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PROPOSTA DE INTEGRAÇÃO DE INSTRUMENTOS DE GESTÃO PARA ESTRATÉGIAS DE ALOCAÇÃO DA ÁGUA DE LONGO PRAZO

PORTO ALEGRE
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Dissertação apresentada ao Programa de Pós-graduação em Recursos Hídricos e Saneamento Ambiental da Universidade Federal do Rio Grande do Sul, como requisito parcial à obtenção do grau de mestre.

Orientador: Prof. Dr. Guilherme Fernandes Marques

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À memória de minha mãe
Normélia Borsoi Dalcin

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RESUMO

A gestão dos recursos hídricos requer o emprego de instrumentos a fim de regular e motivar o uso eficiente da água em uma bacia hidrográfica, materializando, com isso, os objetivos dos usuários. Exemplos de instrumentos incluem os planos de recursos hídricos, as outorgas de direito de uso da água, o enquadramento, a cobrança pelo uso da água, estratégias de alocação, entre outros. Contudo, dado que as decisões de uso da água dependem simultaneamente da quantidade disponível, onde, quando e com que qualidade, a implementação efetiva destes instrumentos exige um bom nível de integração entre os mesmos. Na prática, em diversas regiões e países, políticas de direito do uso da água são geralmente concedidas de acordo com a disponibilidade hídrica e da ordem de solicitação, com pouca ou nenhuma integração entre instrumentos. Este trabalho tem como principal objetivo propor uma abordagem para integrar os instrumentos de gestão, plano de bacia, outorga e enquadramento, e com isso materializar uma estratégia de alocação de água a longo prazo em uma bacia hidrográfica. A pesquisa contou com o desenvolvimento do modelo hidroeconômico VISTA, que é composto por três sub-rotinas de programação: (a) um algoritmo de programação dinâmica para otimizar a alocação de outorgas ao longo de um horizonte de planejamento seguindo o crescimento da demanda da água (alocação temporal); (b) um algoritmo de programação linear multiobjetivo (MOLP) para modelar diferentes políticas hídricas, separando as soluções não dominadas (fronteira de pareto) para encontrar soluções otimizadas de alocação de água para usos econômicos e ambiental (alocação entre usuários de água) e (c) um algoritmo de programação não linear para otimizar a distribuição espacial das outorgas na bacia hidrográfica, sujeita a restrições de qualidade da água de acordo com as metas de enquadramento estabelecidas (alocação espacial). A bacia do Rio dos Sinos no estado do Rio Grande do Sul, Brasil, foi utilizada como estudo de caso. Os resultados apontam que não necessariamente toda água disponível deve ser outorgada do ponto de vista econômico, sendo que políticas hídricas com preferência na proteção ambiental mostraram-se vantajosas economicamente e metas de qualidade menos restritivas de enquadramento não necessariamente trazem maiores benefícios econômicos.

Palavras-chave: modelagem hidroeconômica, política hídrica, instrumento de gestão de água, outorga, enquadramento, plano de bacia, economia dos recursos hídricos, programação dinâmica, otimização multiobjetivo

ABSTRACT

Water management requires effective instruments to regulate and motivate the efficient use of the water in a watershed, materializing the objectives of water users. Examples of instruments include water resources plans, water rights and permitting systems, water allocation strategies, water quality standards, water charges and markets, among others. Given water use decisions depends simultaneously on how much is available, where, when and with which quality, the implementation of these instruments demands their integration. In practice, the integration of different water management instruments is still limited in several regions and countries. Water permits and water rights have long been issued on a first come, first serve basis, often only based on the river water availability. The main goal of this study is to propose an approach to integrate water management instruments, water permits, water quality targets, and water resources plan, in order to materialize a long-term water allocation strategy in a watershed. To accomplish this, the hydro-economic model, named VISTA, was developed combining three sub-routines: (a) a dynamic programming algorithm to optimally allocate water permits through time following user's growth rates (temporal allocation), (b) a multi-objective Linear Programming (MOLP) algorithm to model multiple water policies composing Pareto frontiers and separate non-dominated solutions to allocate water permits to different sectors of economic users and environmental use (user sector allocation), and (c) a non-linear programming algorithm to spatially allocate water permits to users in a watershed, subject to pre-defined water quality standards (spatial allocation). The Sinos River Basin, located in the state of Rio Grande do Sul, Brazil, was used as study area. The results indicate that not necessarily all available water must be allocated to economic users, water policies with environmental protection preference have economic advantages, and less restrictive water quality targets do not necessarily produce higher economic benefits.

