as described in Section 3.4.3 (see Appendix 8.2.8 for detailed code changes).

Trout movement

In order to assess spatial structures of populations in a river network, trout movements between reaches need to be monitored, but the standard inSTREAM model does not currently do this. Monitoring of trout was added to the model by keeping a record of when individuals moved between reaches as a new event. When an event occurs, such as when a trout spawns or dies, they are recorded into an event output file either for the individual fish themselves or for the fish Redds or spawning sites. The output file records the event itself with the date, time, and location it occurs, as well as certain fish characteristics such as a unique identification number, age, length, weight and condition (Railsback et al., 2022). This event file was modified to include a new event to record trout movement between reaches. This event was identified as "Moved from…" and records the reach the trout has left (see Appendix 8.2.9 for detailed code changes).

Trout natal reaches

The natal reach of each fish was also added as a new event in order to monitor where angling stock hatch in the population. A new parameter, the *trout-natal-reach* was introduced, where natal reach is the reach in which the trout was initialised by the model to represent where it hatched. The concept of natal reach was implemented in a sister model of inSTREAM, inSALMO, which models migratory salmon (Railsback, 2021). The existing code from inSALMO was therefore incorporated into inSTREAM (see Appendix 8.2.10 for detailed code changes).

Individual and event output file updates

InSTREAM has the option to select different output files as well as the BehaviorSpace output files. Changes in output files to include both the trout natal reach and movements as events were monitored using the same code as other events within inSTREAM. Additional changes were made to turn off the Redd or spawning site events output file to reduce the file sizes produced by inSTREAM (see Appendix 8.2.11 for detailed code changes).

3.4.7 Spatial structures in a simulated river network

The presence of spatial structures within the simulated river network was confirmed by testing that the network did not behave uniformly across reaches. Evidence that trout are behaving as metapopulations or as patchy populations within the network was also gathered. Here a metapopulation is a population of populations, with individuals mostly confined to one patch but with occasional local extinction and recolonisation (Ovaskainen & Hanski, 2004; Bellard & Hugueny, 2020). Some patches may be highly productive and act as sources of colonists to other patches, whereas other patches may have higher extinction rates and act as sinks (Bellard & Hugueny, 2020). In patchy populations individuals move frequently between habitat patches during their lifetime with high dispersal leaving no patch unoccupied (Ovaskainen & Hanski, 2004; Bellard & Hugueny, 2020). Salmonids typically occupy various habitats depending on their life stage and possibly exhibit spatial dynamics more related to a hybrid of metapopulation and patchy interacting population (Bellard & Hugueny, 2020).

The spatial structure of the river network can then be characterised by birth (or number of hatchlings in the case of trout), death, and movement in the reaches (Pulliam, 1988; Furrer & Pasinelli, 2016). A baseline simulation was run with the river network under no chemical stress and was also used as the control simulation. Spatial structures were summarised across a water year (*sensu* Wasko et al., 2020) or an adjusted 12-month period starting with the month that has the lowest streamflow, a practice used in hydrological monitoring, but also as the census of angling stock was recorded at the end of a water year (September 30th). The net change in abundance (ΔA) in the last water year of the simulation (October 1st – September 30th or year 17, not including the first two years to allow the population to stabilise) was calculated from the difference in number of hatchlings (B) plus immigration into the reach (I) and the number of deaths (D) minus emigration out of the reach (E) (Equation 3.9)

$$\Delta A = B + I - D - E \qquad \text{Equation 3.9 (Pulliam, 1988)}$$

Mean and standard deviations of 10 simulation replicates were calculated for change in abundance, hatchlings per adult, deaths per adult, and net movement ($I - E$) for each reach within the network (Figure 3.11).

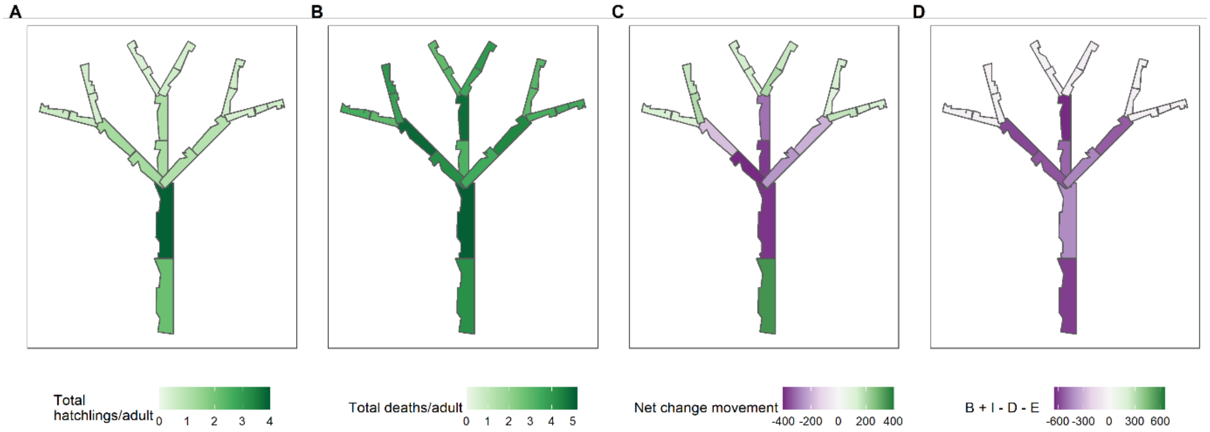


*Figure 3.11 Baseline spatial structure of a simulated river network during the last water year at the end of the simulation period (year 17) for the mean from ten replicate simulations of four characteristics, A) total hatchlings/adult, B) total deaths/adult, C) net change in movement (Immigration – Emigration), and D) net change in abundance (**B**irth i.e. hatchlings in the case of trout + **I**mmigration – **D**eath – **E**migration). Intensity of shading depicts the larger the value. Standard deviation not shown.*

The spatial structure of the populations in the river network is clear with stream orders, branches, and reaches all behaving differently for the four characteristics measured. Hatchlings per adult (Figure

3.11A) was lowest in first order streams and highest in the middle of the river network. Deaths per adult (Figure 3.11B) was similar across the network with highest values upstream in second and third order reaches. Movement was reach dependent with most immigration occurring in the boundary reaches and emigration within the middle reaches of the network (Figure 3.11C). Movement is a size-dependent factor as larger fish are able to move further but the distance moved is not shown here (Railsback et al., 2022). Although many immigration and emigration events took place, many movements involved fish moving back and forth between cells near the border of reaches. Generally all reaches except first order reaches had a net loss of abundance (Figure 3.11D). Although similarly ordered streams behaved in the same way for number of hatchlings and net change in abundance, this was not the same for deaths or net movement, which was also dependent on their location (i.e. either upstream or downstream) within a river branch.

As movement, hatchlings and deaths per adult, and net loss in abundance were all reach dependent, this gives an indication of the presence of source and sink populations. However, as populations persisted throughout the simulation period with no local extinctions and recolonisation events as well as fish were interacting between reach habitats with many movements, populations are possibly patchy and interacting within the river network, a hybrid form of metapopulation. This hybrid form of metapopulation reflects that trout dispersal might be dependent on their life stage and behaviour (e.g. searching for food, spawning sites, and hiding from predators) but exist within distinct patches otherwise (Bellard & Hugueny, 2020).

3.5 Model set-up: deciding on running conditions for simulations

3.5.1 Census days

Census days are common practice in fisheries management to monitor fish population abundance for coarse and game fishing. As an example, in England an Environment Agency survey report found that most respondents fished between the months of October and December (Environment Agency, 2018). A census day at the end of September can therefore provide an estimate of the status of the recreational fishery at the beginning of a fishing season (when angling is at its most popular) and gives a good indicator of the fishery status. For the simulations here, a population census day of September 30th for every year was chosen as this date is post-spawning (i.e. indicating new recruitment into the population), just before the start of the fishing season (October), and at the end of the water year.

3.5.2 Temporal assessment

Following standard protocols, the first two years of simulations, enabled the initial trout population to stabilise in the environment and were not included in subsequent analyses (Railsback & Grimm, 2019). Any chemical exposure is applied after the population stabilisation period.

Suitable time frames for post-stabilisation simulations were determined by running a preliminary simulation of 1-, 5-, and 10- years post-exposure to investigate the influence on the population abundance over time (Figure 3.12) and to identify where the population begins to stabilise. Figure 3.12 shows reduced interannual variation for runs longer than 6 years so a minimum of 6 years post-exposure is suitable for assessment.

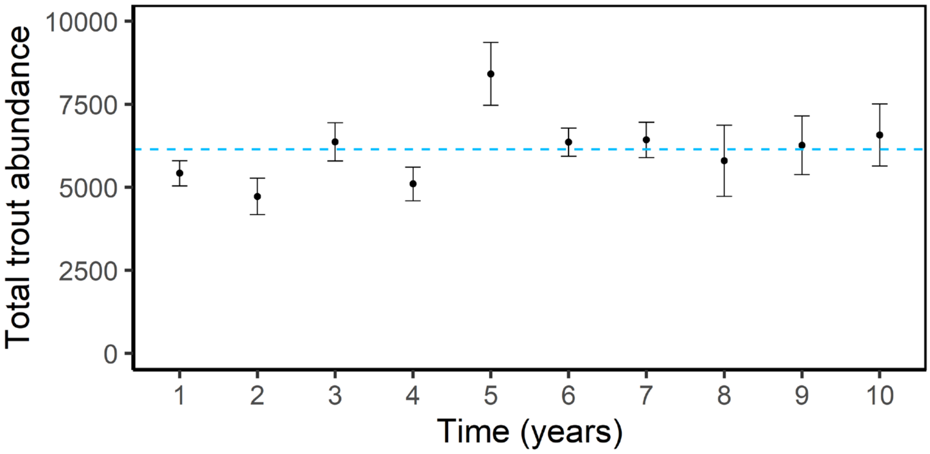


Figure 3.12 Total rainbow trout abundance (Age 0, 1, 2+) for a simulation period of 10 years post-exposure from ten replicate runs. The blue dotted line signifies the mean abundance across the simulation time and identifies where the population appears to stabilise. Error bars denote +/- one standard deviation.

3.5.3 Simulation replicates

InSTREAM guidance (Railsback et al., 2022) suggests that 5 – 10 replicates are sufficient to identify a difference between scenarios. Mean length (Figure 3.13) and age (Figure 3.14) of adult trout (Age 1+), to test the results of a subset of the population, across ten replicates were examined. While there is some variation between the replicates, as expected, the patterns of size structure, age structure, and overall abundance are sufficiently consistent indicating little merit to increase replication further.

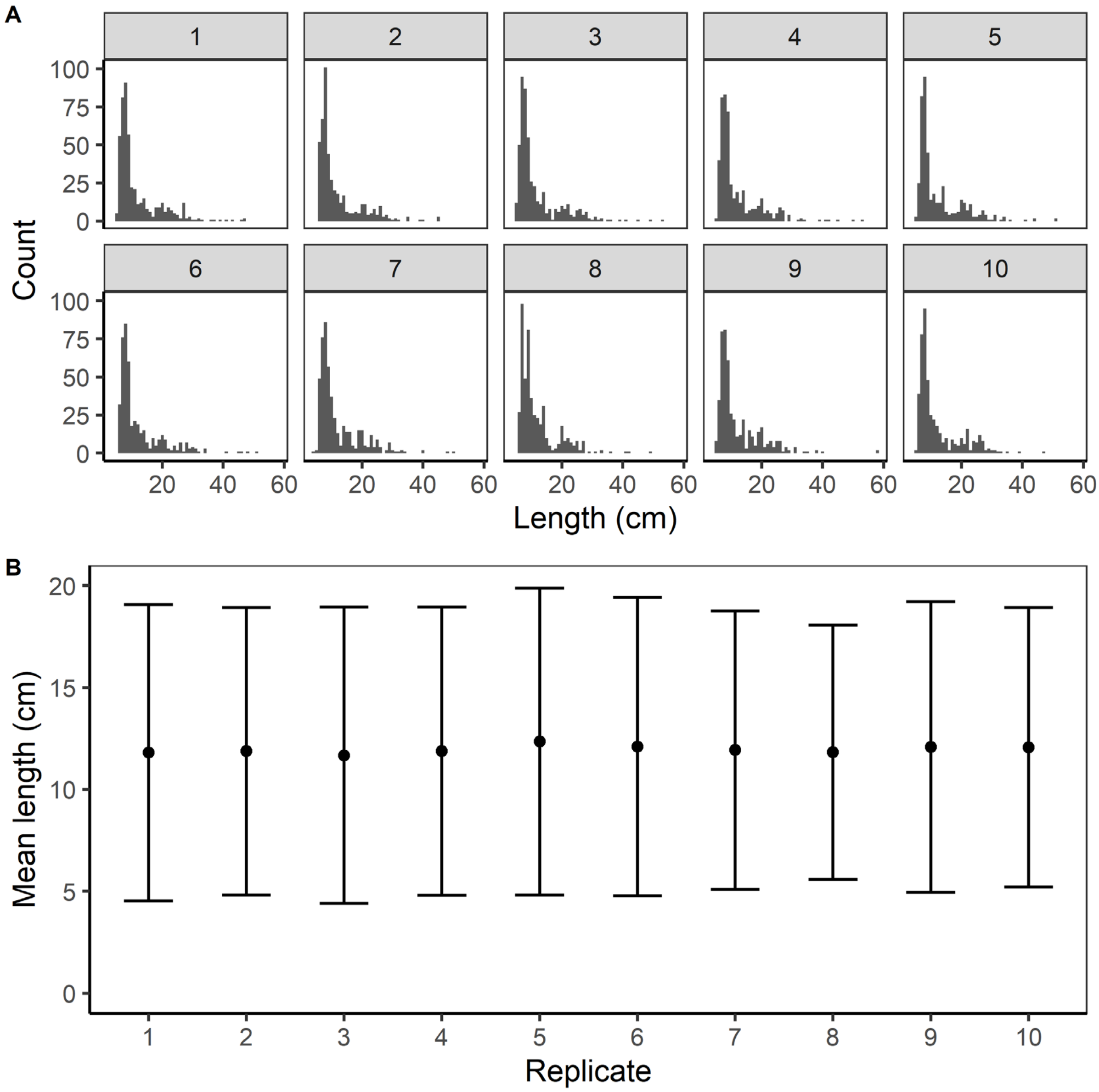


Figure 3.13 Length (cm) of individual rainbow trout 1 year and older in age at the end of a 10-year simulation as A) length density and B) mean length across ten replicates. Error bars denote +/- one standard deviation.

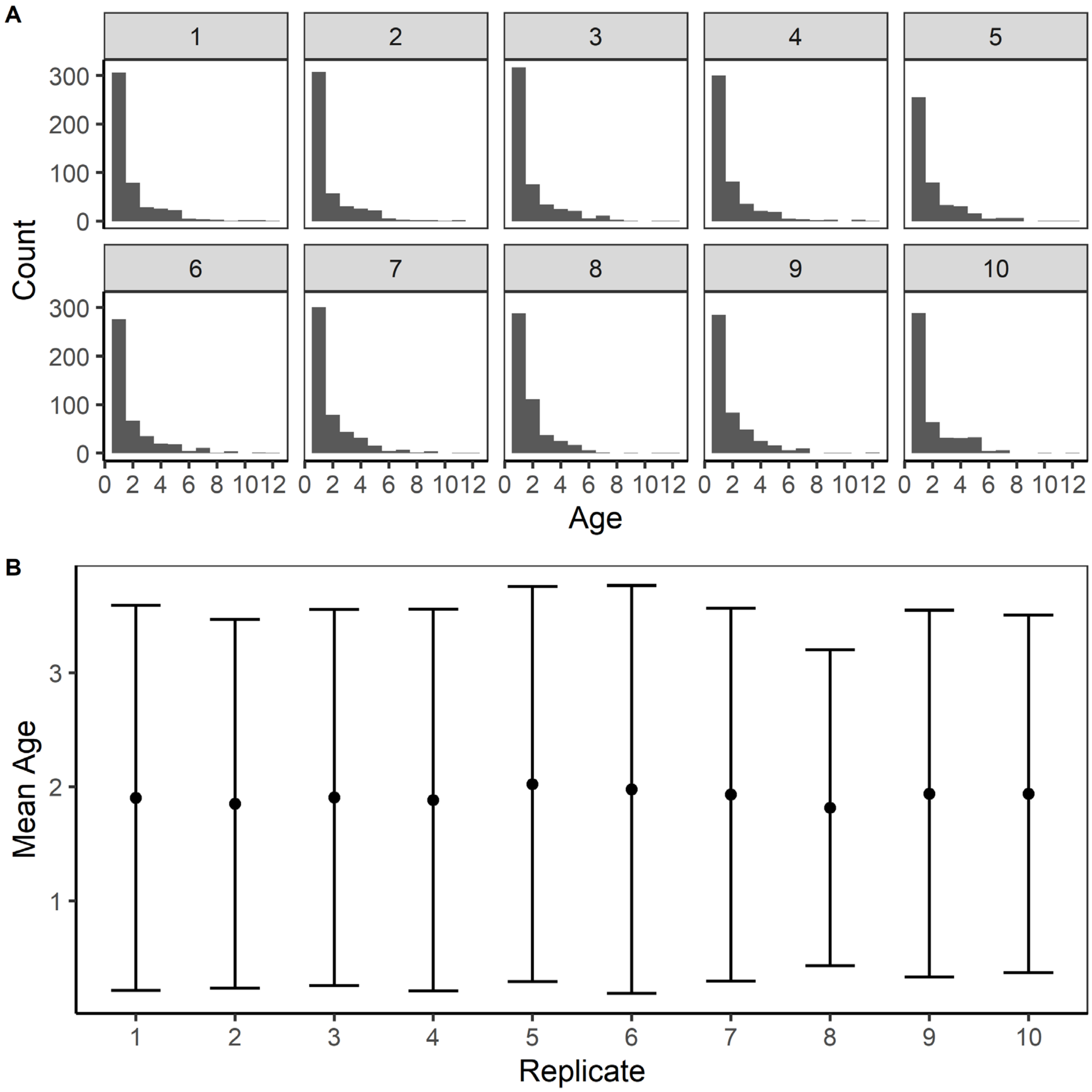


Figure 3.14 Age of individual rainbow trout 1 year and older in age at the end of a 10-year simulation as A) age density and B) mean age across ten replicates. Error bars denote +/- one standard deviation.

To confirm this, the population abundance was compared across replicates. Replicate numbers were considered robust when the difference between mean and standard deviation were independent of replicate number (±5%) using the coefficient of variation (Mintram et al., 2018a) as shown in Table 3.7. Coefficient of variation was within ±5% between 5 – 10 replicates indicating that it is independent of replicate number. Ten replicates were selected to select the higher end of recommended replicates.

Table 3.7 Comparing replicate number with the mean, +/- one standard deviation, and coefficient of variation for abundance of rainbow trout 1 year and older in age.

Replicates	Mean abundance (Age 1+)	± 1 standard deviation	Coefficient of variation (%)
5	466.60	19.93	4.27
6	478.67	15.73	3.28
7	475.57	19.95	4.19
8	467.00	21.75	4.66
9	474.22	18.99	4.00
10	470.80	20.92	4.44

3.5.4 Cloud computing

RONIN cloud computing (https://ronin.cloud/) accessed through the University of Sheffield was used to build six cloud computers of varying sizes in order to run simulations. Chapter 4 simulations were run large virtual computers (minimum 96 RAM, 48 CPUs) with approximately 100 – 120 experiments for single-stress taking between 1 – 3 days to run and 200 – 600 experiments for combined stress taking between 4 – 11 days to run. Chapter 5 simulations were run on even larger virtual computers (192 RAM, 96 CPUs) but restricted to 10 – 20 experiments each taking between 1 – 3 days to run. NetLogo, by default, restricts the amount of memory Java can access to 1024 MB. The Netlogo.cfg file was modified to accommodate simulations on these larger computers by changing the -xmx memory to allow Java to use up to 96GB of memory. Output files were large with a total of 2.7TB of data stored from all simulations but this was dependent on the spatial scale as reach simulations BehaviorSpace files were under 500 MB whilst network-scale simulations were closer to 1GB with the additional 1 – 2GB for events output files for each replicate and experiment.

3.6 Model post-processing: Ecosystem service delivery assessment

3.6.1 Angler satisfaction

As discussed in Chapter 1, Section 1.6.1, an alternative approach to economic valuation is to develop a relative method based on factors known to affect angler satisfaction (Hutt & Jackson, 2008; Beardmore et al., 2015; Birdsong et al., 2021, 2022; Gundelund et al., 2022). Angler satisfaction is a measure of the utility of a recreational fishery through satisfaction or happiness as a reward that anglers receive (Kosenius et al., 2013; Birdsong et al., 2021). Angler satisfaction is influenced by catch-related variables such as catch success, catch rate, and abundance (Birdsong et al., 2021) as well as non-catch-related variables (e.g. aesthetics, crowding, etc.). Angler satisfaction is not usually related to the number of fish caught but instead the interactions with fish such as number of bites (Hutt &

Jackson, 2008), therefore, the likelihood of fish interactions increases with increasing angling stock abundance. In the case of trophy hunters in a catch and release fishery, modelled in this thesis, they are more concerned about the size of fish caught rather than the number of fish caught (Arostegui et al., 2021). This means that angler satisfaction is positively influenced by trout abundance as well as catch success but negatively influenced by the presence of other anglers as this leads to more competition for resources (Pitman et al., 2019). Also, satisfaction is a good measure of the success of a recreational fishery (Beardmore et al., 2015).

To develop an approach to assess angler satisfaction, indices such as the recreational fishery index (Lomnicky et al., 2021) and the sport fishing index (Hickman, 2000) were examined. The recreational fishery index only considers catch-related aspects of species, length, and abundance of game fish and finds the economic value for the recreational fishery (Lomnicky et al., 2021). The sport fishing index calculates both the quantified angler success from angler catch rates and quality based on species population quality such as size structure and angling pressure (Hickman, 2000). However, these are still often related to the quality of a fishery from the perspective of fish in the fishery rather than the net benefit to anglers. Therefore, a more appropriate method of assessing angler satisfaction in a recreational fishery is by calculating both the net benefit and cost to anglers through parameters which influence satisfaction.

Pitman et al. (2019) carried out a survey of angler satisfaction in a catch and release fishery based on interviews, annual catch rate information, and angler effort. The findings of this study were used to propose a conceptual model of the links between angler satisfaction and influencing factors. One of the factors important to anglers was catching fish with catch rates linked to trout abundance. Therefore, higher abundance of trout in the fishery has a positive relationship and increases angler satisfaction. Increased abundance also positively influenced angler effort, however an increase in angler effort can lead to crowding and lower angler satisfaction. A criticism to this connection is that crowding is also non-catch-related and associated with angler density at a site for competition for fishing spots, anglers nearby or seen, and the physical constraints of rivers (Birdsong et al., 2021). Therefore, it is important to consider crowding independently from catch and instead as a result of angler density at the site (Beardmore et al., 2015). Crowding was assessed in Pitman et al. (2019) both by daily angler visits and by interviewing anglers on the number of anglers seen. However, other methods for assessing crowding number of anglers nearby or seen, or competition for fishing spots or visitor number (Birdsong et al., 2021). Ultimately angler effort is the cost for the angler and has a negative relationship with satisfaction and catch rate is the benefit and has a positive relationship with satisfaction. Fisheries regulations both increase or decrease access and angler effort as management can restrict access and decrease satisfaction as well as provide more fishing licenses and increase satisfaction.

Birdsong et al. (2021) conducted a meta-analysis to identify the main components which have strong effects on angler satisfaction. These were categorised as either catch-related or non-catch-related. The most important catch-related components for angler satisfaction was catch rate, size of fish caught, and the fish harvest. Catch rate and fish harvest are both related to the abundance of fish in the fishery, whilst the size of fish caught can be categorised by the angling stock. The most important non-catch-related components were space or access to fishing sites and crowding (Birdsong et al., 2021; Lomnicky et al., 2021). Access and crowding can both negatively or positively influence satisfaction, such as the ability to reach the water can increase satisfaction and limited access with overgrown areas can decrease satisfaction. Crowding can increase competition and decrease satisfaction but can also be reduced if enough fishing sites are available and end up increasing satisfaction. Furthermore, travel is important but often associated with travel cost (Pitman et al., 2019; Birdsong et al., 2021). Instead, travel distance is an alternative metric which was not considered in these two studies (Post et al., 2008; Arostegui et al., 2021). Travel distance has been seen to be negative in noncommitted anglers but positive for committed anglers, for example those willing to travel to catch a particular species of fish (Beardmore et al., 2015).

Therefore, to incorporate the findings of these two studies together, I have adapted the conceptual model developed by Pitman et al. (2019) to include the additional factors suggested in Birdsong et al. (2021): crowding, travel distance, and angling stock (Figure 3.15). All additional factors have the ability to both increase and decrease the angler effort. Coupling the conceptual model with these additional factors allows calculation of angler effort from travel distance and crowding and benefit through catch rate metrics (i.e. the probability of catching large fish and the total fish abundance) to identify where there are impacts to angler satisfaction. The main comparison to determine satisfaction is between the net cost from angler effort versus the net benefit from catch rate. The catch itself had been found as a strong driver of satisfaction, with the size of the largest fish caught critical to catch satisfaction (Birdsong et al., 2022; Gundelund et al., 2022). But for the purpose of this research and to identify the benefit to the anglers, satisfaction can be related to changes in both angler effort and angler catch success to identify an increase or decrease in satisfaction in a recreational fishery under chemical stress. Although important to consider, fisheries regulation is not within the scope of this research as alternative scenarios will not be examined.

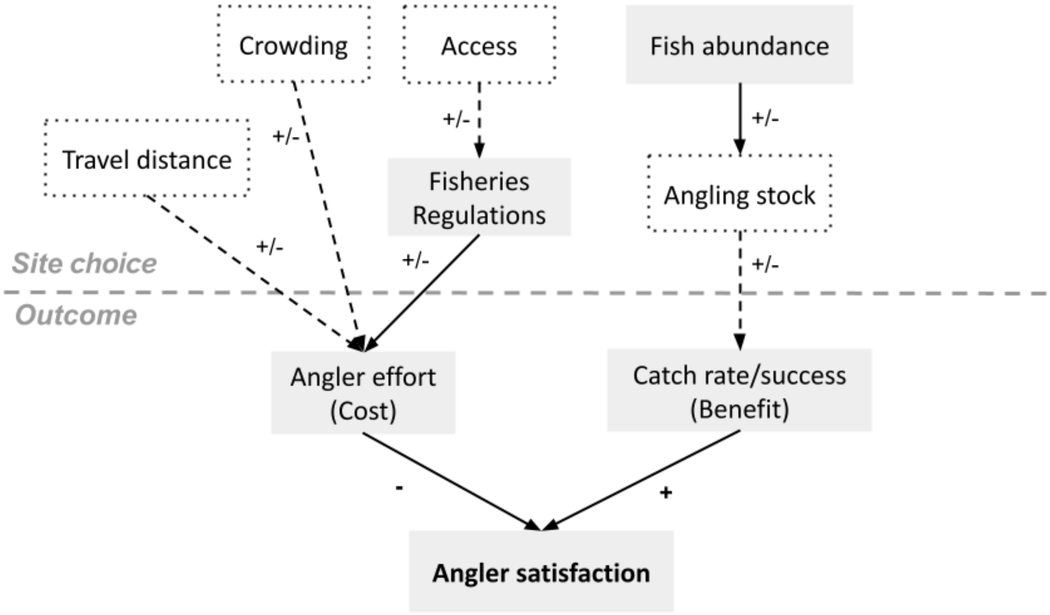


Figure 3.15 Angler satisfaction flowchart linking the positive and negative relationships between catch rate and angler effort adapted from a conceptual model developed for a rainbow trout fishery (Pitman et al., 2019). The grey boxes and solid lines represent the original conceptual model whilst the dotted boxes and lines are added elements. Positive (+) and negative (-) relationships are shown with arrows where they impact but both are also possible depending on how they are impacted (+/-). Access is considered by fisheries regulations and management (access to site, restrictions). Travel and crowding or competition for sites only important on the landscape-scale and added based on (Birdsong et al., 2021)*. Specifying site choice and outcome were also added to the original conceptual model.*

However, not all factors for angler satisfaction are relevant across spatial scales (Figure 3.15) and assessment should be tailored to the local and regional experience for anglers (Food and Agriculture Organization of the United Nations, 2012). On the reach-scale, there is an assumption that access and crowding are homogenous and only catch-related metrics are important as the reach represents one fishing site. A reach is considered a segment of the river where environmental conditions are the same where local factors are more important at the reach-scale than catchment factors (Lomnicky et al., 2021). Population density of anglers in the wider landscape and travel distance are not important as there is no site choice. Therefore, on a reach-scale, angler satisfaction is as described in Figure 3.16 where no cost is included to the angler. The impact on angler satisfaction is considered relative to a baseline (or in the absence of chemical stress) scenario.

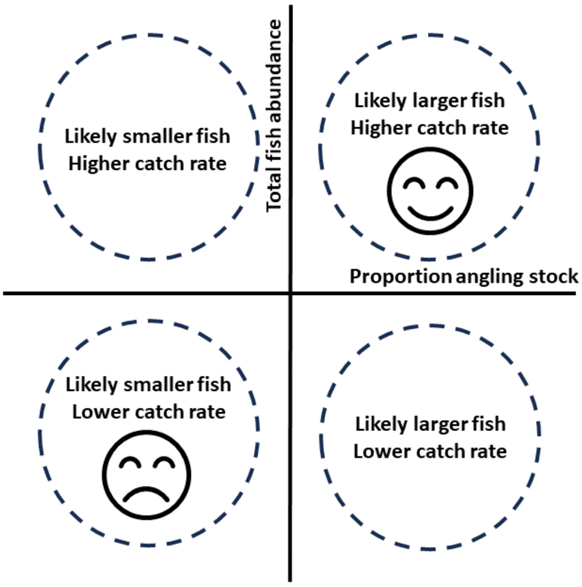


Figure 3.16 Angler satisfaction for a single reach scenario based on the proportion of large adult fish in the river reach (angling stock/total abundance) versus the total fish abundance to show the likelihood of catching a large fish and number of possible interactions. The four quadrants indicate the possible outcomes in satisfaction. The top-right and bottom-left quadrants show either improvement or decrease in satisfaction, respectively. The top-left and bottom-right are particular cases where satisfaction might not be clear. Where the axes cross indicates where the baseline (no effect) would be and the lines separate the four quadrants along this baseline condition.

On the network-scale, the wider landscape relates to the full flowchart including travel distances and population density of anglers (Figure 3.15). Therefore, both angler effort related to non-catch-related factors represent the cost and catch rate related to catch-related factors represents the benefit. Benefit is therefore a function of both angling stock and fish abundance. To capture the likelihood of catching a large fish, the fraction of angling stock from the total fish abundance was calculated (P). However, anglers receive satisfaction not only for catching a large fish, but for catching any fish. This probability of a large catch was then added to the total abundance of fish in the fishery (N). The benefit to an angler is, therefore, a sum of the likelihood of catching a large fish and the total number of fish. The cost to anglers is a sum of the total distance (T) needed to travel to each respective reach within the river network and angler population density (D) as a reflection of crowding (i.e. competition for nearby sites) in the wider landscape. All variables were then scaled to be between 0 – 1 based on the minimum and maximum values for each variable within the landscape. For benefit variables, minimum and maximum values were selected from the 17-year simulation period for the reach. Scaling for the angler satisfaction was based on minimum and maximum values as shown in Equation 3.10

$$\frac{x-min}{max-min} \qquad \text{Equation 3.10 (Rullens et al., 2022)}$$

where x is the value that is being scaled, min is the minimum, and max is the maximum value for each variable. In addition to rescaling each parameter, both elements of catch (benefit) and non-catch (cost) were rescaled to a scale of 0 – 1 by dividing in half assuming that each variable has equal weighting. To determine the satisfaction across the landscape an arbitrary 10 x 10 grid was overlaid onto the river network. The median benefit – cost value for each grid cell in the landscape is summarised in the following Equation 3.11

$$\frac{P_{(0-1)} + N_{(0-1)}}{2} - \frac{T_{(0-1)} + D_{(0-1)}}{2} \qquad \text{Equation 3.11}$$

where $P_{(0-1)}$ is the rescaled probability or likelihood of catching a large fish (number of large fish divided by the total fish abundance), $N_{(0-1)}$ is the rescaled total fish abundance, $T_{(0-1)}$ is the rescaled travel distance to reaches, and $D_{(0-1)}$ is the rescaled angler population density. The change in both cost and benefit is considered relative to baseline or a no chemical stress scenario.

3.7 Conceptual model of thesis

The subsequent data chapters of this thesis (Chapters 4 and 5) are structured as shown in the conceptual model (Figure 3.17). This conceptual model indicates where the ecological model (inSTREAM) adaptations (Section 3.4) have been applied to assess chemical stress on life history parameters and how the inputs and outputs of the model link chemical stress to the output of angling stock abundance. InSTREAM will be utilised to assess both at the reach and network-scales what influence life history parameters have on the subsequent output of angling stock abundance. Angling stock impact is then coupled with the ecosystem service assessment on angler satisfaction (Section 3.6) to determine the impact to the angler (beneficiaries). The differences between the reach-scale (Chapter 4) and network-scale angling stock assessments (Chapter 5) are also highlighted to show that population dynamics are important on the reach-scale versus a hybrid of interacting and patchy population dynamics on the network-scale (Section 3.4.7). Subsequent analysis with angler satisfaction also highlights that only catch-related benefits are relevant on the reach-scale (Chapter 4) whilst non-catch costs are also important on the network-scale (Chapter 5).

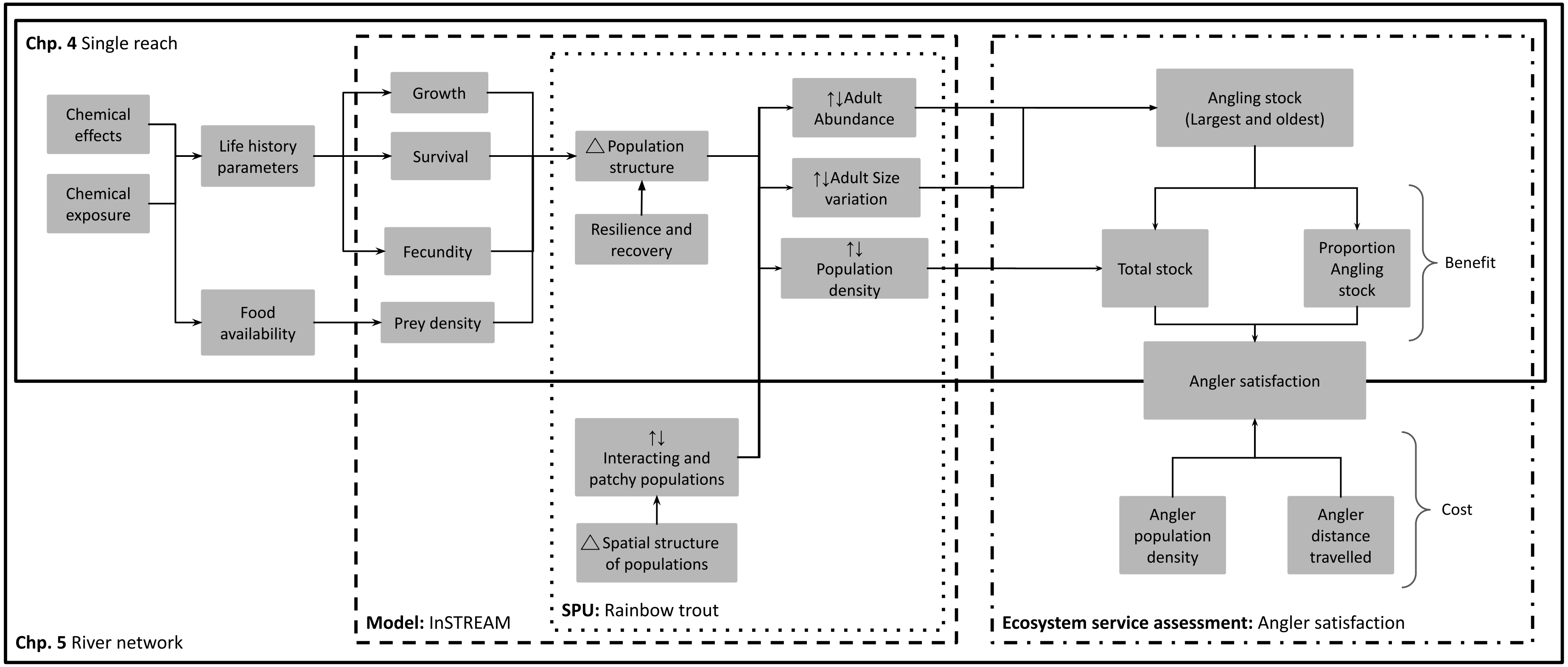


Figure 3.17. Conceptual model of thesis chapters on single reaches (Chapter 4) and a river network (catchment) (Chapter 5) assessment of ecosystem service delivery in a recreational fishery. The dotted boxes show extra information such where inSTREAM is applied and its inputs and outputs, the service providing units or SPUs of the services (rainbow trout) both population and interacting population, and the ecosystem service assessment with angler satisfaction. The solid lined boxes indicate the relevance of each component for either Chapter 4 (reach) or Chapter 5 (network).

3.8 Summary

Overall, this chapter has taken the steps to justify, adapt, and compose a method to assess from chemical effects to angler benefits in a recreational fishery. Coupling an individual-based model to assess impacts on the SPU with an ecosystem service assessment of angler satisfaction both at the reach and network-scale can be applied to address spatial patterns in a recreational fishery. The different elements were highlighted for relevance in each chapter both in terms of spatial patterns of angling stock and angler satisfaction. The methods introduced here will be applied on a reach-scale assessment in Chapter 4 and a network-scale assessment in Chapter 5.

Chapter 4

Chemical stress and angler satisfaction within a river reach recreational fishery

4.1 Introduction

Freshwater recreational fisheries provide a cultural ecosystem service to hundreds of millions of anglers globally and it is estimated that 1 in 10 people practises the sport within developed countries (FAO, 2012; Arlinghaus et al., 2015; Birdsong et al., 2021). Recreational fisheries are considered the dominant use of wild freshwater fish stocks in these countries (Ayllón et al., 2018) and are categorised as either game (i.e. salmonids) or coarse (i.e. non-salmonids) fish fisheries (Winfield, 2016). For a successful recreational fishery, anglers rely on a stable and productive fish population to catch and the service providing unit (SPU, *sensu* Luck et al., 2009) is often regarded as the stock of the target fish. Current fisheries practice is to manage stocks based on estimates of the size, age, and health of fish (O'Higgins et al., 2020; Arostegui et al., 2021). These are useful metrics if the interest is the abundance and biomass of the recreational fishery. However, these measurements do not consider the ecological and social components of recreational fisheries (Arlinghaus et al., 2017).

Recreational fishing or angling is defined as fishing for pleasure (Hughes, 2015) to the benefit of anglers and therefore recreational fisheries have a social significance beyond just an economic value (Liu et al., 2019). There is a complex interaction between the anglers and the freshwater ecological system and stock population within a recreational fishery (Mee et al., 2016). Failing to consider this interaction leads to a disconnect between results from the ecological perspective (investigating the abundance and changes in fish population) and the impact on the anglers themselves. Managing recreational fisheries as a socioecological system relies on not only understanding the dynamics of the resource, or SPU, but also on understanding its users, in this case the anglers (Camp et al., 2018).

Recreational anglers can have different goals when fishing such as catch-oriented anglers are focused on the species type, number of catch, or size of individual catch (Arostegui et al., 2021). For example, trophy anglers wish to catch the largest individuals of a specific species of fish as a trophy (Beardmore et al., 2015). The goals of anglers will determine which aspects of the recreational fishery populations will be valued. For catch-related goals, catch and harvest are important aspects related to the abundance of fish caught or stock size (Birdsong et al., 2021). Other catch-related goals can be the size of the fish caught. The portion of the stock which is of value to the anglers' goals is sometimes referred to as the catchable fish stock (e.g. Durance et al., 2016) but here it will be known as the angling stock to directly connect with anglers. However, the non-monetary value of the fishery is not only dependent on the presence of an angling stock, but also on the experience of its users.

The most common socioecological metric for assessing angler welfare or utility is angler satisfaction (Solomon et al., 2020; Arostegui et al., 2021). Angler satisfaction is the reward or benefit anglers receive from their fishing experience (Gundelund et al., 2022). Angler satisfaction can be measured as an outcome of both catch-related and non-catch-related aspects of angling. Catch-related aspects are those relating to the angling stock such as the rate of catch or size of angling stock whilst non-catch-related aspects concern the fishery itself such as travel distance, competition for sites, and aesthetics (Arostegui et al., 2021; Gundelund et al., 2022). Moreover, non-catch-related aspects are spatially dependent and do not vary on a reach-scale habitat as spatial structures and their uniformity is assumed for a reach. For example, fishery choice related to travel distance and aesthetics are consistent on reach-scale (Mee et al., 2016).

Freshwater recreational fisheries are under increasing pressure from a multitude of anthropogenic stressors including climate change, land use changes, and chemical pollution (Hamilton et al., 2016; Langhans et al., 2019). Chemical pollution has resulted in the loss of game fish from many rivers, including the loss of Atlantic salmon from UK rivers (Hamilton et al., 2016; Winfield, 2016) and large game fish have been found to exceed US EPA tissue limits for mercury (Hughes, 2015). Reducing chemical pollutants in rivers can result in the re-establishment of viable game and coarse fish populations (Winfield, 2016).

Wild fish are often exposed to sublethal concentrations in the wild and therefore identifying the effect of common ecotoxicity endpoints such as growth and reproduction is important to determine these effects on the population (Hamilton et al., 2016). However, current assessment of the risk of chemicals to freshwater fish requires standard toxicity tests on common life history parameters in individuals including growth, reproduction, and survival (Rudén et al., 2017). Individual-level effects are tested with fish acute toxicity tests (survival) and early life stages tests (including hatching, survival, and growth) such as the Fish Short-Term Reproductive Assay (OECD, 2012; Nusbaumer et al., 2021; Hazlerigg et al., 2023). However, data needs to then be extrapolated from individuals to higher levels of biological organisation to understand population-level effects (Forbes et al., 2008; Hamilton et al., 2016; Forbes et al., 2019). In the same way, scaling these to the population-level or even to higher levels such as ecosystem service delivery is difficult with current practices (Forbes et al., 2008).

Chemical effects on individual-level life history parameters can be extrapolated to population-level consequences using population models such as individual-based models (Spromberg & Meador, 2005; Forbes et al., 2008; Hazlerigg et al., 2014). This approach has been used to investigate the effects of endocrine disruptors on hypothetical trout and stickleback populations in hypothetical rivers (Forbes et al., 2019; Hazlerigg et al., 2023). These studies focused on the total fish abundance in a river reach (Hazlerigg et al., 2023) or mature adults in a small river network (Forbes et al., 2019). To date, these studies and similar studies have not considered the benefit to the end user through non-monetary means (i.e. angler satisfaction).

The exposure of recreational fisheries to chemicals varies in time and space. Chemical exposure may be continuous (e.g. pharmaceuticals from wastewaters), or episodic (e.g. pesticides from agricultural land) and ecosystem dynamics can remain altered even after the exposure has ended, causing legacy or carry-over effects (Schäfer et al., 2023). The persistence and stability of a fishery will depend on its ability to resist and recover from chemical exposure (Faber et al., 2021). Population dynamics such as resistance, the ability of populations to remain unchanged under environmental stress, is a function of the amount of a stressor needed to elicit a response and the speed at which the response occurs (i.e. faster changes and lower thresholds indicate low resistance) (Harvey & Railsback, 2011; EFSA Scientific Committee, 2016a; Villéger et al., 2017; Pesce et al., 2023). In the same way, the time pattern of recovery once the stressor has been removed, defines the resilience of a population (Spromberg & Birge, 2005; Luck et al., 2009; Schmutz & Sendzimir, 2018; Faber et al., 2021). In addition to the direct effects of chemicals on fish populations, operating through changes in individual life history parameters, indirect effects can happen through trophic interactions (e.g. reduced food availability due to chemical toxicity to prey species) (Ares, 2003). Studying both direct and indirect effects is important as stressors might not interact, classified as additive (Galic et al., 2018), or interact. Interacting stressors can either increase (synergy) or decrease (antagonistic) their combined impact on fish populations. If they decrease the interaction (antagonistic) this can be either buffering (i.e. one stressor masking the other) or suppression (i.e. one stressors not only masks the other but reverses it) (Tekin et al., 2020).

4.1.1 Aims

This study aims to investigate how the effects of chemical stressors on fish life history parameters (growth rate, fecundity, survival) and fish prey abundance affect angling stock and subsequent angler satisfaction.

This is addressed by:

1. Identifying how resistant angling stock is to chemical stress in a recreational fishery at a reach-scale;
2. Determining how resilient angling stock is to chemical stressors at a reach-scale.

3. Establishing how the combined effects of chemical stress on life history parameters and prey availability affect angling stock;
4. Exploring the implications of 1-3 for angler satisfaction.

The study uses an ecological individual-based model, inSTREAM, to simulate fish population dynamics in a recreational fishery. InSTREAM, or individual-based Stream Trout Research and Assessment Model, is a spatially explicit ecological model developed specifically for non-migratory salmonid species in fluvial environments (Railsback et al., 2022). InSTREAM has been previously used to assess the impact of chemical stressors (Forbes et al., 2019; Hazlerigg et al., 2023) and land use changes (Harvey & Railsback, 2011; Hajiesmaeili et al., 2022) on salmonid populations, but these studies did not consider a reach-scale assessment of angling stock or angler satisfaction. Salmonids such as rainbow trout are used as the target species as they are a standard test species for chemical risk assessment regulations (European Chemicals Agency, 2008) and are an important game fish (Serchuk et al., 1980). Angler satisfaction was developed based on a trophy angler. The SPU was defined to include only the largest and oldest fish and referred to as the angling stock.

4.2 Methods

4.2.1 Simulating chemical stress on trout life history parameters in inSTREAM

Modifications to inSTREAM

Simulation of a hypothetical trout population used inSTREAM version 7.3 (Railsback et al., 2022) hosted in NetLogo (Wilensky, 1999). InSTREAM was modified to include stress targeted at rainbow trout fecundity, growth rate, and survival as well as on prey availability. The fecundity sub-model was modified to include chemical stress on individual fecundity (i.e. number of viable eggs), growth rate sub-model for growth rate and prey density and a new survival sub-model was written. Simulation outputs were changed to record angling stock abundance (trout older than 1 year and larger than 20 cm). To test the resistance and resilience of angling stock, chemical exposure was added to inSTREAM inputs and the code modified to recognise chemical presence or absence within a reach. Two exposure scenarios, continuous and pulse, were used as input data to signify when stress was present in a reach over the simulation period. Two types of individual recovery were added to the model code: permanent inhibition and reversible inhibition. With permanent inhibition, the stress effects remain after exposure to stress has been removed. With reversible inhibition, the individual returns to pre-exposure conditions immediately after exposure is removed. Permanent inhibition was coded by tagging individuals within the reach who were exposed and applying inhibition on these individuals throughout the simulation period. Reversible inhibition was without the tagging. For combined stress simulations, switches were added to select more than one stressed variable (i.e. prey availability with life history parameters). For angler satisfaction, simulation outputs were adapted to count angling stock (defined as trout older than 1 year and larger than 20 cm). The simulations adopted the following assumptions: i) a catch and release fishery with no added mortality from harvest mortality; ii) chemical effects on life history parameters did not vary by age or size class; and iii) anglers are trophy anglers with the primary goal of catching a large fish. For detailed code changes to inSTREAM see Appendix 8.2.