Keywords: hydro-economic modelling, water policy, water management instruments, water permit, water quality target, water resources plan, water allocation, dynamic programming, multi-objective optimization

SUMÁRIO

LISTA DE ABREVIAÇÕES

BOD	Biochemical Oxygen Demand
DBO	Demanda Bioquímica de oxigênio
DO	Dissolved Oxygen
OD	Oxigênio Dissolvido
US	United States
GAMS	General Algebraic Modeling System
IPH	Instituto de Pesquisas Hidráulicas
MATLAB	Matrix Laboratory
RS	Rio Grande do Sul
SRTM	Shuttle Radar Topography Mission
VISTA	Value Integrated Space-Temporal Allocation

CAPÍTULO 1

Introdução

1.1 Introdução

A gestão de recursos hídricos requer o emprego de instrumentos de maneira a regular, garantir e motivar o uso eficiente da água, materializando, com isso, os objetivos de seus usuários. Exemplos de instrumentos incluem os planos de recursos hídricos, as outorgas, o enquadramento, a cobrança pelo uso da água, entre outros. Dado que as decisões de uso da água dependem simultaneamente da quantidade disponível, onde, quando e com que qualidade, todos os instrumentos agem de forma interligada, necessitando de ações interdependentes de cooperação (ALMEIDA, 2003; ANA, 2009). Os planos de recursos hídricos devem atuar como instrumento orientador para a implementação dos demais instrumentos, estabelecendo diretrizes quanto a distribuição de outorgas, metas de qualidade da água e cobrança (ANA, 2013). Na prática, os planos não conseguem orientar as decisões de alocação da água de forma clara (OECD, 2015) e a formulação de tais diretrizes, essenciais para a implementação efetiva dos instrumentos, ainda é limitada por diversos aspectos.

Falta integração entre instrumentos. Políticas de direito do uso da água são geralmente concedidas de acordo com a disponibilidade hídrica e da ordem de solicitação do usuário. É o caso da lei da apropriação regulamentada pela suprema corte dos Estados Unidos em 1922 (Gelt, 1997) e a outorga de direito de uso da água no Brasil, definida pelo Código das Águas de 1934, e formalmente estruturada como instrumento da Política Nacional de Recursos Hídricos pela lei 9.433 de 1997. A abordagem atual de alocação da água pouco avalia os seus reflexos na qualidade da água, de modo a convergir para as metas previstas no enquadramento.

Falta visão de futuro. *Quais são as consequências a longo prazo de conceder uma determinada outorga, em um determinado local, para um determinado usuário?* A abordagem atual da alocação de água carece de uma visão de futuro quanto às consequências econômicas, sociais e ambientais das decisões tomadas no presente. A falta do emprego dos instrumentos em conjunto com uma avaliação do aumento da demanda em um horizonte de planejamento pode acarretar na alocação da água para usos menos eficientes, diminuindo a segurança e confiabilidade dos sistemas hídricos, bem como na dificuldade de alcançar as metas de enquadramento almejadas.

Falta conexão entre os instrumentos de gestão e políticas de desenvolvimento econômico. *Quais são as consequências das decisões de uso da água em termos de qualidade ambiental e benefícios econômicos auferidos com o uso da água?* Um crescimento econômico sustentável deve estimular continuamente o uso eficiente da água, a fim de acomodar novos usuários sob condições de fornecimento cada vez mais limitadas e escassas. Entretanto, políticas mais amplas de desenvolvimento econômico, envolvendo, por exemplo, expansão da geração de energia, expansão industrial, agricultura irrigada, dentre outras, frequentemente são traçadas com pouca ou nenhuma percepção dos seus impactos na bacia hidrográfica. O crescimento econômico traz consequências diretas sobre a gestão dos recursos hídricos, o que torna a compatibilização de políticas regionais ainda mais premente. Contudo, os investimentos são frequentemente realizados de forma pulverizada e nem sempre integrados em uma estratégia de desenvolvimento regional (OECD, 2015). Vale também destacar que investimentos em infraestrutura hídrica, a fim de aumentar a oferta em regiões escassas em recursos hídricos (a exemplo de transposições), podem levar incentivos a setores econômicos intensivos em demanda de água, aumentando futuros conflitos.

Falta definição de uma política hídrica. *Devemos atender a todas demandas econômicas por água? Quanta água devemos deixar para as gerações futuras? Quais as implicações nas demandas ambientais? Como obter e integrar uma política hídrica para nortear essas decisões?* Os planos de bacia não definem de forma clara e explícita as preferências da sociedade quanto ao uso da água na bacia a fim de nortear decisões de desenvolvimento econômico e proteção ambiental.

Como resultado das limitações apresentadas, o sistema de gestão de recursos hídricos perde efetividade. O crescimento das demandas em conjunto com a redução da confiabilidade e segurança dos sistemas hídricos potencializa a ocorrência de conflitos diversos pelo uso da água, limitando, por consequência, o atingimento de metas e objetivos quanto ao desenvolvimento econômico, social e ambiental. Conforme apontado pelo relatório OECD (2015), diversos estados, como a exemplo de São Paulo, Rio de Janeiro e Minas Gerais, tem enfrentado escassez quantitativa e qualitativa da água. A incerteza quanto à disponibilidade atual ou futura de água pode levar os governos locais ou os investidores privados a não aproveitar as potenciais oportunidades de desenvolvimento, o que se traduz em oportunidades perdidas. Ao mesmo tempo, existe o potencial para a capitalização excessiva, na qual os investidores apostam num projeto e depois descobrem que a quantidade ou a confiabilidade da água necessária não está disponível. O relatório ainda consta que metas de qualidade da água, proteção ambiental, expansão agrícola, industrial ou mesmo de expansão energética dificilmente serão atingidas, a menos que uma abordagem estratégica seja adotada para a alocação, com critérios mais bem alinhados com os objetivos sociais e de desenvolvimento mais amplo dos recursos hídricos.

Por fim, o aumento da demanda também reflete em alterações nos regimes de vazão dos corpos hídricos com consequências para os processos fluviais, ecossistemas dependentes e serviços ecossistêmicos relacionados (OECD, 2015). *Até que ponto o uso econômico da água traz benefícios maiores que os custos necessários para mitigar as externalidades geradas (combate à poluição)?*

As consequências apontadas pela falta de efetividade evidenciam a necessidade de serem adotadas estratégias de alocação da água mais efetivas, que proporcionem melhor transparência e conhecimento sobre as perdas-e-ganhos (*trade-offs*) de um determinado objetivo, meta de qualidade ou concessão de uma outorga. Para usos em que a água tem um valor tangível, modelos hidroeconômicos vêm sendo amplamente empregados com o propósito de conhecer os *trade-offs* econômicos de estratégias de alocação de água frente a cenários de conflito e escassez, como a exemplo de Letcher et al. (2004), Pulido-Velasquez et al. (2008) e George et al. (2011a, 2011b). No estudo de Pengelly et al. (2017), desenvolvimento econômico e alocação de água foram integrados em um modelo hidroeconômico, a fim de entender como a escassez de água pode limitar o desenvolvimento de uma determinada região. De forma complementar, em sistemas nos quais a qualidade da água é uma questão premente, alguns estudos incorporam a qualidade da água de forma restritiva ao processo de alocação da água como a exemplo de Moraes et al. (2008) e Davidsen et al. (2015).

Apesar dos avanços, a representação de demandas ambientais em modelos hidroeconômicos ainda apresenta algumas lacunas. A demanda ambiental da água é usualmente tratada nesses modelos na forma de restrição física (HAROU et al., 2009), onde a alocação entre usuários é realizada de maneira a atender uma vazão mínima ambiental no rio. Em outros exemplos, como Tilmant et al (2012), o valor de demandas ambientais foi estimado com emprego de meta análise. Dessa forma, a representação dos benefícios ambientais do ponto de vista econômico ainda se restringe a alguns usos específicos (MOMBLANCH et al., 2016), o que pode ocultar importantes resultados.

Outra limitação é quanto à dificuldade de implementação na prática de tais modelos por operadores ou órgãos de gestão (Harou et al., 2009). Há uma lacuna ainda não explorada pela literatura no que diz respeito à forma como as soluções obtidas pelos modelos poderiam se transformar em gerenciamento efetivo da água por parte dos usuários e gestores, principalmente quanto à formulação de diretrizes. Nós argumentamos que é papel dos instrumentos de gestão entregar as soluções aos usuários.