Simulations

Four reaches, available in GIS Shapefile format from ArcGIS Pro (ESRI, 2021), were simulated (Figure 4.1). Each of the river reach sites has previously been used in individual-based modelling and was calibrated for size class data from census data in the case of sites A (Railsback et al., 2013, 2015) and B, C, D (Harvey & Railsback, 2011)

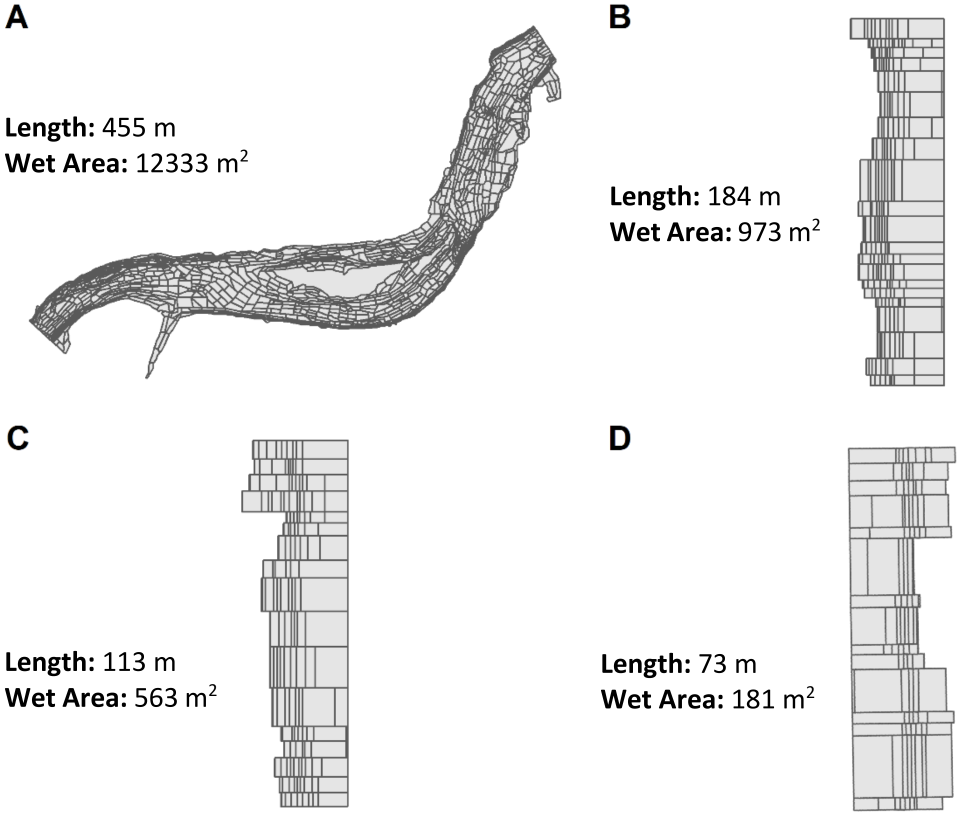


Figure 4.1 GIS Shapefiles with habitat cells; A) Reach A, downloadable with inSTREAM version 7.3, (Railsback et al., 2013, 2015) with delineated habitat cells based on hydrodynamic model mesh and habitat observations, B) Reach B, C) Reach C, and D) Reach D (Harvey & Railsback, 2011) are all from the same river network with habitat cells depicted as rectangular cells. All reaches are based on rivers in California, USA. Not to the same size scale.

Simulations started with an initial population of 1950 rainbow trout individuals in Reach A, 120 in Reach B, 97 in Reach C, and 36 in Reach D. A breakdown by age class for initial populations for reaches

A-D are in Table 3.1. All simulations were run as ten replicates with cloud computing via RONIN web-based management (https://ronin.cloud/) for a time period of 19 years, including a 2-year stabilisation period. A period of 19 years was selected from available historical data for the study sites where temperature and flow were relatively stable with no extreme events or in the case of Reach A, extended from available data. Large adult abundance was subset from the total rainbow trout population as those older than 1 and larger than 20 cm (based on minimum-length catch limits at recreational fishery sites (Ayllón et al., 2018)) and termed the angling stock of the reach. Figure 4.2 summarises the simulations in a conceptual flowchart using reach A as an example.

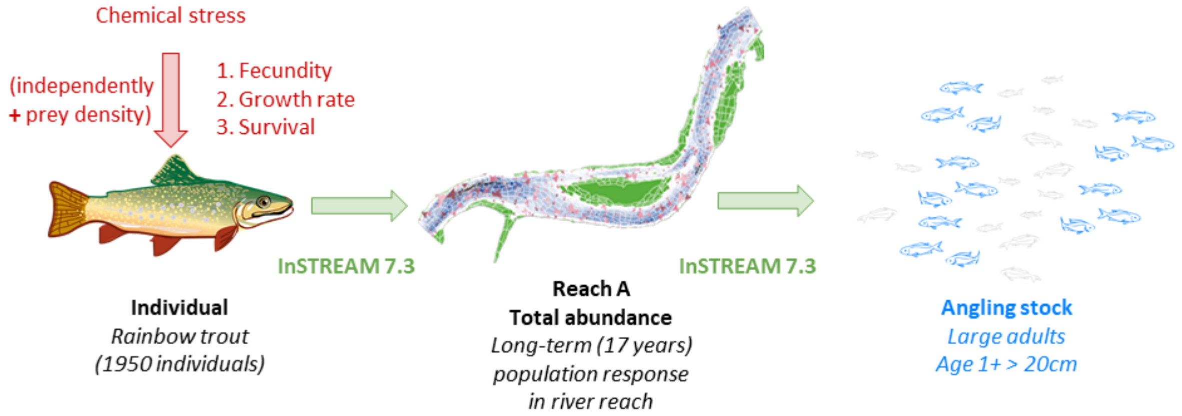


Figure 4.2. Conceptual flowchart of inSTREAM methodology where life history parameters are first simulated independently and then in combination with prey availability to determine the effect on the total abundance and then subset with inSTREAM for the angling stock (or trout older than 1 and larger than 20 cm in length) with reach A as an example.

Angling stock resistance

Angling stock resistance was tested by exposing the reach to continuous chemical stress for 17 years in inhibiting life history parameters (fecundity, growth rate, and survival) separately (Table 4.1) or in combination with inhibition of prey availability (Table 4.4). The inhibition of prey density and three life history parameters of fecundity (number of viable eggs), growth rate (whilst search and drift feeding), and survival (from chemical exposure), ranged from 0 (no inhibition) to 100% inhibition (Table 4.1).

Preliminary analyses (see Figure A4.4) were used to determine the sensitivity of the rainbow trout population to each stressed variable (Table 4.1). Continuous exposure effect thresholds were identified for each life history parameter and were defined as where 1) angling stock abundance diverges from baseline (i.e. less than 75% of data points have overlapping standard deviations over the simulation period) (lower effect threshold) and 2) angling stock collapses and goes extinct (upper effect threshold) (Table 4.2). These thresholds were used in subsequent simulations as the lower and upper bounds of chemical stress treatment levels, with intermediate inhibition levels selected to achieve an evenly spread response for pulse and combined stress simulations.

Table 4.1 Continuous chemical exposure simulations where effects are inhibiting life history parameters (fecundity, growth rate, survival) or an indirect effect on prey density from 0 – 100% in reaches A – D used for sensitivity analyses

Variable	**Inhibition level**
Fecundity	0%, 90%, 91%, 92%, 93%, 94%, 95%, 96%, 97%, 98%, 99%, 100%
Growth rate	0%, 10%, 20%, 30%, 40%, 50%, 60%, 65%, 70%, 80%, 90%, 100%
Survival	0%, 1%, 2%, 3%, 4%, 5%, 10%, 15%, 20%, 100%
Prey density	0%, 10%, 20%, 30%, 40%, 50%, 60%, 70%, 80%, 90%, 100%

Table 4.2 Upper and lower threshold bounds to which angling stock respond to inhibition and where they go extinct from inhibition across life history parameters in reach A (i.e. lower bounds are when 25% lie outside baseline and upper bounds are when angling stock goes extinct).

Life history parameter	**Lower**	**Upper**	**In between**
Fecundity	98%	100%	99%
Growth rate	10%	70%	30%, 50%
Survival	1%	20%	2%, 3%, 4%, 5%, 10%

Angling stock resilience

Angling stock resilience was investigated by exposing fish to chemical stress and inhibiting life history parameters or prey density for 6 years followed by 11 years (with 2 initial starting years pre-exposure to allow the population to stabilise) without stress to allow recovery to occur (Table 4.3). Recovery was described by both how quickly recovery occurs (if possible) and whether the angling stock recovered to baseline conditions. Individual recovery was simulated both with permanent impairment, where the exposed individuals retain the level of inhibition of life history parameters for their entire life, and reversible inhibition, where individuals immediately return to a no-stressed state following removal of stress (Caskenette & Koops, 2018). Recovery was quantified as the time it takes to recover or the number of years for angling stock abundance to reach within one standard deviation of baseline simulations.

Table 4.3 Pulse exposure simulations inhibiting a life history parameter (fecundity, growth rate, survival) or an indirect effect on prey density from 0 – 100% in reaches A – D

Variable	**Inhibition level**
Fecundity	0%, 95%, 96%, 97%, 98%, 99%, 100%
Growth rate	0%, 10%, 30%, 50%, 70%, 100%
Survival	0%, 1%, 2%, 3%, 4%, 5%, 10%, 15%, 20%, 100%
Prey density	0%, 10%, 20%, 40%, 50%, 100%

Combined stressors and response ratios

Inhibition of fecundity, growth rate, and survival were simulated in combination with inhibition of prey density (Table 4.4) and both angling stock resistance and resilience were investigated. Combined stress interactions were investigated using Reach A only.

Table 4.4 Combined stressor simulations inhibiting life history parameter (fecundity, growth rate, survival) and indirect effect in reach A (0% would mean a single stressor so not included)

Variable	**Inhibition level**
Prey density	10%, 20%, 50%, 100%
+ Fecundity	98%, 99%, 100%
+ Growth rate	10%, 30%, 50%, 70%
+ Survival	1%, 2%, 3%, 4%, 5%, 10%, 15%, 20%

Response ratios (R_x) for angling stock abundance were calculated for both single stressor and combined stressors response using Equation 4.1

$$R_x = \frac{W_x}{W_0} \qquad \text{Equation 4.1 (Galic et al., 2018)}$$

where W_x is the response with stress and W_0 is the baseline no effect. Response ratios were calculated from mean angling stock abundance from ten replicate runs. Response ratios were then used in the simple additive/multiplicative or Bliss independence model in Equation 4.2

$$D_A = R_{xy} - R_x R_y \qquad \text{Equation 4.2 (Morris et al., 2022)}$$

where D_A is the deviation from additivity R_{xy} is the combined stressor response ratio, R_x is a single stressor (x) response ratio and R_y is a single stressor (y) response ratio. D_A was then rescaled applying Tekin et al.'s (2020) flowchart for rescaling the deviation from additivity to determine the type of interaction (i.e. synergy, additive, antagonistic buffering, and antagonistic suppression). Rescaling was dependent on positive or negative values of D_A and minimum values or the lowest value from either single stress scenarios (Tekin, 2017; Tekin et al., 2020). For example, the method of rescaling was decided by if $D_A \leq 0$, $D_A > 0$ and $R_{xy} > \min(R_x, R_y)$, or $D_A > 0$ and $R_{xy} < \min(R_x, R_y)$ where min is the minimum values or lowest value from either single stress scenario. Rescaled D_A value was then used to identify the interactions type from the bounds shown in Figure 4.3.

Figure 4.3 Redrawn from Tekin et al.'s (2020) flowchart for classification of interaction types from rescaled deviation from additivity (DA) values and the bounds between interaction types.

4.2.2 Determining impact on angler satisfaction

The two factors that are important to angler satisfaction within a reach are changes in angling stock (adult fish older than 1 year and larger than 20 cm in length), in particular the abundance of large fish, and the changes to catch or the total abundance. A bivariate analysis was applied to identify changes in angler satisfaction (Figure 4.4).

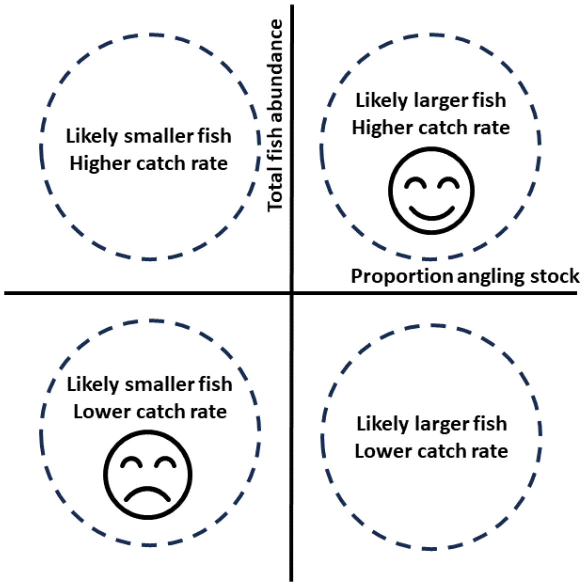


Figure 4.4 Angler satisfaction for a single reach scenario based on the proportion of large adult fish in the river reach (angling stock/total abundance) versus the total fish abundance to show the likelihood of catching a large fish. The four quadrants indicate the possible outcomes in satisfaction. The top-right and bottom-left quadrants show either improvement or decrease in satisfaction, respectively. The top-left and bottom-right are particular cases where satisfaction might not be clear. Where the axes cross indicates where the baseline (no effect) would be and the lines separate the four quadrants along this baseline condition.

Changes in angler satisfaction were determined by comparing angling stock and catch rate relative to a baseline (unstressed reach). Catch rate was assumed to be proportional to the total fish abundance and the likelihood of catching a large fish was assumed to be proportional to the relative abundance of large fish in the population. Relative change from baseline was calculated for both the catch rate (total fish abundance) and the likelihood of catching large fish (proportion of large fish) based on Equation 4.3

$$\frac{(new\ value - baseline)}{baseline} \qquad \text{Equation 4.3 (Döll et al., 2020)}$$

where $new\ value$ is the angler satisfaction under the chemical stress scenarios and $baseline$ is angler satisfaction in absence of chemical stress.

4.2.3 Data analysis

All analyses were performed in R version 4.0.3 (R Core Team, 2024) in R Studio (RStudio Team, 2020) with R packages specific for data manipulation and data visualiation. Mean and ± one standard deviation of angling stock abundance from ten replicate simulations was calculated for the yearly census day (30 September 20XX) or end of the simulations and compared to control abundance of no inhibition in life history parameters. The magnitude of differences between simulations were used to interpret the biological significance of simulation outputs for population dynamics.

4.3 Results

4.3.1 Rainbow trout population density in the absence of chemical stress

All four reaches were able to sustain a rainbow trout population across the whole 17-year simulation (excluding the initial 2 years pre-exposure for the population to stabilise) period with mean population densities up to 3 trout/m^2 (Figure 4.5). Reaches A and D had the lowest population density of (< 1 trout/m^2) and Reach B had the highest population density (2 – 3 trout/m^2). Population density was independent of the reach size, with both the largest and smallest reaches, D and A, respectively, had the same population density. Total trout abundance was reflective of the reach size where the smaller the reach, the less fish (Appendix Figure A4.1).

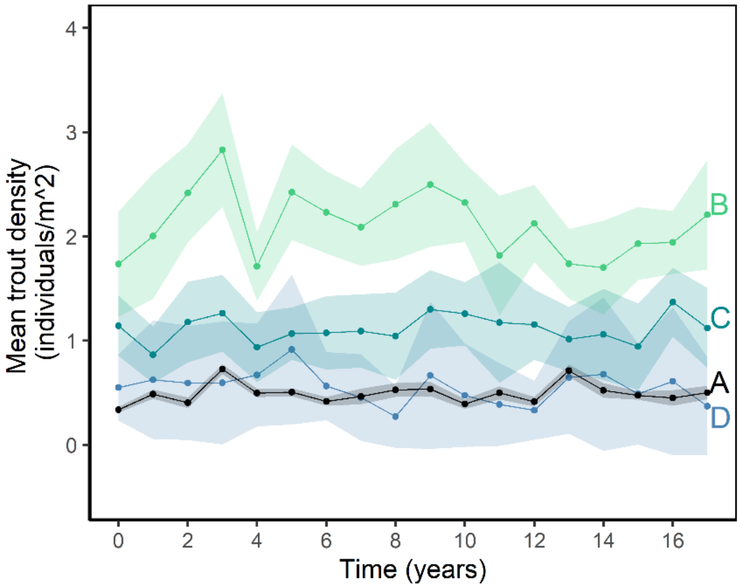


Figure 4.5 Mean trout density (individuals/m^2) from ten replicate simulations of rainbow trout in reaches A – D (labelled). The population density is plotted across a 17-year simulation period (excluding the initial 2 years pre-exposure to allow the population to stabilise) with the value on a census day (September 30th). The shaded ribbon signifies +/- one standard deviation from ten replicate simulations.

4.3.2 Resistance of angling stock population to chemical stress

In the absence of stress, reaches A and B sustained a stable angling stock population (Figure 4.6), whilst reaches C and D did not (Appendix Figure A4.2 and Figure A4.3). This was either because trout were not large enough (reach D, Appendix Figure A4.3) or the mean angling stock abundance was less than five individuals and considered too small to discern patterns in population dynamics (reach C, Appendix Figure A4.2). Reaches C and D were therefore not used in subsequent analyses as only reach A and B are suitable habitats for larger-sized trout.

The time taken for stress effects to be observed on angling stock abundance was four (reach A, Figure 4.6A) or five (reach B, Figure 4.6D) years for fecundity and less than one year for growth rate (Figure 4.6B, E) and survival (Figure 4.6C, F) for both reaches (A, B).

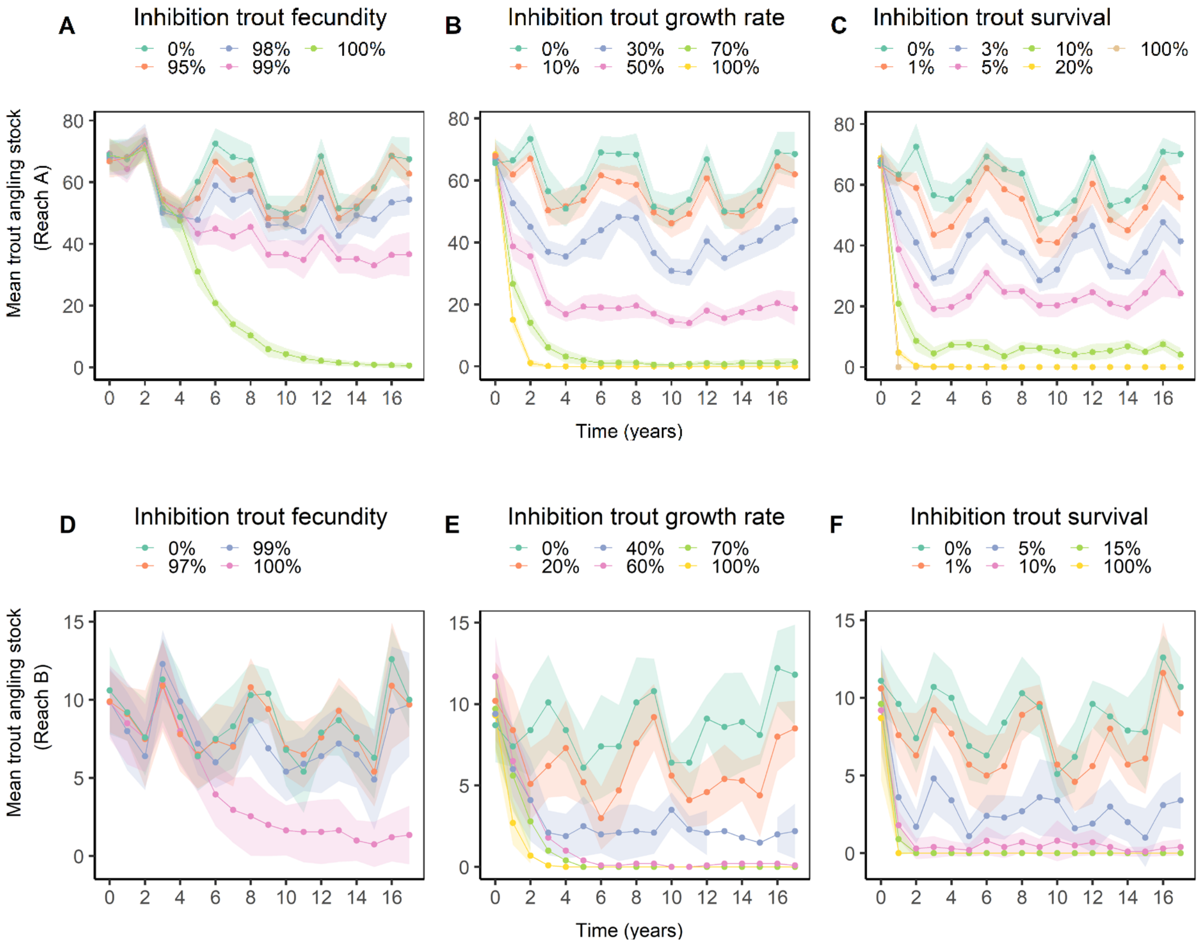


Figure 4.6 A – C) Reach A and D – F) Reach B. Mean trout angling stock (trout older than 1 year and greater than 20 cm in length) abundance from ten replicate simulations on a census day over the simulated chemical exposure period of 17 years, as well as pre-exposure census at year 0. Trout life history parameters were inhibited from 0 – 100% for A, D) fecundity, B, E) growth rate, and C, F) survival. The shaded area depicts +/- one standard deviation from ten replicates. Threshold values of where the stock starts to diverge from baseline and goes extinct as well as selected intermediate values are only shown here. See Appendix Figure A4.4 for full results.

The angling stock population in both reaches were highly resistant to inhibition of fecundity. No responses in the angling stock population were observed until 98% (reach A) or 100% (reach B) inhibition of fecundity (Figure 4.6A, D). At 99% inhibition, angling stock abundance in reach A declined but then stabilised around a mean of 40, indicating that abundance is still at sustainable levels. Angling stock populations were more sensitive to inhibition of growth rate (Figure 4.6B, E) and survival (Figure 4.6C, F). Growth rate inhibition caused reductions in angling stock abundance but populations were able to persist up to 60% (reach B) or 70% (reach A). In contrast, survival inhibition at 1% leads to a decline in angling stock abundance, with 20% inhibition causing angling stock collapse in reach A (Figure 4.6C) and between 10-15% inhibition causing angling stock collapse in reach B (Figure 4.6F).

There was a clear difference in sensitivity of mean angling stock abundance in response to inhibition of the different life history parameters. For both reaches (A, B) fish abundance was most resistant to inhibition of fecundity, followed by growth rate, then survival. Mean angling stock abundance at the end of the simulation period is presented for different treatments in Figure 4.7.

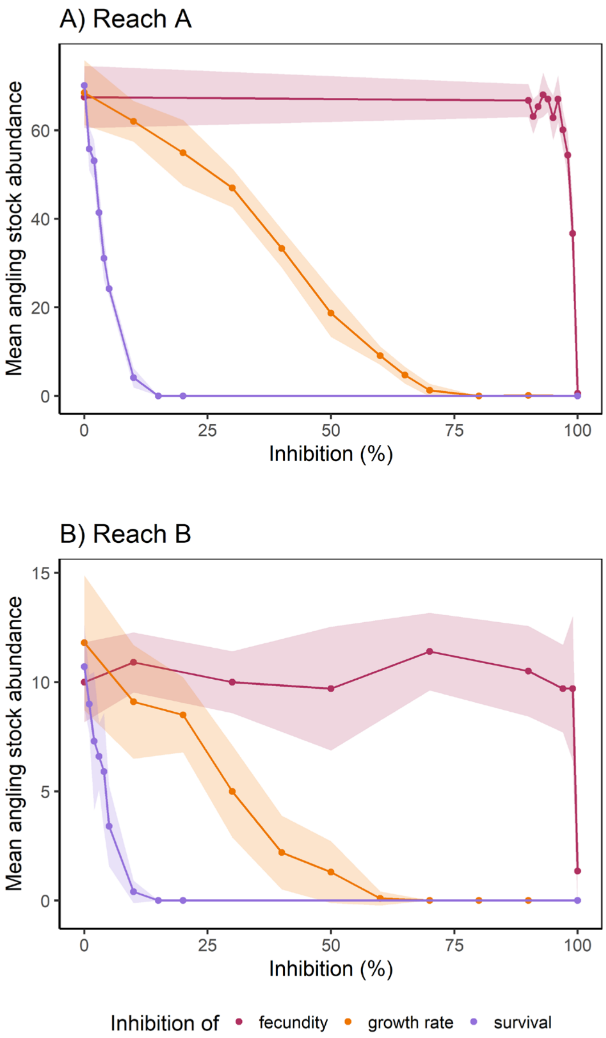


Figure 4.7 A) Reach A B) Reach B. Mean angling stock abundance (number of individuals older than 1 year and larger than 20 cm in length) at the end of a 17-year simulation period from ten replicates. Shading depicts +/- one standard deviation from ten replicates. Inhibition of fecundity (maroon), growth rate (orange), and survival (purple) from 0 to 100%.

Reach A and B were similar to levels of inhibition for both 10% and 50% reduction of angling stock as summarised in Table 4.5. Except the response of rainbow trout in reach B to inhibition of life-history parameters was more variable than for fish in reach A (Figure 4.7). There was slightly more resistance to inhibition of fecundity, survival, and growth rate in reach B than reach A (Table 4.5).

Table 4.5 Inhibition of life history parameters which resulted in 10% and 50% reduction of angling stock in reaches A and B.

Inhibition in reach	**Angling stock reduction**	
	10%	50%
Reach A		
- Fecundity	97%	99%
- Growth rate	10%	40%
- Survival	< 1%	3 – 4%
Reach B		
- Fecundity	> 99%	99 – 100%
- Growth rate	10%	30%
- Survival	<1%	4 – 5%

4.3.3 Resilience and recovery of angling stock population under chemical stress

Regardless of the life history parameter inhibited, if some fish survived the exposure period, the angling stock populations were able to recover once chemical stress was removed from the simulation. The main differences were related to whether individual-level effects were permanent (Figure 4.8) or reversible (Figure 4.9). Under permanent inhibition of exposed individuals, the angling stock population was able to recover from 99% (reach A) or 100% (reach B) inhibition of fecundity (Figure 4. 8A, D). Under reversible inhibition, the angling stock was able to recover from any level of fecundity inhibition (Figure 4.9A, D). There was no difference in the ability of angling stock to recover from growth rate inhibition that was either permanent (Figure 4.8B, E) or reversible (Figure 4.9B, E). However, recovery of the angling stock from either 20% (reach A) and 15% (reach B) inhibition of survival was only possible when individual inhibition was reversible (Figure 4.8C, F). 100% inhibition of survival was the only case where recovery was not able to reach baseline conditions within the simulation period. In instances where no recovery occurred, no trout remained in the simulation.

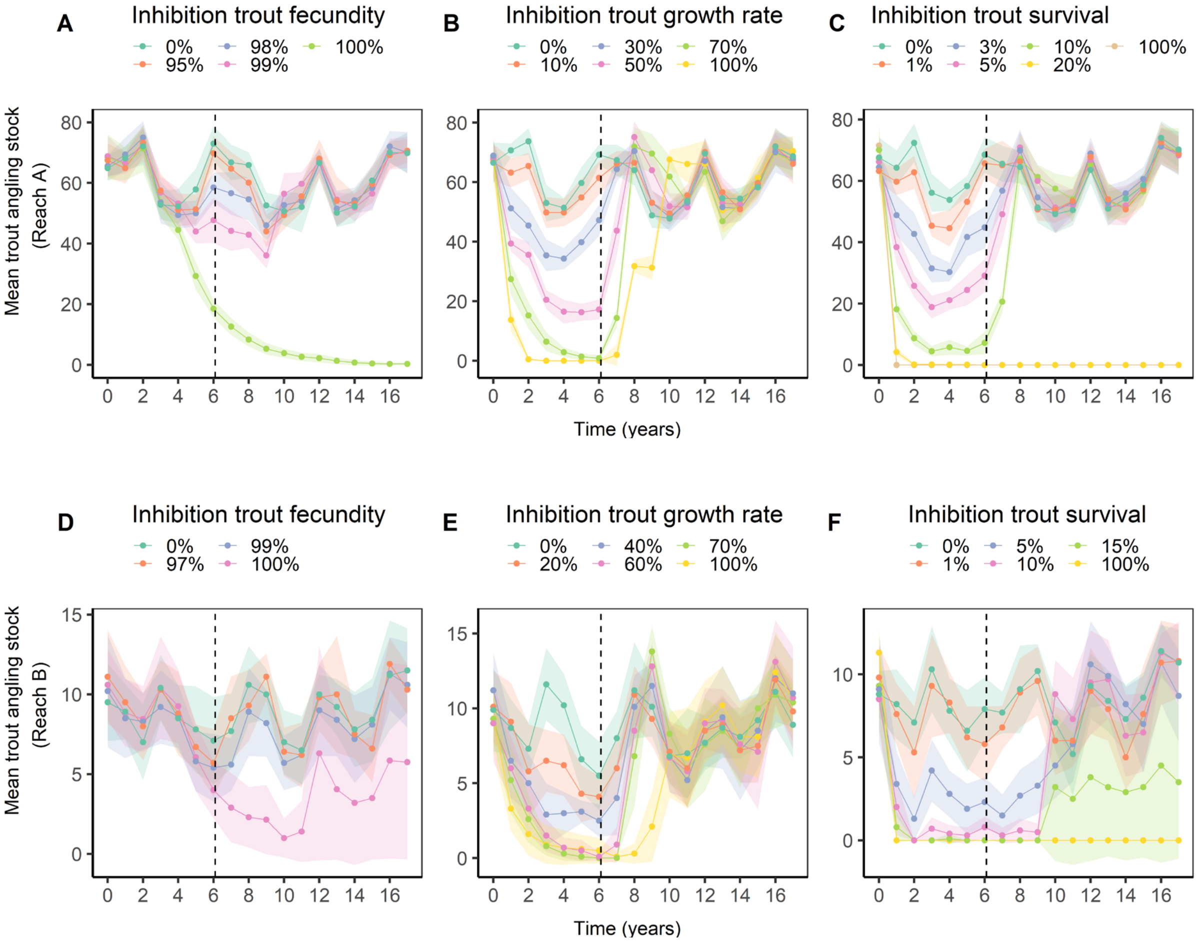


Figure 4.8 A – C) Reach A and D – F) Reach B. Mean trout angling stock (trout older than 1 year and greater than 20 cm in length) abundance from ten replicate simulations on a census day over the simulated pulse chemical exposure period of 6 years, after which the stock recovers, as well as pre-exposure census at year 0. Trout life history parameters were inhibited from 0 – 100% for A, D) fecundity, B, E) growth rate, and C, F) survival. Exposed trout are permanently inhibited for the remainder of their lives. The shaded area depicts +/- one standard deviation from ten replicates.

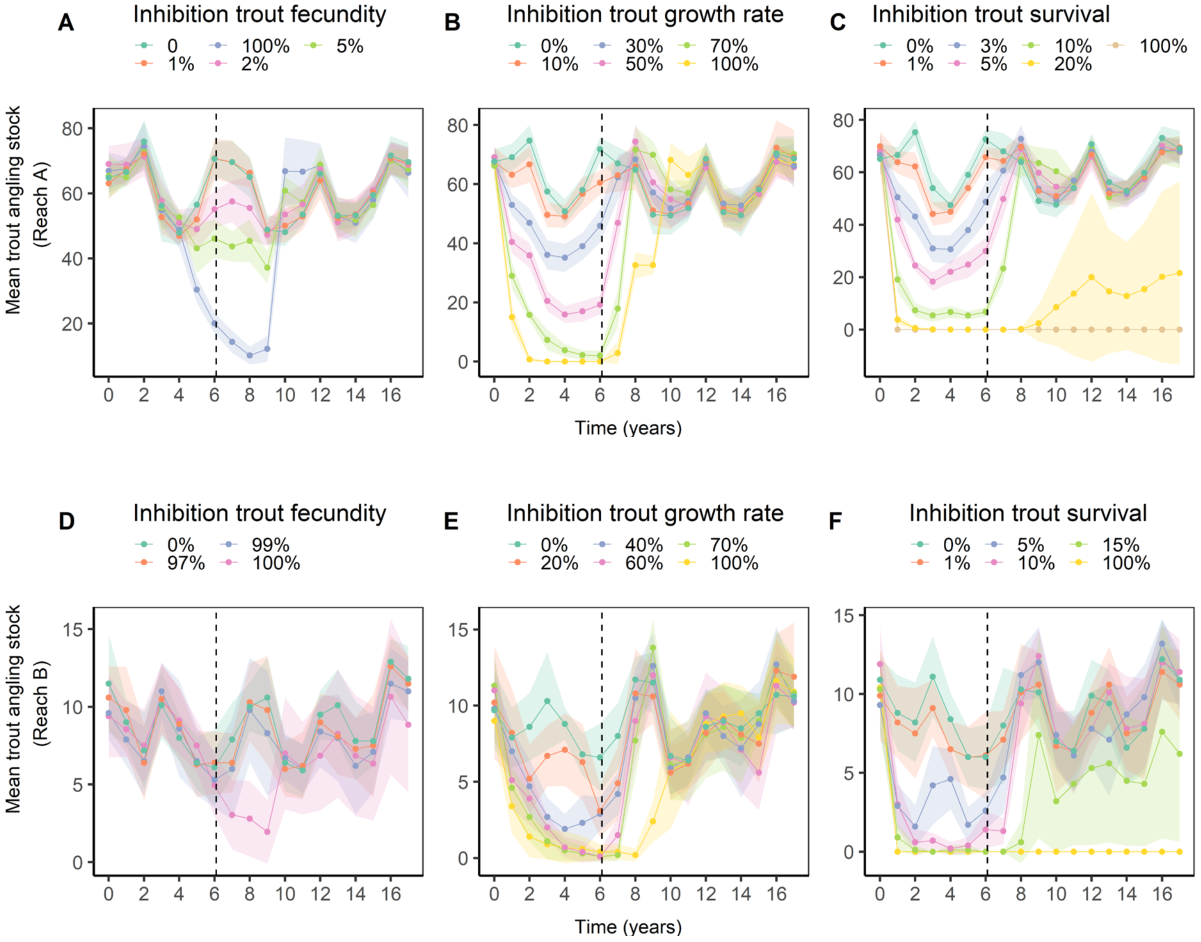


Figure 4. 9 A – C) Reach A and D – F) Reach B. Mean trout angling stock (trout older than 1 year and greater than 20 cm in length) abundance from ten replicate simulations on a census day over the simulated pulse chemical exposure period of 6 years, after which the stock recovers, as well as pre-exposure census at year 0. Trout life history parameters were inhibited from 0 – 100% for A, D) fecundity, B, E) growth rate, and C, F) survival. Exposed trout are reversibly inhibited and recover once exposure is removed. The shaded area depicts +/- one standard deviation from ten replicates.

The time to recovery varied with life history parameter inhibited and usually the higher the level of inhibition the longer the recovery time for both reaches (Figure 4.10). For both reaches (A, B), angling stock was the most resilient to inhibition of growth rate and fecundity as the population was able to recover within 6 (A) or 4 (B) years up to 100% inhibition of both parameters (Figure 4.10A, B). Angling stock was unable to recover above 15% inhibition of survival. Under permanent or reversible inhibition of individuals, overall recovery time decreased in both survival in both reaches and growth rate in reach A but not for fecundity where it remained the same. Overall, the time to recovery was about two years longer in reach A (Figure 4.10A) than in reach B (Figure 4.10B).

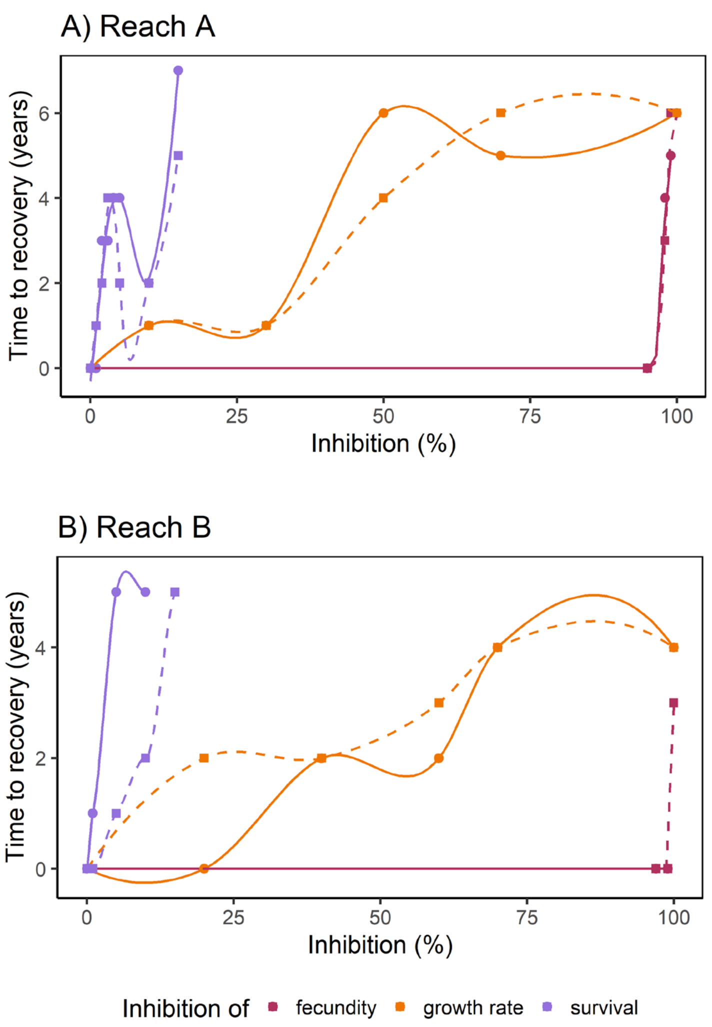


Figure 4.10 A) Reach A B) Reach B. Time to recover for inhibition of fecundity (maroon), growth rate (orange), and survival (purple) from 0 – 100%. Time to recover based on mean angling stock abundance (number of individuals older than 1 year and larger than 20 cm in length) at the end of a 17-year simulation period from ten replicates. Time to recover was determined when abundance reached within +/- one standard deviation of baseline conditions. Solid line and circle points indicates permanent individual recovery, dashed line and square points indicates reversible individual recovery. Smooth automated regressions have been fit to data, for fecundity these have been fit with the formula $y \sim e^x$.

4.3.4 Combined stress interactions: adding inhibition of prey availability

Angling stock was able to persist under 20% inhibition of prey density and recover from 50% inhibition of prey density. Recovery of angling stock was independent of whether the effect of prey inhibition on individual fish was permanent or reversible (cf. inhibition of growth rate) (Figure 4.11).

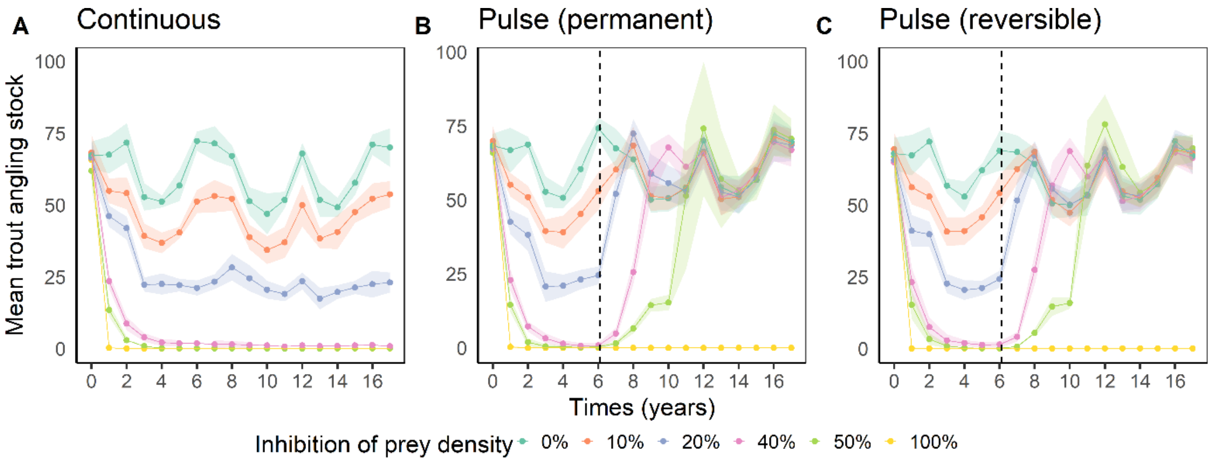


Figure 4.11 Mean trout angling stock (trout older than 1 year and greater than 20 cm in length) abundance from ten replicate simulations on a census day over the course of a 17-year simulation period in reach A. Trout prey density was inhibited from 0 (no effect) to 100% inhibition (no prey density). A) Trout individuals were exposed to continuous reduction of prey density for 17 years B) Trout individuals were exposed to a pulse of 6 years and exposed trout remained permanently inhibited following removal of stress and C) Trout individuals exposed to a pulse of 6 years after which effects were reversible once exposure removed. The shaded area depicts +/- one standard deviation from ten replicates.

Angling stock abundance was more sensitive to inhibition of prey density than growth rate but more resistant than inhibition of survival, based on the effect level at the end of the 17-year simulation period (Figure 4.12A). A 10% inhibition of prey density led to a 30% decrease in angling stock and a 50% inhibition resulted in a 100% decrease in angling stock. Time to recovery was only possibly up to 50% inhibition of prey density and was longer than any of the life history parameters with up to 8 years to recover (Figure 4.12B).

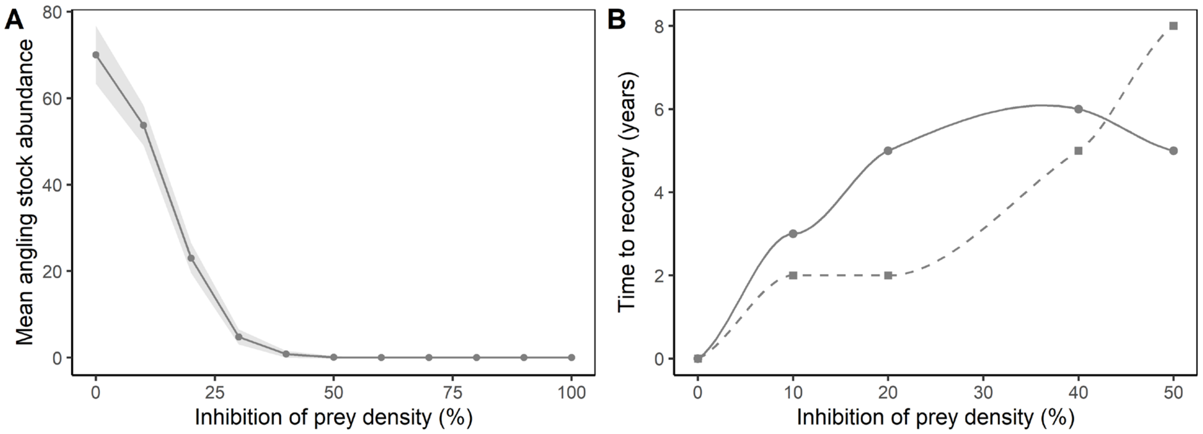


Figure 4.12 A) Mean angling stock abundance (number of individuals older than 1 year and larger than 20 cm in length) at the end of a 17-year simulation period from ten replicates. Shading depicts ± one standard deviation from ten replicates. Inhibition of prey density from 0 to 100%. B) Time to recovery for inhibition of prey density from 0 to 100%. Time to recover was determined when mean angling stock abundance was within ± one standard deviation of baseline conditions. Permanent (solid line and circle point) recovery in individuals and reversible (dashed and square point) recovery in individuals.

The outcomes of the combined stressor interactions, i.e. inhibition of prey density and either inhibition of fecundity, growth rate, or survival, are presented in Figure 4.13. No interaction (i.e. additive) was the most the most common outcome (11 out of 20 stress combinations), followed by antagonistic buffering, where the effect of the combined stress is less than expected (5 out of 20 stress combinations), and synergy, where the effect of the combined stress is greater than expected (3 out of 20 stress combinations). Antagonistic buffering occurred with > 20% prey density inhibition plus either 100% inhibition of fecundity (Figure 4.13A), 10% growth inhibition (Figure 4.13B), and 1% inhibition of survival (Figure 4.13C). Synergy occurred with higher levels of growth rate inhibition (> 50%) or survival inhibition (1%) and low (10%) to medium (20%) inhibition of prey density. Inhibition of survival at low levels (1%) with low inhibition of prey density (10%) interacted antagonistically with suppression as survival had a stronger effect on angling stock over prey density. This is further demonstrated as increasing prey density inhibition by 20% led to antagonistic buffering. Therefore, more inhibition of prey density was needed for the impacts to not be overshadowed by inhibition of survival.

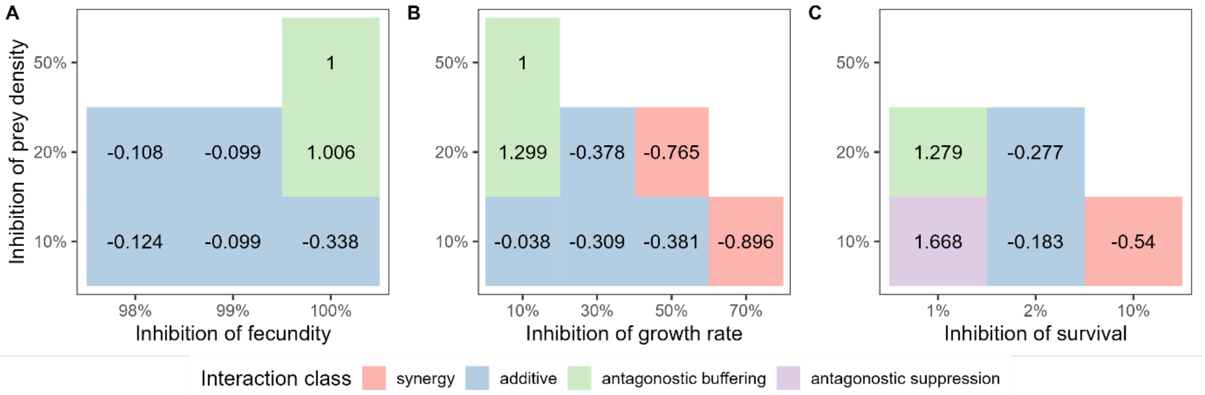


Figure 4.13 Heat map of the combined stress interaction class between inhibition of life history parameters and feeding through prey density inhibition with rescaled deviation from additivity values shown. Combined stressor analysis was only possible when angling stock abundance > 0 and those simulations where angling stock abundance < 0 are blank within the heat map. Angling stock abundance values were taken from the end of the 17-year period of continuous exposure.

4.3.5 Subsequent impacts of chemical stress on angler satisfaction

Angler satisfaction did not improve with any level of inhibition of life history parameters (Figure 4.14). Inhibition of growth rate and survival reduced the catch rate and increased the likelihood of catching smaller fish. Inhibition of fecundity increased the likelihood that larger fish would be caught but decreased the catch rate. So, although trophy anglers would have a higher chance of the fish they catch being large they would possibly need to fish for longer. Angler satisfaction did not fully recover to baseline satisfaction under any level of inhibition of life history parameters indicating permanent shifts in the angling stock population structure as a result of chemical stress exposure.

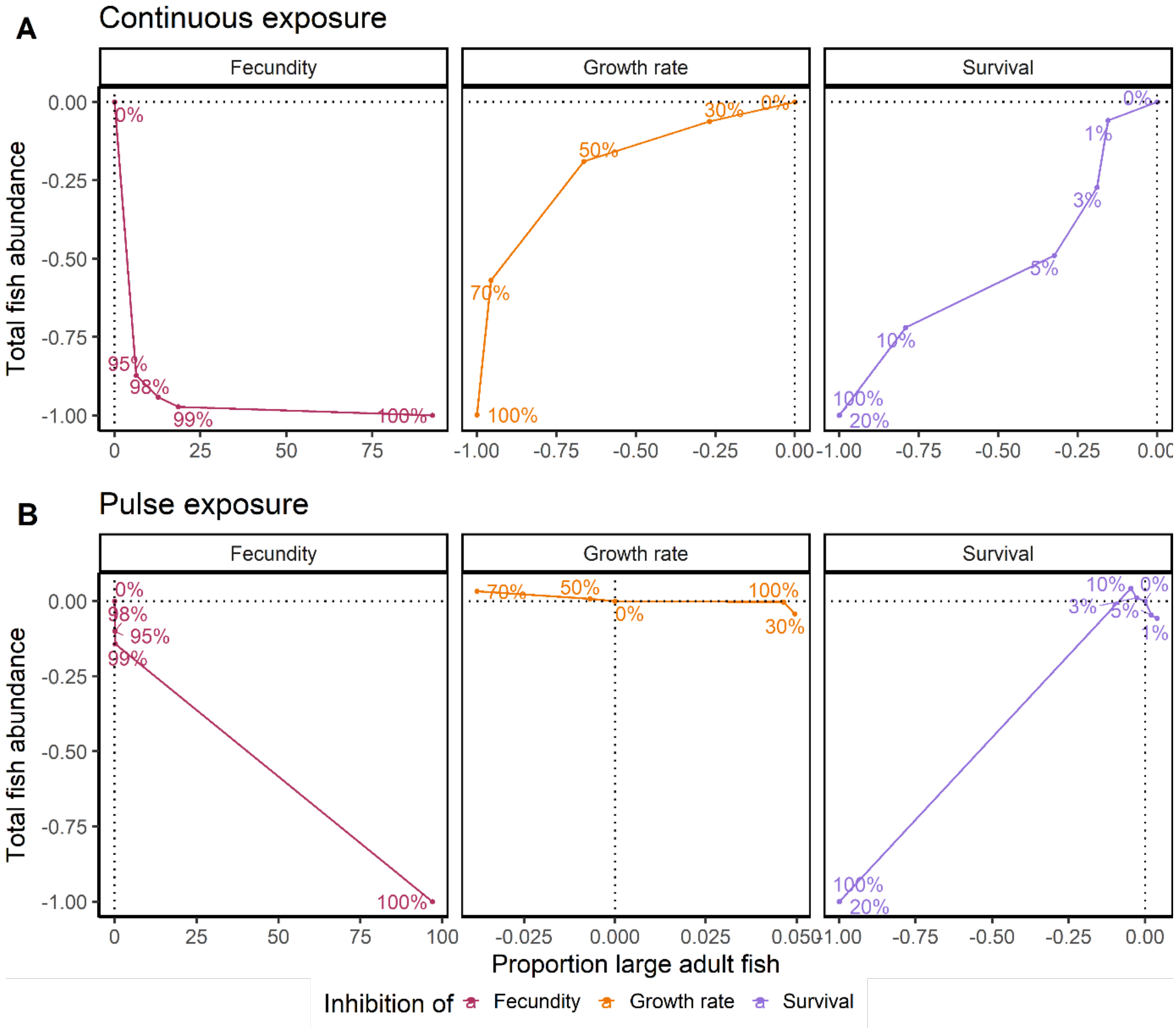


Figure 4.14 Reach site A angler satisfaction at the end of the 17-year simulation period. Angler satisfaction is based on the mean (ten replicates) total trout abundance within the reach and proportion of large fish within the site relative to baseline conditions. Trajectories show the direction of change with level of inhibition. A) Shows continuous exposure for inhibition of fecundity, growth rate, and survival B) shows pulse exposure (permanent inhibition in exposed individuals) for inhibition of fecundity, growth rate, and survival.

Angler satisfaction decreases under all combined stressor scenarios, with increasing prey density inhibition resulting in reduced catch rates (lower fish abundance) and reduced likelihood of catching large fish (lower proportion of large fish) (Figure 4.15).

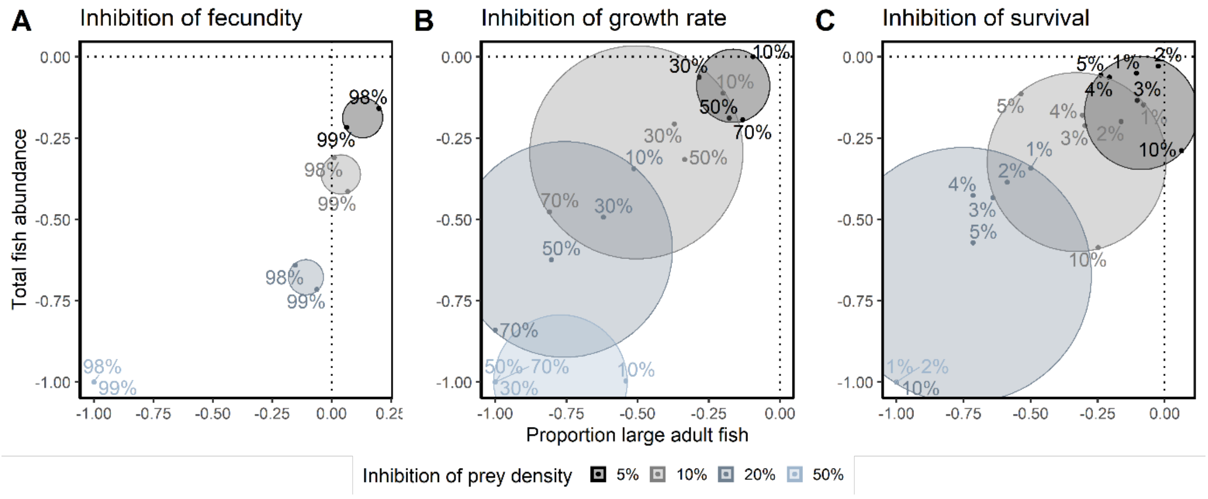


Figure 4.15 Inhibition of A) fecundity, B) growth rate, and C) survival. Reach site A angler satisfaction at the end of the 17-year simulation period under combined stress with inhibition of prey density. Angler satisfaction is based on the mean (ten replicates) total trout abundance within the reach and proportion of large fish within the site relative to baseline conditions. Results are from continuous exposure for inhibition of fecundity, growth rate, and survival in combination with prey density. Clusters annotate groupings of points for level of prey density inhibition.

4.4 Discussion

This study aimed to answer how the effects of chemical stressors on fish life history parameters (growth rate, fecundity, and survival) and fish prey abundance affect angling stock and angler satisfaction at the reach-scale. This was done by identifying how resistant and resilient angling stock was to chemical stressors, establishing the combined effects on life history with prey availability, and finding the subsequent implications for angler satisfaction. Population-level resilience and recovery of angling stock to chemical stress were dependent on the life history parameter with the highest resistance and resilience to fecundity and lowest to survival and combined stress mostly additive with no interaction. Angler satisfaction was seen to decrease with population-wide permanent shifts under all chemical stress scenarios.

4.4.1 Angling stock resistance to chemical stress in a recreational fishery on a reach-scale

Angling stock resistance to chemical stress depended on the life history parameter inhibited at the individual-level. This is in line with previous studies on life history impacts of fish populations (Spromberg & Birge, 2005; Carim et al., 2017; Vaugeois et al., 2020b) where population-level impacts were dependent on population dynamics and individual-level stressor response. Not only did angling stock exhibit population dynamics in response to chemical stress, but responses were dependent on the life history parameter. Angling stock was most resistant to inhibition of fecundity and most sensitive to inhibition of survival.

Major impairment of fecundity (i.e. 98%) had little impact on angling stock abundance. This contrasts with a similar study with inSTREAM which showed that the level of effect on fecundity combined with male spawning impacted adult trout abundance with increasing stress (Forbes et al., 2019). However, the level of impact was dependent on the trout species where greenback cutthroat trout was more sensitive than brown trout but high exposure to an endocrine disrupted caused only a 10% decrease in abundance. The difference in response could indicate that larger rainbow trout are more resistant to inhibition of fecundity than other trout species' adults and why angling stock abundance showed little response with very high inhibition was needed (99%) to reduce angling stock numbers. On a population-level study, with a sister model, inSTREAM-Gen (Ayllón et al., 2016), copper inhibition on fecundity had less impact on population-level effects due to density-dependence on trout within their first year albeit individual-level effects (Janssen et al., 2024). As a result of density-dependence a decrease in fecundity may not translate into population-level effects in the field (Hamilton et al., 2016). For endocrine disrupting chemicals, individual-level impacts have been observed, with low exposure impacting embryonic development and higher exposure leading to increased mortality (Schultz et al., 2003). However, extrapolating to population-level responses found that effects on individuals will lead to a decline in population abundance but it may take several generations of exposure until populations are substantially reduced (Hamilton et al., 2016).

Existing literature indicates that population-level impacts would be expected as a result of fecundity inhibition from chemical stress in rainbow trout (Accolla et al., 2019). However, this was not apparent in a subset of the population, the angling stock. Mechanisms of targeting fecundity in inSTREAM could explain these results. As fecundity was not dependent on size or life stage of rainbow trout, only rainbow trout of a reproductive age would be impacted. However, fecundity is usually body-size dependent, and within inSTREAM the number of eggs does depend on both the number of viable eggs a trout can produce and the length of the spawning individual where longer individuals produce more eggs (Galic et al., 2017; Railsback et al., 2022). Egg size has also been linked to pesticide tolerance, where small eggs are more tolerant than larger eggs (Nusbaumer et al., 2021), however, this mechanism is not built into inSTREAM. This means that angling stock, as the largest fish within the population, will produce more eggs than smaller fish when inhibited. As targeting fecundity did not inhibit the reproductive success of adult rainbow trout but instead the number of viable eggs to hatch, only the number of hatchlings entering the population was reduced. These numbers are not reflected when only looking at angling stock abundance. Also, spawning only occurred between two dates (1st April – 30th June) within inSTREAM, meaning inhibition of fecundity is only possible three months out of the entire year. This would explain why it took four years for any impacts to become visible in the angling stock population (Railsback et al., 2022)

Angling stock was most sensitive to inhibition of survival. This is in line with previous studies on mortality in other fish species exposed to pesticides causing an abrupt decline in population biomass in cyprinid fish (Accolla et al., 2022). Another study applying population modelling in a river population of rainbow trout in Canada (Caskenette & Koops, 2018) found that the population (growth) was most sensitive to the vital rate of survival but this was seen to decrease with the age of the fish. Although, this current study did not consider age or size-related impacts of survival and the individual response was the same, angling stock responded the same as would be expected for the entire population so the impacts are translatable. Survival was not modelled on the impact of survival of eggs. This reflects the findings of an empirical study which found that a pesticide has no significant effect on embryo survival at high exposure concentrations (Nusbaumer et al., 2021). Survival without chemical stress was influenced by different factors depending on the life stage of rainbow trout. Whilst spawning and hatching survival are generally considered constant, survival for juvenile to adult stages are often dependent on temperature, growth, and movement and for adult fish species additional fishing mortality (Hayes et al., 2009). Although fishing mortality was not modelled in inSTREAM, this reflects the choice to not include chemical stress on early-life stage fry but also shows those at most risk to survival are the juvenile trout. However, this was not modelled in inSTREAM, instead survival was assumed the same irrespective of life stage.

Mortality and growth have been seen as the most sensitive in juvenile brook trout at the individual-level (Janssen et al., 2024). On the population-level, growth rate and survival were the early life history parameters which had the largest influence on isolated cutthroat trout populations (Carim et al., 2017). This differs from the results found here as survival was the most sensitive and growth was in between the sensitivity of fecundity and survival. However, inhibiting growth rate was more sensitive than fecundity as it led to a longer time for individuals to reach maturity and therefore inadvertently also impacted fecundity and reproductive success of the individuals within the population (Vaugeois et al., 2020). Unlike for survival, angling stock reduction was probably a result of the increase in smaller-sized fish. Modelled continuous copper exposure to brook trout growth resulted in smaller-sized trout especially for the oldest fish individuals (Janssen et al., 2024). This loss of the largest and oldest portion of the population as a result of chemical stress in this study is in line with other studies on the persistence of isolated populations where size structures tend to be smaller individuals compared to connected populations (in the absence of stress) (Carim et al., 2017). Growth is the one life history parameter simulated in this study which has hypothesised influences on survival and fecundity as it can influence reproductive success due to smaller fish not reaching maturity size as well as increased likelihood for predation (Hamilton et al., 2016). Growth rate was also not dependent on life-history stage in inSTREAM, although in the field, growth has been seen to slow down with increasing age of fish (Hazlerigg et al., 2023). Also, growth impacts have been seen to differ between trout species where brown trout had a larger percentage decrease than greenback cutthroat trout (Accolla et al., 2019).

The size of the reach simulated was an important factor for angling stock abundance. This is so as habitat size can reflect quality where larger habitats, or reaches in the case of this study, should theoretically support larger populations (Neville et al., 2006). A large habitat can have more resources and promote more population abundance and fish growth both needed for a healthy angling stock. This was confirmed with the results as not all reaches could provide a habitat for large fish. Theories behind this often relate habitat depth to fish distribution based on individual size (Halvorsen et al., 2020). Carim et al. (2017) hypothesised that smaller and lower quality habitats promote a lower population growth rate but found no association between length and viability of cutthroat trout populations. Therefore, although reaches might not produce angling stock they can still produce a sustainable (but small) populations. Considering reach B was more than half the size of reach A, a minimum size range of 9000 – 12000 m^2 is predicted as suitable to host angling stock, giving an indication that habitat size is important for producing larger fish desired by anglers. To confirm this, additional investigations could include simulations of both smaller and larger reaches.

4.4.2 Angling stock resilience to chemical stress in a recreational fishery on a reach-scale

Angling stock was able to recover as long as rainbow trout were present within the reach simulated. Older adults have been shown to recover quickly within six years from stressors on fecundity and survival with matrix modelling for several fish species (Spromberg & Birge, 2005). Nearly all simulations of angling stock recovered within six years, but this signifies that that several generations of hatchlings (i.e. new recruits) were required before recovery was possible. The results also demonstrate that once chemical stress is removed from the simulation, the individuals also recover. Recovery from growth and fecundity effects from chemicals such as endocrine disrupters have also been considered reversible once stress has been removed (Hazlerigg et al., 2023). This varies for pesticides where effects on fecundity caused a 98% decrease in eggs where recovery was unlikely at high exposure levels (Accolla et al., 2022). However, the simulations considered permanent inhibition as well and at 100% inhibition of fecundity, angling stock was not able to recover. Reversible inhibition in individuals only slightly reduced the time to recovery compared to permanent inhibition only in the case of survival and growth rate. Differences in individual-level and population-level responses are usually comparable for stressors (Accolla et al., 2019) but the time to recovery did not indicate this as population recovery was more important than individual recovery in the reach. Generally, the lower the inhibition, the shorter the recovery, and the higher the inhibition, the longer the recovery.

Density dependence, where population growth relies on population density, is an important predictor of population resilience (Hazlerigg et al., 2014; Mintram et al., 2018b). Life history traits such as individual growth rate, reproduction, and survival can be influenced by the number of fish already present (Mintram et al., 2018b). Concerning the life history parameter of survival and growth, these are both dependent on population density within the field but this is often not well understood in fish species (Hamilton et al., 2016). Density dependence can lead to reduced population-level effects compared to individual-level effects indicating that extrapolation from individuals to populations may be overprotective (Mintram et al., 2018b). However, survival from fish predation is one of the only components in inSTREAM with direct density dependence where large trout can themselves become predators and reduce juvenile trout survival (Railsback et al., 2022). The other is for newly hatched trout where there is a critical survival time and competition for resources (Railsback et al., 2022). Trade-offs between fish size and abundance can still occur due to density dependence for growth and survival even though they might not be directly simulated in inSTREAM (Mee et al., 2016). For example, reduced population density can increase the growth and survival of remaining individuals and aid in recovery (Hazlerigg et al., 2014). This was seen in the results for time to recovery as angling stock was able to quickly recover. However, if size structure was changed completely, this could slow recovery (Solomon et al., 2020). This was also seen in the time for recovery as the higher the level of inhibition on growth rate, the slower the recovery, indicating that the size structure was more likely to be influenced with higher levels of stress.

Reach size influenced the recovery time of the angling stock where a larger population of both rainbow trout and angling stock needed more time to reach pre-exposure levels. Habitat quality such as the availability of spawning sites can allow quicker recovery (Ensign et al., 1997). The spawning sites might differ between the reaches but the reproduction potential of rainbow trout was the same in both reaches post-exposure. Reach A had a higher abundance of angling stock and each fish replicates the same, therefore more time is needed to reach these higher numbers (Schnaser & Mundahl, 2022). Recovery is also dependent on the threshold limits of populations where even a high-quality habitat could not allow recovery if too few fish were present to reach effective population growth rates (Kristan, 2003). This was reflected in recovery to inhibition of survival where angling stock was unable to recover in the larger reach (A) compared to the smaller reach (B). However, overall, recovery time was more dependent on the life history parameter inhibited than the reach simulated.

4.4.3 Combined stress interactions with the addition of prey availability

Combined stress interactions were mostly additive. This was as expected as additive is mostly likely to occur when stressors exhibit different modes of action which is the case for the different life history parameters inhibited (Morris et al., 2022) or with chemical stress combined with temperature (Galic et al., 2017). However, other modelling studies have suggested that population-level combined stress is rarely only additive (Galic et al., 2018). From the results of this current study, there was some indication that combined stress interactions were dependent on the level of chemical stress. This can be explained by an understanding of the mechanisms underlying these results (Schäfer et al., 2023). For example, by inhibiting fecundity and prey density, the resulting antagonistic buffering was a result of reduced competition (Gallagher et al., 2022) as only few angling stock remained and further reduction of prey density had no additional impact as enough resources were available for the individuals remaining. Fish behaviour can also change as a result of stress, where feeding tests on the boldness of brown trout showed that fish with a larger mass were more bold in their search (McGlade et al., 2022). Although boldness is not simulated in inSTREAM, larger-sized fish were able to search over a larger area. Therefore, reduced prey availability may be less influential for angling stock than smaller fish. Food restriction and starvation stress can also decrease trout size (leading to increased risk for predation) and growth to mature sizes (limiting their reproductive success). This can lead to a source of additional mortality from starvation as survival is also dependent on food availability (Kristan, 2003). The combined stress with survival interaction resulting in suppression is primarily due to minimal influence of prey density restriction when there are fewer individuals present, which reduces intraspecific competition for food (Vlaeminck et al., 2022). Simulations of isolated rainbow trout populations in lakes have found that restricted food availability leads to better population growth than what is expected in nature after 10 years of simulation (Serchuk et al., 1980). Inhibition of prey availability led to increased competition and limited food supply available for the population. This indicates there is a threshold of inhibition on prey availability that angling stock in a recreational fishery at the reach-scale can manage at 40% (Vaugeois et al., 2020).

4.4.4 Implications for angler satisfaction in a recreational fishery under chemical stress

Angler satisfaction was dependent on the life history parameter inhibited but did not improve under any level of stress as the size-structure state of the fishery changed, resulting in a loss of satisfaction. It was also dependent on both the presence of large fish and the abundance of fish. Fish body-size is dependent on stressor and whether it impacts on changes in fecundity, behaviour, and survival (Ahti et al., 2021) and size structure of a fish population of interest to fisheries management (Andersen, 2020). Satisfaction was therefore consistent with catch-related aspects as on a reach-scale there is no influence of non-catch-related aspects (Birdsong et al., 2021).

However, mortality from angling was not taken into account despite recreational fishing being an additional source of mortality as anglers exert additional pressure on fishery stocks (Stoeven, 2014;

Arlinghaus et al., 2017). Mortality from fishing practices can alter the spatial heterogeneity and distribution of populations, making them more sensitive to environmental stress such as climate change (Hsieh et al., 2010). Although this study considered a catch and release fishery, a percentage of mortality resulting from this practice was not included. Size-related restrictions on fishing are also common, but studies have found that size-selective fishing practices alter the size-age structure of the population causing destabilisation (Hsieh et al., 2010). Selective recreational fishing has been shown to have significant long-term impacts on the structures of freshwater populations (Ochwada-Doyle et al., 2022). Fishing mortality is therefore an additional source of stress and although prey availability was simulated in this study, this could be expanded to include fishing mortality to determine combined stress impacts with chemical stress. Another version of inSTREAM, inSTREAM-Gen (Ayllón et al., 2016) has already addressed the pressures recreational fishing practices under climate change place on salmonid populations and found declines in biomass but this was not in the presence of chemical stress. A previous study also implemented the influence of size-related catch and release mortality but still resulted in modelling a stable fish population (Forbes et al., 2019). Albeit additional mortality was not considered in this study, it is often regarded as an insufficient indicator of fishing impacts. Instead, the population structure can be a sufficient indicator of impacts on a fishery (Hsieh et al., 2010). Additionally, from an environmental risk assessment perspective, the effects of chemicals on the fish population are important to understand before the additional influence of fishing mortality.

4.4.5 Implications of findings

The implications of the level of effects of chemical stressors on fish life history parameters and fish prey abundance impacts on angling stock can be understood from a risk assessment perspective, as no observed effect concentration for population-level effects ($NOEC_{pop}$). The $NOEC_{pop}$ (*sensu* Janssen et al., 2024) can be determined from where the highest level of inhibition has no effect larger than 10% on the abundance based on the EC10 “safe thresholds” of EU regulation (European Chemicals Agency, 2008). As population-level effects are usually based on survival, growth, and reproduction translated from individual-level effects, a similar approach can be adopted for angling stock effects (Rudén et al., 2017). Estimating the effect-response relationships at both 10% and 50% reduction of angling stock, better informs risk assessment as they provide regulatory context to the simulation results (Table 4. 5). For example, understanding where a 10% reduction in angling stock occurs could enable decision-makers to set acceptable limits for a fishery exposed to chemical stress (Hanson & Stark, 2011). This can also be further translated to accepted impacts on angler satisfaction to be within 10% loss of satisfaction.

Time to recovery can also be used by fisheries management to determine recovery targets in angling stock. Recovery targets are based on the population density (individuals per area) and sometimes adult density (adults per area) as minimum viable populations of adult abundance (Caskenette & Koops, 2018). Results of angling stock recovery showed that as long as there is angling stock present within the system, they are predicted to recover once the stress is removed showing that angling stock density can be used to set recovery targets. However, angling stock might not be present in numbers to sustain angler satisfaction post-exposure and the fishery would not be functional for several (1-6) years once all chemical exposure has been removed. Also, it is recognised that the scenario used in the modelling (i.e. stress removed instantaneously and fully) is unlikely in reality although permanent individual inhibition still showed recovery was possible. However, the results reported in this study provide a useful illustration of how modelling approaches such as inSTREAM could be applied in a management context.

Overall, this study highlighted the importance of considering a reach-scale to address an isolated angling stock. From a conservation perspective, a reach-scale is the most valuable unit as most work is done on stream reaches at the local scale for e.g. habitat restoration (Peterson & Dunham, 2009; Blevins et al., 2014). Research suggests that reach-scale habitat qualities are influential to biodiversity in fish communities and stream fauna and studying reach-scale processes in response to anthropogenic stress is important to understand the health of the fish population (Blevins et al., 2014). Recreational fisheries are also spatially structured sometimes with patchy stocks across landscapes (Mee et al., 2016) with managed areas such as no-take zones and recreational-only fishing areas to sustain game fish populations (Arostegui et al., 2021). These practices can lead to isolated populations such as through introducing barriers as measures to avoid infection with e.g. non-native or invasive species (Carim et al., 2017). As a result of habitat isolation, fragmented and isolated populations are at greater risk of collapsing and may not recover from population bottlenecks such as those from environmental stressors (Carim et al., 2017). However, although angling stock populations may exist in isolation, they also do make up part of a more complicated landscape dynamic not considered here as spatial and temporal patterns across a landscape are also important for biological processes (Neville et al., 2006). Therefore, scaling this approach to the landscape-scale can outline spatial heterogeneity in recreational fisheries (Arlinghaus et al., 2017).

4.5 Conclusion

Chemical stressors on life history parameters can be an effective way to extrapolate impacts on individuals to population-level impacts. In a recreational fishery, however, the entire population is not of interest, instead the angling stock or the subset of the population with the largest and oldest adults represent the quality of the fishery stock. Angling stock impacts was dependent on the life history parameter inhibited. The findings here have shown that angling was resistance to changes in fecundity and sensitive to survival from chemical stress but overall was resilient and able to recover as long as trout were present in the fishery. However, in the same way population-level impacts can be characterised, so can a subset of the population, although the impacts on the population do not translate to the angling stock.

This study has connected chemical effects to ecosystem service delivery within a recreational fishery by translating stress on individual rainbow trout to angling stock and applying this as a proxy for angler satisfaction on a river reach-scale. This is a more effective way of describing the possible changes for recreational anglers as opposed to the current approaches by fisheries management of assessing fishery productivity through population abundance or biomass and in risk assessment with standardised toxicity testing which only examines individual-level effects. These are important findings in the understanding of the application of individual-based models and angler satisfaction at a reach-scale which can be used to assess different spatial scales within recreational fisheries.

Chapter 5

Spatially scaling an ecosystem services approach: assessing chemical stress to recreational fisheries in a river network

5.1 Introduction

Freshwater recreational fisheries are an important cultural ecosystem service enjoyed by millions of anglers globally with estimates of between one and four million recreational anglers in England alone (Environment Agency, 2018). However, freshwaters such as rivers are under threat from multiple stressors, of which chemical pollutants are one of the most widespread (Birk, 2022). Although assessment of the impacts of chemical stressors is typically focused on individual taxa, or overall biodiversity, there is increasing interest in extending such assessment to examine the effects on ecosystem services (Forbes & Calow, 2012; Munns et al., 2016; Maltby et al., 2018a, 2021a). Whilst links between individual biotic components of an ecosystem, and the delivery of ecosystem services can be complex (Maltby, 2013), in recreational fisheries the benefits to anglers are closely related to abundance, size structure, and accessibility of fish. Recreational fisheries, therefore, provide a good case study for investigating the effects of toxic chemicals on ecosystem services.

One challenge with ecological risk assessment, and in particular with the incorporation of ecosystem services into it, is that impacts can occur, and services may be delivered, in a spatially heterogeneous manner. This problem of variation in space has been identified as a challenge for mapping services and chemical exposure and makes it a priority to identify the relevant spatial scales on which assessment should be made to increase their ecological relevance (Maltby et al., 2018a; Faber et al., 2019). Furthermore, spatial scale is an important dimension to consider when specifying, for example, specific protection goals (SPG) (EFSA Scientific Committee, 2016b) which guide ecological risk assessments (Maltby et al., 2017). Many recreational fisheries, particularly those involving game fish, are located in river ecosystems, which comprise networks of channels. These networks can extend across significant areas of the landscape and fish populations, chemical inputs, and ecosystem services can occur, and be used, in different parts of that network. Here, the landscape refers to the relevant scale of ecosystem service assessment such as the catchment or sub-catchment surrounding the recreational fishery in the river network. Extending risk assessment across a river network requires consideration of a wider range of ecological processes than when assessing a single reach. In particular across the river network there are potential movements of individuals, differences in habitats along the river, and varied locations of stressor occurrence and ecosystem service delivery.

Fish habitats are distributed in patches across a river network and the connectivity between these habitats can influence the viability of populations through migration, dispersal, residency, and source and sink dynamics as well as necessary habitats for life-stage development (Cooke et al., 2016). Source and sinks are one form of metapopulation dynamics where sources are high-quality habitats, with a net surplus of individuals, which provide immigrants to the surrounding ecosystem and sinks are lower-quality habitats with net population loss where populations cannot persist without immigration from other habitat patches (Pulliam, 1988; Heinrichs et al., 2016). A river network can house stream fish spatial structures of multiple patchy populations with high dispersal and movement or metapopulations with infrequent interactions across reach and stream order boundaries where some areas will be more productive and habitable than others (Ovaskainen & Hanski, 2004; Bellard & Hugueny, 2020). Theses spatial structures are dependent on habitat connectivity which will influence the patterns of fish movement across a river network (Radinger & Wolter, 2014; Cooke et al., 2016). However, this connectivity can be jeopardised through fragmentation, for example, as a result of human activity, reducing population persistence and possibly leading to extinction events (Carim et al., 2017; Bellard & Hugueny, 2020; Birk, 2022; García-Vega et al., 2022). Understanding the effects of habitat fragmentation on stream fish spatial structures can be challenging due to confounding variables and logistics of long-term observations on fish metapopulation across spatial scales (Harvey & Railsback, 2011). It is also difficult to investigate where effects across a landscape affect fish spatial structures. Modelling can help to instead predict population persistence and spatial dynamics in response to stressors in the landscape (Bellard & Hugueny, 2020).

Recreational fishery species, such as salmonids, also exhibit spatial patterns and are categorised by a variety of life history movement patterns. These include anadromy, where fish migrate between the sea and freshwater, potamodromy, where they migrate exclusively in freshwater, and maintaining residency (Ferguson et al., 2019; Birnie-Gauvin et al., 2021; García-Vega et al., 2022). However, these movement patterns are not mutually exclusive. For example, rainbow trout remain in freshwaters throughout their life cycle and when their movement is anadromous they are known as steelhead trout (Pitman et al., 2019). Furthermore, salmonids can exhibit both long and short distance migratory patterns, or they may remain entirely resident within a reach (Ferguson et al., 2019). In potamodromy, there is still a practice of upstream and downstream movement depending on fish behaviour. Residency is typically a behavioural choice when growth is prioritised, whilst movement is usually driven by the need to spawn and reproduce elsewhere (Kendall et al., 2015). The balance between movement and residency is often for growth potential in other habitats (e.g. enhanced feeding)

(Ferguson et al., 2019). Fish make decisions to move for feeding and spawning initiated by environmental conditions, such as water flow, where disruptions to fish decisions can reduce the connectivity of populations (García-Vega et al., 2022). Spawning often occurs in smaller streams and tributaries of rivers known as natal streams whereas when fish grow and age, they emigrate to downstream reaches to grow more rapidly (Leathe et al., 2014). Research has shown that the time of these movements can be highly variable among juvenile fish (Leathe et al., 2014). These movements and their spatial patterns can vary between individuals and populations (Ferguson et al., 2019). Estimating these movements can better help to understand fish behaviour within complex environments such as river networks.

In freshwater ecosystems of rivers, species dispersal is influenced by landscape connectivity and suitability (Bertassello et al., 2022) but knowledge of the spatial patterns of fish movement from the field is limited (Radinger & Wolter, 2014). Studies have identified patterns and variables that may be responsible and have applied them to fish models that are capable of simulating fish movements across spatial and temporal scales (Grimm et al., 2005; Grimm & Railsback, 2012). Models can better help to understand how populations and interacting population dynamics occur across spatial scales. A key predictor of fish movement within a river network is that distance is positively related to fish length (Radinger & Wolter, 2014), a pattern which underpins models such as the individual-based model inSTREAM (version 7.3) (Railsback et al., 2022). Therefore, the largest fish within a system will hypothetically move the furthest whilst small fish will be restricted to site choices more locally. In addition to this, larger streams are typically higher stream orders and are able to accommodate more fish (Neville et al., 2006; Carim et al., 2017; Halvorsen et al., 2020). These indicate that the largest reaches would be higher stream orders with the greatest number of fish. Overall, spatial patterns of fish populations within river networks are dependent on movement, interactions with their environment and other populations, and habitat connectivity of the river network. This means spatial patterns can be unique to the river and diverse across a river network.

This background of patchy habitat distribution and fish movement in river networks can then be subject to the occurrence of chemical stressors, which themselves are spatially patterned (Maltby et al., 2018). To determine the effects of chemical stress on spatial patterns of fish in freshwaters, defining the spatial scales can help to define the biological levels which are at risk, e.g. trout populations in a recreational fishery of a reach versus trout metapopulations in fisheries across a river network. Modelling has been applied to identify the effect of chemicals on individuals within the fish population of a recreational fishery of a small network (Forbes et al., 2017; Forbes et al., 2019). This did not show how the river landscape was influential as it did not consider fish spatial structures or target different areas of the river network and stream orders. Modelling will help to identify whether reach location or stream order influences spatial structures of populations and how.

For spatial structures of a recreational fishery in applying the ecosystem services concept in risk assessment, both the impact of chemical stress on ecosystem service delivery and the beneficiaries need to be considered. The fish spatial structures make up one part of the recreational fishery landscape which influences ecosystem service delivery, whereas anglers are the beneficiaries of the landscape (O'Higgins et al., 2020). Anglers, such as trophy anglers, are generally not primarily interested in the entire fish population, but instead the largest (and, therefore, oldest) individuals in the population (Food and Agriculture Organization of the United Nations, 2012), here termed the angling stock. Identifying chemical impacts on angling stock can be done at the individual-level and then translated to what this means for the angling stock (Forbes et al., 2019; Galic et al., 2019). However, spatial patterns related to angling stock dynamics can better help to determine how they respond to chemical impacts. The impact on anglers depends on both the benefits, such as abundance and size of stock and accessibility to the site, as well as the effort or cost, including travel distance to the fishery and crowding or competition for fishing sites (Post & Parkinson, 2012; Arostegui et al., 2021; Birdsong et al., 2021; Lomnicky et al., 2021). Both angler benefits and costs can vary or covary across a network with the quality of a fishery leading to sites which are more favourable than others which will ultimately determine the angler satisfaction (Food and Agriculture Organization of the United Nations, 2012). As angler satisfaction depends not only on angling stock but also on the angler effort or cost required to reach the recreational fishery, understanding the impacts of chemical stress on angling stock can determine gains or losses in satisfaction.

This study will therefore answer how chemical stress on a recreational fishery and angling stock spatial patterns influence angler satisfaction in a river network by investigating:

1. How does angling stock respond to chemical stress on common life history parameters (growth, survival, fecundity) when simulated as part of a network versus a single reach in isolation?;
2. How and where do chemicals targeting common life history parameters in fish impact angling stock dynamics and angler satisfaction across the network?

As angling stock entails the largest and oldest fish within the network, their location and behaviour can be predicted based on reach and network characteristics mainly related to habitat quality and reach size.

5.2 Methods

An individual-based model, inSTREAM version 7.3 (Railsback et al., 2022), was used to simulate exposure to chemical stress in populations of a game fish (rainbow trout*, Oncorhynchus mykiss*) in a river network comprising up to third-order watercourses and twenty reaches. The simulation outputs were assessed for the outcome on both populations and delivery of a final ecosystem service: angler satisfaction.

5.2.1 Study site

The simulated river network was compiled of mapped sections of first to third order streams from the Little Jones Creek watershed (Harvey & Railsback, 2011) available in GIS Shapefile format from ArcGIS Pro (ESRI, 2021) (Figure 5.1). Reaches were varying lengths of 73 m draining 2 km^2, 115 m draining 15 km^2, 184 m draining 26 km^2 for third, second, and first order streams, respectively. Reaches were connected to make up a network of six tributary branches made up of two copies of first order reaches, three forks made up of two copies of second order reaches, and a mainstem segment of two copies of a third order reach. The length of the entire network is 1.934 km of streams. No physical or environmental barriers were simulated between the reaches. Each reach has previously been calibrated in a prior application with inSTREAM within a network for observed trout abundance and size based on census data (Harvey & Railsback, 2011). Mean daily flow and temperature were estimated from continuous flow gauge and monitoring measurements, respectively (Harvey & Railsback, 2011). Depths of cells within the reach were based on a hydraulics sub-model as flow changes.

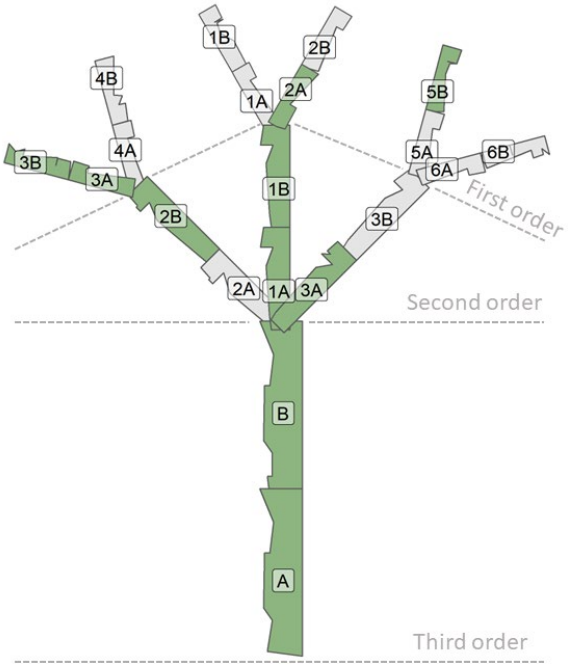


Figure 5.1 The simulated river network for experiments in inSTREAM (version 7.3). The river network was made up of 20 river reach sites composed of replicates of first order (12), second order (6), and third order (2) reach sites. Reach copies for each stream order were identical environmentally and with initial starting trout populations except for their position within the network. Reaches were identified by their position in the network (order), branch in their order (1-6), and position within the branch (A or B, downstream and upstream, respectively). Reaches within the network were randomly selected using stratified sampling (Section 5.2.3) to identify reaches targeted for chemical stress (green).

5.2.2 InSTREAM: model rational, set-up, and modifications

InSTREAM, individual-based Stream Trout Research and Assessment Model (Railsback et al., 2022), a spatially explicit individual-based model for trout populations, hosted in NetLogo (Wilensky, 1999) and run on cloud computing via RONIN web-based management (https://ronin.cloud/), was applied to simulate rainbow trout. InSTREAM is driven by time-varying variables and habitat variables which influence fitness and adaptive behaviour in individual trout to predict their population status in streams. The tool was originally developed for stream trout management, specifically addressing questions of how habitat changes can affect trout population persistence and productivity. In this study productivity will refer to the abundance of trout or size of the stock in the river as opposed to the biomass produced (Food and Agriculture Organization of the United Nations, 2012).

All simulations of the river network or reaches were run with inSTREAM (version 7.3) (Railsback et al., 2022). Each simulation spanned 19 years with ten replicates, where each year, the simulation was monitored on a census day of September 30th. The first two years of the simulations were omitted from analysis to eliminate the impact of initial conditions and allow the model to stabilise so a simulation period of 17 years was analysed (Railsback & Grimm, 2019; Railsback et al., 2022). Due to parts of the model being stochastic in nature, guidance on applying inSTREAM suggests between five and ten replicates as sufficient to identify differences between scenarios (Railsback et al., 2022). Preliminary trials investigating replicate numbers concluded using ten replicates. Each reach has been previously calibrated for initial trout population inputs (Table 3.1) (Harvey & Railsback, 2011) which resulted in the river network initiated with a rainbow trout population with 1254 individuals of varying age classes (ages 0, 1, and 2+). A baseline (control) run, comprising ten replicates, was used to determine how the trout behaved within the network under no chemical stress. This was used as the control simulation to compare with the impacts of chemical effects.

Chemical exposure was added to inSTREAM inputs by adapting code to recognise chemical presence or absence within reaches. Chemical effects were then added to inSTREAM by inhibiting trout life history parameters of fecundity, growth rate, and survival. The most appropriate target in inSTREAM parameters or sub-models was identified for each life history parameter. The fecundity sub-model was modified to include chemical stress on individual fecundity (i.e. number of viable eggs), growth rate sub-model for growth rate and prey density and a new survival sub-model was coded. Simulation outputs from inSTREAM were changed to record angling stock abundance (i.e. trout older than 1 year and larger than 20 cm), record the natal reach of angling stock, as well as to record the movement of trout individuals between reaches to identify the time angling stock individuals spent exposed to chemical stress. The simulations adopted the following assumptions: i) a catch and release fishery with no added mortality from harvest mortality; ii) chemical effects on life history parameters did not vary by age or size class; and iii) anglers are trophy anglers with the primary goal of catching a large fish. For detailed code changes to inSTREAM see Appendix 8.2.

Simulating angling stock dynamics in absence of chemical stress: Determining baseline spatial structure

Before any chemical stress was applied to the river network, a simulation of ten replicates of the river network in the absence of chemicals stress was simulated. This was to determine the angling stock dynamics in the absence of chemical stress to use as a baseline or control run to compare to chemical stress scenarios. Not only was this simulation used as a baseline but the angling stock density across the river network reaches and angling stock natal reaches were also determined.

The trout spatial structure was also determined with these simulations, with preliminary results (Figure 3.11) indicating a hybrid form of metapopulation with both patchy and interacting populations but some evidence of source-sinks (Falke & Fausch, 2010). Compared to a metapopulation, or a population of populations where individuals are mostly confined to one patch with occasional local extinction and recolonisation (Ovaskainen & Hanski, 2004; Bellard & Hugueny, 2020), a patchy population consists of individuals moving frequently between patches with high dispersal (Ovaskainen & Hanski, 2004; Bellard & Hugueny, 2020), where a hybrid of both is often seen in salmonids. This was reflected as trout dispersal depended on their life stage as hatched trout were diverse across the river network. Also, immigrations occurred at the ends of the river network and emigrations in the middle of the network so trout were seen to exist and move within distinct patches (Bellard & Hugueny, 2020). Some traditional metapopulation dynamics of source and sink habitats (*sensu* Pulliam, 1988) was also present as most trout hatched in third order reaches with overall positive net change in abundance in first order reaches and negative in second and third order reaches.

5.2.3 Simulating chemical stress on trout life history parameters in inSTREAM

Selecting reaches exposed to chemical stress

Reaches were selected to simulate chemical exposure in the river network where chemical stress on individual life history parameters of growth rate, fecundity, and survival were inhibited. For the purpose of this study, exposure is related to chronic exposure at localised reaches which degrades between reaches, therefore, only the reach exposed exhibits chemical stress. One reach was targeted for chemical stress at a time. To balance computational power and ecological realism, a subset of the river network reaches was selected. Preliminary analysis of a baseline run of the river network spatial structure determined groupings which were applied to select the target reaches (Figure 3.11). Groupings were based on similarity in results between third order and first order streams, as well as between second order downstream (A) and upstream (B) reaches. These four main groupings defined the strata to randomly sample from for reach site selection with a stratified random sampling approach (% sampled in Table 5.1). A total of ten reaches (making up 50% of the network) were randomly selected to span evenly across the branches and orders making sure to have more than one site in the same group for comparison. Reach selection was done with R version 4.0.3 (R Core Team, 2024) in R Studio (RStudio Team, 2020) (Table 5.1).

Table 5.1 A summary of the randomly selected reaches within the network for targeted chemical stress.

Group	Total number	No. Sampled (and %)	Reach names
First	12	4 (33%)	2A, 3A, 3B, 5B
Second (Upper)	3	2 (66%)	1B, 2B
Second (Lower)	3	2 (66%)	1A, 3A
Third	2	2 (100%)	A, B

Selecting chemical effect endpoint levels in trout individuals

Common life history parameters of growth rate and fecundity as well as an added survival sub-model were inhibited (0 – 100%) when a reach was exposed to chemical stress. Preliminary analysis on a reach-scale determined the sensitivity of angling stock to chemical stress. Depending on the sensitivity of angling stock to inhibition of life history parameters, a low, medium, and high effect level was specified for inhibition (Table 5.2). A control run with no chemical effect (0% inhibition of all parameters) served as a baseline for comparison between the effect and no effect simulations.

Table 5.2 A summary of the low, medium, and high levels of inhibition of common life history parameters used for inSTREAM simulations. N/A indicates where this range is not applicable for the life history parameter.

Endpoint	Low	Medium	High
Fecundity	N/A	90%	99%
Growth rate	20%	50%	80%
Survival	1%	5%	N/A

Simulating reaches in isolation and as part of the entire network

To examine the question of how angling stock responded to chemical stress on common life history parameters (growth, survival, and fecundity) when simulated as part of a network versus a single reach in isolation, reaches were simulated both in isolation and as part of a network, as summarised in Table 5.3 with ten replicates for each simulation. As the river network was comprised of three stream orders, the first, second, and third order reaches were simulated in isolation for each life history parameter inhibited at varying inhibition levels (0 – 100%). Within the network, each of the reaches selected as exposed to chemical stress was simulated for each life history parameter and inhibition level (0 – 100%). This simulation of the whole river network was used for subsequent analysis of angling stock dynamics.

Table 5.3 A summary of reaches simulated in isolation and as part of a river network with life history parameter inhibited (endpoint) of fecundity, growth rate, or survival as well as the level of inhibition.