Diante do contexto apresentado, a hipótese assumida neste trabalho considera que integrar os instrumentos de gestão sob uma visão de planejamento de longo prazo aumenta a efetividade na gestão dos recursos hídricos, auxiliando também a reconciliar o desenvolvimento econômico com a qualidade ambiental. Dessa forma, como objetivo é proposto uma abordagem para integrar os instrumentos de gestão, **plano de bacia**, **outorga** e e**nquadramento**, e como isso materializar uma estratégia de alocação de água de longo prazo. A integração também visa contribuir para o emprego de **instrumentos econômicos,** a fim de estimular o uso racional da água, bem como visa possibilitar englobar a avaliação da demanda ambiental na forma de **política hídrica**.

Para realizar esta proposta de integração, foi desenvolvido, no âmbito desta pesquisa, o modelo VISTA (*Value Integrated Space-Temporal Allocation*), o qual é composto por três módulos de programação que operam de forma integrada. O **módulo I** apresenta um algoritmo de programação dinâmica para alocar outorgas no tempo seguindo curvas de crescimento da demanda (alocação temporal). O **módulo II** apresenta um algoritmo de programação linear multiobjetivo (MOLP) para modelar múltiplas políticas hídricas e encontrar soluções otimizadas de distribuição de outorgas entre usuários econômico e ambiental (alocação entre usuários). Já, o **módulo III** utiliza um algoritmo de programação não linear para encontrar soluções ótimas de distribuição espacial de outorgas na bacia hidrográfica de maneira a seguir metas de enquadramento (alocação espacial).

A ferramenta desenvolvida pode ser empregada tanto na elaboração e na revisão de planos de recursos hídricos, bem como em outros processos de planejamento trazendo como principal contribuição para a área de estudo a possibilidade de identificar e criar diretrizes para os instrumentos de gestão, o que é essencial para permitir a integração e melhora da efetividade do sistema de gestão de recursos hídricos como um todo. Dessa forma, quando um usuário solicita uma nova outorga, a autoridade competente poderá checar as diretrizes no Plano de bacia de acordo com a Política hídrica e meta de enquadramento adotada, a fim de decidir se a outorga deve ser concedida naquele local (onde), naquele momento (quando) e na quantidade solicitada (quanto).

Por fim, a bacia do Rio dos Sinos localizada no estado do Rio Grande do Sul, Brasil, foi utilizada como bacia alvo de estudo de maneira a explorar a hipótese assumida e ilustrar o emprego do modelo hidroeconômico VISTA desenvolvido.

1.2 Objetivos

Esta dissertação de mestrado tem como principal objetivo:

Propor uma abordagem para integrar os instrumentos de gestão, plano de bacia, outorga, enquadramento, e com isso materializar uma estratégia de alocação de água de longo prazo.

Mais especificamente, esta pesquisa visa:

a) Desenvolver uma metodologia para compor diferentes políticas hídricas representando níveis de preferência ótimos de alocação entre setores usuários de água econômico e ambiental;
b) Desenvolver um modelo de otimização/simulação da qualidade da água que possibilite integrar o instrumento enquadramento (através de metas de qualidade) e avaliar as implicações na qualidade da água resultantes da alocação espacial de água na bacia hidrográfica;
c) Desenvolver um modelo hidroeconômico que integre a avaliação de diferentes políticas hídricas, as demandas de água de usuários econômicos (tendo como base políticas de desenvolvimento) e as implicações de metas de enquadramento para compor trajetórias de alocação de água economicamente ótimas no tempo (horizonte de planejamento) e no espaço (bacia hidrográfica);
d) Aplicar o modelo hidroeconômico em uma bacia alvo de estudo utilizando diferentes cenários para avaliação dos resultados;
e) Avaliar os benefícios resultantes da abordagem integrada na gestão de recursos hídricos proposta em relação à abordagem atual fragmentada.