Reach in isolation	Reach as part of network	Endpoint	Inhibition level
First order	2A, 3A, 3B, 5B	Fecundity	90%, 99%
Second order	1A, 1B, 2B, 3A	Growth rate	20%, 50%, 80%
Third order	A, B	Survival	1%, 5%

5.2.4 Determining the impact on angler satisfaction

Ecosystem service delivery, particularly where there is no straightforward mapping onto economic value, can be difficult to quantify, and therefore, alternative measures are often used as a means of determining impacts (Logsdon & Chaubey, 2013). To determine how angler satisfaction is impacted across the network, angler satisfaction was used as a measure of utility or the happiness that anglers receive from their angling experience (Kosenius et al., 2013; Birdsong et al., 2021). Angler satisfaction is based on the benefits and costs for an individual angler to determine if there is a subsequent gain or loss in angler satisfaction across the landscape at the end of the simulation period (17 years). Variables influential to benefits and costs were determined from an adapted conceptual model of angler satisfaction (Pitman et al., 2019) where angler effort indicates the cost for the angler and catch rate or the benefit from the reaches. Angler satisfaction is dependent on both non-catch-related factors, which represent the cost or angler effort, and catch-related factors represent the benefit or catch rate for anglers. Angler effort is related to crowding and travel distance whilst benefit is a function of both catching a large fish and fish abundance in the fishery site (Post & Parkinson, 2012; Arostegui et al., 2021; Birdsong et al., 2021; Lomnicky et al., 2021). These factors were used to develop a cost-benefit analysis of angler satisfaction. To capture the likelihood of catching a large fish, the fraction of angling stock from the total fish abundance was calculated (P). This probability was then added to the total abundance of fish in the fishery (N) to determine if the fishery has large fish and if there are many fish that could be caught. Both factors can influence angler satisfaction as not only is there is satisfaction for catching a fish but additional satisfaction if the fish caught is large. The benefit to an angler is, therefore, a sum of the likelihood of catching a large fish and the total number of fish. The cost to anglers is a sum of the total distance (T) needed to travel to each respective reach within the river network and angler population density (D) as a reflection of crowding (i.e. competition for nearby sites) in the wider landscape. All variables were then scaled to be between 0 – 1 based on the minimum and maximum values for each variable within the landscape. For benefit variables, minimum and maximum values were selected from the 17-year simulation period for the reach. Scaling for the angler satisfaction was based on minimum and maximum values as shown in Equation 5.1

$$\frac{x-min}{max-min} \qquad \text{Equation 5.1 (Rullens et al., 2022)}$$

where x is the value that is scaled, min is the minimum, and max is the maximum value for each variable. In addition to rescaling each parameter, both elements of catch (benefit) and non-catch (cost) were rescaled to a scale of 0 – 1 by dividing in half to assume that each variable has equal weighting. To evaluate the satisfaction across the landscape an arbitrary 10 x 10 grid was overlaid onto the river network to represent the starting locations for an angler in the landscape. As there are 20 possible fishery sites (reaches) within the river network a median for these values was used for each respective grid cell (100 in total) to identify the median value from each starting location within the network. The median benefit – cost value for each grid cell in the landscape is summarised in Equation 5.2

$$\frac{P_{(0-1)} + N_{(0-1)}}{2} - \frac{T_{(0-1)} + D_{(0-1)}}{2} \qquad \text{Equation 5.2}$$

where $P_{(0-1)}$ is the rescaled probability or likelihood of catching a large fish (number of large fish divided by the total fish abundance), $N_{(0-1)}$ is the rescaled total fish abundance, $T_{(0-1)}$ is the rescaled travel distance to reaches, and $D_{(0-1)}$ is the rescaled angler population density. The change in both cost and benefit is considered a relative measure to baseline or a no chemical stress scenario. Although both benefit and cost were scaled again by dividing by two to be between 0 – 1, angling stock proportion ($P_{(0-1)}$) and total abundance ($N_{(0-1)}$) values from stressed scenario could be outside the bounds defined by these minimum and maximum values across the 17-year simulation period. This indicates that values of best and worst (1 and 0, respectively) from baseline simulations might not be the best and worst when compared to chemical stress simulations.

Simulating angler population density

Anglers must travel from their starting point to reach a recreational fishery site. Angler density in this residential population can impact the likelihood of crowding at fishery sites due to competition for fishing sites. To reflect the influence of potential crowding at fishery sites, three population scenarios for angler density were developed: 1) a uniform population density, 2) a large densely populated area downstream, and 3) several sparsely populated areas upstream. These scenarios reflect the possible crowding in reaches as actual site visitation is not possible in the absence of specific fishery data. Scenario 1 is an extreme case where there is no influence of angler population density in the landscape; this also serves as a baseline scenario. Scenario 2 (Figure 5.3B) is another extreme case with a large densely populated area is located downstream of the river network to signify distance from the most beneficial fishery sites (Figure 5.2). Angler population density is dispersed from a central point with maximum density and dissipates by half in each grid cell that is further away from the centre. This reflects the possible negative influence on catch rates that can occur from competition originating from urban epicentres (Arlinghaus et al., 2008; Post et al., 2008). Finally, scenario 3 (Figure 5.3C) is a collection of random areas in the upstream portion of the landscape with low angler population density at about one-tenth of the maximum density. This signifies the presence of less anglers found in more rural areas closer to the most attractive fishery sites (within this network the upper second order streams in the centre of the river network) (Kaemingk et al., 2020) as shown in Figure 5.2 with angler benefit. The overall benefit (Figure 5.2) and cost (Figure 5.3) under no chemical stress was calculated for each scenario.

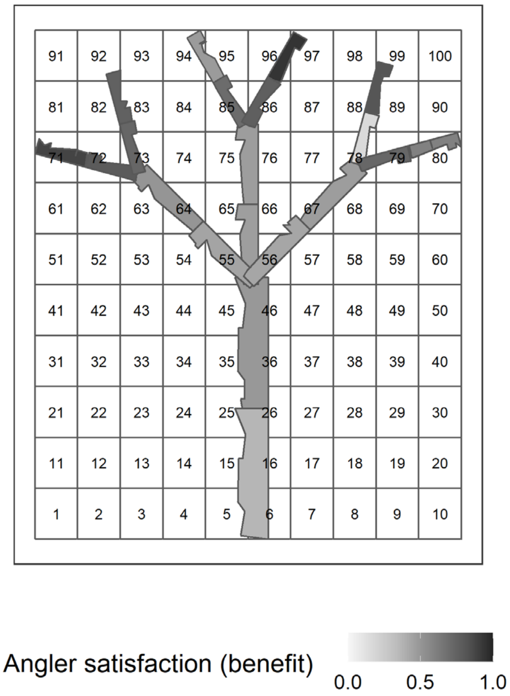


Figure 5.2 Angler satisfaction based on the median benefit (scaled 0 – 1) or a summation of the probability of a large catch (0 – 1) and the total fish abundance (0 – 1) scaled for each reach from their min and max values over a 17-year simulation. The values are the benefit at the end of the simulation.

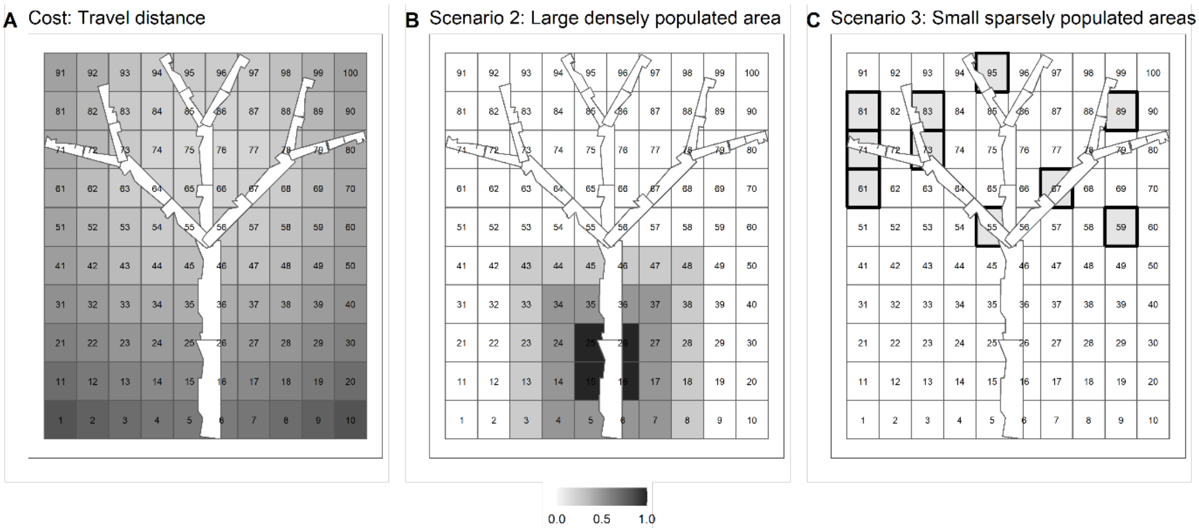


Figure 5.3 Angler satisfaction median cost for A) travel distance (0 – 1) to reach fishery sites from the minimum to maximum distance across the whole landscape and B – C) angler population density scaled from 0 – 1. B) Scenario 2 is the additional cost related to a large densely populated area that is downstream of the river network. C) Scenario 3 is the additional cost when there are ten randomly placed areas in the top 50% of the landscape (outline in black) which are sparsely populated within a grid upstream of a simulated river network.

5.2.5 Data analysis

All data analyses were run with R version 4.0.3 (R Core Team, 2024) in R Studio (RStudio Team, 2020) with R packages for data manipulation and data visualiation. Spatial analyses were run with ArcGIS shapefiles and R with package sf (Pebesma, 2018; Pebesma & Bivand, 2023). Mean and ± one standard deviation from ten simulation replicates were calculated for angling stock abundance for the reaches simulated in isolation and as part of the river network. Results were compared to baseline (control) simulations and considered a different result if standard deviations (± one standard deviation) do not overlap with means between simulations. The total angling stock in the river network was summarised with mean, median, and interquartile ranges across ten replicates for each simulation with stressed reaches and inhibition of life history parameters.

Median angler satisfaction (benefit – cost) and interquartile ranges for each grid cell in the landscape was calculated as an indicator of central tendency in the landscape as angler satisfaction distributions were skewed. The value for each grid cell was then classified based on whether satisfaction was lost or gained (Table 5.4) to determine if patterns in grid cells change across the landscape (Qian et al., 2022). The overall median benefit – cost without variation in angler population density was used as a baseline measure to compare the change in the percentage of landscape grid cells. The percentage of the landscape (% of 100 grid cells) for each classification was then used to summarise the change in angler satisfaction.

Table 5.4 Range of angler satisfaction (median benefit – cost of baseline scenario), where X is the median value under baseline (no stress), IQR is the interquartile range, Q1 is the 25% percentile, and Q3 is the 75th percentile.

Classification	**Gain**	**Loss**
High gain/loss	[Q3 + 1.5*IQR, Inf]	[Q1 – 1.5*IQR, -Inf]
Gain/Loss	[X, Q3 + 1.5*IQR]	[X, Q1 – 1.5*IQR]
Gain/loss within IQR	[X, Q3]	[X, Q1]
No change	= X	= X

To summarise the changes at the landscape level summary statistics of median, interquartile range, and mean were also calculated for angler satisfaction for the entire landscape (i.e. all 100 grid cells) across the three angler population density scenarios based on a similar approaches of quantifying change across a spatial landscape (Qian et al., 2022).

5.3 Results

5.3.1 Angling stock structural dynamics in absence of chemical stress

Figure 5.4A shows that angling stock did not hatch uniformly throughout the river network. Most angling stock hatched within upstream third order streams, then second, and fewest in first order streams or downstream third order streams. For angling stock at the end of the simulation (Figure 5.4B), they were unevenly spread across the river network with second order upstream reaches as the most productive and first order streams as least productive in the absence of chemical stress. This indicates the underlying spatial patterns and structure of angling stock.

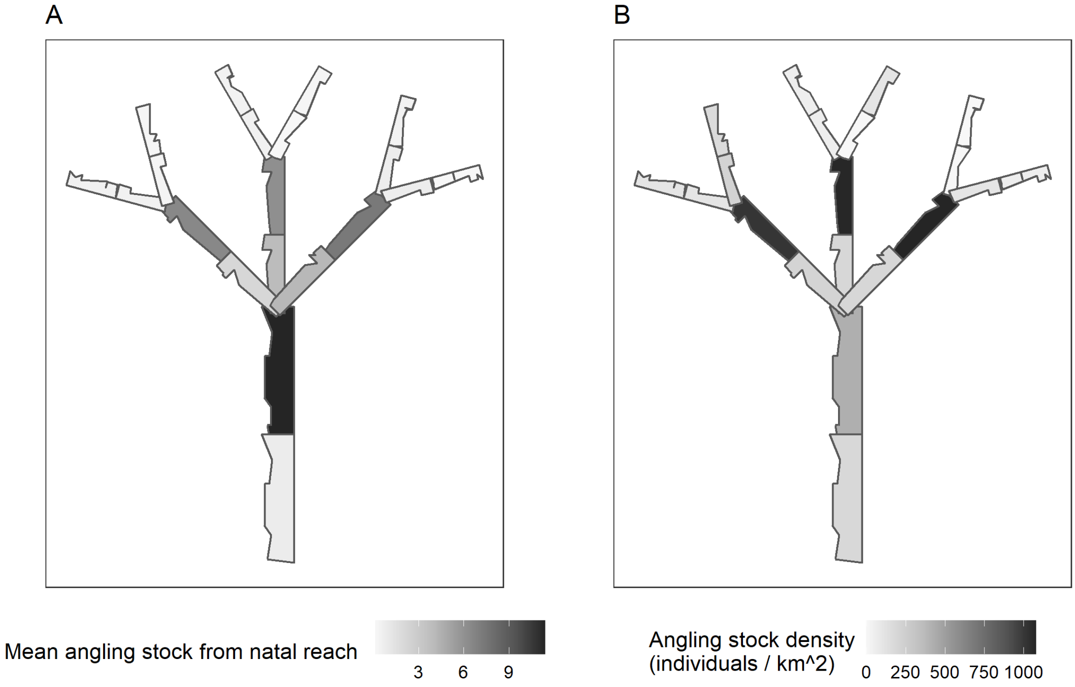


Figure 5.4 A) Mean abundance of angling stock (number of individuals older than 1 year and larger than 20 cm in length) which hatched in natal reaches from ten replicates at the end of a 17-year simulation period after a 2-year initial period to allow the population to stabilise. The intensity of grey depicts an increase in hatched angling stock abundance within the reach compared to others. B) Baseline angling stock density (individuals per km^2) at the end of the simulation period of a simulated river network. The intensity of grey depicts the angling stock density value within the reach compared to other reach values.

5.3.2 Angling stock simulated in an isolated reach versus a reach as part of a network

Changes in angling stock were assessed across a river network in response to inhibiting fecundity, growth rate, and survival in trout individuals across ten reaches in three stream orders by looking at abundance within reaches simulated in isolation and reaches simulated as part of a river network. Initial investigation of angling stock abundance in reaches (see Appendix Figure A5.1 – A5.3) revealed similar patterns between stream orders as well as upstream (A) and downstream (B) reaches within these stream orders so results for streams of the same order and their position upstream or downstream within a branch were combined accordingly.

First order streams

All first order stream reaches responded with the same pattern of low numbers of mean angling stock abundance (a mean of less than one individual per reach) in response to inhibiting fecundity, growth rate, and survival (Figure 5.5). This is irrespective of their location upstream or downstream within a first order branch and whether the reach is simulated as part of the network or in isolation. Low angling stock numbers meant that no identifiable patterns related to inhibition of life history parameters were possible. One exception to this was angling stock in first order streams with high inhibition of growth rate (80%) with means above one individual, but this was not enough to discern any patterns.

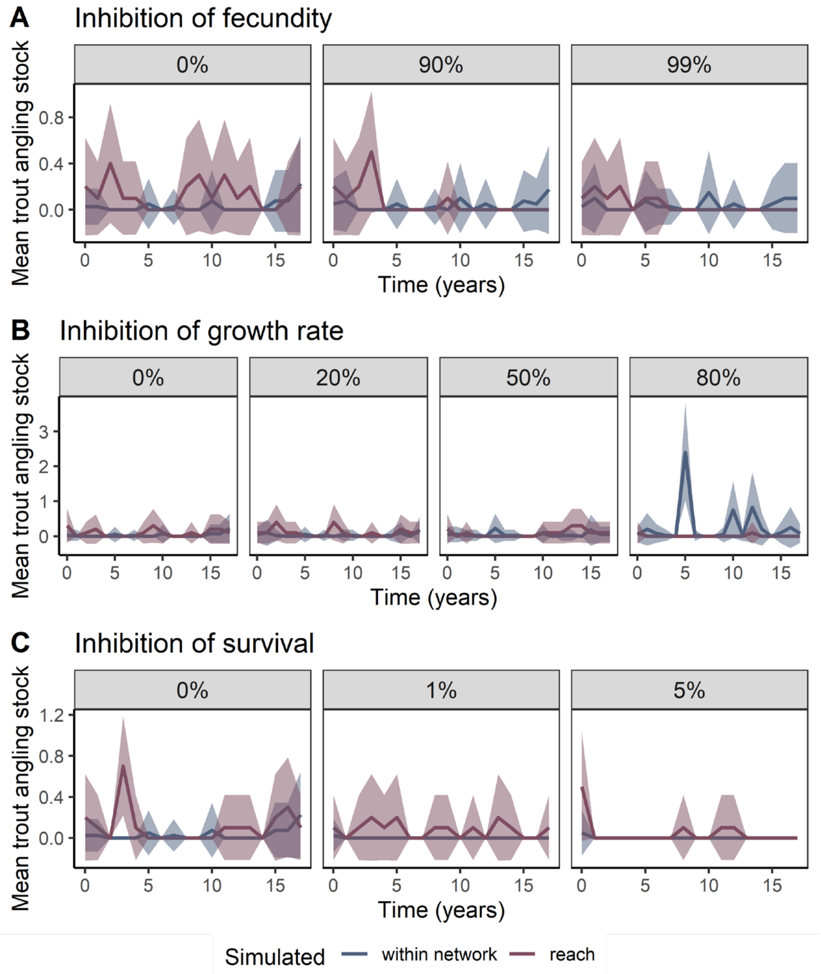


Figure 5.5 Mean angling stock abundance (number of individuals older than 1 year and larger than 20 cm in length) across the river network at the end of the simulation from ten replicates and four first order streams exposed to inhibition (0 – 100%) of A) fecundity, B) growth rate, and C) survival, run in isolation (red) and run within a river network (blue). Shading signifies +/- one standard deviation from replicates.

Second order streams

Second order streams, when simulated as a single reach in isolation, were low in angling stock productivity with a mean of less than three individuals (Figure 5.6). Again, as with first order streams, there were not enough trout in the angling stock to discern any patterns in response to inhibiting fecundity, growth rate, or survival, albeit abundance does decrease. When simulated as part of the river network, angling stock in second order streams responded differently depending on whether they were in an upstream (B) or downstream (A) reach within their branch. Angling stock in downstream second order reaches responded the same irrespective of when the reach was simulated as part of the network or in isolation with a mean of less than three individuals across the 17-year simulation. An exception to this is angling stock in downstream second order reaches were more resistant to inhibition of fecundity (up to 99%) and growth rate (up to 50%) than when simulated alone.

In contrast, upstream second order reaches were more productive for angling stock than their downstream second order reaches with baseline runs having means of 3 – 13 mean individuals when simulated as part of a network. The angling stock in these reaches were also resistant to inhibiting fecundity up to 99% but responded to inhibiting growth rate and survival decreasing steadily with increasing inhibition. Nonetheless, mean angling stock abundance remained higher when simulated as part of a network as opposed to in isolation except for high inhibition of survival. These angling stock in downstream second order reaches experienced larger changes as a result of life history parameter inhibition than angling stock in upstream reaches. Also, upstream second order reaches, when simulated as part of a network, were more productive than when simulated in isolation. Overall, second order streams were more productive than first order streams and across all second order streams angling stock was resistant to changes in fecundity up to 99% inhibition.

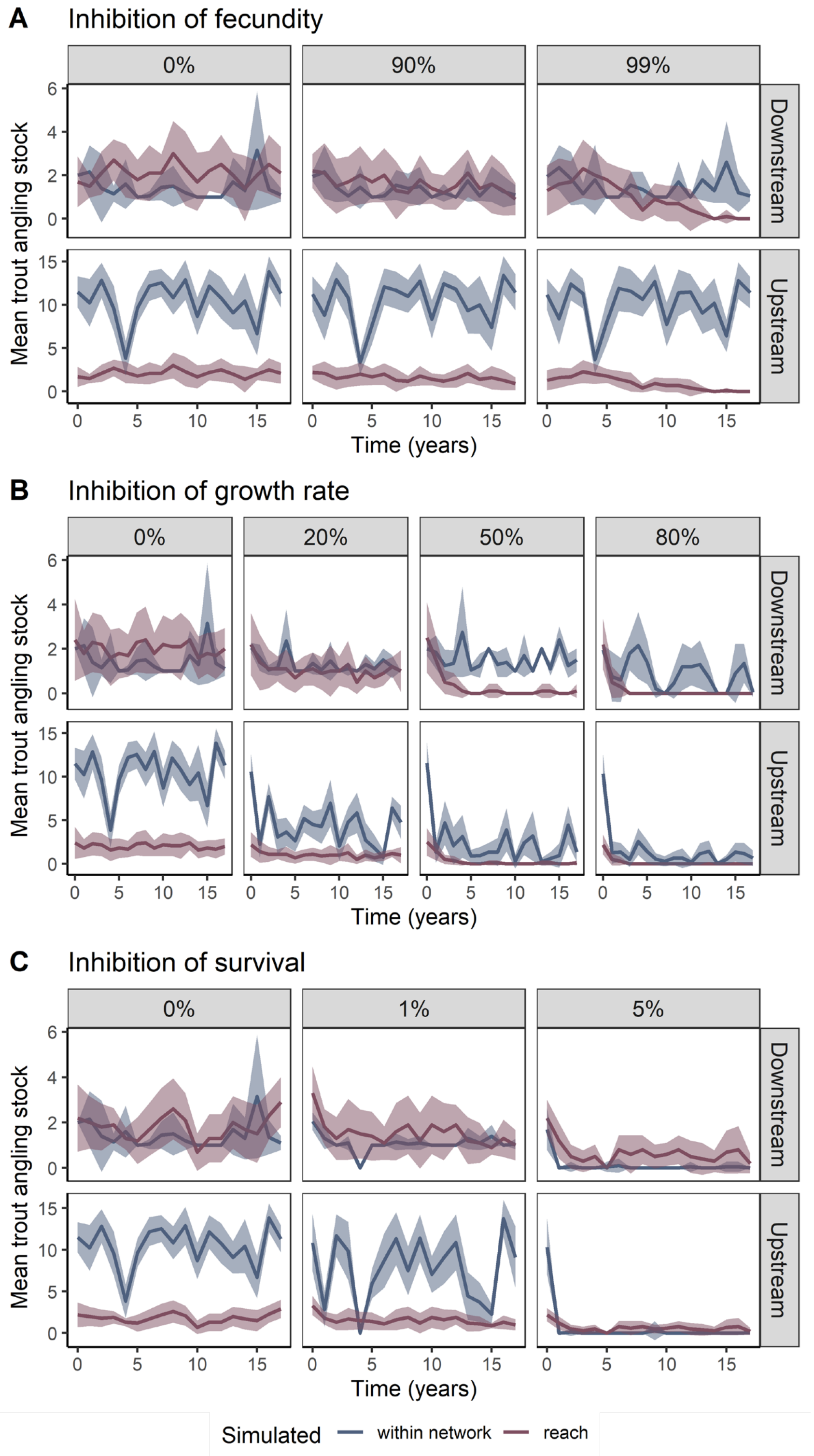


Figure 5.6 Mean angling stock abundance (number of individuals older than 1 year and larger than 20 cm in length) across the river network at the end of the simulation from ten replicates and four second order streams exposed to percentage inhibition (0 – 100%) of A) fecundity, B) growth rate, and C) survival, run as a single reach in isolation (red) and run within a river network (blue). Second order streams are separated into their downstream reaches and upstream reaches. Shading signifies +/- one standard deviation from replicates.

Third order streams

Third order streams, when simulated as a single reach in isolation, were the most productive reach for angling stock with a mean of 6 – 12 individuals over the 17-year simulation (Figure 5.7). Angling stock was resistant to inhibition of fecundity up to 99%, but decreased with growth rate up to 50% and survival up to 5%, where in both cases angling stock no longer inhabited the reach. When inhibition of life history parameters was simulated in third order streams as part of a river network, angling stock responses depended on whether they were in an upstream (B) or downstream (A) reach within their branch. All third order streams were less productive when simulated as part of a network as opposed to as an isolated reach. Apart from a peak mean of 13 (upstream) and 9 (downstream) individuals at 4 years exposure, mean angling stock ranged between 0 – 6 individuals. Angling stock in third order reaches, irrespective of location in the branch, were resistant to inhibiting fecundity up to 99%, decreased with growth rate inhibition up to 20% and survival up to 5%. For 50% inhibition of growth rate, angling stock was more abundant when simulated within a network than when simulated alone with the downstream (A) third order reach more abundant than its upstream (B) counterpart. At 80% inhibition, downstream (A) and upstream (B) reaches responded differently. The downstream reach angling stock abundance increased to a mean above 8 individuals over the simulation period whilst the upstream reach continued to decrease.

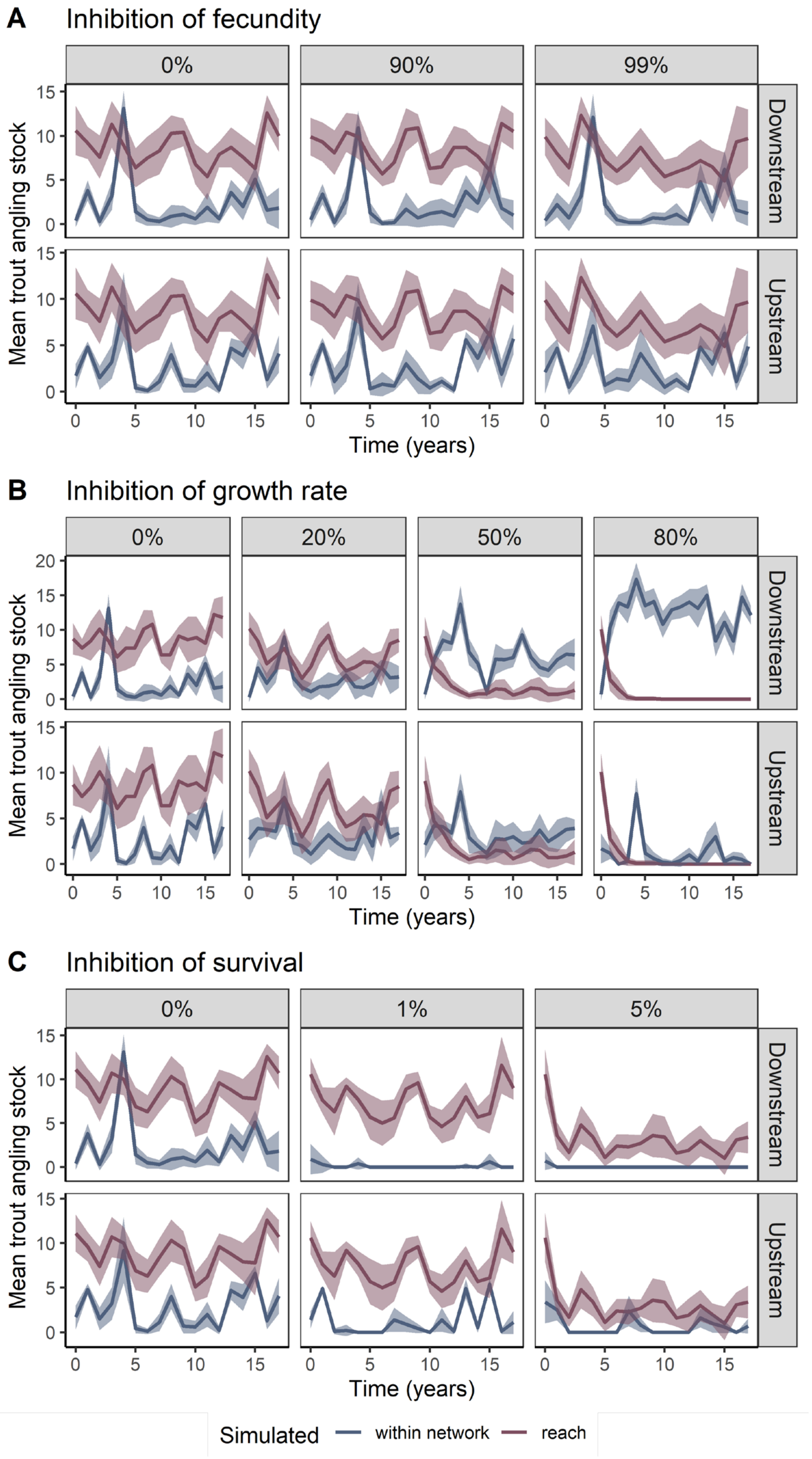


Figure 5.7 Mean angling stock abundance (number of individuals older than 1 year and larger than 20 cm in length) across the river network at the end of the simulation from ten replicates and two third order streams exposed to inhibition (0 – 100%) of A) fecundity, B) growth rate, and C) survival, run as a single reach in isolation (red) and run within a river network (blue). Third order streams are separated into their downstream reaches and upstream reaches. Shading signifies +/- one standard deviation from replicates.

Summary across the river network

Angling stock was summarised across the river network for inhibition of fecundity (Figure 5.8), growth rate (Figure 5.9), and survival (Figure 5.10) in ten exposed reaches (as preliminary analysis across time did not discern differences shown in Appendix Figure A5.4). Inhibiting fecundity by 99% in trout individuals, and sometimes with 90% (First-3A, 5B), decreased both the mean and median angling stock abundance across a river network (Figure 5.8). However, angling stock were dependent on the reach exposed. For example, in second order reaches, median angling stock decreased, but the median angling stock was still within the baseline interquartile range. But this was not the case for two first (2A, 5B) and third order streams (A, B) for 99% inhibition of fecundity, where median angling stock, decreased below the interquartile range. A 90% inhibition of fecundity was insufficient to completely change angling stock throughout the river network meaning spatial dynamics largely mitigated these effects. Overall, the variation between results from the ten replicates increased with inhibition level.

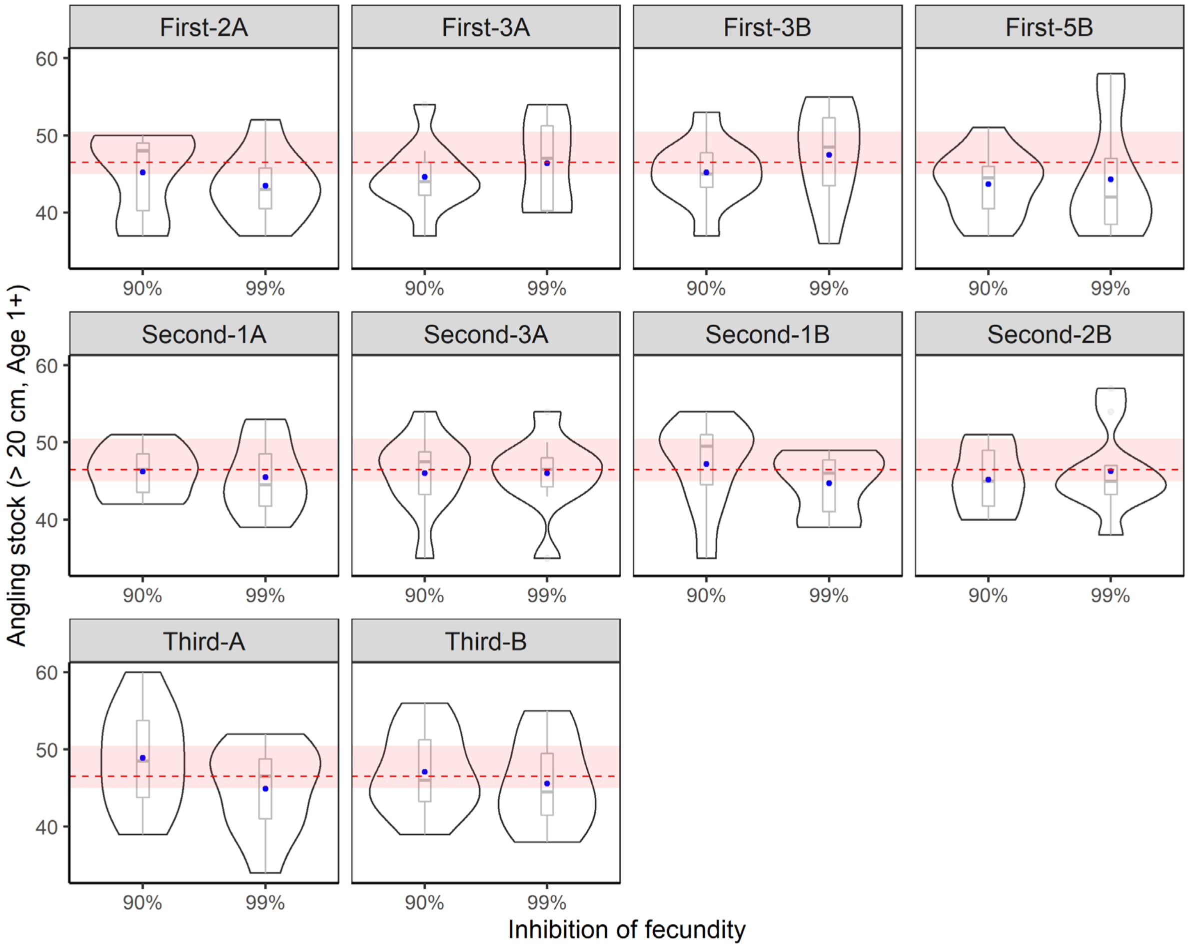


Figure 5.8 Violin plots of angling stock abundance (number of individuals older than 1 year and larger than 20 cm in length) across the network from ten replicates with internal boxplots of the median and interquartile range when inhibiting fecundity (90-99%) across ten exposed reaches at the end of a 17-year simulation. The red line indicates the median angling stock abundance with no chemical effects and red shading indicates the interquartile range (Q1-Q3). The blue points are the mean values.

When third order reaches were subjected to growth rate inhibition of 20% or more, angling stock abundance, at the level of the whole river network, substantially decreased from baseline conditions (Figure 5.9). The same cannot be said for first and second order streams. For first and second order downstream (A) reaches, the abundance decreases in some reaches at 20% inhibition, but increased with higher rates of inhibition (80%) but remained within the interquartile range of baseline conditions with the exception of the First-3A reach where angling stock increased.

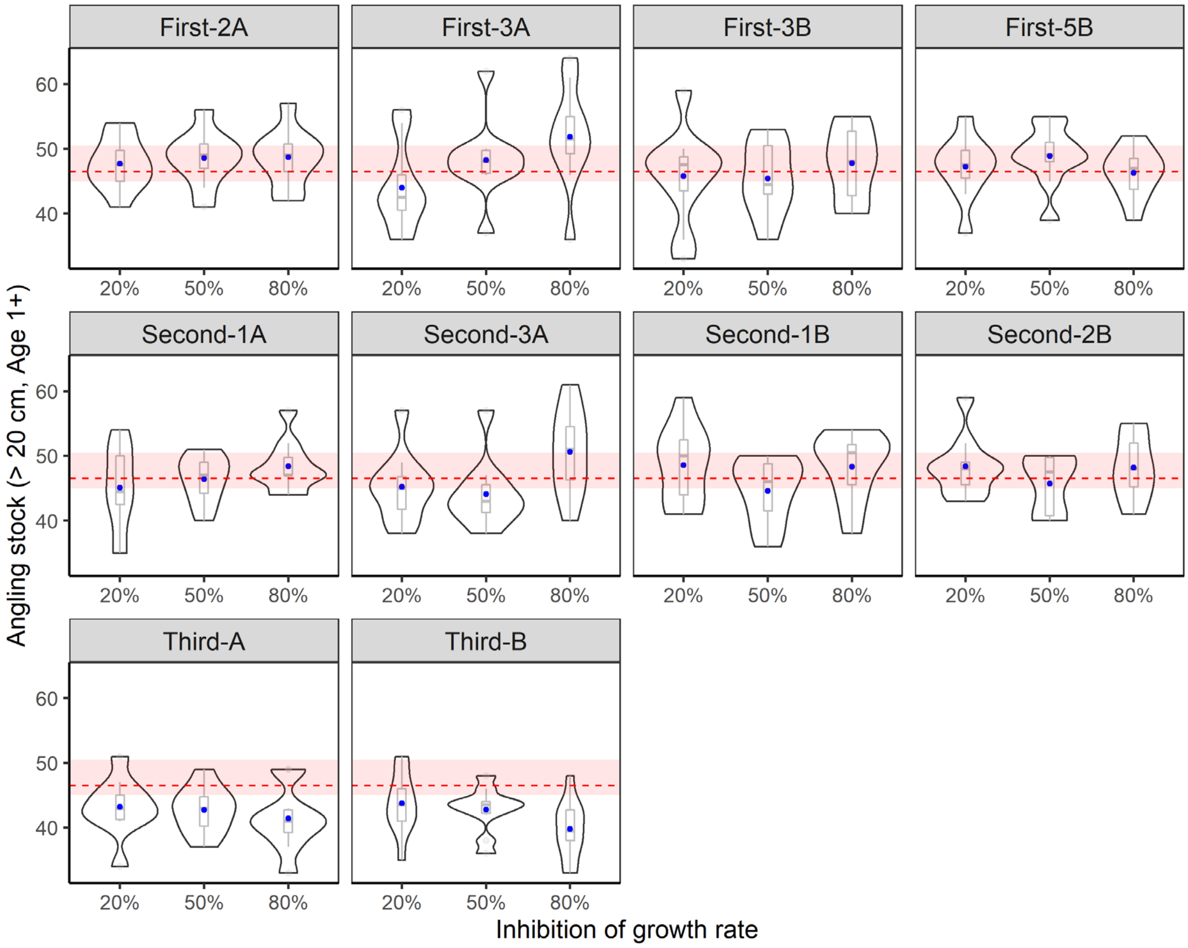


Figure 5.9 Violin plots of angling stock abundance (number of individuals older than 1 year and larger than 20 cm in length) across the network from ten replicates with internal boxplots of the median and interquartile range when inhibiting growth rate (20-80%) across ten exposed reaches at the end of a 17-year simulation. The red line indicates the median angling stock abundance with no chemical effects and red shading indicates the interquartile range (Q1-Q3). The blue points are the mean values.

Inhibiting survival by 5% (and in some cases 1%) decreased median abundance below baseline when second and third order reaches were targeted. First order streams were, in some cases, below baseline but for the most part they were within the interquartile range.

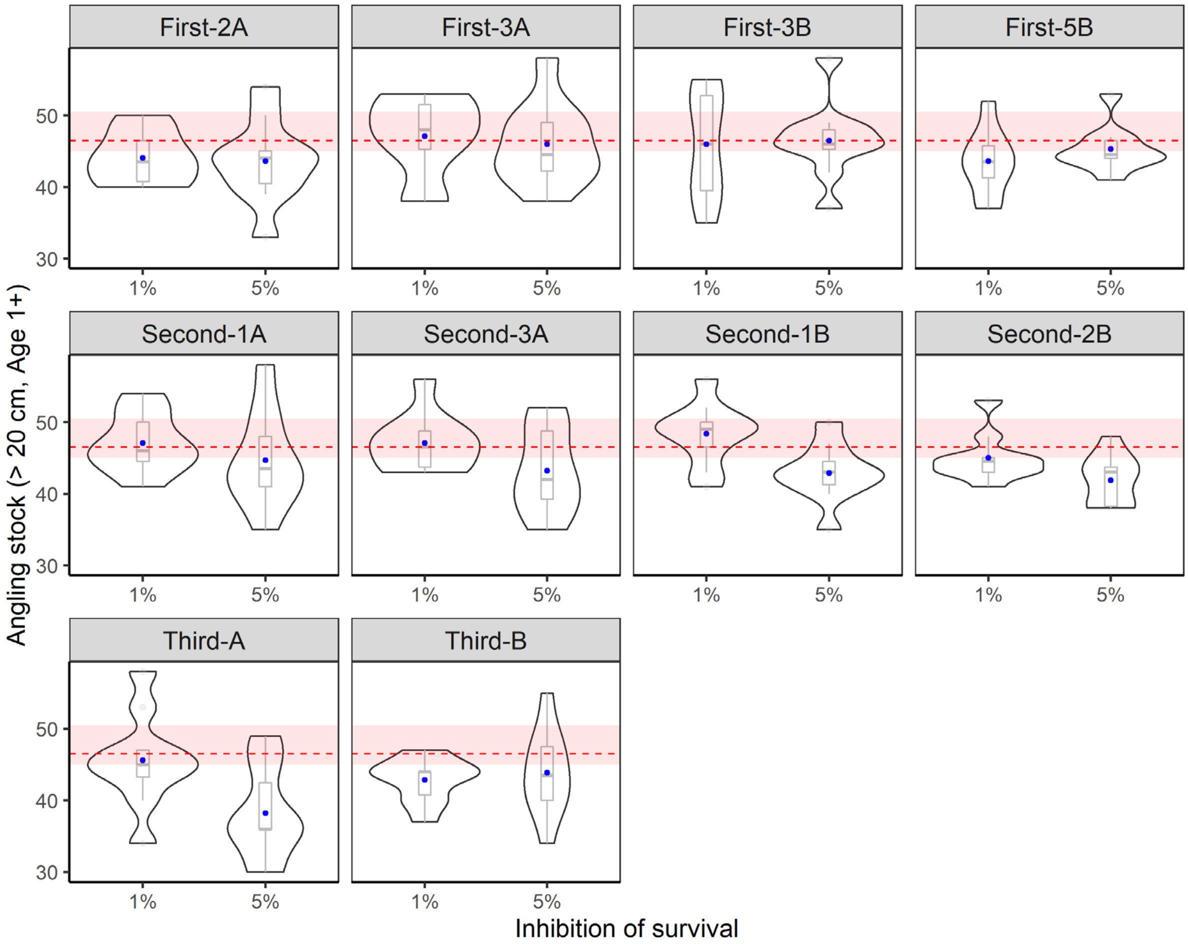


Figure 5.10 Violin plots of angling stock abundance (number of individuals older than 1 year and larger than 20 cm in length) across the network from ten replicates with internal boxplots of the median and interquartile range when inhibiting survival (1-5%) across ten exposed reaches at the end of a 17-year simulation. The red line indicates the median angling stock abundance with no chemical effects and red shading indicates the interquartile range (Q1-Q3). The blue points are the mean values.

Angling stock exposure time

The results for the time trout individuals from the angling stock spent in exposed reaches in their lifetime were summarised for both upstream (B) and downstream (A) stream order reaches, as well as combined for all third order reaches since the results were the same (Figure 5.11). Without any inhibition of life history parameters, angling stock individuals spent the majority of their lifetime in third and second order reaches.

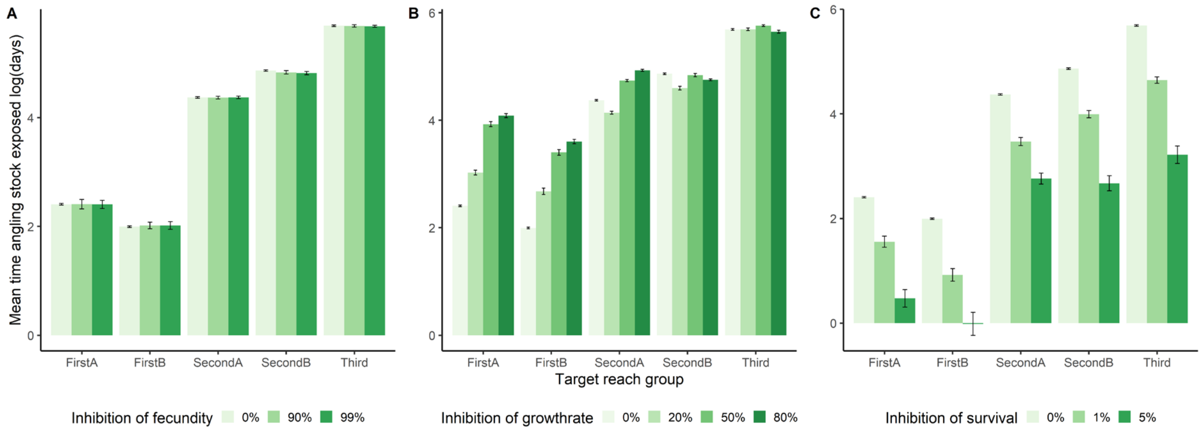


Figure 5.11 Mean time spent in exposed reach across ten replicate simulations and reach location with stream order branch (upstream or downstream) for inhibition of A) fecundity (0-99%) B) growth rate (0-80%), and C) survival (0-5%) over the entire 17-year simulation period. Error bars indicate +/- one standard deviation from ten replicates.

Angling stock movement did not respond to inhibiting fecundity (Figure 5.11A) but did respond to inhibiting growth rate (Figure 5.11B) and survival (Figure 5.11C) in trout individuals. For inhibition of growth rate, there was a slight increase in the time angling stock spent in exposed reaches as the inhibition of growth rate increased for first order reaches and second order downstream reaches. The same was not observed for second order upstream or third order reaches. As shown in Figure 5.5 – Figure 5.7, abundance in exposed reaches sometimes increased with increased inhibition of growth rate, but this was not reflected in the time angling stock was exposed. Finally, as inhibition of survival increased, individuals spent less time in exposed reaches as a result of their shorter lifespans.

5.3.3 Angler satisfaction in absence of chemical stress

The surrounding landscape was summarised by the median value and interquartile range for each grid cell (100 in total) in the landscape. This was based on the costs of the distance from each of the 20 reaches and angler population density in each cell, which were subtracted from the benefits derived from both the probability of catching a large fish and the likelihood of catching a fish from the angling stock. The results of angler satisfaction under no chemical stress for the three angler population density scenarios are shown in Figure 5.12. The landscapes for all three scenarios had the highest values in the grid cells closer to the middle second order reaches of the river network. Angler population density decreased satisfaction in around third order reaches in the second scenario with a large densely populated area downstream, but small sparsely populated areas in scenario 3 were not enough to show any visible changes in the landscape.

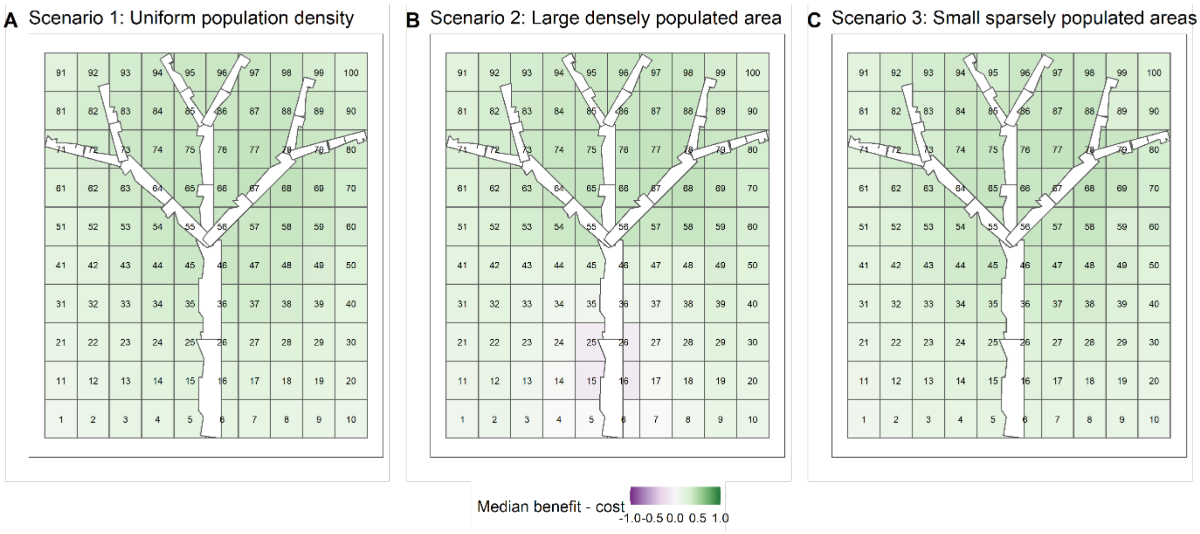


Figure 5.12 Angler satisfaction (based on median benefit – cost) across the landscape in the absence of chemical stress (control) for the three angler population density scenarios. A) scenario 1 with a uniform angler population density, B) scenario 2 with a large, populated area downstream and C) scenario 3 with ten randomly places small sparsely populated areas upstream.