1.3 Organização do trabalho

Este trabalho é estruturado na forma de artigos, separados por capítulos, conforme descrito:

Capítulo 2: *Reconciling water policies with broader economic development policies through integrated water management instruments*

Capítulo 3: *Integrating water permits and quality targets to establish a long-term spatial water allocation strategy*

Capítulo 4: *Avaliação de um modelo simplificado de simulação da qualidade da água para o Rio dos Sinos*

O artigo do capítulo 2, *Reconciling water policies with broader economic development policies through integrated water management instruments*, aborda a formulação e configuração do modelo VISTA desenvolvido no âmbito desta pesquisa como ferramenta para possibilitar integrar os diferentes instrumentos de gestão na avaliação de estratégias de alocação de água de longo prazo numa bacia hidrográfica. O modelo é então aplicado em uma bacia alvo do estudo, Bacia do Rio dos Sinos - RS, na qual são apresentados os dados de entrada utilizados, os cenários modelados e os resultados obtidos, sendo os resultados espaciais da modelagem abordados separadamente no capítulo 3, *Integrating water permits and quality targets to establish a long term spatial water allocation strategy*.

O artigo do capítulo 4, *Avaliação de um modelo simplificado de simulação da qualidade da água para o Rio dos Sinos*, apresenta a formulação e verificação do modelo simplificado de qualidade da água utilizado como parte integrante do modelo hidroeconômico VISTA.

Por fim, no capítulo 5, são apresentadas as conclusões gerais e principais aprendizados obtidos com o desenvolvimento do trabalho.

CAPÍTULO 2

Reconciling water policies with broader economic development policies through integrated water management instruments

2.1 Introduction

Water management requires effective instruments to regulate and motivate the efficient use of the water in a watershed, materializing the objectives of water users. Examples of instruments include water resources plans, water rights and permitting systems, water allocation strategies, water quality standards, water charges and markets, among others. Given water use decisions depends simultaneously on how much is available, where, when and with which quality, the implementation of these instruments demands their integration. For example, the water resources plans should provide clear directives for distribution of water permits and definition of water quality standard targets. Such directives depend on a negotiation between users, which has the water allocation at its center.

In practice, the integration of different water management instruments is still limited in several regions and countries. Water permits and water rights have long been issued on a first come, first serve basis, often only based on the river water availability. This has been the case with the prior appropriation law ruled by the US Supreme Court in 1922 (Gelt, 1997) and water concessions defined in the Brazilian Water code of 1934, formally structured as a water management instrument with the water permit in the National Water Resources Policy law of 1997. Water permitting systems such as the Brazilian one, where users fill out requests to the management authority and those requests are processed primarily based on their individual filling order lack future vision about the long-term consequences on how the water is allocated, bearing no connection with broader development policies that would be fostering a particular demand in a particular watershed. As result, the water management authority trails behind the problem; approving permits mostly based on the water availability and missing the opportunity to use the water permit as a real instrument to reevaluate and improve water allocation and management in tune with other development policies that affect water demands (e.g. energy, food, environment).

A sustainable economic growth would foster (and enforce) increasingly efficient water use in order to accommodate new users under limited (and scarce) water supplies. This brings the water policy as a key element that represents society's preferences and priorities towards environmental quality. Choosing a water policy is part of the water management activities.

However, very often broader development policies are drawn with very little perception of their impact to the rivers in a watershed. The resulting economic growth increases the stress over the environment, compromising environmental quality. When local inhabitants and affected users push back to tighten water control, licensing and river protection, it conflicts with the broader development policy. In this situation, there is no water policy clearly defined, and even the call for improved water management is often disconnected from the broader development goals in the region.

In this context, the negotiation between users is limited, often due to lack of knowledge on the trade-offs associated to a given objective, water quality standard or water allocation to a given group of users. For example, tighter water quality standards require higher investments in wastewater treatment and the impossibility to issue environmental permitting to some types of industries or uses. Such trade-offs should be organic to the negotiation process so that users can more clearly understand the present, and especially the future outcomes of decisions on water allocation and water quality targets. In order to accomplish that, the water management instruments need to be integrated: the water resource plans should clearly identify the water policy for the watershed, which will set the directives for river water quality standards and the amount of water that should be allocated to environmental demands and to other competing users. Based on these goals, other instruments, such as water permitting, will define when, where and how much water will be made available to meet the watershed demands. When a new user requests a water permit, the river basin authority or department of water resources will check on the water policy directives to decide if that permit should be issued on that particular location, at that moment in time and in the required amount.