5.3.4 Angler satisfaction under chemical stress

The three scenarios (1 – 3) for angler satisfaction show the cost (influence of travel distance and angler population density) with the angling stock benefit (proportion of large fish and total abundance) when ten reaches in a simulated river network were exposed to chemical stress which inhibited trout individual fecundity, growth rate, and survival. Firstly, all grid cells within the landscape were classified as either gained or lost satisfaction compared to a control (i.e. scenario 1 with no chemical stress). Then the entire landscape was summarised for angler satisfaction for each of the three scenarios.

Scenario 1: Uniform population density

The proportions of cells in each category under inhibition of life history parameters in target reaches are summarised in Figure 5.13. Under inhibition of fecundity (Figure 5.13A), growth rate (Figure 5.13B), and survival (Figure 5.13C), there was an increased loss in satisfaction in the landscape. Loss in satisfaction varied across simulations in 3 – 52% of cells when fecundity is inhibited, in 1 – 63% of cells when growth rate is inhibited, and in 1 – 78% of cells when survival is inhibited. The range widened with increasing sensitivity to inhibition of life history parameters; fecundity, growth rate, and then survival. Two exceptions were when a second order downstream reach (3A) was exposed to high (80%) levels of inhibition of growth rate and for a third order upstream reach (B) exposed to 5% inhibition of survival led to a gain in satisfaction across the landscape. This signifies that under any level of chemical stress on trout fecundity and growth rate, there were visible changes at the landscape-level. Only in exceptional circumstances was there a gain in satisfaction. There was no identifiable pattern between where the chemical stress is applied according to stream order type or branch location.

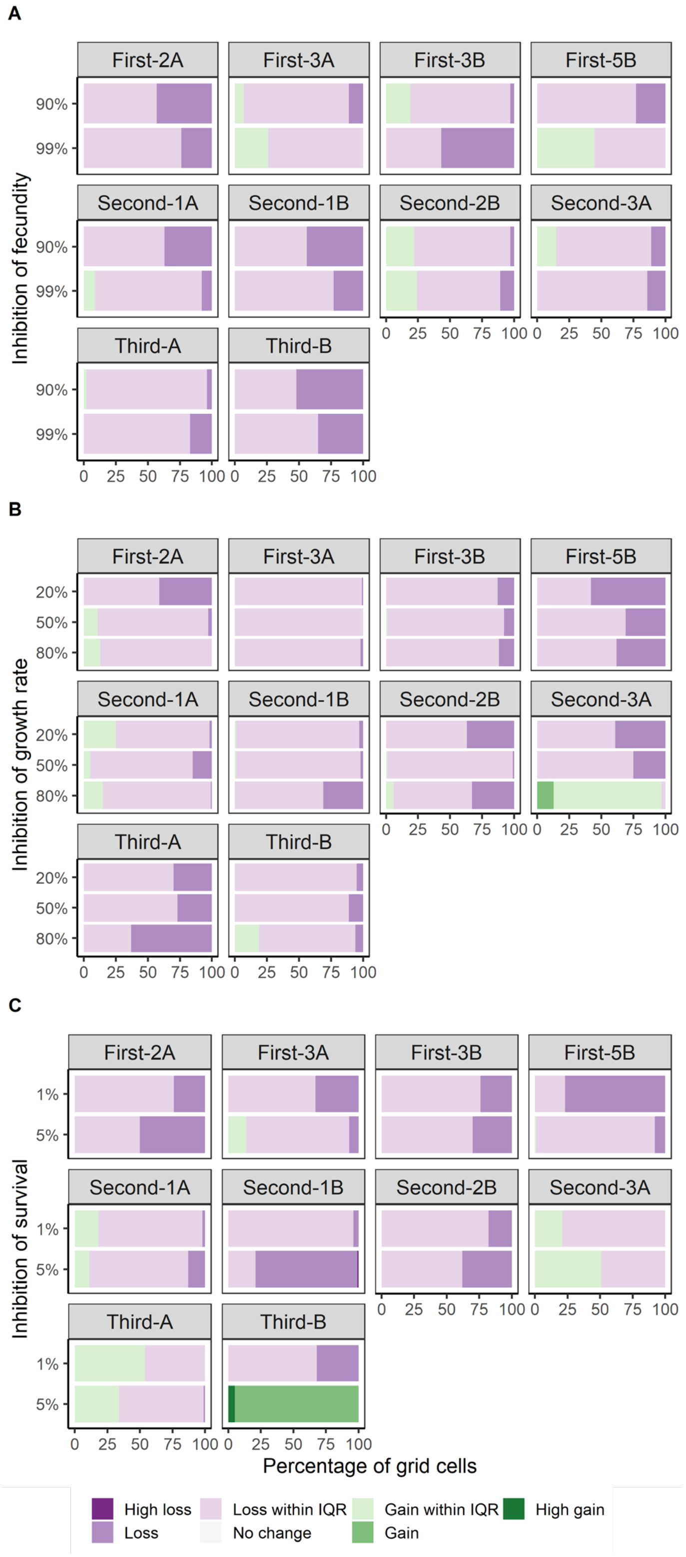


Figure 5.13 Percentage of grid cells within the landscape according to the classification (Table 5.4) of loss (purple) and gain (green) in angler satisfaction (based on median benefit – cost in grid cells) where the intensity of the colour is higher loss or gain. The grid cells are for scenario 1 across inhibition of A) fecundity, B) growth rate, and C) survival in regard to the reach exposed. IQR is the interquartile range based on the baseline (no chemical stress) grid cells in the landscape. Results are from the end of the simulation period and ten replicates.

Angler satisfaction (benefit – cost) was then summarised for the landscape impact of inhibiting fecundity (Figure 5.14), growth rate (Figure 5.15), or survival (Figure 5.16). The distribution of the values in grid cells expanded but generally, the summary statistics of median, mean, and interquartile range did not diverge from the interquartile range of baseline conditions, except for when a third order upstream (B) reach was exposed to survival inhibition at 5%. This reflects what was seen in Figure 5.13 where the entire landscape gained satisfaction. Changes to satisfaction on a grid cell-level in the landscape were not necessarily reflected as the median remained similar or changes were not large enough to impact the overall landscape summary. Therefore, in some simulations loss in satisfaction of part of the landscape was balanced with a gain in satisfaction in other parts of the landscape.

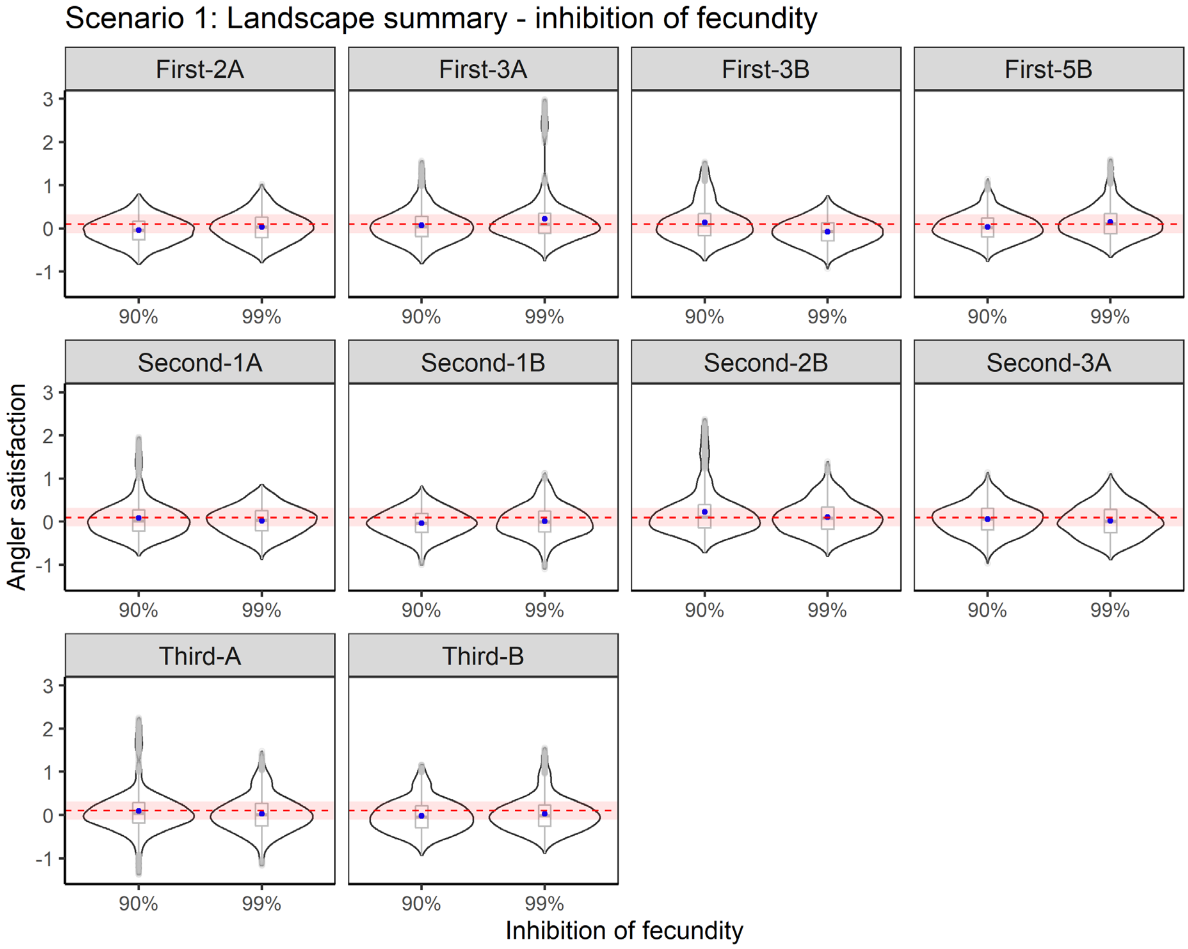


Figure 5.14 Violin plot of the distribution of median angler satisfaction (benefit – cost) across a landscape with fecundity inhibited (90 – 99%) within ten exposed reaches. An additional internal boxplot summarising median and the interquartile range (Q1-Q3). The blue point signifies the mean value. The red dashed line is the median with no effects and the shading shows the interquartile range (for comparison). Results are from the end of the simulation period and ten replicates.

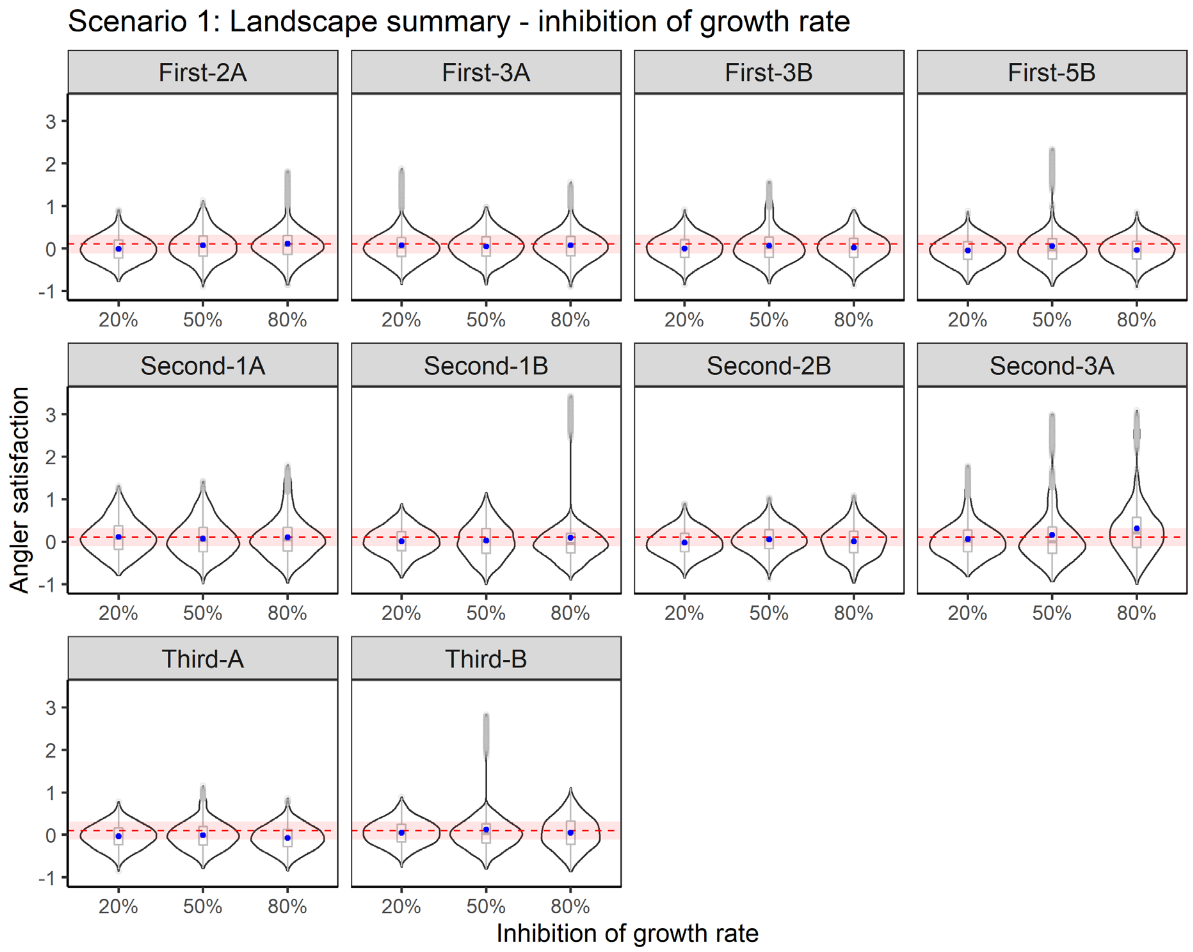


Figure 5.15 Violin plot of the distribution of median angler satisfaction (benefit – cost) across a landscape with growth rate inhibited (20 – 80%) within ten exposed reaches. An additional internal boxplot summarising median and the interquartile range (Q1-Q3). The blue point signifies the mean value. The red dashed line is the median with no effects and the shading shows the interquartile range (for comparison). Results are from the end of the simulation period and ten replicates.

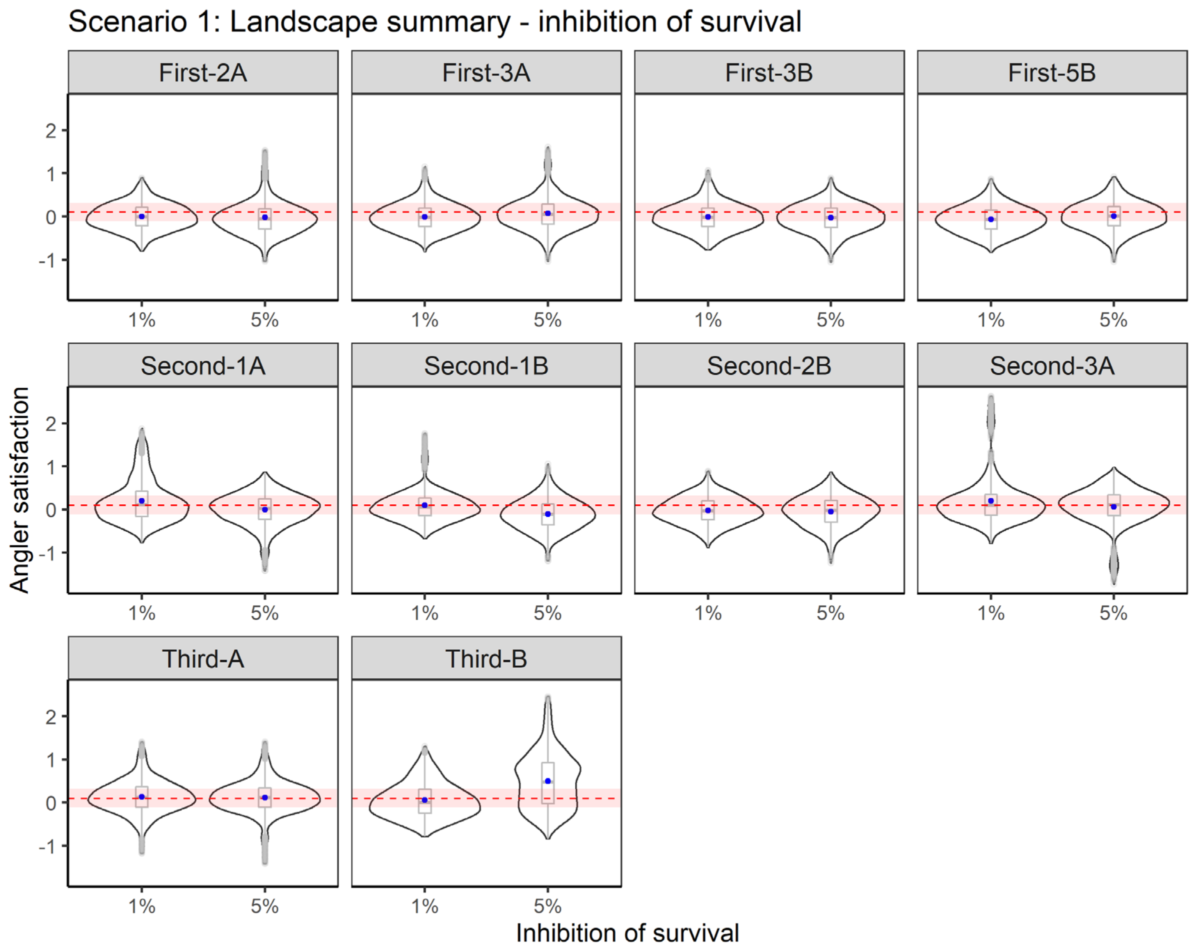


Figure 5.16 Violin plot of the distribution of median angler satisfaction (benefit – cost) across a landscape with survival inhibited (1 – 5%) within ten exposed reaches. An additional internal boxplot summarising median and the interquartile range (Q1-Q3). The blue point signifies the mean value. The red dashed line is the median with no effects and the shading shows the interquartile range (for comparison). Results are from the end of the simulation period and ten replicates.

Scenario 2: Large densely populated area

The landscape pattern for loss or gain in satisfaction was summarised for inhibition of fecundity (Figure 5.17A), growth rate (Figure 5.17B), and survival (Figure 5.17C). In order to assess the impact a densely populated area with anglers had on the landscape, classifications of gain or loss were compared to scenario 1 baseline (uniform population density). The landscape experienced a greater loss over all life history parameters inhibited with some of the landscape cells categorised as high loss. The level of loss was between 17 – 40% when fecundity was inhibited, 18 – 44% when growth rate was inhibited, and 6 – 58% when survival was inhibited. The range widened with increasing sensitivity to inhibition of life history parameters; fecundity, growth rate, and then survival. The level of high loss was 9 – 20% when fecundity was inhibited, 7 – 20% when growth rate was inhibited, and 4 – 23% when survival was inhibited. There were two exceptions: a second order downstream reach (3A) at high (80%) levels of inhibition of growth rate led to a slight gain in satisfaction, and a third order upstream reach (B) at high (5%) inhibition of survival led to an overall gain in satisfaction across the landscape. Not only was there landscape-wide loss, showcasing that any level of chemical stress on trout fecundity and growth rate will result in a loss of angler satisfaction, but an increase in angler population density also led to landscape wide losses.

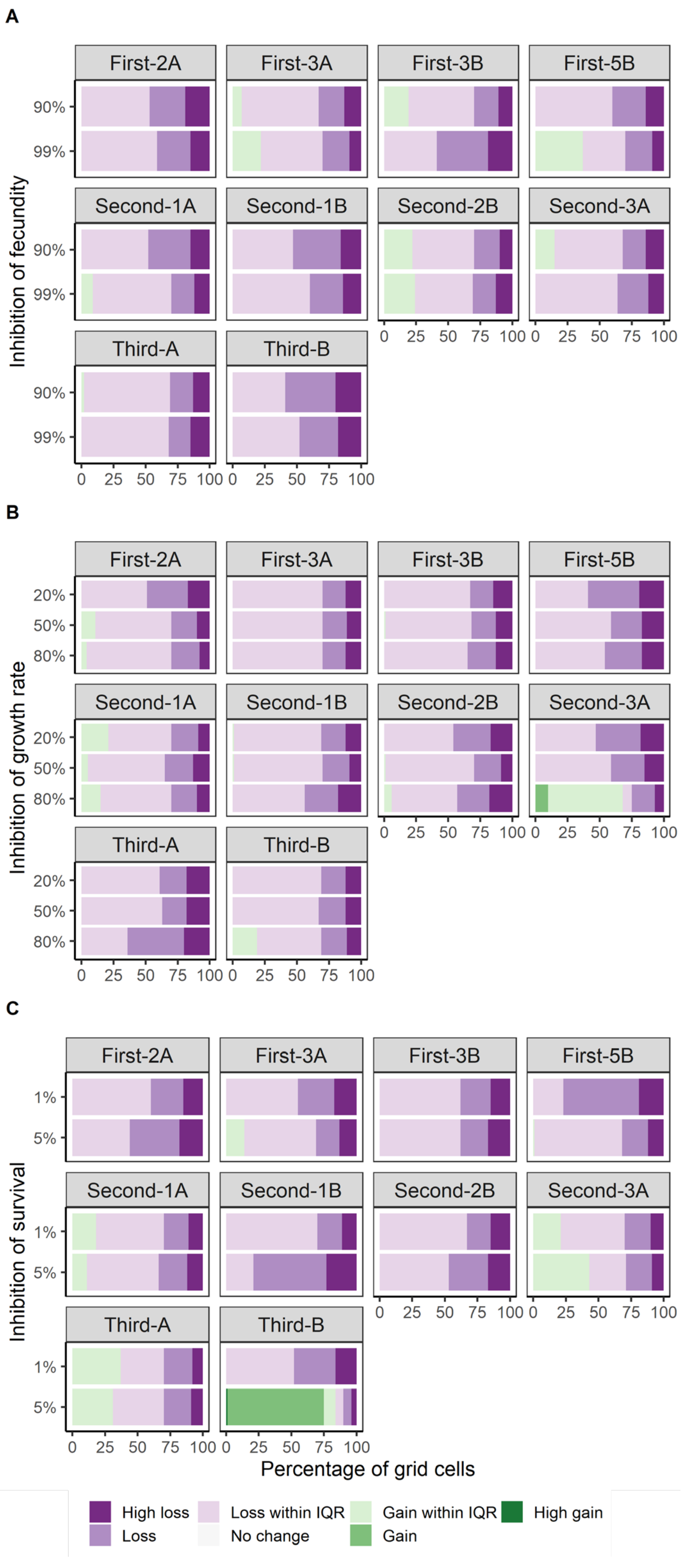


Figure 5.17 Percentage of grid cells within the landscape according to the classification (Table 5.4) of net cost (purple) and net benefit (green) where the intensity of the colour is higher cost or benefit. The grid cells are for scenario 2 (densely populated area) across inhibition of A) fecundity, B) growth rate, and C) survival in regard to the reach exposed. Categories were based on scenario 1 (baseline) to deduce the impact of angler population density on angler satisfaction. Results are from the end of the simulation period and ten replicates.

Summarising the landscape in scenario 2 showed that gain or loss in satisfaction at the grid cell level for fecundity (Figure 5.18), growth rate (Figure 5.19), and survival (Figure 5.20) were masked. The distribution of the grid cell values appeared to become more varied, especially as inhibition increased when some reaches were exposed. For example, 99% inhibition of fecundity on a first order downstream reach (3A) and growth rate inhibition in some second order reaches (1B and 3A) showed wider distribution of angler satisfaction in the landscape. Median angler satisfaction was lost across most simulations to the limits of the bottom quartile of the baseline simulation. The only landscape which gained satisfaction was a third order upstream reach (B) when survival was inhibited by 5%.

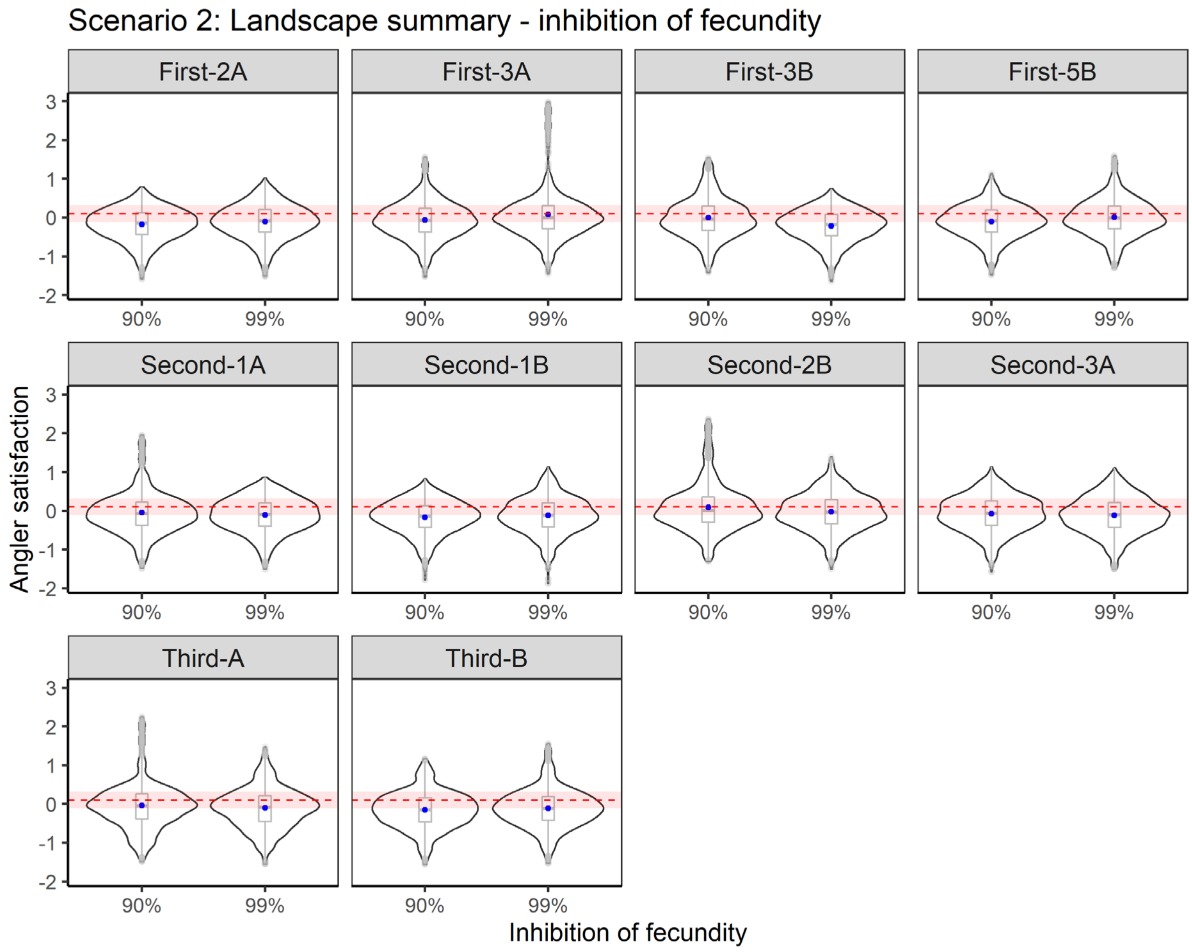


Figure 5.18 Violin plot of the distribution of median angler satisfaction (benefit – cost) across a landscape with fecundity inhibited (90 – 99%) within ten exposed reaches in a landscape with a large densely populated area. An additional internal boxplot summarising median and the interquartile range (Q1-Q3). The blue point signifies the mean value. The red dashed line is the median of scenario 1 (baseline) with no effects and the shading shows the interquartile range (for comparison). Results are from the end of the simulation period and ten replicates.

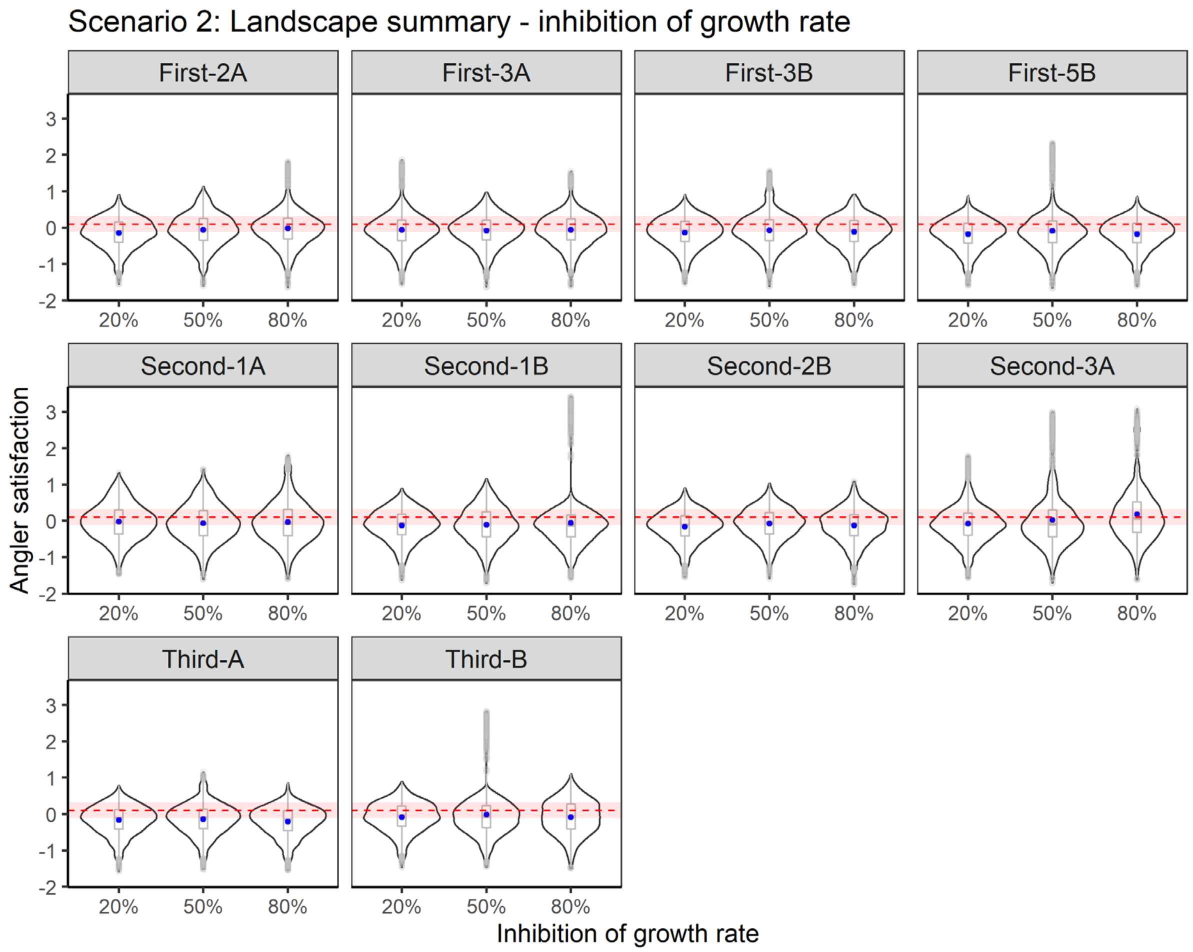


Figure 5.19 Violin plot of the distribution of median angler satisfaction (benefit – cost) across a landscape with growth rate inhibited (20 – 80%) within ten exposed reaches in a landscape with a large densely populated area. An additional internal boxplot summarising median and the interquartile range (Q1-Q3). The blue point signifies the mean value. The red dashed line is the median of scenario 1 (baseline) with no effects and the shading shows the interquartile range (for comparison). Results are from the end of the simulation period and ten replicates.

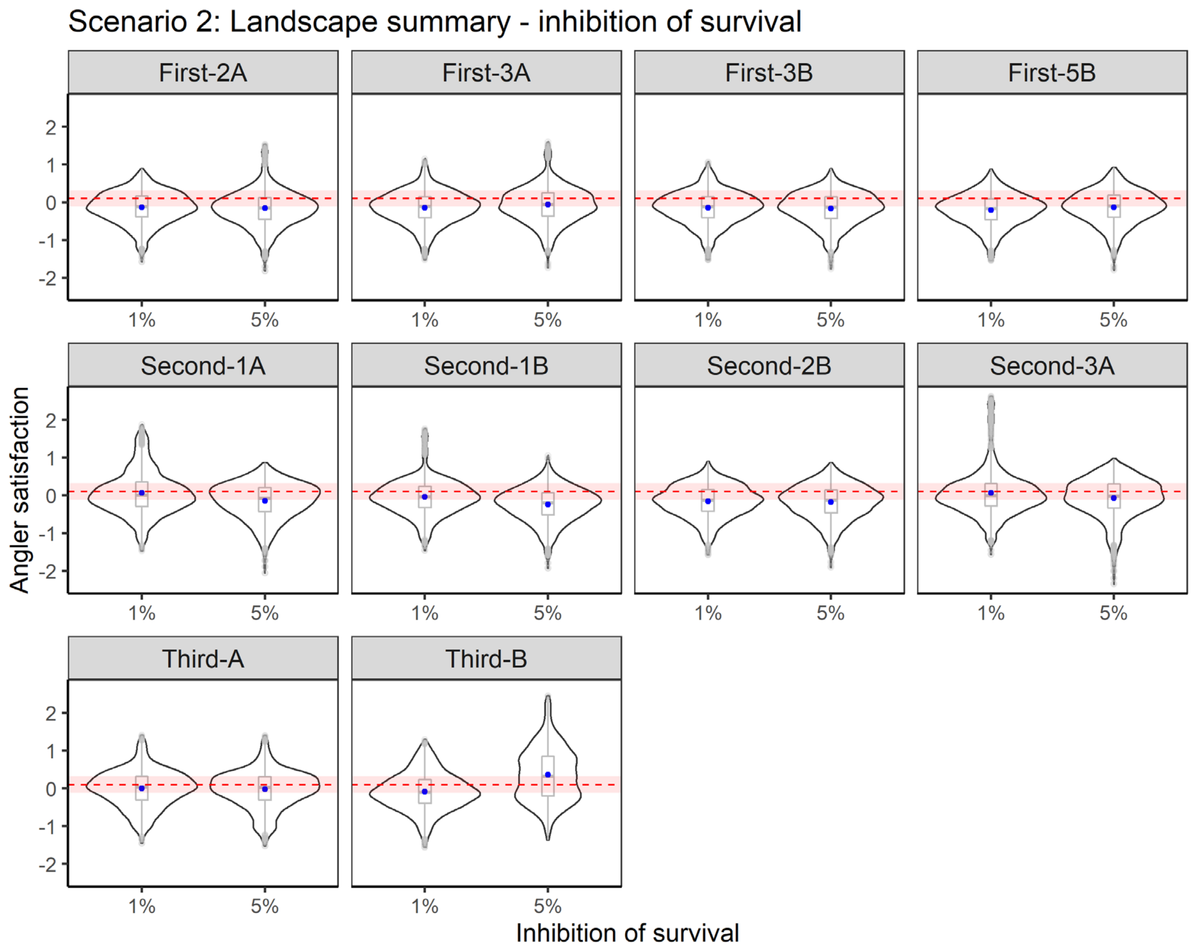


Figure 5.20 Violin plot of the distribution of median angler satisfaction (benefit – cost) across a landscape with survival inhibited (1 – 5%) within ten exposed reaches in a landscape with a large densely populated area. An additional internal boxplot summarising median and the interquartile range (Q1-Q3). The blue point signifies the mean value. The red dashed line is the median of scenario 1 (baseline) with no effects and the shading shows the interquartile range (for comparison). Results are from the end of the simulation period and ten replicates.

Scenario 3: Small sparsely populated areas

The addition of small sparsely populated areas upstream did not change the landscape pattern (Figure 5.21) as well as the summary of the landscape under inhibition of fecundity (Figure 5.22), growth rate, (Figure 5.23) and survival (Figure 5.24) compared to a uniform population density (scenario 1). The addition of sparsely populated areas was not enough to alter the patterns seen across all reaches and life history parameters targeted.

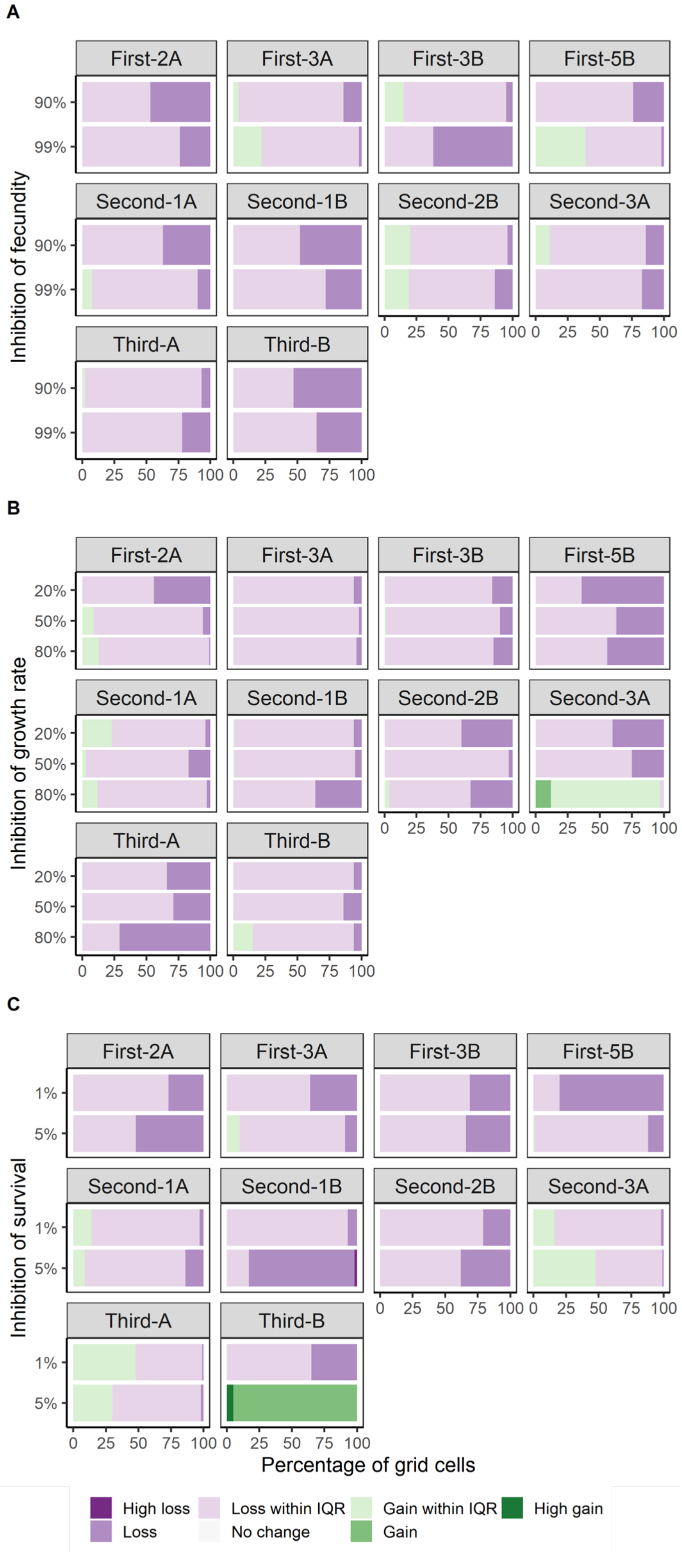


Figure 5.21 Percentage of grid cells within the landscape according to the classification (Table 5.4) of net cost (purple) and net benefit (green) where the intensity of the colour is higher cost or benefit. The grid cells are for scenario 3 (sparsely populated areas) across inhibition of A) fecundity, B) growth rate, and C) survival in regard to the reach exposed. Categories were based on scenario 1 (baseline) to deduce the impact of angler population density on angler satisfaction. Results are from the end of the simulation period and ten replicates.

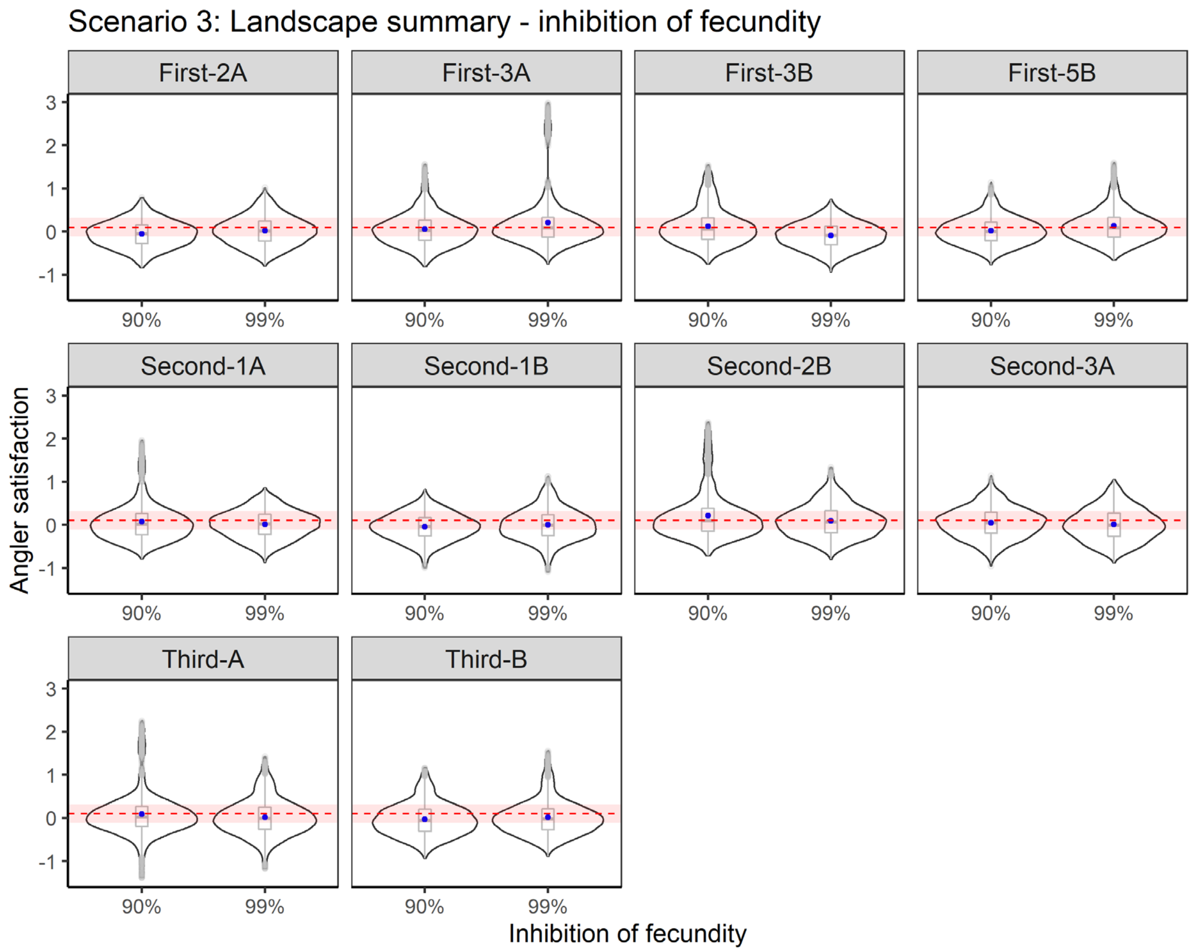


Figure 5.22 Violin plot of the distribution of median angler satisfaction (benefit – cost) across a landscape with fecundity inhibited (90 – 99%) within ten exposed reaches in a landscape with dispersed sparsely populated areas. An additional internal boxplot summarising median and the interquartile range (Q1-Q3). The blue point signifies the mean value. The red dashed line is the median of scenario 1 (baseline) with no effects and the shading shows the interquartile range (for comparison). Results are from the end of the simulation period and ten replicates.

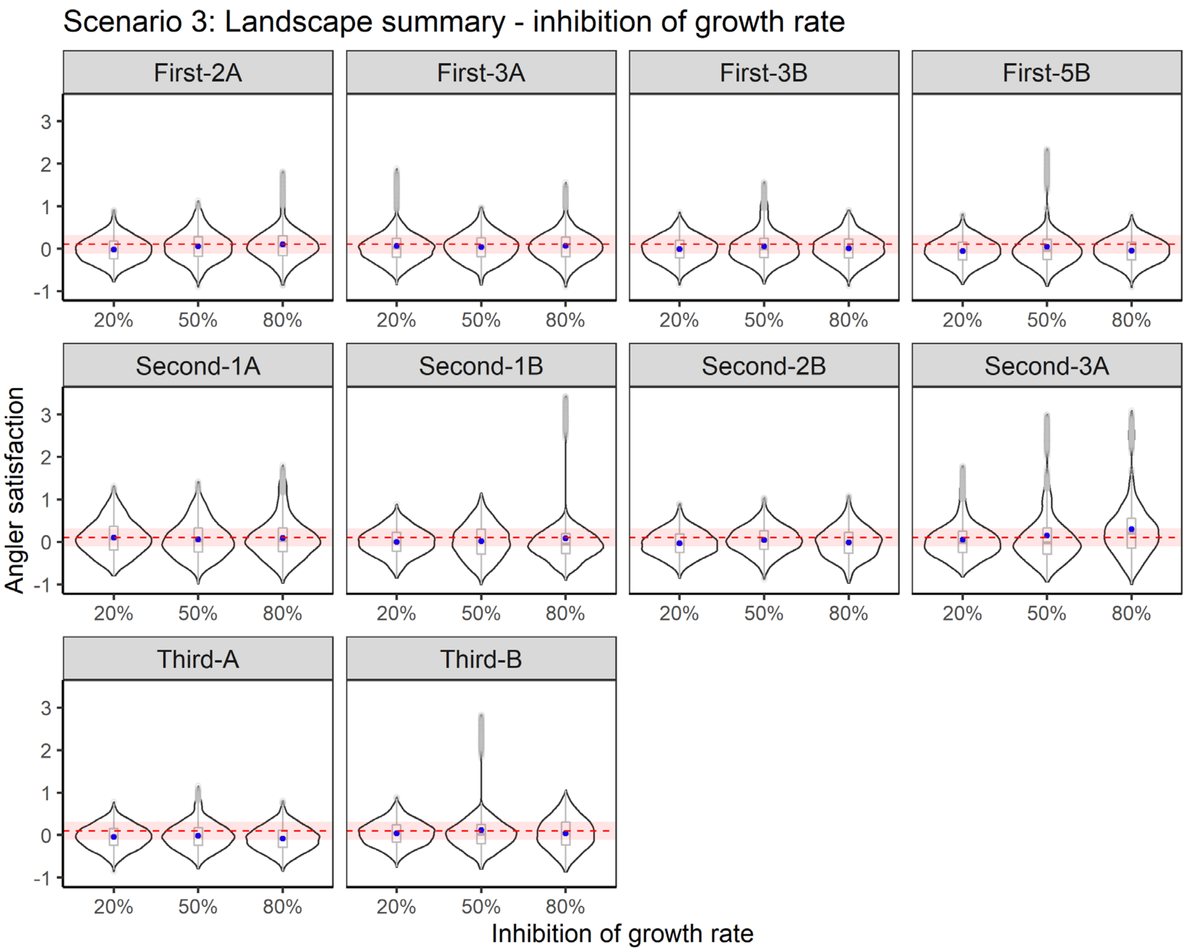


Figure 5.23 Violin plot of the distribution of median angler satisfaction (benefit – cost) across a landscape with growth rate inhibited (20 – 80%) within ten exposed reaches in a landscape with dispersed sparsely populated areas. An additional internal boxplot summarising the median and the interquartile range (Q1-Q3). The blue point signifies the mean value. The red dashed line is the median of scenario 1 (baseline) with no effects and the shading shows the interquartile range (for comparison). Results are from the end of the simulation period and ten replicates.

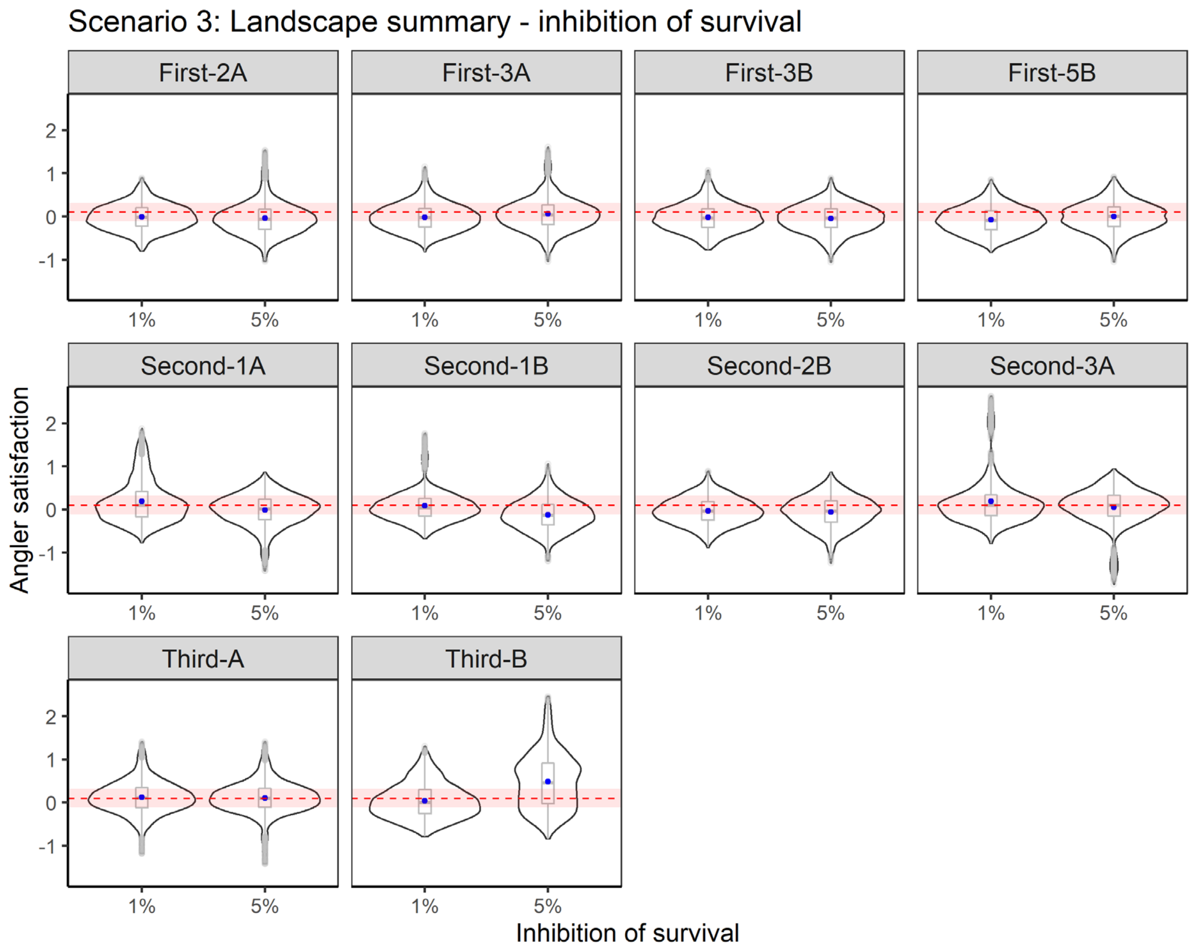


Figure 5.24 Violin plot of the distribution of median angler satisfaction (benefit – cost) across a landscape with survival inhibited (1 – 5%) within ten exposed reaches in a landscape with dispersed sparsely populated areas. An additional internal boxplot summarising the median and the interquartile range (Q1-Q3). The blue point signifies the mean value. The red dashed line is the median of scenario 1 (baseline) with no effects and the shading shows the interquartile range (for comparison). Results are from the end of the simulation period and ten replicates.

5.4 Discussion

This study aimed to answer how chemical stress on a recreational fishery and angling stock spatial patterns influenced angler satisfaction in a river network. This was done by identifying angling stock responses to chemical stress on common life history parameters in reaches simulated as part of a network or in isolation and examining where exposure in the network influenced angling stock and angler satisfaction. Angling stock simulated in a reach exposed to stress, responded differently than when simulated as part of a network depending on the reach stream order and position in the branch. Across the whole network, median angling stock remained largely unchanged as interacting populations were able to mitigate chemical effects. In the same way, angler satisfaction was lost or gained at the grid cell-scale but summarising the landscape-scale satisfaction did not show any change, except in specific scenarios of reaches exposed to stress or when there was a dense population of anglers in the landscape. Overall it is clear that spatial scale matters and is important to consider when assessing the impact of chemical stressors on ecosystem service delivery in a recreational fishery.

5.4.1 Angling stock response to chemical stress within a network versus a reach

Angling stock responded differently to chemical stress when simulated as part of a river network compared to isolated as a single reach. Reach stream order resulted in different productivity of angling stock abundance in a river network with the greatest difference related to the most productive reach. First order streams were not capable of producing rainbow trout of sufficient size to be classified as angling stock both when simulated in isolation and as part of a network. Third order reaches were the most productive for angling stock when simulated in isolation, whereas second order upstream (B) reaches were the most productive when simulated as part of a network. This suggests that the addition of habitat choice and connectivity within a river network influences where angling stock would be most abundant. The third order reaches were the largest in size and were, therefore, hypothesised to be the most suitable habitats for larger-sized fish based on simulations of the reaches in isolation. Theories have equated fish distribution as a function of individual size with habitat depth (Halvorsen et al., 2020) and stream width (Cantin et al., 2021) where a larger environment is theoretically more complex and provides more resources for fish to grow as a result of density dependence and less competition (Nevoux et al., 2019). However, angling stock preferred the middle of the river network which were not the largest reaches in the network. A previous study also hypothesised that the distribution of fish would be related to the quality and size of habitat but did not find an association between habitat length and distribution in cutthroat trout (Carim et al., 2017). This indicates that the largest site may not always be the preferred habitat for the largest fish. In this study, angling stock did not select the largest reaches as they did not need to focus on growth in the same way as juvenile fish would, where larger juveniles are usually more influenced by larger stream sites as small sites have little impact on juvenile growth (Railsback et al., 2015). Stream order is often used, as a surrogate for habitat characteristics, to predict the location of ecological components (Gilvear et al., 2016). However, the findings of this current study show that predictions based on stream order may not be transferable to the network-scale. When reaches were connected, the angling stock responded differently due to movement and habitat selection. This reflects that reach status does not necessarily result in the same status at the catchment-level (Kuemmerlen et al., 2019).

Reach position within the stream branch as either upstream (B) or downstream (A) was also important for angling stock productivity. For second order reaches, a downstream position was less productive than upstream, because downstream reaches neighbour larger and more preferred third order streams with more habitat choice. In contrast, the upstream reach was located next to smaller first order streams, making it the more favoured choice. This was observed both with and without chemical stress. Second order streams flowing downstream into streams third order or higher have been found to support more diverse fish communities compared to those flowing into other second order streams, a consequence of fish moving into tributaries (Shao et al., 2019). This possibly explains the differences observed in the branches. In other words, the upstream location near a tributary increases the abundance of fish entering the upstream second order reach in this study simulation (as opposed to diversity). However, for other stream orders, location in the branch was not as important.

Third order streams were the most productive when simulated in isolation as a reach because they were the largest habitat. When simulated as part of a network, the reach productivity was the same and only when a life history parameter was inhibited, did the location of the third order reach in the branch then matter. This could be because the downstream reach had only one neighbouring reach to mitigate effects of chemical stress whilst the upstream reach had support from both third and second order reaches. In combination, these effects mean that angling stock can withstand substantial impacts of stressors in a river network due to interacting populations and trout movement not present in an isolated reach. Upstream (moving with the flow) populations are more likely to interact with downstream (against the flow) reaches (Cooke et al., 2022), especially larger fish which are more capable of swimming further and upstream, except in the event of migration. Migration is not currently modelled in inSTREAM but studies on other salmonids, have found that most upstream movement in potamodromous brown trout occurred between October to December (García-Vega et al., 2022) after the census date (September 30th) so would not influence the abundance monitored in this thesis.

This method of simulating both parts of a river and the whole river has been applied in other studies to a main channel stream to identify habitat “hot spots” or the most productive habitats (Railsback et al., 2015). In addition, stream characteristics are often used to predict distribution and abundance of trout (Cantin et al., 2021). However, the simulations here showed that impacts on angling stock in a reach simulated in isolation did not directly translate to the network-scale. Instead, a more appropriate method would be to consider both scales in an assessment, incorporating the important ecological principles which can explain the difference in results between scales (Kuemmerlen et al., 2019). Kuemmerlen et al. (2019) found that assessments on the reach-scale might be in a “good” state but the whole river network state could be “bad” as the status of one reach does not reflect the overall state of the entire catchment. The same has been shown in this study, that on a reach-scale angling stock is more at risk to chemical stress targeting growth rate and survival, however, on a network-scale the risk is lowered as angling stock is more likely to persist in exposed reaches. This allows a more effective conclusion to predict the spatial patterns exhibited by angling stock in a river network.

5.4.2 Angling stock dynamics under chemical stress in a river network

Although chemical exposure was localised to specific reach, changes in angling stock was seen across the river network. The effects of chemical stress on life history parameters were visible across the river network as the summary of angling stock in reaches changed in response as well as variation across replicates. The median angling stock abundance across the network was mostly dependent on the life history parameter inhibited and at times the reach location. For inhibition of fecundity, angling stock abundance was generally seen to reduce with high levels (99%) of inhibition in first and third order streams, but this pattern was not as clear for second order streams. The high angling stock abundance in second order streams suggests that these streams may play a key role in supporting overall angling stock abundance across the whole network. For inhibition of growth rate, the increasing level of growth rate inhibition resulted in an increase in angling stock in the river network when first order streams were exposed, but decreased when third order streams were exposed. This may indicate the availability of resources in these reaches, where inhibited growth rate in smaller streams supporting smaller fish reduces competition for resources for larger fish in the rest of the network (Nevoux et al., 2019). Finally, for inhibition or survival, angling stock abundance decreased with high (5%) levels of inhibition across all stream orders. Spatial patterns of fish populations in recreational fisheries have been seen in response to temperature and water flow, but chemical contaminants have been highlighted as an additional stressor to investigate (Li et al., 2018). A previous study exposed an entire network to endocrine disruptors on trout fecundity but did not consider that the impact can change depending on where exposure occurs in the network (Forbes et al., 2019). The findings of this current study are novel as angling stock dynamics under chemical stress across a network have not been previously simulated.

Angling stock life history strategy of potamodromy (i.e. movement) or residency (i.e. staying within their habitat) are driven by a need for energy and resources and are influenced by environmental factors (Birnie-Gauvin et al., 2021). Although trout in inSTREAM cannot sense the presence of chemical stress as might be possible in nature (Cooke et al., 2022), they can respond to the impact stress has on the population size structures in exposed reaches and make decisions to remain or leave the exposed reach. Therefore, angling stock movement reflects their response to stressed reaches in the river network (Section 5.3.2 *Angling stock exposure time*). As angling stock includes the largest fish in the simulation they are also capable of moving furthest (Railsback et al., 2022). The time angling stock spent exposed to chemical stress gives an indication of angling stock choices as either residency or movement between reaches. For example, angling stock time in exposed first order reaches increased with increasing inhibition of growth rate. This suggests that they were exposed during their early life-stages, as angling stock abundance was low to none in first order streams. It also indicates that fish, which eventually become part of the angling stock, remained resident in first order reaches for longer as growth was slower but resources remained the same. This was not as expected as slower growth would usually encourage potamodromy as it takes fish longer to mature at slower growth (Kendall et al., 2015). However, upstream movement can be size-selective and change this size-age maturity response and indicate why fish remained resident (Ferguson et al., 2019). In addition to this, young trout with reduced growth have been shown to utilise a poor environment through behavioural changes such as early sexual maturation (Nevoux et al., 2019). Residency is a common behaviour when growth is chosen as a behavioural choice and under growth rate inhibition the size structure of present populations would be smaller and although they grow slower, there is reduced competition for available resources. On the other hand, movement is a choice for behaviour, such a spawning, where potamodromous populations return to spawning grounds between autumn and winter (García-Vega et al., 2022). However, under chemical exposure, there was no difference in the time angling stock spent in a reach whether chemical stress was present or not. This could be reflected by the time of spawning in the model (April – May) and the census day used (September 30th) (Railsback et al., 2022) or that the size structure did not change significantly for trout to respond. Potamodromy is also influenced by survival where movement can improve chance of survival (Cooke et al., 2016; Nevoux et al., 2019). Although the time angling stock spent in reaches exposed to inhibition of survival decreased, this could also be as they were more likely to die.

Movements between reaches also show that the river network is not acting as isolated populations but as an interacting populations (Section 5.3.1). Angling stock spatial structure reflected the spatial structure of the entire trout population as a hybrid metapopulation form of both patchy and interacting populations (Ovaskainen & Hanski, 2004; Falke & Fausch, 2010). This was clear as angling stock hatched more in distinct reaches (i.e. sources) which were different to the reaches where angling stock was most dense (Pulliam, 1988). Angling stock density also reflects that the population is not uniform throughout the river network and distinct populations of angling stock are present within the reaches. As all reaches were occupied, although first order reaches did not have more than one angling stock, this strongly suggests a patchy metapopulation model (Falke & Fausch, 2010). Examining the metapopulation concept in response to impacts of stressors has been considered for chemical risk assessment especially in regards to the spatial scale of effects as population-level approaches do not consider possible spatial patterns (Galic et al., 2010; EFSA Scientific Committee, 2016b; Rohr et al., 2016).

Predicting impacts to angling stock in exposed reaches, where at the reach-scale impacts would be expected to be detrimental such as inhibition of survival leading to extinction events (Section 5.3.2, was not the same as stress was mitigated at the network-scale. Although reduced, a stable population of angling stock persisted in reaches under chemical stress simulated as part of a river network across the simulation period. This mimics what is expected in nature where toxicants might damage local populations but effects are mitigated by connectivity and movement from neighbouring populations, or in this case angling stock in neighbouring reaches (Ares, 2003).

Not only do interacting populations in river networks mitigate effects of chemicals but river networks are known to reduce pollution of receiving water bodies through the retention of sediment and nitrate (Karki et al., 2023). This is due to the functional and structural connectivity of the river network as well as habitat availability across the network (Shao et al., 2019). Dispersal pathways for fish species are strongly influenced by branches of the river system which, in turn, affect the dynamics of interacting populations. For example, fish diversity tends to increase with larger stream sizes due to an increase in habitat variability and size (Shao et al., 2019). Assessing water quality, such as chemical pollution, is dependent on the arrangement of reaches which cannot be determined when assessing isolated reaches (Schuwirth, 2020). In terms of the modelling approach applied here, only one type of network structure was simulated: a dendritic network structure. Using branching connections in river network models, such as in a dendritic structure, has been shown to have greater metapopulation persistence (Larsen et al., 2021). Branching reflects the availability of tributaries which support abundance of fish in the river network and they differ in terms of habitat providing unique refuge habitats from the main stem of a river (Shao et al., 2019). A previous study has evaluated the influence of the branching type (i.e. simple versus complex) as well as implementing several stream orders finding that network structure can influence population dynamics (Larsen et al., 2021). It is therefore important to consider that the placement of reaches and branches used in this current study with a simple branching network can also influence the results of this study.

5.4.3 Angler satisfaction across the surrounding landscape

Stress induced inhibition of life history parameters mostly resulted in reduced angler satisfaction in the grid cells surrounding the river network. But this loss was not observed when summarising the median satisfaction across the whole landscape. This signifies that the overall satisfaction in the wider catchment remained unchanged as there was not enough loss to change the median angler satisfaction. Unlike angling stock, angler satisfaction was not as responsive to inhibition of life history parameters on the individual as angling stock only made up one factor in assessing satisfaction. Changes in satisfaction were, however, observed due to specific combinations of life history parameter and the reach exposed. For example, exposing a third order downstream reach (A) with inhibition of survival at 5% led to overall gain in the landscape satisfaction. In contrast, higher angler population density was associated with a high loss in a percentage of the landscape. Ultimately satisfaction was dependent on benefit and cost where factors in this assessment determine the supply (angling stock) and demand (angler population density).

A survey on the non-market values associated with angling in England found that anglers preferred fishery sites with medium- to large-sized fish (Environment Agency, 2018). The most beneficial fishery sites were those with the highest density of angling stock. These are the sites with the best supply for anglers which were second order streams and under most chemical stress scenarios these remained the most favourable highlighting why chemical stress on individual trout generally did not reduce satisfaction. Utility has been used as a metric for assessing angler benefit and usually in relation to angler effort (Golden et al., 2022). The benefit – cost analysis applied here for angler satisfaction addresses both catch rate and angler effort as a function of satisfaction. However, incorporating fish population dynamics through the likelihood of catching a large fish and total abundance helps to determine the subsequent size structure of the fishery stock. Even in trophy anglers, with intentions to catch a large fish at a site, catching a fish will still contribute to satisfaction (Birdsong et al., 2021) and although the focus was on trophy, in reality anglers are diverse in their behaviour and their angling goals (Ward et al., 2013).

Angler density highlights the demand for angling sites within the landscape where demand for angling is spatially variable (Cantin et al., 2021) as shown by the three angler population density scenarios and their influence on angler satisfaction. The main effects of angler population density are observed when there is a substantial concentration of anglers resident in a particular area, rather than numerous smaller clusters of anglers distributed across the landscape. However, fishery quality is typically better in remote areas and poorer near densely populated areas (Post & Parkinson, 2012). The possible crowding resulting from higher density of anglers travelling to the fishery sites increases the angler effort or cost in those sites especially as urban anglers are more likely to extend to fishing areas in rural areas (Arlinghaus et al., 2008). Remote areas or populations of smaller sizes were not enough to influence any changes in additional loss of satisfaction. These spatial patterns indicate supply and demand assessments which can be used for spatial prioritisation and to guide fisheries management (Kuemmerlen et al., 2019; Cantin et al., 2021).

5.4.4 Implications of findings for risk assessment

It is important to consider where chemical impacts are occurring in a river network and to understand how spatial scale influences the assessment of risk to ecosystem service delivery. By examining the influence of spatial scale on simulations of chemical stress on a recreational fishery, it is clear that impacts vary across spatial scales. In this study many different scales have been adopted for angling stock and angler satisfaction including the reach – network – landscape grid cell – landscape. If the median angler satisfaction (benefit – cost) for the entire landscape is used to summarise angler satisfaction, in general, no overall changes in angler satisfaction were observed. In contrast, if the percentage of the landscape which resulted in a gain or loss in satisfaction is used to summarise angler satisfaction, changes in angler satisfaction were identified at the local or grid-cell level. Therefore, whereas across the whole landscape, angler satisfaction is balanced by both loss and gain, certain areas within the landscape are impacted. This observation is very important when deciding which scale to use in a risk assessment especially as spatial scales tend to be predefined (Paetzold et al., 2010; EFSA Scientific Committee, 2016b; Wu et al., 2022).

Risk assessment of recreational fisheries requires identifying the spatial scale of the fishery (Peterson & Dunham, 2009). Chemical exposure such as agricultural applications can vary greatly along a river network. Therefore, it is crucial to identify productive reaches and locations most at risk to exposure along a network in addition to angler satisfaction. This can be problematic as shown with the results, where the spatial scale of assessment produced different outcomes. This is also true of fisheries management, where restrictions could be localised to a segment of a river (or lake) or have licenses available along a whole stretch of the river network (Arostegui et al., 2021; O'Higgins et al., 2020; Pitman et al., 2019). Considering the riverscape is essential for population assessment and fisheries management as local, small-scale assessments do not capture the large-scale habitats and processes that are vital for population migration (Cooke et al., 2016). However, from this study the riverscape can also mask local stress events, which might be the site of an important fishery. Many freshwater ecosystems are evaluated at the local or reach-scale for habitat properties to identify impacts from local surroundings but there is incentive to consider the wider network (Kuemmerlen et al., 2019). The proof of concept applied here can be used for both ecological risk management and also to fisheries management.

5.5 Conclusions

This study aimed to answer how chemical stress on a recreational fishery and angling stock spatial patterns influence angler satisfaction in a river network. Chemical stress on a recreational fishery resulted in spatial patterns visible across angling stock populations in the river network and angler satisfaction in the wider landscape. But the chemical stress impacts on angling stock and angler satisfaction observed were dependent on the spatial scale of assessment. For example, angling stock responded differently to chemical stress on common life history parameters (growth, survival, and fecundity) when simulated as part of a network as opposed to as a single reach in isolation. It was clear that the spatial scale of chemical stressor assessment on angling stock influenced the results as isolated reaches could not mitigate impacts, whereas interacting populations present in reaches simulated as part of a network could. The study also investigated where chemicals targeting common life history parameters in fish impact angling stock dynamics and angler satisfaction across the network had some impacts such as on location upstream or downstream in a branch as well as stream order. The location of chemical stress in a river network was important for angling stock dynamics. However, this did not necessarily translate to angler satisfaction, where the biggest impact on satisfaction was the presence of a highly dense population of anglers in the surrounding landscape. This highlights the need to consider the spatial patterns of not only the SPU in the ecosystem services concept in risk assessment but also the spatial patterns of beneficiaries. This study also shows evidence that the impacts of chemical stress do not translate across scales where stress is more apparent at the local reach-scale but the impacts at the river network and landscape-scale are not as visible. Risk assessments is typically done for a predefined spatial scale; however, this study emphasises the advantages of conducting both local and broader-scale assessments. Together these assessments can complement each other and provide insights into the level of protection required for the SPU at both the reach and river network scales as well as for the beneficiaries in the landscape.

Chapter 6

General discussion: spatial patterns and chemical risk assessment

6.1 Main findings

The ecosystem services concept has promising applications in ecological risk assessment from better defining SPGs, providing a more ecologically relevant risk assessment, and connecting the impact of pressures to benefits for end users (Maltby et al., 2018; van den Brink et al., 2021). However, many ecosystem services like recreational fisheries exist on multiple spatial scales from a reach to the wider river network (Food and Agriculture Organization of the United Nations, 2012; Arlinghaus et al., 2017). This thesis addresses the question of how the effects of chemical stressors on ecosystem service providers at local scales relate to the delivery and risk assessment of ecosystem services at the catchment-scale, using a recreational fishery as a case study. Overall, I found that the impacts of chemical stress on angling stock and angler satisfaction were variable at the local scale whilst at the catchment-scale, the presence of neighbouring reaches could mitigate impacts showing that impacts on the reach-scale did not directly translate to impacts at the catchment-scale.

In the first half of this thesis, I approach the first thesis objective 1) to establish what ecosystem services tools and methods are currently available for freshwater catchments which could be used to apply an ecosystem services concept in ecological risk assessment (Chapters 2 and 3). I identify what tools were available for applying the ecosystem services concept in ecological risk assessment (Chapter 2) by conducting a rapid evidence assessment. I then assessed the tools found from the rapid evidence assessment and their capability of assessing different spatial scale and pressures, with a specific focus on chemical pollution (Chapter 2). It was clear from the results of a rapid evidence assessment that the literature on tools for chemical stress on ecosystem service delivery is limited as eight studies out of 332 studies assessed chemical pollution and three tools were applied in more than one study (Chapter 2). For assessing chemical stress, no common method was found and a decision-tree was used to select the most appropriate for ecosystem service assessment. Results from the review contrasted findings from databases on tools and ecological production functions (EPFs), highlighting the importance of conducting reviews to identify the current state of literature, especially as some tools are deprecated. Then, using one of the approaches found in Chapter 2, I adapted the individual-based model inSTREAM. I then coupled this with an assessment on angler satisfaction as a method to assess the effects of chemicals on individuals to ecosystem service delivery (Chapter 3)

The second half of this thesis applies the method developed in Chapter 3 to determine spatial patterns of angling stock and angler satisfaction on the reach and catchment-scales (Chapters 4 and 5, respectively). Spatial patterns were identified by investigating the two thesis objectives 1) how the effects of chemical stressors on fish life history parameters (e.g. growth rate, fecundity, survival) and fish prey abundance affect angling stock and subsequent angler satisfaction (Chapter 4) and then 2) how chemical stress on a recreational fishery and angling stock spatial patterns influence angler satisfaction in a river network (Chapter 5). Spatial patterns important in risk assessment were identified such as isolated populations versus interacting populations and in relation to local anglers versus anglers in the catchment. On the reach-scale, growth rate and survival inhibition impact the provision of angling stock, but angling stock is highly resistant to chemical stress on individual fecundity (Chapter 4). In contrast, impacts were not as prominent within the river network in response to stress applied at the reach-scale. From the landscape perspective angler satisfaction is only impacted with areas of higher angler population density (Chapter 5). This is because the spatial structures and population dynamics such as connectivity and interacting populations mitigate the impacts in the landscape (Chapter 5).

In this chapter, I will address the thesis objectives by synthesising results from the previous chapters around the two themes of i) spatial patterns and recreational fisheries (Section 6.1.1) and ii) adapting ecological models to apply the ecosystem services concept in ecological risk assessment (Section 6.1.2). I will then address the final thesis objective by considering the implications of the thesis findings for ecological risk assessment by discussing how to implement spatial patterns in a recreational fishery into chemical risk assessment and decision-making (Section 6.2).

6.1.1 Spatial patterns and recreational fisheries

The spatial scale of assessment was important in investigating spatial patterns in a recreational fishery. For example, on a reach-scale the effects of chemical stressors on angling stock and angler satisfaction was dependent on the fish life history parameter (e.g. growth rate, fecundity, survival) or indirect effect (e.g. prey abundance) inhibited. Whereas on a catchment-scale, the effects of chemical stressors on angling stock patterns and angler satisfaction was dependent on the spatial structures and patterns of angling stock and anglers in a recreational fishery. Overall, the spatial patterns of angling stock and angler satisfaction observed in a recreational fishery under chemical stress were dependent on the life history parameter inhibited, the spatial structures of rainbow trout in the fishery, the location of chemical exposure in the river network, and the angler dynamics in the catchment.

For both the reach and catchment-scale assessments, chemical stress on angling stock is influenced by the particular life history parameter inhibited in trout individuals (Chapters 4 and 5). This is a potentially important finding in that it may be important to know what chemical stressors target in individuals to predict how their effects will be seen at the population, or angling stock, level. This aligns with other simulation studies which found population-level effects were dependent on life history target of chemicals (Mintram et al., 2018a; Accolla et al., 2019; Vaugeois et al., 2020). However, the angling stock response to inhibition of life history parameters is dependent on the spatial scale of assessment, as impacts were more prominent at the reach-scale than the catchment-scale. Angling stock is able to recover from chemical stress at the reach-scale due to angling stock resilience and mitigate the impacts at the river network-scale due to habitat connectivity and movement (Chapters 4 and 5). Fish adaptive capacities such as resistance, resilience, and recovery, as examined in Chapter 4, extend beyond biological levels of organisation and, therefore, across spatial scales (Pesce et al., 2023). This is so as the fish behaviour and life history parameters influence how they interact with their environment (Peterson & Dunham, 2009). For example, fish will remain resident in order to grow, but move for spawning and if ability to grow or spawn are inhibited due to chemical stress their responses can also be impacted as a result (Leathe et al., 2014). Without the option to remove themselves from an isolated reach, angling stocks are confined to the stressed habitat and the population resilience and resistance is tested. On a catchment-scale, literature suggests that fish survival, movement, and life history is dependent on habitat conditions (Hayes et al., 2009). With the options of other habitats, angling stock movement can respond to chemical stress within the river network.

Habitat connectivity and fish movement mitigated the impact of chemicals on angling stock in a river network. For example, inhibition of survival and growth impacts on angling stock was reduced in a reach exposed to chemical stress when simulated as part of a network compared to in isolation, with the exception of when fecundity was inhibited and angling stock resistance was the same. In general, factors which influence sensitivity to chemical stress vary as physiology, chemical potency, and exposure scenario are important at the individual-level, whilst fecundity, density dependence, and environmental conditions are important at the population-level (Mintram et al., 2018b; Accolla et al., 2019). In other words, at the reach-scale in an isolated population, the fish physiological response to stress is more important to angling stock sensitivity as seen by their response to the life history parameter inhibited (Chapter 4). But at the river network-level, sensitivity in rainbow trout varies at population-level dependent on factors which are influenced by the spatial scale such as size distribution of angling stock and habitat connectivity in the case of this thesis. In contrast, in a river network, individual-level variation leads to movement and distinct structures of populations dependent on the reaches and interactions which was only possibly to observe in a river network (Radinger & Wolter, 2014; Cooke et al., 2016; Nevoux et al., 2019; Cantin et al., 2021; Cooke et al., 2022). This was seen as the median angling stock in the network did not change substantially in response to chemical stress on fecundity, growth rate, and survival in individuals, but was dependent on the reach exposed and level of stress so impacts were not limited to the stressed reach and mitigated by interacting populations. Connectivity reflects trout movement across spatial scales related to residency, dispersal, and migration (Cooke et al., 2016). Movement was also monitored when reaches were exposed showing that there were changes in the way fish moved around the river network as a result of chemical stress. In the field, with little evidence of fish movement, modelling has increased in use to instead predict movement (Cooke et al., 2022) which was only possible to monitor at the network-scale (Chapter 5). Changes in movement reflect the interactions between reach populations in the network as a result of habitat connectivity which allow the impacts of chemical stressors on angling stock abundance to be mitigated across the network.

The impacts of chemical stress on angling stock spatial structures was also dependent on the location of chemical exposure in the river network and did not translate between scales. For example, stream order and location of chemical exposure as either upstream or downstream within a branch influenced the angling stock response in second and third order reaches, where each branch and stream order are otherwise environmentally identical (Chapter 5). Furthermore, as seen in Chapter 5, although angling stock displayed similar sensitivities and resilience to inhibition of common life history parameters, angling stock dynamics were not consistent when a reach was simulated in isolation compared to as part of the network. River network topology can influence the provision of ecosystem services where a catchment's structure is an important factor in the spatial distribution of ecosystem services (Karki et al., 2021, Larsen et al., 2021; Bertassello et al., 2022; Karki et al., 2023). This aligns with a previous study scaled local scale population dynamics to the landscape-scale for rainbow trout in a catchment and found that increasing stream order increased the quality of stream habitat (Cantin et al., 2021). The river network structure and the exposed reach both influenced the angling stock spatial structures.

In relation to angling stock spatial structures, the structure of the network drives habitat connectivity and interactions between populations. Literature suggests junctions of tributaries provide particularly favourable habitats for a variety of aquatic species (Karki et al., 2023). The productivity of reaches reflected this difference between the spatial scales as on a network-scale, productivity was highest in the middle of the network closest to a tributary (Chapter 5) but on a reach-scale, it was in the highest stream order (Chapter 4). Network structure influenced habitat choice of angling stock on a network-scale, whilst at the reach-scale, reach productivity of angling stock is dependent on habitat provision

(Chapter 4). These angling stock spatial patterns arose from reach placement in the network as rivers themselves are considered connected pathways with many possible geometric shapes (Karki et al., 2021; Larsen et al., 2021). Here the angling stock in the wider network was able to respond to chemical stress in an exposed reach. Angling stock is influenced by the river network structures, so future considerations on the influence of other river network structures on the spatial patterns and productivity of angling stock could be beneficial.

Spatial patterns of angler satisfaction highlight recreational anglers' interactions with fish in a recreational fishery. These interactions are impacted by angler behaviour and skill as well as fish behaviour and productivity (Harmon et al., 2018) and are dependent on the distribution of anglers and fish abundance within a landscape (Post, 2013). Fisheries themselves can also be linked through angler mobility as anglers interact with local fish angling stock (Arlinghaus et al., 2017). Chapter 5 showed that angler distribution influenced angler satisfaction across the landscape when a large densely populated area was simulated. Anglers have been known to choose a fishery site based on minimising travel time, resulting in fishing occurring nearer to densely populated areas and in poorer quality fishery sites (Post & Parkinson, 2012). On a landscape-scale, grid cells close to large populations of anglers suffered reduced satisfaction due to the additional cost of increased competition (Chapter 5). This is also reflected in literature where demand for urban recreation is driven by population density and not the quality of recreation service itself (Andrew et al., 2015). When observing angler satisfaction at a river network-scale, a location within the landscape, or landscape-scale, all concluded varying levels of satisfaction across the scales (Chapter 5). These angler spatial patterns highlight "hot spots" or areas of net benefit and net cost which could be useful to consider in management.

Recreational fisheries need to consider the spatial heterogeneity of aquatic systems in landscapes, as highlighted in Chapters 4 and 5 (Shao et al., 2019; Cantin et al., 2021). Especially as recreational fisheries can be managed over larger landscapes but managing fish species is usually done on smaller spatial scales focusing on specific fish populations and the relationship between fish and their habitats (Rahel & Nibbelink, 1999; Cantin et al., 2021). Fisheries management does not regularly apply landscape-patterns as much as wildlife habitat management does (Cantin et al., 2021). Ecologists have also identified the need for analyses spanning scales and levels of organisation, particularly in relation to extrapolating individual to system dynamics (Grimm & Railsback, 2012). This is as larger scale outcomes are predicted to be a result of multiple local scale interactions between anglers with angling stock and their habitat (Arlinghaus et al., 2017). However, Chapters 4 and 5 showed that there is not a direct translation between scales because reach-scale results did not directly translate to network-level results as interacting populations and connected habitats helped to mitigate the impacts of chemicals. Therefore, spatial patterns of anglers and fish should be considered together in fisheries management as opposed to disconnected assessments of anglers at the landscape-scale (visitor numbers, travel distance) with local scale assessments of the reach for habitat quality and specific populations. Overall, the spatial patterns identify the resistance and resilience of a recreational fishery to chemical stress as reflected by both the catch and non-catch-related aspects of angler satisfaction and can determine if a fishery is vulnerable to collapse (Post, 2013).

6.1.2 Adapting ecological models to apply the ecosystem services concept in ecological risk assessment

Ecological modelling has been promoted as a method to not only extrapolate from toxicity test data to service providing units (SPUs) (Maltby et al., 2021) but to help reduce the gap between measurement and assessment endpoints in ecological risk assessment, more specifically between SPUs and ecosystem services (van den Brink et al., 2021). Modelling also limits the need for costly experimental data and animal testing (Rohr et al., 2016). Ecological models such as individual-based models are well suited for risk assessment as they are capable of extrapolating from impacts measured from individual-level data, readily available from standardised tests, to the field (Grimm & Martin, 2013; Ibrahim et al., 2014). However, previous reviews have investigated the inclusion of population-level ecological models in chemical risk assessment and found that several were not developed with risk assessment in mind but have the potential to be used for that purpose (Galic et al., 2010). Adapting ecological models utilises a suite of tools not originally intended for chemical risk or ecosystem service assessments (Chapter 2). Adapting tools for risk assessment, such as mechanistic population models, in particular individual-based models, is possible through the inclusion of environmental parameters, species-types, and habitat quality; and stressors either related to environmental factors or to chemical stressors themselves (Schmolke et al., 2018; Forbes, 2024). In this thesis, reviewing literature found that models were limited and did not extrapolate effects on the individual to the beneficiary (Chapter 2). Therefore, I selected and adapted a model to develop a method to investigate the spatial patterns of a recreational fishery.

On the basis of modelling a recreational fishery as a case study, a decision tree suggested modelling interactions between nature and society with individual-based models (Harrison et al., 2018). Individual-based models are also capable of extrapolating effects from the individual to the population-level and widespread use in chemical risk assessment (Forbes et al., 2008; Galic et al., 2010; Forbes & Calow, 2012; Grimm & Martin, 2013; Martin et al., 2014; Forbes et al., 2019). For these reasons, inSTREAM (Railsback et al., 2022) was selected and adapted to include chemical stress on common life history parameters of fecundity, growth rate, and survival across spatial scales. Similar adaptations of inSTREAM to model chemical stress on individual trout growth rate and fecundity were also developed by Hazlerigg et al. (2023) and in an older version of inSTREAM (Forbes et al., 2019) but the code for these adaptations was not publicly available from previously published studies or was hosted on a different platform, respectively. Therefore, the work in this thesis was done independently. Adaptations to target chemical stress were done by inhibiting fecundity, growth rate, and prey density through relevant sub-models or model parameters already present in inSTREAM. For survival, inSTREAM only considers survival from high temperature, stranding, low condition, terrestrial predation, and fish predation as well as egg survival in redds. A new sub-model needed to be added in order to simulate survival from chemical stress and was the first instance of adding survival from chemical stress to inSTREAM. Furthermore, inSTREAM can utilise existing ecotoxicological data for specific exposure scenarios by simulating a known chemical stressor as done in previous applications (Forbes et al., 2019; Hazlerigg et al., 2023), but this was not the approach used in this thesis. Instead a hypothesis-based approach was applied to test the sensitivity of fisheries to serve as the first step towards adapting a model for application in risk assessment (Kramer et al., 2011).

One of the positives of adapting models such as inSTREAM for use in chemical risk assessment is that it reduces experimental data collection on animals. Current risk assessment relies on animal testing (Marx-Stoelting et al., 2023) and the approach applied here in this thesis contributes to the reduction of animals used in testing (Directive 2010/63/EU), especially reducing animal tests on vertebrates such as fish. Regarding fish testing under REACH regulation, the standard Fish, Early-life Stage Toxicity Test (OECD 210) requires about 700 fish per test where rainbow trout is a recommended test species. As 15 – 25% of substances have required an OECD 210 test, estimates of fish for this test alone range between 0.77 to 1.29 million fish (Rovida et al., 2023). Here, in the research for this thesis, no sentient fish were used to test for the effects of chemical pollutants on populations upward of 1000 fish in each replicate of ten and their generations through the full 19-year simulation period as well as in over 100 experiments.

Guidance on good modelling practices has been published to encourage model development for regulatory purposes (European Food Safety Authority, 2014) but few models are currently approved for risk assessment (e.g. plant protection products, Larras & Charles, 2022). There have been many efforts in literature to promote their application in risk assessment (Accolla et al., 2021) as well as the production of guidance such as Pop-GUIDE to assist risk assessors with model use and development (Raimondo et al., 2021). From a regulatory perspective, there is currently guidance related to specific mechanistic models, such as the toxicokinetic/toxicodynamic (TK/TD) models or population models for bees (BEEHAVE) and reptiles and amphibians (Larras et al., 2022), but these have yet to be extended to population models for other species. A study assessed the actual use of models in plant protection product registrations and found that the effects of only a few active substances have been modelled with population models (Larras et al., 2022). So, although promising, the applications of models in risk assessment are scarce (Forbes, 2024). To add to this, population models are applied less in risk assessment than when compared to other fields where they are widely applied (e.g. conservation, fisheries, environmental management) and therefore risk assessment relies on adapting models used for other purposes (Larras & Charles, 2022). However, as guidance on applying population models is lacking (Forbes, 2024), this can lead to inconsistencies in (population) model adaptation and implementation (Hazlerigg et al., 2023). In the case of adapting inSTREAM to use in risk assessment, this has required adapting inSTREAM for each study which has applied the model (Forbes et al., 2019; Hazlerigg et al., 2023) and this thesis, which can result in inconsistencies between applications.

Validating ecological models, especially adapted versions, is a concern for risk assessors (Forbes, 2024). Some individual-based models, such as inSTREAM, are developed with a pattern-oriented modelling approach, an important tool to improve confidence in model predictions. Pattern-oriented modelling is also an accepted form of model development for good modelling practice in risk assessment (European Food Safety Authority, 2014). Pattern-oriented modelling makes bottom-up modelling more comprehensive by focusing on essential information about complex systems (Grimm et al., 2005) and develop mechanistically correct representations (Gallagher et al., 2021). This is important as it implements observed patterns across different levels of biological organisation as opposed to related to direct correlations (Grimm et al., 2005; Gallagher et al., 2021). Patterns are not only important in model development but are also produced by simulations, such as the spatial patterns assessed in this thesis and population patterns of response to life history parameters in various reaches. In other words, although pattern-oriented modelling is used in designing, selecting, and calibrating models, it can also be used as a method to assess existing models to identify important processes and play a role in making predictions (Grimm & Railsback, 2012; Grimm et al., 2020b). For example, spatial patterns can be utilised to test simulation outputs spatiotemporally (Gallagher et al., 2021). Spatial patterns from Chapters 4 and 5 are not only useful for risk assessment but for the adaptation of inSTREAM, understanding what spatial patterns occur as a result of model mechanisms. For example, the similar angling stock patterns observed across different reaches in response to life history parameter inhibition and spatial patterns related to chemical exposure location in a river network.

Despite the limited adoption of individual-based modelling approaches in risk assessment, there are few notable successes, one of which is BEEHAVE, a honeybee model which simulates colony behaviour. BEEHAVE shows a promising future for adapted models in risk assessment as the model was adapted to include the impacts of pesticides for risk assessment in the module BEEHAVE$_{\text{ecotox}}$ (Preuss et al., 2022). In the same way as BEEHAVE, inSTREAM (version 7.3) (Railsback et al., 2022) is freely available online, regularly updated, and well-documented to better help users and risk assessors understand the model. InSTREAM is published with guidance to encourage model use in risk assessment such as an Overview, Design concepts and Details (ODD) protocol (Grimm et al., 2020a) which facilitates model evaluation (Forbes, 2024). InSTREAM is a great candidate for the development of a ecotoxicology module as it has already has sister models, inSTREAM-Gen for investigating evolutionary dynamics and the impacts of angling (Ayllón et al., 2016) and inSALMO for migratory salmon (Railsback, 2021). It could also become an integral part of next-generation risk assessment and has already been suggested for use in risk assessment (Forbes et al., 2019; Hazlerigg et al., 2023) as has its sister model inSTREAM-Gen has been used for risk assessment of copper (Janssen et al., 2024). For BEEHAVE, the $\text{BEEHAVE}_{\text{ecotox}}$ module was added following criticism to a lack of an ecotoxicology module and validation of model additions (Preuss et al., 2022). This is also a concern with inSTREAM as currently users are required to add their own adaptations to simulate chemical stress which do not necessarily align between applications. This thesis takes the first step towards developing an ecotoxicology module for a previously non-chemical assessment tool by applying sensitivity testing of the angling stock at various spatial scales. Adapting inSTREAM to have an ecotoxicology module would allow it to be a part of future of standardised ecological models used in risk assessment (Forbes, 2024).

6.2 Implementing spatial patterns into chemical risk assessment and decision-making

Given the complexities of considering chemical impacts of ecosystem service delivery at the catchment-scale, what prospect is there for taking these into account in risk assessment? Specifically, how do these complexities bear on decisions which are made for risk assessment concerning selection of relevant scales, species, endpoints, and beneficiaries, as well as how to consider their spatial patterns.

6.2.1 Selecting relevant scales for ecosystem service assessment in risk assessment

Recreational fisheries not only exist across spatial scales themselves (Chapters 4 and 5) but are dependent on the underlying mechanisms which are all at their own various scales. For example, angling stock is spatially structured as habitats support populations, populations interact across a catchment, and recreational angler effort is heterogenous across the landscape (Food and Agriculture Organization of the United Nations, 2012; Mee et al., 2016). It has been suggested that landscape structure, function, and processes should be coupled with ecological risk assessment to identify overlaps between the various scales relevant for a fishery (Lin et al., 2021). However, whilst spatial structure and landscape patterns are sometimes considered separately in ecosystem service assessment through landscape planning and ecology, respectively (Torres et al., 2021), this thesis and other studies have highlighted that landscape and spatial structures of ecosystem components are important (Lin et al., 2021). Simulating the landscape and different scales is important as a functioning ecosystem is dependent on interacting populations in a landscape (Kramer et al., 2011). Landscape structure of recreational angler potential was simulated from different angler population densities within the landscape (Chapter 5) whilst spatial structures of populations (Chapter 4) and interacting populations (Chapter 5) in the reach and river network, respectively, were determined by angling stock changes. As spatial distribution of ecosystem service delivery is often determined by environmental factors and supply and demand of ecosystem services is determined by human factors (Wei et al., 2017; Lin et al., 2021), this also encapsulates the socioecological nature of a recreational fishery. Selecting the appropriate spatial scale for risk assessment is important in order to consider all parts relevant for ecosystem service delivery.

Concerning scales there is also the added influence of temporal framing of the scale. Risk assessment can be either retrospective, evaluating risk after stressors are known or suspected in the environment (i.e. reactive), or prospective, evaluating risk before stressors enter the environment (i.e. proactive) (Hope, 2006). However, there is a shift for assessments to be more predictive in order to cover wider spatial and temporal scales (Pesce et al., 2023). Although the assessment applied in Chapters 4 and 5 used simulations with historical data, the assessment can be used to predict potential risk of chemical stress to ecosystem service delivery. Additionally, chemical and ecological data is rarely collected in landscapes, so historical data is not always available for retrospective analyses. Within a regulatory context, there is an interest in basing prospective risk assessment on modelling (Larras & Charles, 2022). Simulations can still allow predictions based on historical or hypothetical data to assess the risk to future provision of ecosystem services.

6.2.2 Selecting relevant simulation species and target endpoints in modelling

Risk assessment of environmental contaminants typically relies on standardised testing with relevant test species. Testing is usually with the most sensitive and biologically relevant species to derive predicted no-effect concentrations (PNEC) (Benfenati, 2022). These sensitive species are used as a proxy in standardised toxicity testing in order for ecological modelling to be useful in risk assessment (Devos et al., 2019). Furthermore, selecting a species to model in risk assessment has its difficulties as these can be dependent on available data from either laboratory or field studies as well as ensuring the species is ecologically relevant (Reed et al., 2016). However, literature suggests that species should be selected based on their vulnerability; including not only species sensitivity but susceptibility to exposure and population resilience (Ibrahim et al., 2014; Maltby et al., 2021). Standardised test species used in laboratory tests may not always be the most vulnerable species to apply in modelling (European Food Safety Authority, 2014).