Such decisions however depend on some performance criteria to determine what is “best” in terms of water allocation. For those uses where the water has a tangible economic value, hydro-economic models have a track record as a useful tool for evaluating economic trade-offs of water allocation strategies and searching for improved (economic more efficient) water use operations. In a context of competition over scarce water resources, knowing the trade-offs among different water allocation strategies provide valuable information to negotiators. Recent examples of such studies include Rosegrant et al. (2000), Lund; Cai; Characklis (2006), Pulido-Velazquez et al. (2008), Kondili; Kaldellis; Papapostolou (2010), George et al. (2011a), George et al. (2011b) , Wang et al. (2015), Roozbahani; Schreider; Abbasi (2015), Hu et al. (2016a), Hu et al. (2016b), Ghosh et al. (2017), Xu et al. (2019), Letcher; Jakeman; Croke (2004), Grafton et al. (2011). Wang et al (2009) proposed an optimization model for allocating water resources in a river basin over the long term, through combination of forecasting method to predict domestic and industrial water demands.

In water systems where water quality is a pressing issue, the definition of water quality standards is tightly associated with water availability and it is likely to reflect on other instruments, like water permits. Some studies incorporate water quality analyses in the water allocation approach, as in the example of Azevedo et al. (2000) who applied the water quality model QUAL-2E-UNCA to simulate the water quality resulting from a water allocation optimization model. Wang; Yang; Chang (2019) used an one-dimensional quality model based on mass balance and depuration equations to simulate the water quality resulting from the water allocation optimization process of a multi-water resources. Cai; Mckinney; Lasdon (2003) developed an integrated hydrologic-agronomic-economic model that includes quality simulation of salinity in the river basin network and salt balance in crop root zones, which results in penalty taxes based on salt discharge.

Other works also include water quality requirements as a constraint in optimal water allocation models besides water quantity. Tu (2006) developed a model to optimize water allocation in a water-distribution system considering sources of varied water quality to users with different water quality requirements. Hemmat et al. (2007) proposed a water allocation model of a river-reservoir system based on user's quantity and quality requirements applying genetic algorithm optimization and water quantity/quality simulation models. Moraes et al. (2008) developed a hydro-economic model, which incorporates water quality restrictions based on Streeter–Phelps equation for OD and BOD constituents, to determine optimal water allocation and effluent allocation to fertigation of sugar-cane areas. Ahnadi et al. (2012) developed a model to provide water to downstream users following quality and quantity requirements based on mass balance equations while maximizing agricultural production of upstream land. In Molinos-Senante et al. (2014) the allocation model determines the volume of water that should be supplied to each demand unit and identifies from which source should be supplied the water following quality requirements.

Water quality standards and downstream users' requirement for a certain quality also influence decisions on wastewater treatment investment, which should be evaluated along with other economic benefits. For example, Davidsen et al. (2015) proposed an optimization model that enables to compare economic impacts of complying with various water quality grades thought inclusion of optimal pollution discharge and water treatment in the water allocation problem using Streeter–Phelps equation to compute OD concentrations in the river. Martinsen et al. (2019), proposed a water allocation model that optimizes water delivery to end-users according to quantity and quality requirements while minimizing total costs of groundwater pumping and surface cleaning when the quality of the source do not comply with the quality demand.

In recent years, the environmental demand and protection has emerged as a key element in the allocation process. With new amendments (as the 1972 amendment to the US Clean

Water Act) there has been an increased focus on understanding the environmental and socio-economic benefits of leaving water in streams, rivers and aquifers rather than extracting it for consumptive use (Momblanch et al., 2016). However, Harou et al. (2009), who review 80 hydro-economic modeling from 23 countries dating back 45 years, point out that environmental water uses, such as ecological minimum in-stream flows are usually not represented economically. Momblanch et al. (2016) reviewed 95 studies applying hydro-economic models and documented how the environmental demands is represented, along with the methods used to value environmental costs and benefits. The authors concluded that about two thirds considered environmental aspects in physical terms, mostly as constraints to realizing other use values. The third which valued at least one environmental impact in economic terms were mostly limited to a single environmental aspect or included very broad or vague environmental aspects. Recreation, commercial fishing and salt dilution were the most frequently valued in hydro-economic models. Considering trade-offs analyses between ecosystem and societal needs for water, Zamani et al. (2019) developed a spatial planning framework that enables conservation practitioners to strategically allocate incentives for water conservation actions to balance ecosystem and societal needs for water based on a Pareto tradeoff curve between meeting societal water needs and environmental flow goals.