Fish are good ecological indicators from an ecological and ecotoxicology perspective as they can be monitored and are long-lived (Burnett et al., 2021). From a risk assessment perspective, rainbow trout is a standardised test species and considered an SPU for plant protection products (European Commission, 2024), but recreational fishery stock can either be other game fish species as well as coarse fish (Curtis & Breen, 2017). A previous study investigated population resilience of European fish species to identify the least resilient species to pesticide exposure compared to life history parameters (Ibrahim et al., 2014). Rainbow trout are not native to Europe so were not included in the study, but brown trout were included. Brown trout were not identified as the least or most resilient for any life history parameters of fecundity and survival, indicating that compared to other species it would not necessarily be the most vulnerable species. However, brown trout were less resilient to fecundity than to adult survival. Although not included in this previous study, rainbow trout have been introduced into European rivers and are a recommended test species for risk assessment (European Chemicals Agency, 2008). As suggested in Chapter 4, rainbow trout could have been the most tolerant species to inhibition of fecundity compared to brown trout and greenback cutthroat trout in previous studies so might not be the most vulnerable to chemical stressors on fecundity (Forbes et al., 2019). When rainbow trout were used within these simulations (Chapters 4 and 5), angling stock populations were resilient to inhibition of fecundity, but were sensitive to survival, similar to brown trout. InSTREAM is capable of modelling other salmonid species such as cutthroat trout and brown trout and another version of the model, InSALMO (Railsback, 2021), can simulate salmon. Furthermore, there are other individual-based models already developed to include chemical stress which can be applied to European standard test species such as the three-spined stickleback (Mintram et al., 2020a). The methodology applied in this thesis can be used to assess the resilience and sensitivity of other relevant game fish species to identify the most vulnerable species for either game (inSTREAM) or coarse fish recreational fishery in a certain location (Ibrahim et al., 2014). Comparing simulations of various salmonids in inSTREAM and other individual-based models for fish can determine the population structures and spatial patterns of relevant fishery species to then decide the most vulnerable species used for risk assessment depending on, for example, the chemical mode of action.

Possible life history parameters, which are endpoints for chemical stress, should also be evaluated when identifying a species as vulnerable (Maltby et al., 2021). For example, in this thesis, the common life history parameters selected as targets for chemical stress in simulations, fecundity, growth rate, and survival (Chapters 2 – 5) all expressed varying vulnerability of rainbow trout when inhibited. One of these, fecundity, which rainbow trout was particularly resistant to, is an important parameter as reproduction is the most commonly applied sublethal endpoint in standardised testing (Martin et al., 2014). Here, fecundity or the number of viable eggs (Chapter 3), is the specific reproductive output used although other relevant mechanisms were present in inSTREAM for reproduction. Fecundity is an important target endpoint as it reflects both the reproductive health of individuals as well as underlying mechanisms such as growth and development (individuals must be mature to reproduce) (Martin et al., 2014). Growth rate and indirect impacts through prey density were selected as they both influence an individual's ability to grow within its habitat. Finally survival forms the basis of forming laboratory toxicity tests to determine lethality of a chemical (Focks et al., 2014). Therefore, all endpoints considered are relevant to risk assessment (Chapter 3). InSTREAM does also have the additional capacity to investigate other impacts such as fish behaviour, which can be difficult, costly, and time-consuming to examine in laboratory or field conditions (Ford et al., 2021). Individual-based models allow simulation with a hypothesis-based testing approach, by testing the sensitivity of fishery species and endpoints which will not only allow more focus assessment but reduces the need for animal testing (Kramer et al., 2011). This results in not only a more ecologically relevant assessment on vulnerable species, but also can determine the vulnerability of fishery species and their impacts dependent on specific mode of actions of chemicals they might be exposed to.

6.2.3 Selecting relevant beneficiaries: the recreational angler

Selecting a relevant measure of the SPU in a recreational fishery can be framed in various ways from the angler focus to the fisheries management. This thesis is framed from the perspective of a trophy angler in a catch and release fishery with rainbow trout (Chapters 3 – 5) where the focal SPU is angling stock. Many different measures can be used to assess the quality of angling stock in a recreational fishery. In the case of a trophy hunter, the length and age of the fish were the most important, but if the aims of the angler are different then the angling stock can be redefined. For example, weight is another factor that is used for assessing catch quality where not the largest but the heaviest are preferred (Ropars-Collet et al., 2021). Although not applied in this thesis, fish weight is a relevant measure of catch success for trophy anglers and is possible to measure from inSTREAM outputs instead of large fish related to length. Another example, abundance, often used in stock assessments of fisheries, could have still been a relevant endpoint for example, for a non-trophy angler who is more concerned with the number of fish caught (Food and Agriculture Organization of the United Nations, 2012). As in this thesis, considering a subset of the population has been done in a previous study with inSTREAM on adult fish (Forbes et al., 2019). Quantifying the abundance of adult fish does not consider the size structure of adult fish, which can grow smaller under some forms of stress, but it is a useful metric to consider the health of the adult population in a fishery.

Beneficiaries also have an added influence on the recreational fishery which need to be considered in assessment. For example, anglers have different fishing practises which can apply additional pressure or mortality. A catch and release fishery was modelled in this thesis which regulates fishing style but can lead to hooking mortality, although the practise is often promoted to encourage the potential of trophy fish growth by releasing fish and allowing them to continue to grow (Sass & Shaw, 2020). Further research including mortality due to catch and release practises would be a next step to see if it would alter any of the impacts seen in the results of this thesis. Although mortality from catch and release considered in previous studies (Ayllón et al., 2016; Forbes et al., 2019) still resulted in simulating stable fish populations. Furthermore, anglers place additional pressures as they compete for fishery sites either due to crowding or input from urban areas which can lead to race-to-fish practises, or short-term competition, and decrease the overall quality of fishing for anglers and satisfaction (Ropars-Collet et al., 2021). For example, if a fishery has a short fishing season with angler overcrowding, there is competition for anglers to race-to-fish before the season is finished. Although this was not explicitly simulated in this thesis, the principle is the same as including population density in the angler satisfaction calculation (possible angler demand) for nearby sites. Overall, anglers are diverse and have different interests which are important to consider. However, in this thesis I have simplified angler interest. A study using machine learning found that angler decisions can be predicted based on fish size – though in this case it was across species – but this was not the only factor in angler decisions to harvest (Kaemingk et al., 2020). Kaemingk et al. (2020) found that urban versus rural as well as spatial pattern of anglers and water bodies were all important factors for anglers. So although anglers are a diverse group, there are certain predictors which are consistent between anglers (e.g. fish length, number of fish caught, and distance travelled) and these were applied within this thesis.

Another finding is the importance of considering the non-monetary value for beneficiaries. Incorporating non-monetary valuation of ecosystem services was only established relatively recently (Torres et al., 2021). As a result, there has also been a shift from economic to more cultural perspectives. The ecosystem services concept is still evolving, but more recent typologies such as nature's contributions to people (NCP) consider culture and communities more directly (Ellis et al., 2019). Non-monetary valuation and focusing on cultural perspectives can take fisheries management information as well as chemical standardised testing data and compile them to identify influences within a spatially explicit landscape scenario. In this thesis, I have applied a non-monetary valuation with hypothetical scenarios and testing model responses (Chapters 4 and 5), the proposed way to incorporate impact on ecosystem services (Benfenati, 2022). If the assessment in this thesis had been focused on monetary valuation, only elements of angler satisfaction could be valued, for example, the angling stock and travel distance could instead be quantified as travel cost (Pitman et al., 2019; Birdsong et al., 2021). A monetary valuation would apply alternative socioeconomic assessments such as stated preference (e.g. surveying willingness to pay) and revealed preference (e.g. observed behaviour) (Wainger & Boyd, 2009). However, these valuations do not express the utility of anglers and both the catch-related and non-catch-related factors are important for anglers as included in angler satisfaction (Food and Agriculture Organization of the United Nations, 2012; Birdsong et al., 2021). For relevant ecosystem service assessments, the impact on beneficiaries needs to be considered. A previous review found that only 49% of ecosystem service assessments considered the benefits of ecosystem services from monetary value with 7% of assessments considering non-monetary value (Mandle et al., 2021). The study found that value (both monetary and non-monetary) was assessed by considering supply and benefit separately with ecological production functions (EPFs) and socioeconomic utility functions (SUF), respectively (Mandle et al., 2021). That approach was taken in this study: the assessment of supply (SPU) is based on ecological modelling with inSTREAM but post-processing of angler satisfaction can also be known as the SUF measuring angler satisfaction. These two translations are crucial in ecosystem service assessment and both need to be included in ecological risk assessment of services, which often only considers the EPF.

6.2.4 Implementing spatial patterns into risk assessment

Current risk assessment typically predefines spatial scales (Paetzold et al., 2010; EFSA Scientific Committee, 2016b; Wu et al., 2022). For SPGs, EFSA suggests the relevant ecosystem services, SPUs, and level of protection (spatial scale of biologically relevant effects for potential stressors) are defined in relation to spatial recovery of non-target organisms (EFSA Scientific Committee, 2016b). However, in this thesis, I have shown that when predefining a spatial scale for assessment, the scale choice can affect the results.

A solution to this would be to combine spatial scales in risk assessment. Rohr et al. (2016) evaluated the advantages and disadvantages of data from varying levels of biological organisation and suggested that both a lower-level (e.g. individual) and higher-level (e.g. landscape) should be included in screening. Addressing multiple scales can help to identify overlaps between ecological risks and ecosystem services (Huang et al., 2023). As shown in this thesis it is important to consider both the wider landscape-scale assessment as well as the habitat-level of the target SPU. Adopting a similar approach in risk assessment for spatial patterns of ecosystem services would allow what differs between scales to be highlighted. Moving from only site-based risk assessment to landscape-level risk assessments is more ecologically relevant as it incorporates the influence of spatial scale (Rohr et al., 2016) and modelling the landscape relates exposure to ecology such as populations (Benfenati, 2022). However, the landscape-scale cannot relate effect to lower levels of biological organisation, such as the individual, a lower-level assessment is needed as well at a smaller spatial-scale. Considering different scales, integration of models across scales applied in the same analysis has also been suggested as a consideration for risk assessment (Chen et al., 2013). Integrating and adapting methods (Chapters 2 and 3) can provide an alternative to the current one model – one outcome in risk assessment (Maltby et al., 2021; Qian et al., 2023).

Another solution to carrying out risk assessment at both spatial scales, would be adopting a tiered assessment approach where spatial scales are considered in a stepwise approach, similar to tiered risk assessments in which population-level or higher-level effects are only considered in risk assessment if there is unacceptable risk to individuals (Kramer et al., 2011). The results from Chapter 4 concluded that on a reach-scale there are unacceptable effects to angling stock across multiple levels of inhibition of several life history parameters, indicating the need for a higher-tier assessment at a larger spatial-sale. Chapter 5 builds onto this identified risk at the reach-scale to then address how this translates to the angling stock in a river network and to the anglers in the landscape. This would not only address multiple spatial scales but can address where effects are most prominent and what is best protected. In the case of the thesis, the level of protection would be at the reach-scale. This is because population-level effects were seen on a reach-scale both when simulated in isolation and as part of a network but angling stock abundance across the river network remained stable, suggesting that local effects are masked at the larger scale. The reach-scale would be required to protect, especially if reaches exposed to chemical stress are the most productive for angling stock.

However, current approaches in chemical risk assessment do not consider the reversibility or dynamic nature of systems and assume impacts result in permanent changes (Rohr et al., 2016). In reality, many species have the ability to recover to a certain extent whether that is within the population or within a landscape with the influence of emigration and immigration of individuals. The approach within this thesis has shown both these capacities for recovery in a population or interacting populations and alternative ways in which ecological risk assessment can be improved with ecological modelling methods not yet commonplace in risk assessment. But addressing the dynamic nature of spatial patterns and structures still needs to be considered in risk management and decision-making.

6.2.5 Implementing spatial patterns into risk management and decision-making

Ultimately, risk assessment assesses and provides scientific advice for risk management (Benfenati, 2022). Going beyond risk assessment is crucial when considering the risk to ecosystem service delivery where other factors not currently considered are important. These factors include trade-offs of ecosystem services, non-monetary valuation, supply and demand, and the long-term provision of a service (Maes et al., 2012; Maltby, 2013; Wolff et al., 2015; Cord et al., 2017; Z. Wang et al., 2021). Although information from risk assessment, whilst primarily directed towards risk managers, is important, other stakeholders are impacted by pollution in a recreational fishery. These include fisheries management, management across spatial scales such as local communities, regional, or national, and environmental management as well as the beneficiaries, which all have an influence on the fate of a recreational fishery (Food and Agriculture Organization of the United Nations, 2012). Chen et al. (2013) stated that risk assessments are concerned with ecological systems but risk evaluation and decision-making is concerned with socioecological conditions. As discussed, socioecology methods are important for ecosystem service assessment especially to capture the non-monetary value to beneficiaries. In addition to this, socioecological systems allow the potential to address trade-offs (Pesce et al., 2023) which are important in ecosystem service delivery. Integrating both ecological and socioecological assessment and going beyond risk assessment will be necessary to better utilise the findings of this thesis in the context of decision-making.

A way that this can be utilised is by considering a recreational fishery under chemical stress as a dynamic system by characterising supply and demand (Bagstad et al., 2013b; Andrew et al., 2015). A supply and demand ratio is a useful indicator of spatial mismatches or imbalances within a location (Z. Wang et al., 2021). Supply and demand also addresses the ecosystem service flows from provision to beneficiaries or better yet the spatial dynamics of ecosystem services (Bagstad et al., 2013b). Quantifying the supply of cultural services is dependent on biophysical and social indicators such as species abundance, travel time, and distance to service (Wei et al., 2017). Demand, on the other hand, is often related to use, preference, and perception (Wei et al., 2017). In this thesis, I have addressed how ecosystem service delivery in a fishery is utilised by beneficiaries as well as demand as a result of angler population density (Chapter 5), but further research would be needed to quantify and highlight the supply and demand. Addressing supply and demand is important as it can determine the sustained provision of the recreational fishery. However, provision alone does not necessarily indicate the quality of a recreational fishery and methods to maintain provision of ecosystem services are scarce (Andrew et al., 2015). Therefore, an ecosystem service-based risk assessment coupled with the supply-demand ratio can apply the resulting spatial patterns to categorise the quality of the fishery (Z. Wang et al., 2021). These spatial patterns can be related to the fishery SPU themselves (angling stock) or the anglers (angler satisfaction) as well as supply and demand patterns. As I have addressed population, interacting populations, and socioecological dynamics in a recreational fishery, this would involve taking the spatial patterns of these to determine dynamics of the recreational fishery.

Incorporating supply and demand with spatial patterns, recreational fishery dynamics can be used to determine the long-term provision of a fishery that is exposed to chemical stress. Chemical stress can result in low fish productivity or local extinction events in recreational fisheries but they are also subjected to other pressures such as overfishing in areas of high angler population density or demand for fishing (Cantin et al., 2021). This balance between identifying which areas are most at risk to chemical stress or the spatial patterns and the supply and the demand of the fishery for anglers is crucial to understanding a recreational fishery as a socioecological system (Mee et al., 2016; Solomon et al., 2020). Social and ecological processes have been poorly integrated in the past which has led to lack of understanding the dynamic nature of recreational fisheries (Post, 2013).

Moving from assessing ecosystem service supply as a static pattern to a dynamic one is novel and has been applied only previously to forest ecosystem services (Snäll et al., 2021). Ecosystem service dynamics include the ecosystem, SPU, and management or regulations acting on the ecosystem services. In the case of recreational fisheries, which have fishery season impacts, angler and angling impacts, and overall exhibits a dynamic state, this would integrate supply and demand as well as management changes by considering the dynamics of the service. In this way, the spatial patterns identified for the underlying changes in angling stock for recreational fisheries, as well as the benefit and costs, can be integrated further to develop a sound image of the recreational fishery dynamics to include management.

6.3 Conclusion

This thesis shows that understanding spatial patterns in ecosystem service assessment is important to determine the impacts of chemical stress on angling stock. As observed, impacts on angling stock and anglers are more variable at the local and reach-scale based on chemical stress targets in the individual as impacts are diffused and possibly mitigated due to interacting populations in the river network. Angler population density in the wider landscape only impacted angler satisfaction when simulating a highly dense area whilst sparse and spread out populations did not change satisfaction. This thesis has identified the need to address spatial patterns and the varying conclusions that can be drawn in relation to selecting spatial scales for chemical pressures within a recreational fishery. However, future research is also needed to determine the most vulnerable species of angling stock and address angler diversity and network structure to see how these impact the spatial patterns seen in this thesis.

With an ecosystem service assessment applying socioecological methods to understand the benefits to the angler, there is also an importance to assess the quality and the long-term provision of a fishery under chemical stress. Future work can couple the results of this thesis with an assessment of supply and demand (e.g. supply and demand ratio) to address the quality of the fishery under chemical stress, “hot spots”, and the dynamic nature of the recreational fishery. Experts in both risk assessment and decision-making should consider various spatial scales of assessment and spatial patterns of SPUs and beneficiaries as they represent the dynamic nature of underlying components to ecosystem service delivery.

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Appendices

8.1 Appendix A2: Supplementary tables and figures for Chapter 2

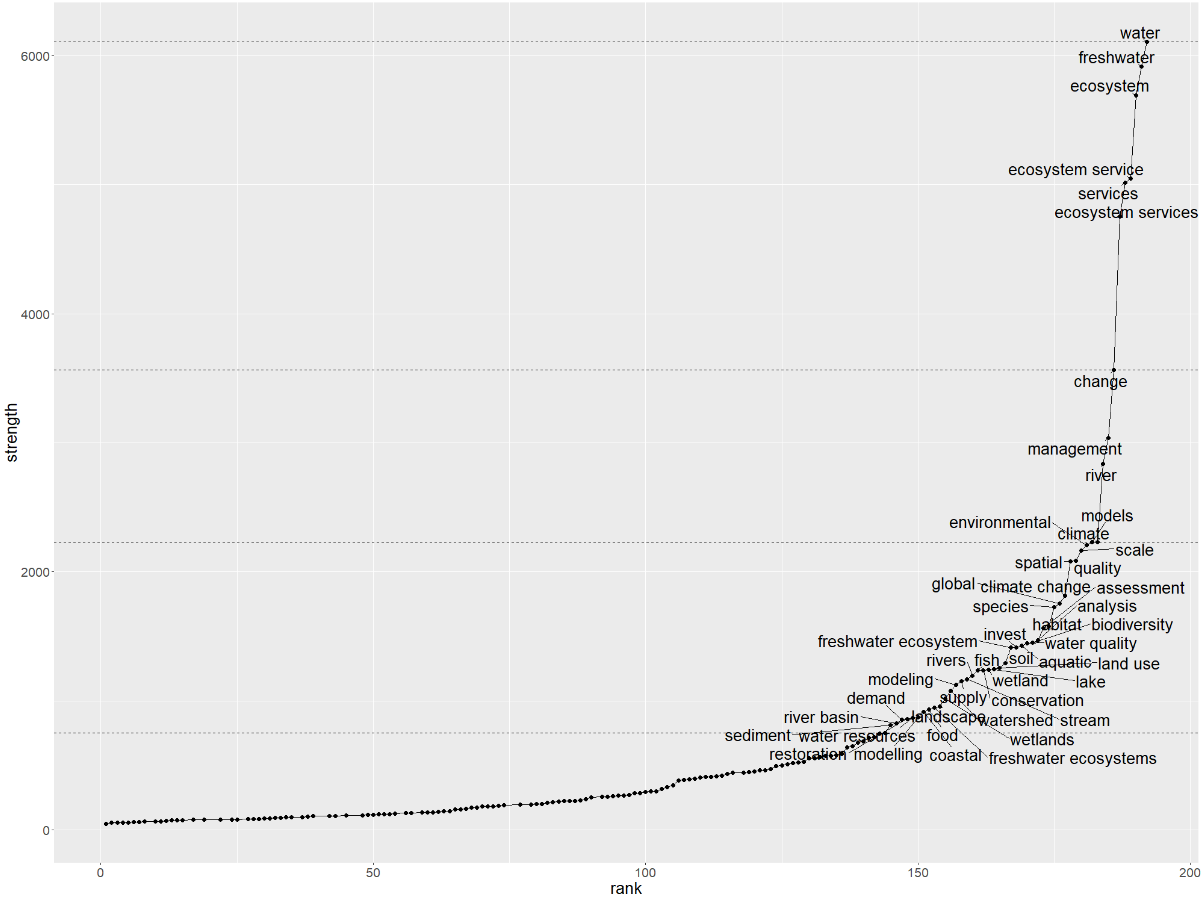


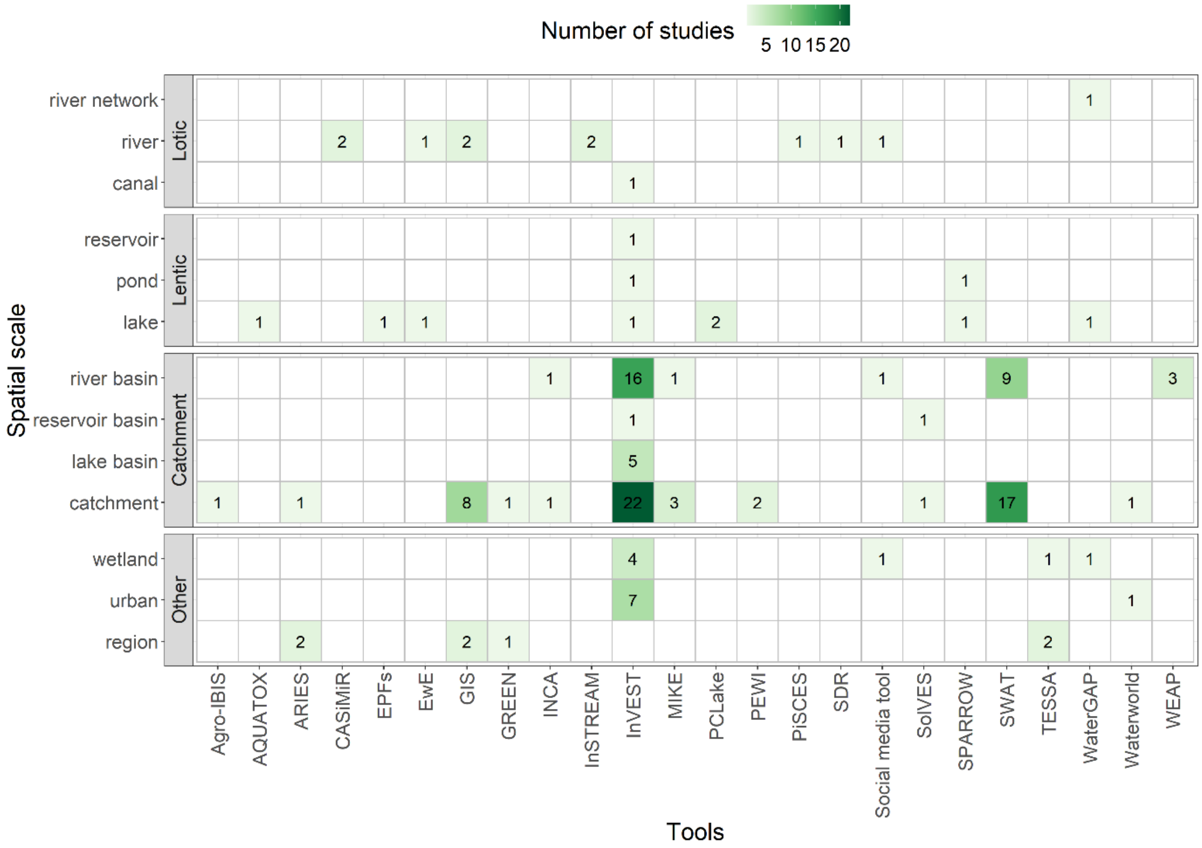
Figure A2.1 LitSearchR package in R (Grames et al., 2019) was used to generate an automated search string from keywords found with a naïve search in SCOPUS (ecosystem services AND freshwater AND models). LitSearchR scanned the title, abstract, and keywords from 327 literature and identified words ranked by a co-occurrence network across literature. The highest ranked keywords are shown here by strength or in other words their relative importance across the naïve literature search. Keywords are shown for those above the first cut-off threshold (where the result strength changes) with over > 752 strength (used for the construction of the automated search term). Some irrelevant terms such as "coastal" were returned. After removing these terms the suggested search string predicted with LitSearchR is ("ecosystem services") AND (biodiversity OR fish OR food OR habitat OR sediment OR services OR species OR \"water quality\" OR \"water resources\") AND (aquatic OR freshwater OR lake OR river OR stream OR water OR wetland) AND (analysis OR assessment OR change OR conservation OR invest OR global OR landscape OR management OR modeling OR modelling OR models OR quality OR restoration OR scale OR spatial OR supply OR valuation).*

Table A2.1 CICES V5.1 classification of ecosystem services was framed for the relevant freshwater ecosystem services. The framed ecosystem services are divided according to their section, division, and group following the CICES organisation (Haines-Young & Potschin, 2018).

Section	**Division**	**Group**
Provisioning (biotic)	Biomass	Cultivated aquatic plants Reared aquatic animals Wild plants Wild animals
	Genetic material from biota	Plants, algae or fungi Animals Organisms
Regulatory & Maintenance (biotic)	Transformation of biochemical inputs to ecosystems	Mediation of wastes or toxic substances Nuisances on anthropogenic origin
	Regulation of chemical, biological conditions	Regulation of baseline flows Lifecycle maintenance Pest and disease control Regulation of soil quality Water conditions Atmospheric conditions (carbon)
Cultural (biotic)	Direct, in-situ and outdoor interactions with living systems that depend on presence in the environmental setting	Physical and experiential interactions with natural environment Intellectual and representative interactions with natural environment
	Indirect, remote, often indoor interactions with living systems that do not require presence in the environmental setting	Spiritual, symbolic, and other interactions with natural environment Other biotic characteristics that have a non-use value
Provisioning (abiotic)	Water	Nutrition Materials Energy
Regulatory & Maintenance (abiotic)	Transformation of biochemical or physical inputs to ecosystems	Mediation of waste, toxics and other nuisances by non-living processes Mediation by other chemical or physical means
	Regulation of physical, chemical, biological conditions	Regulation of baseline flows and extreme events

Table A2.2 Ecosystem service (ES) Group (from CICES V5.1, Haines-Young & Potschin, 2018) not assessed by review tools

ES Group (CICES V5.1)	Category code not found
Cultivated aquatic plants for nutrition, materials or energy	Plants cultivated for nutritional purposes Fibres and other materials for direct use or processing Plants cultivated as an energy source
Reared aquatic animals for nutrition, materials or energy	Fibres and other materials for direct use or processing Animals reared an energy source
Wild plants (terrestrial and aquatic) for nutrition, materials or energy	Wild plants for nutrition Fibres and other materials for direct use or processing Wild plants used as a source of energy
Wild animals (terrestrial and aquatic) for nutrition, materials or energy	Wild animals as a source of energy
Genetic material from plants, algae or fungi	Seeds, spores and other plant materials collected for maintaining or establishing a population Higher and lower plants (whole organisms) used to breed new strains or varieties Individual genes extracted from higher and lower plants for the design and construction of new biological entities
Genetic material from animals	Animal material collected for the purposes of maintaining or establishing a population Wild animals (whole organisms) used to breed new strains or varieties
Genetic material from organisms	Individual genes extracted from organisms for the design and construction of new biological entities
Mediation of wastes or toxic substances of anthropogenic origin by living processes	Bio-remediation Filtration/sequestration/storage/accumulation
Mediation of nuisances of anthropogenic origin	Smell reduction
Regulation of baseline flows and extreme events	Buffering and attenuation of mass movement
Lifecycle maintenance, habitat and gene pool protection	Pollination Seed dispersal
Pest and disease control	Pest control (including invasive species) Disease control
Regulation of soil quality	Decomposition and fixing processes and their effect on soil quality
Atmospheric composition and conditions	Regulation of temperature and humidity, including ventilation and transpiration
Mediation by other chemical or physical means (e.g. via Filtration, sequestration, storage or accumulation)	
Regulation of baseline flows and extreme events	Liquid flows



Spatial scale		Agro-IBIS	AQUATOX	ARIES	CASiMiR	EPFs	EwE	GIS	GREEN	INCA	InSTREAM	InVEST	MIKE	PCLake	PEWI	PiSCES	SDR	Social media tool	SolVES	SPARROW	SWAT	TESSA	WaterGAP	Waterworld	WEAP
Lotic	river network																						1		
	river				2		1	2			2					1	1	1							
	canal											1													
Lentic	reservoir											1													
	pond											1								1					
	lake		1			1	1					1		2						1			1		
Catchment	river basin									1		16	1					1			9				3
	reservoir basin											1							1						
	lake basin											5													
	catchment	1		1				8	1	1		22	3		2				1		17			1	
Other	wetland											4						1				1	1		
	urban											7												1	
	region			2				2	1													2			

Figure A2.2 Spatial scale assessed by twenty-four tools. Spatial scales are categorised as lotic or flowing surface waters, lentic or standing surface waters, catchment, and other (including wetlands, urban ecosystems and regions such as nations). Classifications within spatial scales are identified as how they were presented from the studies. Unclassified catchment relates to an unspecified catchment or cases where it was not clear whether the catchment surrounds a lake, river, urban area, or considers catchment with many freshwater surface water bodies. Urban depicts a city area. Region depicts a larger area such as on the national-scale. Green signifies the number of studies where the tool was applied to assess the spatial scale, with increasing number shown with intensity of green.

Table A2.3 CICES classes, codes, and number of ecological production functions (EPFs) for freshwater ecosystems from Faber et al. (2021).

CICES V5.1 Class	Code	Number EPFs	EPFs
Aquatic plant biomass	2.2.1.1 -1.1.2.3	2	- Aquatic plant biomass in response to site exposure, plant growth form, water depth, and water transparency. - Algal species diversity can enhance total biomass production.
Aquatic animal biomass	2.2.1.1 - 1.1.4.3	6	- Aquaculture of farmed crustaceans, molluscs and finfish - Ratio of wild fisheries inputs to farmed fish output affected by feed composition - Environmental performances among local and exotic genetic resources in fish species - Microorganisms and small eukaryotes improve the culture of fish and shrimp. - Coral aquaculture offers potential for metabolites and production of bacterial biomass for drugs - Microalgae as sources of proteins, lipids, and bioactive compounds in aquaculture of fish
Filtration, Sequestration, storage, accumulation by micro-organisms, algae, plants, and animals	2.1.1.2	4	- Filter feeding molluscs improve water quality by removing nutrients - Aquatic filter-feeding insects reduce sestonic bacteria from stream water - Soil mineral N related to plant functional groups - Aquatic invertebrates retain nutrients
Control of erosion rates	2.2.1.1	1	- Net-spinning caddisflies contribute to fine-gravel substrate stability
Hydrological cycle and water flow regulation	2.2.1.3	3	- Natural river ecosystems regulate flood flows and low flows. - Macrophytes reduce flow velocity relative to the incoming flow velocity - Stream herbivore grazing regulates periphyton biomass and physiognomic structure
Nursery populations and habitats	2.2.2.3	1	- Contributions to recruitment of critical habitats to commercial and recreational fishery can be valued across spatial scales.
Regulation of the chemical condition of freshwaters by living processes	2.2.5.1	5	- Wastewater nutrient removal related to plant biomass production rate. - Purification of lakes by macrophytes, microorganisms, macrofauna and fish. - Macrophytes enhance retention of particulate matter. - Mussel aggregations regulate nutrient cycling. - Fragmentation and decomposition of organic matter is enhanced by aquatic insects.

Table A2.4 List of tools from US EPA SMaRT database (U.S. Environmental Protection Agency, 2024) search when searching "ecosystem services" and environmental topic areas as "water".

Name	Acronym
EnviroAtlas	EnviroAtlas
Causal Analysis / Diagnosis Decision Information System	CADDIS
AQUATOX	AQUATOX
EcoBox - Toolbox for Ecological Risk Assessors	EcoBox
Information Collection Rule Most Probably Number Calculator	ICR MPN
EPA H2O	EPA H2O
Rapid Benefit Indicators Approach-Checklist Tool	RBI Checklist
Rapid Benefit Indicators Approach-Spatial Analysis Toolset	RBI Spatial
Unit Emergy Value Database	UEV
TRAITS - Freshwater Biological Traits Database	Traits
VELMA ecohydrological model and decision support framework	VELMA
Hydrologic Micro Services	HMS
Ecological Database and Interactive Visualizations of Evidence Records	EcoDIVER
Exposure Analysis Modeling System	EXAMS
Storm Water Plugin (not freshwater)	SWMM-Plugin

8.2 Appendix A3: InSTREAM code adaptations

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All additions to inSTREAM (version 7.3) code required applying changes to either the NetLogo code itself or to the parameter input file (.txt file) which defines the value(s) for each input variable. The modifications are reflected as either in the "Parameter input file" and "NetLogo code". Source code markers for (;;) and notes (;) are included to signify where the code is within the files and additional notes on the code additions, respectively. Modified code additions are highlighted where they have been added in the specified sub-models of inSTREAM (starting with to-report). Some source code from inSTREAM (version 7.3) is provided to signpost where these modifications, in the form of additions, have been made.

8.2.1 Fecundity

Parameter input file

```
; Trout and redd parameters
…
set trout-spawn-egg-viability      (list     0.8     )
set spawn-chemical-effect          (trout-spawn-egg-adjuster) ; SMM 20-07-
2022 adding spawn chemical effect
```

NetLogo code

```
  ; Trout and redd parameters -- values set in set-parameters (which is in
the parameter file)
  ; These are lists with values for each species
  ; New parameters must be declared here, initialized in set-parameters,
and added to the procedure test-parameters

  spawn-chemical-effect ; SMM 20-07-2022 adding for chemical effect to
reproduction
```

```
to-report num-viable-eggs
; A trout procedure to calculate the number of viable eggs a female lays
let chem-exposure [reach-toxconcentration] of patches-reach

report ifelse-value ((fecundity-effect? and chem-exposure > 0) or (fecun-
dity-effect? and color = violet)) ; true then
  [
   round ((item trout-spp-index trout-spawn-fecund-mult) *
  (trout-length ^ (item trout-spp-index trout-spawn-fecund-exp)) *
  ((item trout-spp-index trout-spawn-egg-viability) * spawn-chemical-ef-
fect))
]
[ ; value if no chemical exposure - no effect on survival
  round ((item trout-spp-index trout-spawn-fecund-mult) *
  (trout-length ^ (item trout-spp-index trout-spawn-fecund-exp)) *
  (item trout-spp-index trout-spawn-egg-viability))
]

end
```

8.2.2 Growth rate

Parameter input file

```
; Trout and redd parameters
  …
set trout-energy-density          (list    5900   )
set growth-chemical-effect        growth-rate-adjuster ; SMM 23-05-2023
growth aduster effect
```

NetLogo code

```
; Trout and redd parameters -- values set in set-parameters (which is in
the parameter file)
; These are lists with values for each species
; New parameters must be declared here, initialized in set-parameters, and
added to the procedure test-parameters

growth-chemical-effect ; SMM 09-06-2022 adding for chemical effect to
growth rate

; Reach parameters -

chemical-effect                ; SMM 09-06-2022 adding parameter for chemical-
effect added here as not a list, not added to test-parameters to avoid is-
list?
```

```
to-report growth-rate-for [ a-fish a-cell an-activity ]
…
; SMM 28-02-2023
let chem-exposure [reach-toxconcentration] of patches-reach

let growth-rate-effect ifelse-value ((growth-rate-effect? and chem-expo-
sure > 0) or (growth-rate-effect? and a-fish = violet)) ;true then
[ ; value if chemical exposure present
    growth-chemical-effect
]
[ ; value if no chemical exposure no effect on survival
    1.0
]

  ifelse (an-activity = "drift")
  [
    …
    set the-growth (drift-net-energy-intake / fish-energy-density) *
growth-rate-effect
  ] ; an-activity = "drift"
  [
   …
     ; SMM 09-06-2022 () * chemical-effect parameter effect on growth rate
for percentage effect on search feeding
     set the-growth (search-net-energy-intake / fish-energy-density) *
growth-rate-effect
    ] ; an-activity = "search"
```

8.2.3 Survival

Parameter input file

```
; Trout and redd parameters
…
set mort-condition-S-at-K5          (list    0.98    )
set time-tox-test                   time-tox-test-adjuster ; days the
survival rate is over
```

NetLogo code

```
; Trout and redd parameters -- values set in set-parameters (which is in
the parameter file)
; These are lists with values for each species
; New parameters must be declared here, initialized in set-parameters, and
added to the procedure test-parameters

time-tox-test        ; SMM 31-01-2023 time of the survivorship from chemical
toxicity
```

As survival to chemical toxicity is added as a new survival sub-model, the NetLogo code was updated in both the *to-survive* model as well as in the new *survival-chem-tox-for* model.

```
to survive
…
survival-chem-tox-for self patches-cell    ;SMM 30-06-2022 chemical tox-
icity submodel
)
file-close
]
; End of test output code
…
if (random-float 1.0) > (survival-chem-tox-for self patches-cell) ^ step-
length ; if 1.0 then not die as unlikely number greater in bernolli
[ ; Trout died of chemical toxicity
  save-event "died of chemical toxicity"
    die
]
end

to-report non-starve-survival-for [ a-fish a-cell an-activity ]
; The observer procedure that calculates overall daily survival probability
; except for condition (starvation). Used in habitat and activity selec-
tion.
; Valid values of an-activity are "hide", "drift", and "search"

report (survival-high-temperature-for a-fish a-cell) *
;   (survival-high-velocity-for a-fish a-cell an-activity) *  REMOVED
    (survival-stranding-for a-fish a-cell) *
    (survival-fish-pred-for a-fish a-cell an-activity) *
    (survival-terr-pred-for a-fish a-cell an-activity) *
    (survival-chem-tox-for a-fish a-cell) ; SMM 30-06-2022 adding chemical
toxicity submodel

end
```

```
;SMM 06-02-2023 Updating chemical toxicity sub-model to time extrapolated
simple with date of exposure
to-report survival-chem-tox-for [a-fish a-cell]
; An observer (or trout, cell) procedure to report survival probability for
chemical toxicity.

;SMM 04-07-2022 reach tox concentration included
let chem-exposure [reach-toxconcentration] of patches-reach
let adjusted-time-survival 14 ; SMM 21-02-2023 observed time for survival
(e.g. 14 day standard toxicity test)

; if have a known chemical, exposure, and LC50 unsilence the following code
; survivorship for the whole population between 0 - 1.0 based on toxicity
model for time period of model run
let chemical-survival ifelse-value ((survival-effect? and chem-exposure >
0) or (survival-effect? and color = violet)); true then
[ ; if known LC50, exposure, and LC10 can use the following model and
silence the other code (SCHAMPHELAERE AND JANSSEN 2004 model) ; SMM check
code
;  1.0 / ( 1.0 + ((chem-exposure / exp (ln (chem-tox-LC50)) ^ chem-
slope))) ; or ( ln( 1 / 9 ) / (ln(chem-tox-LC10) - ln(chem-tox-LC50)))))
; this is the survival rate seen in a toxicity test e.g. over 72 hours/14
days this needs to be extrapolated to a daily rate
; adjust to time extrapolation Loti 2022 DOI: 10.2307/j.ctv2n37k6t.9

(chemical-survival-percent) ^ (adjusted-time-survival / time-tox-test)

]

[ ; value if no chemical exposure no effect on survival and beyond time of
test length
  1.0
]

report chemical-survival

end
```

8.2.4 Prey availability

Parameter input file

```
;; Reach-scale habitat parameters
set prey-chemical-effect         (prey-density-adjuster) ; SMM 20-07-2022
adding for chemical-effect to prey-density, chemical effect % change on
prey density
```

NetLogo code

```
; Reach parameters -- values set in set-parameters (which is in the parame-
ter file)
; These are lists with values for each reach
; New parameters must be declared here, initialized in set-parameters, and
added to the procedure test-parameters
…
prey-chemical-effect         ; SMM 20-07-2022 chemical-effect % change on
prey density
```

```
to-report growth-rate-for [ a-fish a-cell an-activity ]
…
; SMM 28-02-2023 prey density effect
let chem-exposure [reach-toxconcentration] of patches-reach

let prey-chemical-effect-run ifelse-value ((prey-density-effect? and chem-
exposure > 0) or (prey-density-effect? and a-fish = violet)) ; true then
[ ; value if chemical exposure is present
  prey-chemical-effect
]
[ ; value if no chemical exposure present
  1.0
]
…
ifelse (an-activity = "drift")
[
  ; Do the energy balance and convert J energy intake to g growth
  let drift-net-energy-intake (drift-intake * (([reach-prey-energy-density]
of the-reach) * prey-chemical-effect-run)) - drift-resp ; SMM 28-02-2023
…
] ; an-activity = "drift"
[
ifelse (an-activity = "search")
[
…
  ; Do the energy balance and convert J energy intake to g growth
  let search-net-energy-intake (search-intake * (([reach-prey-energy-
density] of the-reach) * prey-chemical-effect-run)) - search-resp ; SMM 28-
02-2023
…
] ; an-activity = "search"
```

8.2.5 Chemical exposure scenarios

Parameter input file

```
;; Time series reach variables
reach-toxconcentration        ; SMM 12-10-2022 adding toxicity
concentration
```

NetLogo code

```
to build-reaches

…

;; Initialize dynamic reach variables
set reach-toxconcentration  -999               ; SMM 12-10-2022

to update-habitat
…

set reach-toxconcentration ts-get reach-time-series-input mid-step-time
"concentration" ; 12-10-2022 SMM

to-report growth-rate-for [ a-fish a-cell an-activity ]
; [reach-toxconcetration] of the-reach ; SMM 20-06-2022 added to align with
rest of model but silenced in model
```

```
to redd-survive
…
; Test output for redd survival.
…
[reach-name] of patches-reach
[reach-temperature] of patches-reach
[reach-flow] of patches-reach
[reach-toxconcentration] of patches-reach ; 12-10-2022 SMM
depth

to update-output
…

; Do the brief population output
if brief-pop-output?
[
  if brief-pop-outfile-name = "b-p-o-n" [ build-output-file-named brief-
pop-outfile-name ]
  file-open brief-pop-outfile-name

   foreach reach-names [ next-reach-name ->
    let next-reach one-of reaches with [reach-name = next-reach-name]
    let the-flow [reach-flow] of next-reach
    let the-temperature [reach-temperature] of next-reach
    let the-turbidity [reach-turbidity] of next-reach
    let the-concentration [reach-toxconcentration] of next-reach ; SMM 12-
10-2022

to-report growth-rate-for [ a-fish a-cell an-activity ]
…
let chem-exposure [reach-toxconcentration] of patches-reach
```

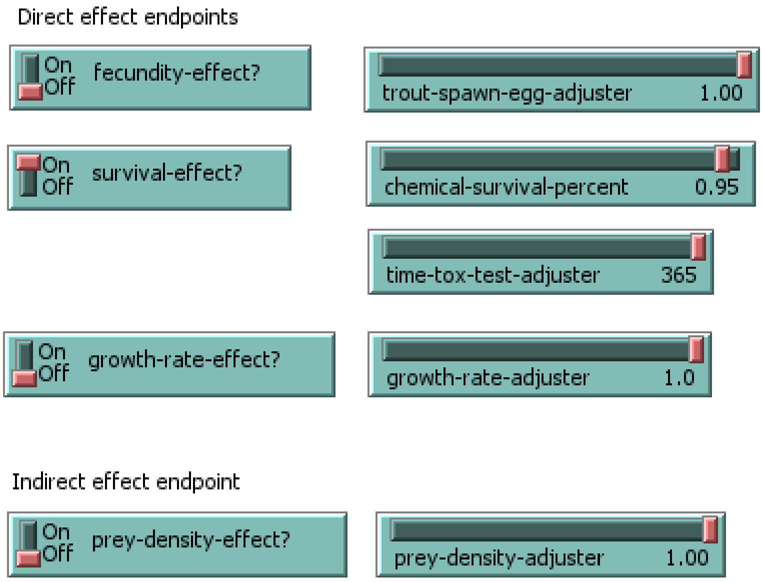
The model will first check that chemical exposure is present in the system (at > 0) before applying any inhibition on life history parameters. An example of this is shown below (silenced).

```
to select-activity-and-habitat
…

 ;  SMM 13-03-2023 exposed fish inhibited until their death
  let chem-exposure [reach-toxconcentration] of patches-reach

  ; if chem-exposure > 0 ; SMM 27-07-2023 introduce if exposure to chemi-
cals cause inhibition in trout for their entire life
  ; [ set color violet ]
```

The second part of the code (`if chem-exposure > 0`) would be unsilenced (by removing ;) if activating a chemical impact and if the chemical impact is inhibiting for the entirety of the individual's life then the third part of the code `[ set color violet ]` is unsilenced.

8.2.6 NetLogo interface and BehaviorSpace simulations

In order to investigate the impacts of a direct inhibition of life history parameters the following interface was added to inSTREAM (version 7.3).



Direct effect endpoints

On Off fecundity-effect?

trout-spawn-egg-adjuster 1.00

On Off survival-effect?

chemical-survival-percent 0.95

time-tox-test-adjuster 365

On Off growth-rate-effect?

growth-rate-adjuster 1.0

Indirect effect endpoint

On Off prey-density-effect?

prey-density-adjuster 1.00

BehaviorSpace experiments were run with the following code, those parts highlighted (grey) were turned on (true) or off (false) depending on the simulation and a list of parameter values was added where there is an X (highlighted in grey). This is how combined stressors were run simultaneously.

```
["brief-pop-output?" false]
["detailed-pop-output?" false]
["shade-variable" "off"]
["update-plots?" false]
["events-output?" true]
["redd-output?" true]
["fecundity-effect?" true/false]
["survival-effect?" true/false]
["growth-rate-effect?" true/false]
["prey-density-effect?" true/false]
["trout-spawn-egg-adjuster" X] ; include if fecundity-effect?
["chemical-survival-percent" X] ; include if survival-effect?
["time-tox-test-adjuster" 365] ; include if survival-effect?
["growth-rate-adjuster" X] ; include if growth-rate-effect?
["prey-density-adjuster" X] ; include if prey-density-effect?
```

8.2.7 Angling stock outputs

Angling stock was determined as the largest (>20 cm length) and oldest (Age 1+) trout. BehaviorSpace outputs were updated to calculate the abundance of angling stock.

NetLogo code

```
;; BehaviorSpace outputs, updated in update-BehaviorSpace-outputs. Add any
variables to be reported from BehaviorSpace
  RT-adult-abund       ; SMM 05-08-2022
  RT-angling-stock-10     ; SMM 05-08-2022
  RT-angling-stock-20     ; SMM 05-08-2022
  ;RT-angling-stock-30     ; SMM 05-08-2022 if interested in larger stock
  ;RT-angling-stock-50     ; SMM 05-08-2022 if interested in larger stock
```

Code for angling stock in a reach simulated independently is shown below.

```
to update-behaviorspace-outputs
…

; SMM 07-12-2022 results for whole network/reach
let RT-age-0s trout with [trout-age = 0 and trout-species = "Rain-
bow"]       ; Age 0 Rainbow trout ; SMM 24-11-2022 Rainbow example
let RT-age-1s trout with [trout-age = 1 and trout-species = "Rain-
bow"]       ; Age 1 trout Rainbow trout
let RT-age-2s trout with [trout-age >= 2 and trout-species = "Rain-
bow"]      ; Age 2 and older Rainbow trout

; Angling stock amounts for entire network/reach SMM 05-08-2022
let RT-adults trout with [trout-age >= 1 and trout-species = "Rain-
bow" ]      ; SMM 05-08-2022 Adult trout Age 1+
set RT-adult-abund sum [trout-superind-rep] of RT-
adults                     ; SMM 05-08-2022 Abundance of Adult rainbow
trout

let RT-stock-10 RT-adults with [trout-length > 10 and trout-species ="Rain-
bow" ] ; and [trout-weight > minimum-weight-catch-fishery]  ; SMM 05-08-
2022 Stock of fish for angling with length (cm) and weight (g).
set RT-angling-stock-10 sum [trout-superind-rep] of RT-stock-
10                     ; SMM 05-08-2022 Abundance of Adult angling stock
over > 10 cm

let RT-stock-20 RT-adults with [trout-length > 20 and trout-species ="Rain-
bow" ] ; and [trout-weight > minimum-weight-catch-fishery]  ; SMM 05-08-
2022 Stock of fish for angling with length (cm) and weight (g).
set RT-angling-stock-20 sum [trout-superind-rep] of RT-stock-
20                     ; SMM 05-08-2022 Abundance of Adult angling stock
over > 20 cm

; if interested in a larger angling stock
; let RT-stock-30 RT-adults with [trout-length > 30 and trout-species
="Rainbow" ] ; and [trout-weight > minimum-weight-catch-fishery]  ; SMM 05-
08-2022 Stock of fish for angling with length (cm) and weight (g).
; set RT-angling-stock-30 sum [trout-superind-rep] of RT-stock-
30                     ; SMM 05-08-2022 Abundance of Adult angling stock
over > 30 cm

; let RT-stock-50 RT-adults with [trout-length > 50 and trout-species
="Rainbow" ] ; and [trout-weight > minimum-weight-catch-fishery]  ; SMM 05-
08-2022 Stock of fish for angling with length (cm) and weight (g).
; set RT-angling-stock-50 sum [trout-superind-rep] of RT-stock-
50                     ; SMM 05-08-2022 Abundance of Adult angling stock
over > 50 cm
```

Code for angling stock in each reach of a river network is shown below. As an example only one reach/habitat from the river network is shown here but each habitat/reach would require the same code repeated by changing the name.

```
to update-behaviorspace-outputs
…
; SMM 07-12-2022 results for each reach
; "LJC-LowerA"
let RT-age-0s-LJC-LowerA trout with [trout-age = 0 and trout-species =
"Rainbow" and ([reach-name] of patches-reach) = "LJC-LowerA"]
let RT-age-1s-LJC-LowerA trout with [trout-age = 1 and trout-species =
"Rainbow" and ([reach-name] of patches-reach) = "LJC-LowerA"]
let RT-age-2s-LJC-LowerA trout with [trout-age >= 2 and trout-species =
"Rainbow" and ([reach-name] of patches-reach) = "LJC-LowerA"]
set RT-age-0-abund-LJC-LowerA sum [trout-superind-rep] of RT-age-0s-LJC-
LowerA
set RT-age-1-abund-LJC-LowerA sum [trout-superind-rep] of RT-age-1s-LJC-
LowerA
set RT-age-2+-abund-LJC-LowerA sum [trout-superind-rep] of RT-age-2s-LJC-
LowerA
set RT-age-0-length-LJC-LowerA ifelse-value (any? RT-age-0s-LJC-LowerA)
[mean [trout-length] of RT-age-0s-LJC-LowerA] [""]
set RT-age-1-length-LJC-LowerA ifelse-value (any? RT-age-1s-LJC-LowerA)
[mean [trout-length] of RT-age-1s-LJC-LowerA] [""]
set RT-age-2+-length-LJC-LowerA ifelse-value (any? RT-age-2s-LJC-LowerA)
[mean [trout-length] of RT-age-2s-LJC-LowerA] [""]

let RT-stock-10-LJC-LowerA RT-adults with [trout-length > 10 and trout-
species ="Rainbow" and ([reach-name] of patches-reach) = "LJC-LowerA"] ;
and [trout-weight > minimum-weight-catch-fishery]  ; SMM 05-08-2022 Stock
of fish for angling with length (cm) and weight (g).
set RT-angling-stock-10-LJC-LowerA sum [trout-superind-rep] of RT-stock-10-
LJC-LowerA                   ; SMM 05-08-2022 Abundance of Adult angling
stock over > 10 cm
let RT-stock-20-LJC-LowerA RT-adults with [trout-length > 20 and trout-
species ="Rainbow" and ([reach-name] of patches-reach) = "LJC-LowerA"] ;
and [trout-weight > minimum-weight-catch-fishery]  ; SMM 05-08-2022 Stock
of fish for angling with length (cm) and weight (g).
set RT-angling-stock-20-LJC-LowerA sum [trout-superind-rep] of RT-stock-20-
LJC-LowerA
…
end
```

Angling stock density was also important so inSTREAM was used to calculate the area of the wet cells in the stream.

NetLogo code

```
;; BehaviorSpace outputs, updated in update-BehaviorSpace-outputs. Add any variables to be reported from BehaviorSpace
…
; 15-08-2023 Wet cells area
Area-network
Area-LowerA
Area-LowerB
Area-Upper1A
Area-Upper1B
Area-Upper2A
Area-Upper2B
Area-Upper3A
Area-Upper3B
Area-Weejak1A
Area-Weejak1B
Area-Weejak2A
Area-Weejak2B
Area-Weejak3A
Area-Weejak3B
Area-Weejak4A
Area-Weejak4B
Area-Weejak5A
Area-Weejak5B
Area-Weejak6A
Area-Weejak6B
```

```
to update-behaviorspace-outputs
…
; SMM 15-08-2023
; Area (cm2) of wet cells in network and reaches
set Area-network sum [cell-area] of cells with [depth > 0]
set Area-LowerA sum [cell-area] of cells with [depth > 0 and ([reach-name]
of patches-reach = "LJC-LowerA")]
set Area-LowerB sum [cell-area] of cells with [depth > 0 and ([reach-name]
of patches-reach = "LJC-LowerB")]
set Area-Upper1A sum [cell-area] of cells with [depth > 0 and ([reach-name]
of patches-reach = "LJC-Upper1A")]
set Area-Upper1B sum [cell-area] of cells with [depth > 0 and ([reach-name]
of patches-reach = "LJC-Upper1B")]
set Area-Upper2A sum [cell-area] of cells with [depth > 0 and ([reach-name]
of patches-reach = "LJC-Upper2A")]
set Area-Upper2B sum [cell-area] of cells with [depth > 0 and ([reach-name]
of patches-reach = "LJC-Upper2B")]
set Area-Upper3A sum [cell-area] of cells with [depth > 0 and ([reach-name]
of patches-reach = "LJC-Upper3A")]
set Area-Upper3B sum [cell-area] of cells with [depth > 0 and ([reach-name]
of patches-reach = "LJC-Upper3B")]
set Area-Weejak1A sum [cell-area] of cells with [depth > 0 and ([reach-
name] of patches-reach = "LJC-Weejak1A")]
set Area-Weejak1B sum [cell-area] of cells with [depth > 0 and ([reach-
name] of patches-reach = "LJC-Weejak1B")]
set Area-Weejak2A sum [cell-area] of cells with [depth > 0 and ([reach-
name] of patches-reach = "LJC-Weejak2A")]
set Area-Weejak2B sum [cell-area] of cells with [depth > 0 and ([reach-
name] of patches-reach = "LJC-Weejak2B")]
set Area-Weejak3A sum [cell-area] of cells with [depth > 0 and ([reach-
name] of patches-reach = "LJC-Weejak3A")]
set Area-Weejak3B sum [cell-area] of cells with [depth > 0 and ([reach-
name] of patches-reach = "LJC-Weejak3B")]
set Area-Weejak4A sum [cell-area] of cells with [depth > 0 and ([reach-
name] of patches-reach = "LJC-Weejak4A")]
set Area-Weejak4B sum [cell-area] of cells with [depth > 0 and ([reach-
name] of patches-reach = "LJC-Weejak4B")]
set Area-Weejak5A sum [cell-area] of cells with [depth > 0 and ([reach-
name] of patches-reach = "LJC-Weejak5A")]
set Area-Weejak5B sum [cell-area] of cells with [depth > 0 and ([reach-
name] of patches-reach = "LJC-Weejak5B")]
set Area-Weejak6A sum [cell-area] of cells with [depth > 0 and ([reach-
name] of patches-reach = "LJC-Weejak6A")]
set Area-Weejak6B sum [cell-area] of cells with [depth > 0 and ([reach-
name] of patches-reach = "LJC-Weejak6B")]
```

8.2.8 Targeting stress in a network reach

An individual time series file was drafted for the target reach and added into the model code parameter file as shown.

Parameter file input

```
; Reach parameters -- lists with one value per reach
;; Reach initialization parameters
…
set time-series-input-files
(list "LJC-Network_Input/LJC-Lower_TimeSeriesInput.csv"  ; LJC-LowerA
      "LJC-Network_Input/LJC-Lower_TimeSeriesInput.csv"  ; LJC-LowerB
      "LJC-Network_Input/LJC-Upper_TimeSeriesInput.csv"  ; LJC-Upper1A
      "LJC-Network_Input/LJC-Upper_TimeSeriesInput.csv"  ; LJC-Upper1B
      "LJC-Network_Input/LJC-Upper_TimeSeriesInput.csv"  ; LJC-Upper2A
      "LJC-Network_Input/LJC-Upper_TimeSeriesInput.csv"  ; LJC-Upper2B
      "LJC-Network_Input/LJC-Upper_TimeSeriesInput.csv"  ; LJC-Upper3A
      "LJC-Network_Input/LJC-Upper_TimeSeriesInput.csv"  ; LJC-Upper3B
      "LJC-Network_Input/LJC-Weejak_TimeSeriesInput.csv"  ; LJC-Weejak1A
      "LJC-Network_Input/LJC-Weejak_TimeSeriesInput.csv"  ; LJC-Weejak1B
      "LJC-Network_Input/LJC-Weejak_TimeSeriesInput.csv"  ; LJC-Weejak2A
      "LJC-Network_Input/LJC-Weejak_TimeSeriesInput.csv"  ; LJC-Weejak2B
      "LJC-Network_Input/LJC-Weejak3A_TimeSeriesInput.csv"  ; LJC-
Weejak3A ; SMM 31-08-2023 targetting chemical stress in Weejak 3A
      "LJC-Network_Input/LJC-Weejak_TimeSeriesInput.csv"  ; LJC-Weejak3B
      "LJC-Network_Input/LJC-Weejak_TimeSeriesInput.csv"  ; LJC-Weejak4A
      "LJC-Network_Input/LJC-Weejak_TimeSeriesInput.csv"  ; LJC-Weejak4B
      "LJC-Network_Input/LJC-Weejak_TimeSeriesInput.csv"  ; LJC-Weejak5A
      "LJC-Network_Input/LJC-Weejak_TimeSeriesInput.csv"  ; LJC-Weejak5B
      "LJC-Network_Input/LJC-Weejak_TimeSeriesInput.csv"  ; LJC-Weejak6A
      "LJC-Network_Input/LJC-Weejak_TimeSeriesInput.csv"  ; LJC-Weejak6B
  ) ; Name of time series input file ; SMM 13-07-2022 new site
```

Each reach also had their own output parameters.

```
;; BehaviorSpace outputs, updated in update-BehaviorSpace-outputs.
…
; SMM 05-05-2022 Angling stock for entire network/reach
RT-adult-abund       ; SMM 05-08-2022
RT-angling-stock-10     ; SMM 05-08-2022
RT-angling-stock-20     ; SMM 05-08-2022
RT-angling-stock-30     ; SMM 05-08-2022
RT-angling-stock-50     ; SMM 05-08-2022
; SMM 14-06-2023 Angling stock for each reach in a network
RT-angling-stock-20-LJC-LowerA
RT-angling-stock-20-LJC-LowerB
RT-angling-stock-20-LJC-Upper1A
RT-angling-stock-20-LJC-Upper1B
RT-angling-stock-20-LJC-Upper2A
RT-angling-stock-20-LJC-Upper2B
RT-angling-stock-20-LJC-Upper3A
RT-angling-stock-20-LJC-Upper3B
RT-angling-stock-20-LJC-Weejak1A
RT-angling-stock-20-LJC-Weejak1B
RT-angling-stock-20-LJC-Weejak2A
RT-angling-stock-20-LJC-Weejak2B
RT-angling-stock-20-LJC-Weejak3A
RT-angling-stock-20-LJC-Weejak3B
RT-angling-stock-20-LJC-Weejak4A
RT-angling-stock-20-LJC-Weejak4B
RT-angling-stock-20-LJC-Weejak5A
RT-angling-stock-20-LJC-Weejak5B
RT-angling-stock-20-LJC-Weejak6A
RT-angling-stock-20-LJC-Weejak6B
```

8.2.9 Trout movement

It is important to make sure that the save-events? output file option is turned on at the NetLogo interface to produce the event file and record trout movement. This .csv file will then mark every instance a fish has moved away from its natal reach (or the reach it was initialised in) and subsequent reaches it resides in.

<u>NetLogo code</u>

```
to use-cell-and-activity [ a-cell an-activity ]
; A trout procedure to move to and conduct the activity in a cell
; Procedure moves fish to somewhere within a cell's patches,
; updates fish variable for activity, and
; depletes any resources used by the fish

let old-reach patches-reach ; SMM 30-06-2023 Steve suggestion identify
previous reach before movement

move-to one-of [cells-patches] of a-cell

if patches-reach != old-reach  ; SMM 30-06-2023 identify if reach changed
[save-event (word "Moved from " ([reach-name] of old-reach)) ] ; SMM 30-06-
2023 save movement as event, events file will have natal and current reach
and this is the old reach

set trout-activity an-activity
```

8.2.10 Trout natal reaches

The following code changes were added from inSALMO for trout natal reaches (Railsback, 2021).

<u>NetLogo code</u>

```
trout-own

…

  trout-natal-reach        ; SMM 14-07-2022 (from inSALMO) to record
movement in reaches

to build-initial-populations
…
; Create the fish for each row

create-trout the-num-fish
[
  let the-length random-triangular the-len-min the-len-mode the-len-max
  ; Use this separate procedure to keep all fish initialization code in one
place
  initialize-trout-with the-species the-age the-length 1
  move-to one-of wet-patches
  set trout-natal-reach [reach-name] of patches-reach  ; SMM 14-07-2022
(from inSALMO) to record fish movement
  save-event "initialized"
] ; create-trout
```

```
to grow
…

; Save the state of the fish here for output later, if individual fish
output is on.
if individual-fish-output? and trout-tracked?
[
    set trout-output-list (list behaviorspace-run-number formatted-sim-time
light-phase ([reach-name] of patches-reach)
          who trout-species trout-age trout-length trout-weight trout-
condition trout-superind-rep trout-activity trout-natal-reach ; SMM 30-06-
2023 add trout-natal-reach
          (first growth-memory-list) ; Growth in current time step.
          (first survival-memory-list) ; Survival probability for current
time step.
      patches-cells-ID (([xcor] of patches-cell) * world-resolution)
(([ycor] of patches-cell) * world-resolution) ([reach-temperature] of
patches-reach) depth velocity
          light ([cell-num-hiding-places] of patches-cell) ([cell-
available-hiding-places] of patches-cell))
]

to redd-emergence

…

; Create the trout that have superindividual size = maximum
let my-species redd-species
let my-index redd-spp-index
hatch-trout num-new-trout
[
  let the-length random-triangular (item my-index trout-emerge-length-min)
    (item my-index trout-emerge-length-mode)
    (item my-index trout-emerge-length-max)

  initialize-trout-with my-species 0 the-length superind-rep
  save-event "emerged"
  move-to one-of [cells-patches] of patches-cell
  set trout-natal-reach [reach-name] of patches-reach ; SMM 14-07-2022
(from inSALMO) to record movement
]

; Now create the last superindividual, which represents fewer individuals
if last-trout-rep > 0
[
  hatch-trout 1
  [
    let the-length random-triangular (item my-index trout-emerge-length-
min)
      (item my-index trout-emerge-length-mode)
      (item my-index trout-emerge-length-max)

    initialize-trout-with my-species 0 the-length last-trout-rep
    save-event "emerged"
    move-to one-of [cells-patches] of patches-cell
    set trout-natal-reach [reach-name] of patches-reach ; SMM 14-07-2022
(from inSALMO) to record movement
  ]
]
```

8.2.11 Individual and event output file updates

Here the events output file was modified to add in the current reach the individual is in as well as their natal reach. In order to turn off redd events (otherwise produced as well) this has been commented out of the code as seen (;).

NetLogo code

```
to build-output-file-named [a-file-name]
…

; Create the individual fish output file.
if a-file-name = "f-o-n" ; This is the value of the uninitialized file name
[
  set fish-outfile-name (word "IndFishOut-" outfile-name-base)
  if file-exists? fish-outfile-name [ file-delete fish-outfile-name ]
  file-open fish-outfile-name
  file-print (word "InSTREAM-7 individual fish output file, Created " date-
and-time)
  file-print "Growth and survival reported here were calculated using
length and weight from end of previous time step!"
  file-print "BehavSp-Run,End of time step,Light
phase,Reach,FishID,Species,Age,Length,Weight,Condition,SuperindRep,Activity
,NatalReach,Growth,Survival,Cell,Cell-X,Cell-
Y,Temperature,Depth,Velocity,Light,TotalHidingPlaces,AvailHidingPlaces" ;
SMM 30-06-2023 added NatalReach
  file-close
]
```

```
to save-event [an-event-type]
; A turtle (trout, redd, cell) procedure to save events for the event
output files.
; an-event-type is a character string that says what event happened.
; Save the event only if events output is turned on.

if events-output?
[
  if breed = trout
  [
    set fish-events-list lput (csv:to-row (list
      behaviorspace-run-number
      formatted-sim-time
      light-phase
      [reach-name] of patches-reach ; SMM 30-06-2023 current reach note for
self
      trout-species
      who
      patches-cells-ID
      trout-natal-reach ; SMM 19-07-2022 add to monitor movements from
natal reach in event output
      trout-age
      trout-length
      trout-weight
      trout-condition
      trout-superind-rep
      an-event-type
    )) fish-events-list
  ]

; SMM 30-06-2023 comment out redd save-events from file, un-comment to add
back
 ;   if breed = redds
 ;   [
 ;     set redd-events-list lput (csv:to-row (sentence
 ;       behaviorspace-run-number
 ;       formatted-sim-time
 ;       light-phase
 ;       [reach-name] of patches-reach
 ;       redd-species
 ;       who
 ;       patches-cells-ID
 ;       an-event-type
 ;    )) redd-events-list
 ;   ]
]

end

; SMM 30-06-2023 comment out so not to have output file
;  if not empty? redd-events-list
;  [
;    if redd-events-outfile-name = "r-e-o-n" [ build-output-file-named
redd-events-outfile-name ]
;    file-open redd-events-outfile-name
;    foreach redd-events-list [ the-event -> file-print the-event ]  ;
Write each event to the file
;    file-close
;    set redd-events-list (list)                ; Clear the event list
;  ]
]
```

8.3 Appendix A4: Supplementary figures for Chapter 4

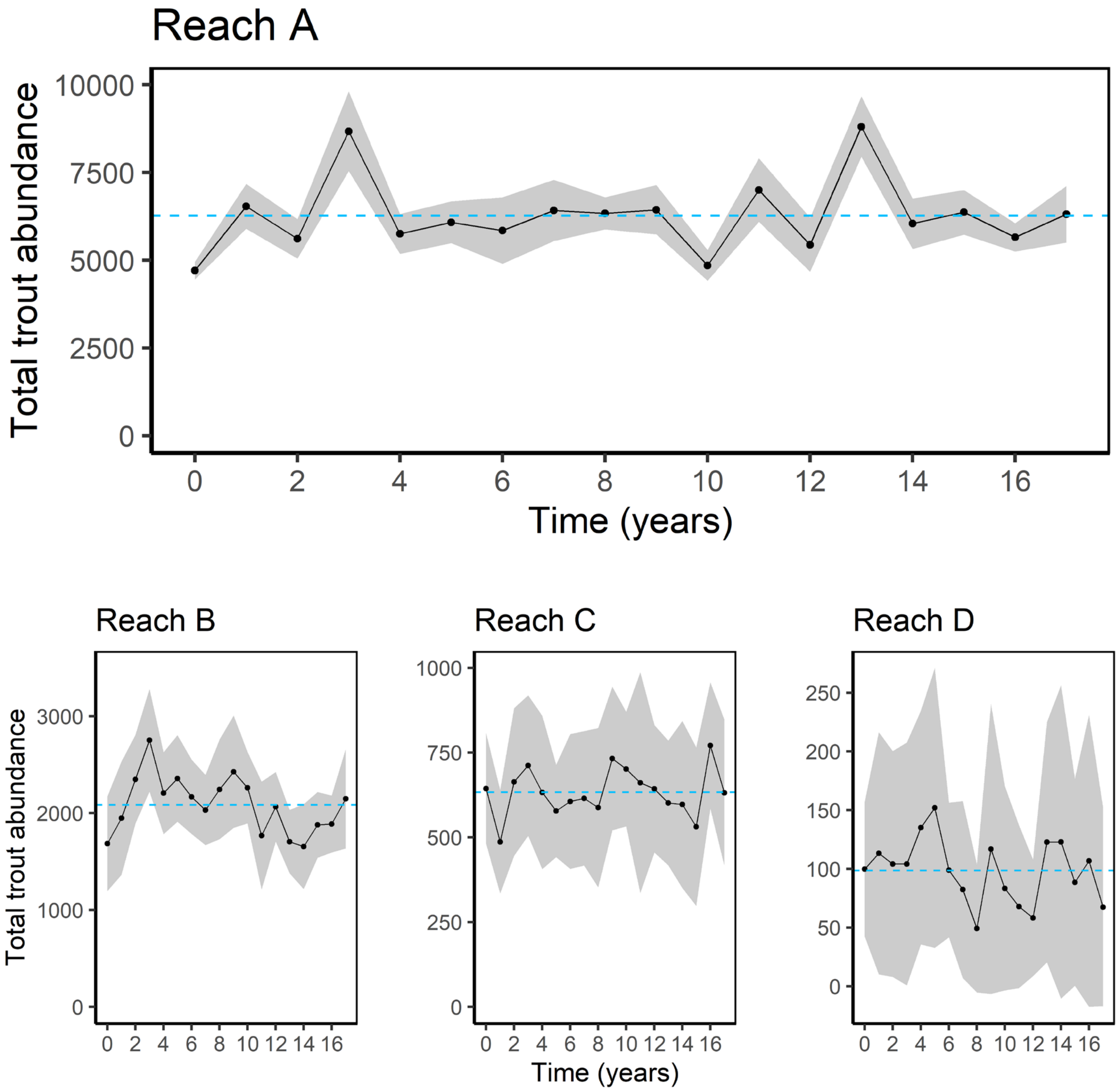


Figure A4.1 Mean total abundance of rainbow trout in river reach site A (top), B (bottom left), C (bottom middle) and D (bottom right) from ten replicate simulations. The abundance is plotted across a 17-year simulation period with the value on a census day (September 30th). The mean blue line indicates the mean abundance across the 17-year simulation period. The shaded area indicates +/- one standard deviation.

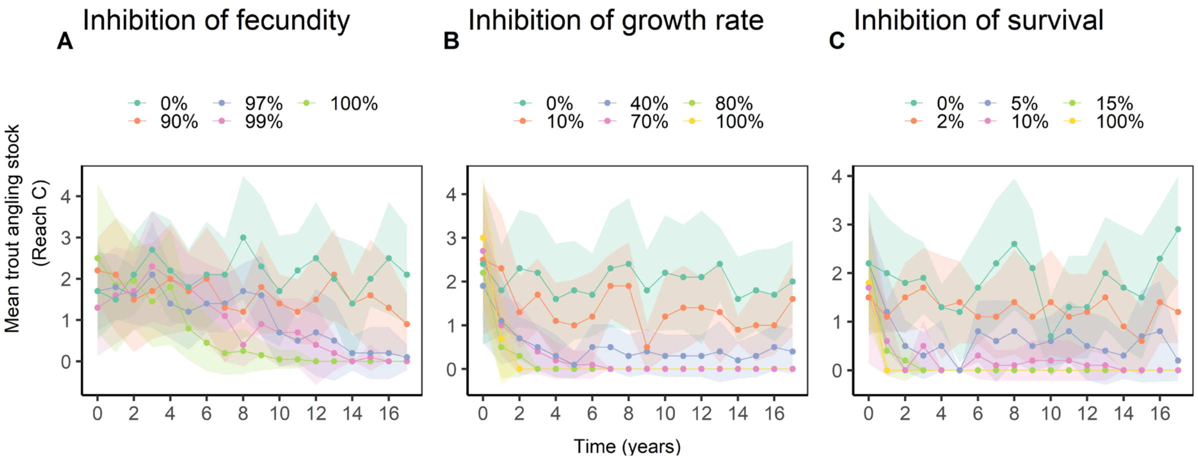


Figure A4.2 Reach C. Mean trout angling stock (Age 1+ and > 20 cm in length) from ten replicate simulations on a census day over the simulated chemical exposure period of 17 years, as well as pre-exposure census at year 0. Trout life history parameters were inhibited from 0 – 100% for A) fecundity, B) growth rate, and C) survival. The shaded area depicts +/- one standard deviation from ten replicates.

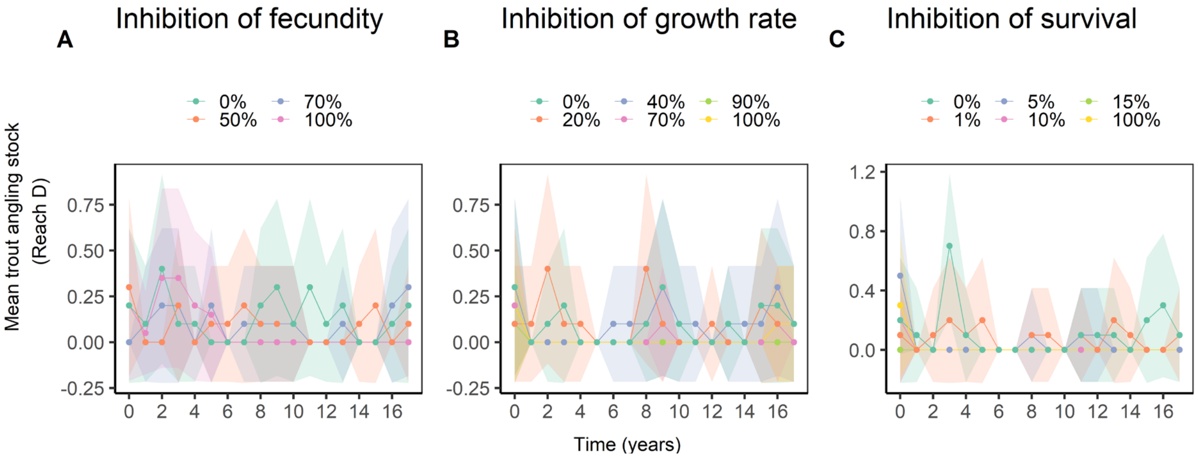


Figure A4.3 Reach D. Mean trout angling stock (Age 1+ and > 20 cm in length) from ten replicate simulations on a census day over the simulated chemical exposure period of 17 years, as well as pre-exposure census at year 0. Trout life history parameters were inhibited from 0 – 100% for A) fecundity, B) growth rate, and C) survival. The shaded area depicts +/- one standard deviation from ten replicates.

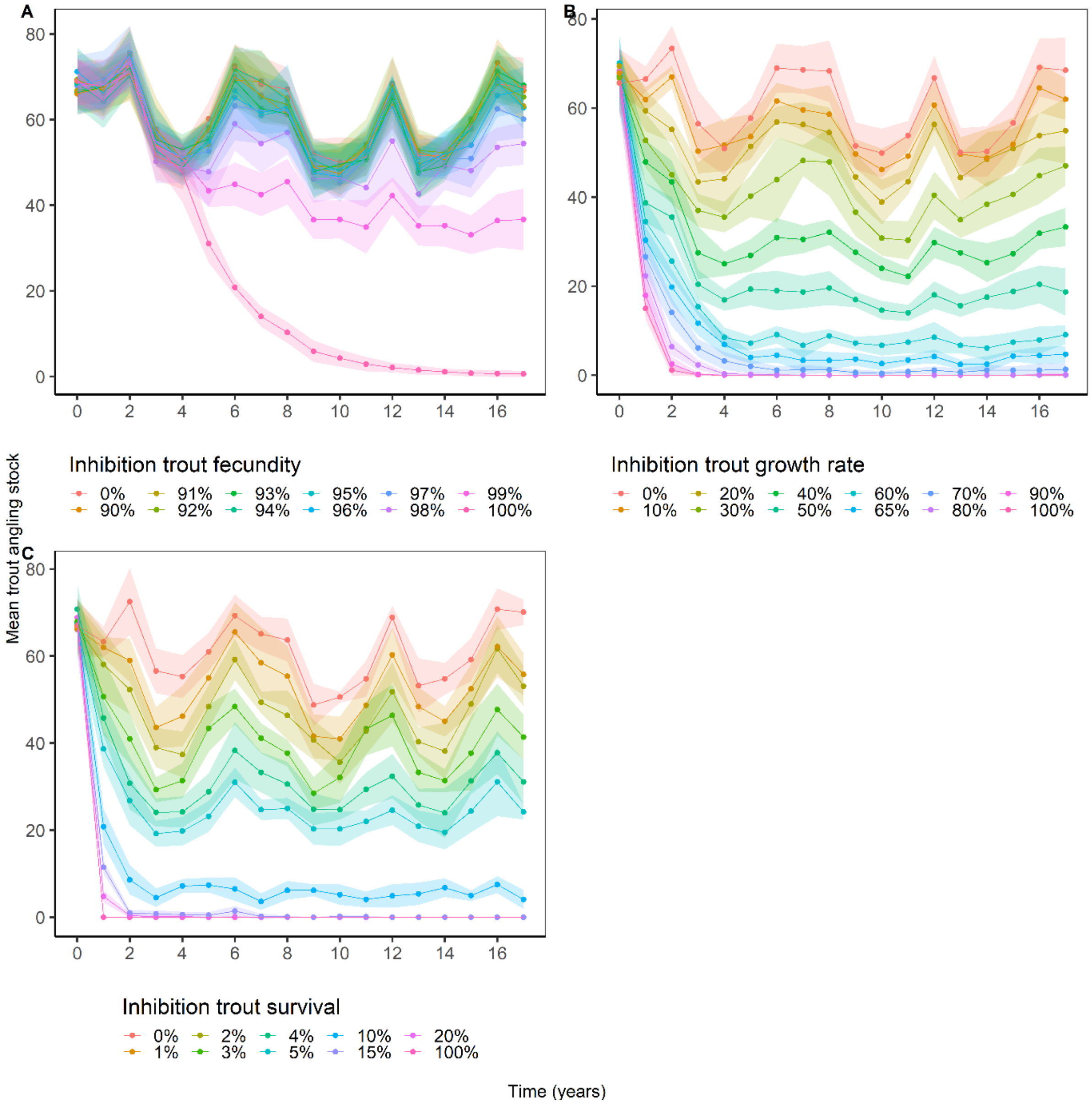


Figure A4.4. Reach A. Full results, before selecting thresholds, of mean trout angling stock (Age 1+ and > 20 cm in length) from ten replicate simulations on a census day over the simulated chemical exposure period of 17 years, as well as pre-exposure census at year 0. Trout life history parameters were inhibited from 0 – 100% for A) fecundity, B) growth rate, and C) survival. The shaded area depicts +/- one standard deviation from ten replicates.

8.4 Appendix A5: Supplementary figures for Chapter 5

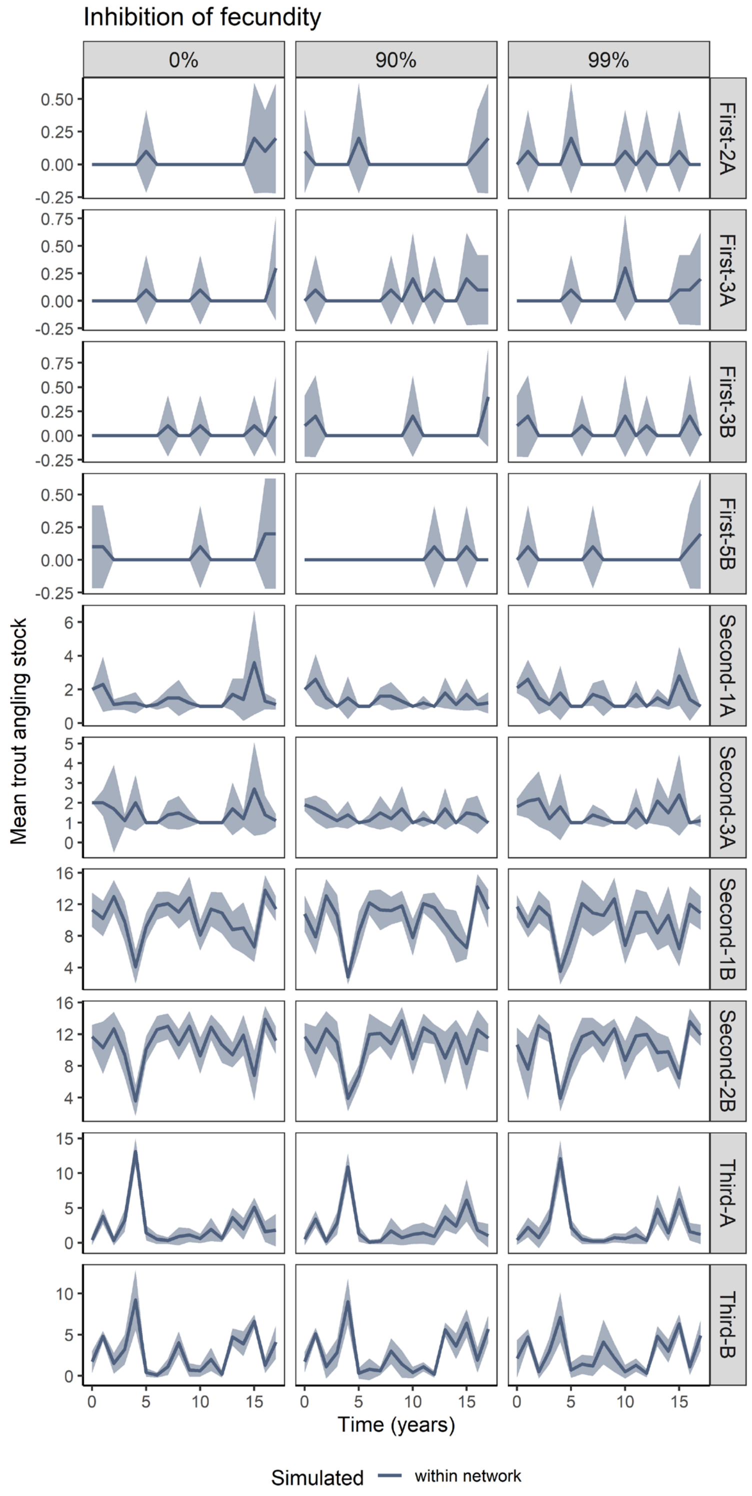


Figure A5.1 Mean angling stock abundance across the river network (number of individuals) at the end of the simulation from ten replicates and for all ten reaches exposed to chemical stress inhibition (0 – 100%) of fecundity within a river network (blue). Shading signifies +/- one standard deviation from replicates.

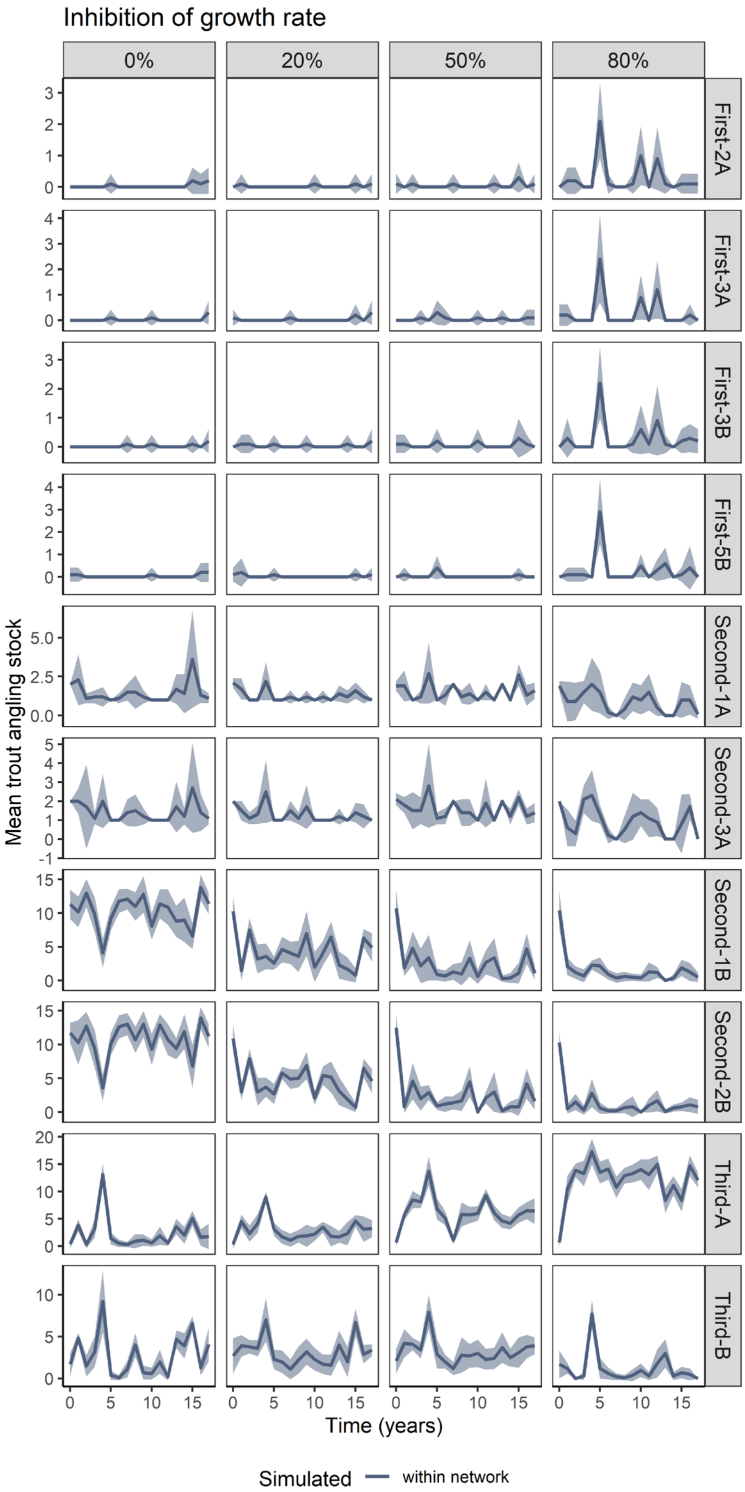


Figure A5.2 Mean angling stock abundance across the river network (number of individuals) at the end of the simulation from ten replicates and for all ten reaches exposed to chemical stress inhibition (0 – 100%) of growth rate within a river network (blue). Shading signifies +/- one standard deviation from replicates.

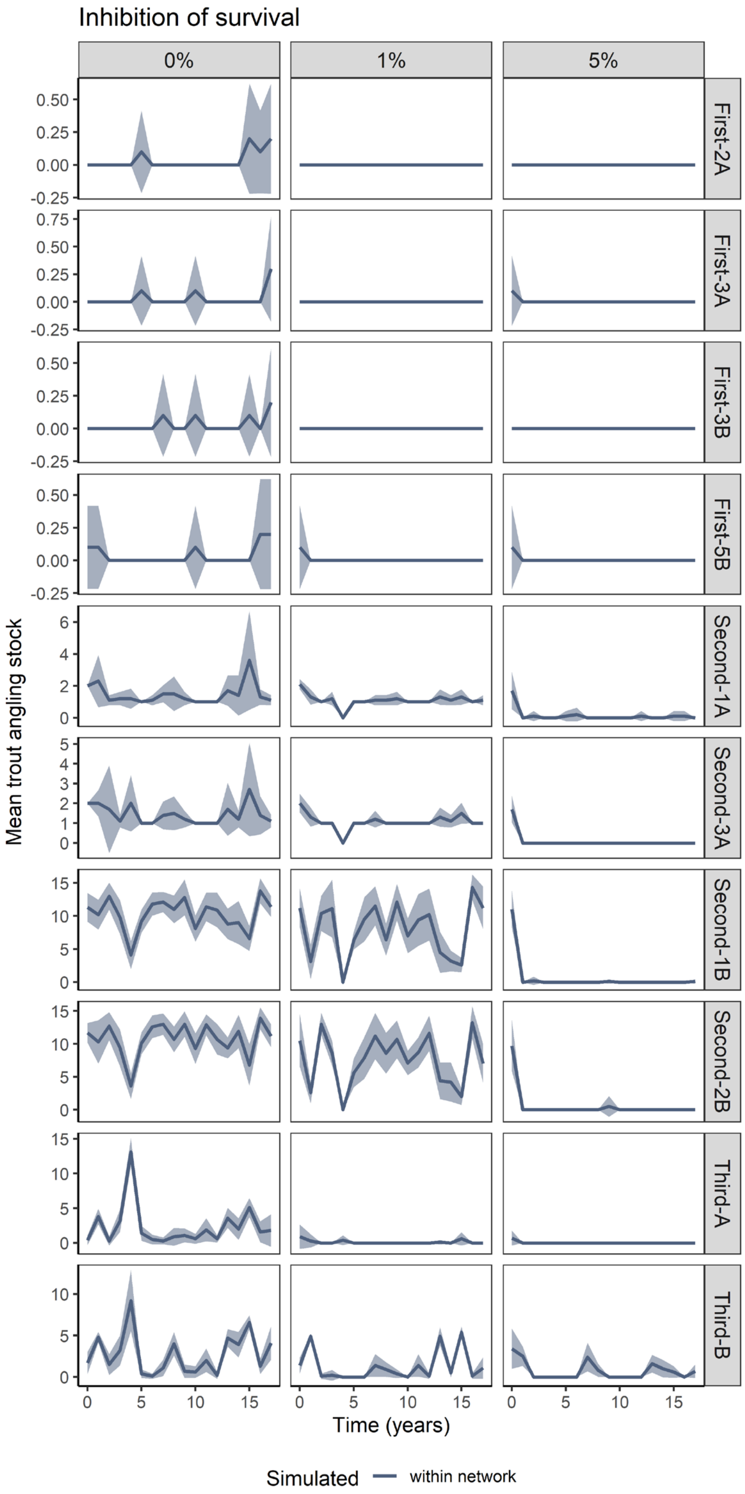


Figure A5.3 Mean angling stock abundance across the river network (number of individuals) at the end of the simulation from ten replicates and for all ten reaches exposed to chemical stress inhibition (0 – 100%) of survival within a river network (blue). Shading signifies +/- one standard deviation from replicates.

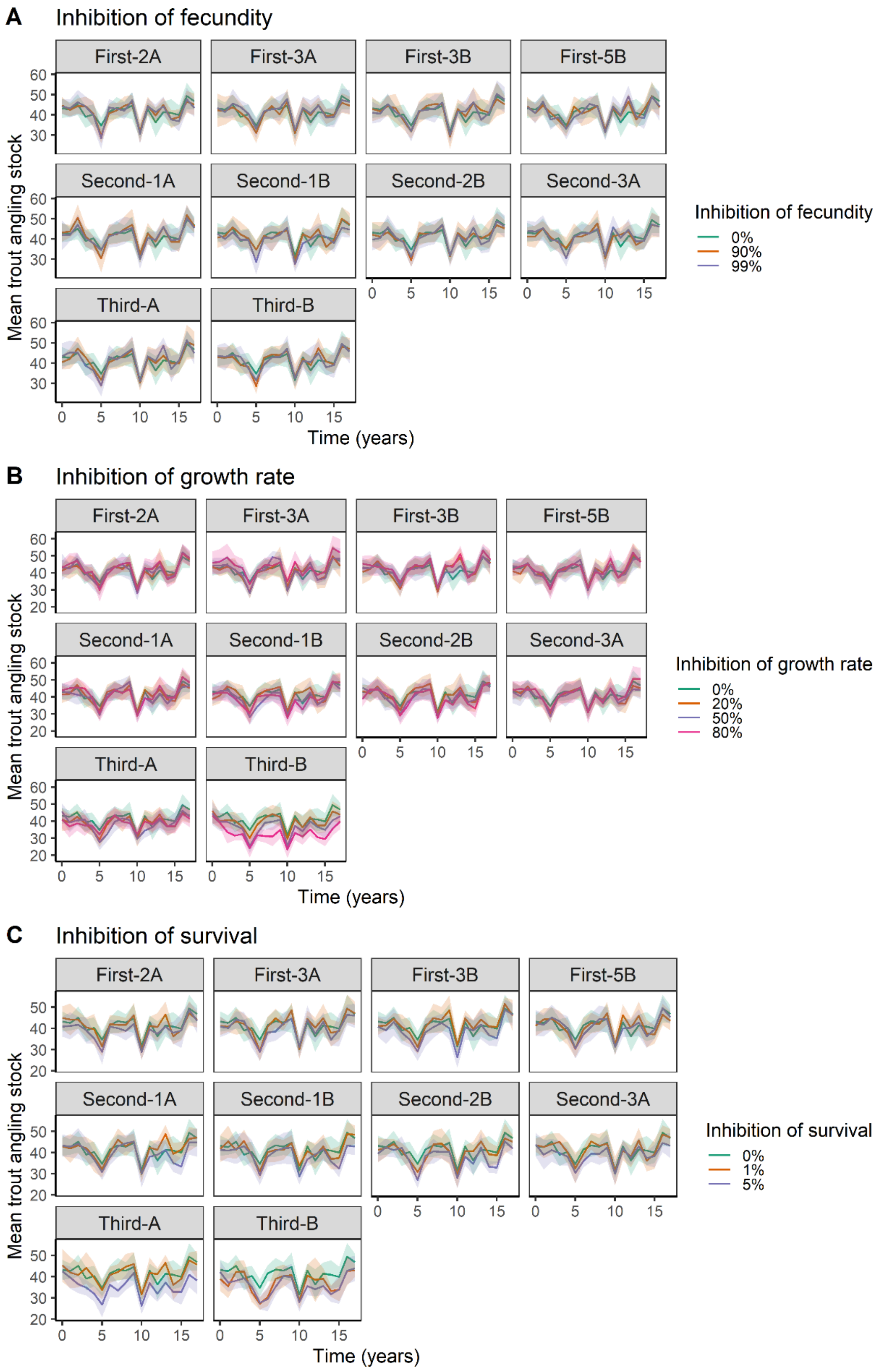


Figure A5.4 The total mean angling stock abundance (number of individuals older than 1 year and larger than 20 cm in length) in a river network over a 17-year simulation period and ten replicates for all ten reaches exposed to chemical stress inhibition (0 – 100%) of A) fecundity, B) growth rate, and C) survival. Shading signifies +/- one standard deviation from ten replicates.