However, while existing literature in the field have explored at length the outcomes of various water allocation strategies choices with optimization models, there is still a gap on how to deliver those solutions to the users in the field. We argue it is the role of the water management instruments to accomplish this task, but to our best knowledge, there is still no work that proposes how to integrate water management instruments coherently under a long-term planning vision to improve their effectiveness. In addition, the studies addressing water quality and quantity conflicts generally have in common a fixed network of users, not considering the incorporation of new ones, or the increase in existing uses in the system. Finally, while the integrated water resources management (IWRM) concept covers wide terms and issues (Biswas, 2004), what we focus here is the integration of water management instruments and their implementation in operational terms.

This paper presents a novel approach to reconcile water management actions, and it provides an original contribution to the field by filling the integration gap among selected water management instruments. The water management actions addressed here include (i) the definition of the water policy; (ii) the distribution of water permits and (iii) the evaluation of water quality targets with a broader development policy, which drives user's water demand growth. To accomplish this, the paper proposes a methodology that combines (a) explicit modeling and identification of different water policies composing Pareto frontiers with two objectives (non allocated water for environmental demands and water for economic uses), (b) an exogenous development policy and (c) search for economically optimal water allocation solutions both in time and space that meet prescribed water quality standards in the watershed. An important aspect is that the solutions provided in (c) has the water policy as boundary condition, which can be changed and adjusted along the planning horizon. Also, the water allocation is provided with a hydro-economic model that treats water demand growth, and the water economic values (economic demand) separately. Thus, instead of running a hydro-economic model to derive optimal economically efficient solutions as in Pulido-Velazquez; Andreu; Sahuquillo (2006) our approach considers multiple Pareto-efficient solutions first, each one representing a possible water policy, followed by optimization of economic returns to allocate water in time and in space. This approach removes the limitation of the economic optimization not including other water values that are difficult (of for which there is limited data) to evaluate, such as water demands for certain ecosystem services. Instead, it represents those with water volume units in the Pareto frontier. The approach proposed here uses water quality simulation and economic optimization integrated in the long-term planning (together with development policies), enabling one to explore optimal spatial allocation of water permits at minimum wastewater treatment costs to accomplish river and user's water quality requirements.

The methodology combines three sub-routines: a dynamic programming algorithm to optimally allocate water permits through time following user's growth rates (temporal allocation), a multi-objective Linear Programming (MOLP) algorithm to model multiple water policies and separate non-dominated solutions to allocate water permits to different sectors of economic users and environmental use (user sector allocation), and a non-linear programming algorithm to spatially allocate water permits to users in a watershed, subject to pre-defined water quality standards (spatial allocation). All algorithms are integrated in a single model, named VISTA *(Value Integrated Space-Temporal Allocation).*

By combining an explicit declaration of users' preferences in a water policy, with a user growth rate and optimal water allocation and water quality solutions, our approach allows the investigation of solutions that can contribute to reconcile goals of economic growth and environmental quality. This may be useful in finding negotiation solutions for water allocation and management that depart from the traditional conflict between economic development and environmental quality. Most importantly however, the results provided by the approach proposed here should help water managers and users to identify directives to water management instruments that will deliver improved water allocation and water quality standard solutions to the field.

2.2 Methodology

2.2.1 Policies definition

In this paper we adopt three import policy definitions for modeling purposes. First, we define ***development policy*** as the main driver that determines how a region will attain economic growth, generate employment, and fight poverty, among other objectives. The decisions which characterize such policies define how new business, investment and opportunities are attracted to the region. This could be made with investment in key infrastructure (airports, ports, roads, power transmission lines, and such), tax easements and deductions, subsidies and other. The number of water users and their activity level (e.g. crop acreage or industrial output) may increase as consequence of this growth, which will reflect on water demands.

Second, we define ***water policy*** as the level of preference between two main water uses: economic and environmental. This definition is discretized in a scale from 1 to 10. 10 being strictly environmental and 1 strictly economic. The closer the water policies are to 1, the more available water is allocated to economic uses (i.e. urban, agricultural, and industrial). The closer the water policies are to 10, the less water is allocated to economic uses and more is left in the river to fulfill environmental water demands and protection (this is referred to here as "flow allocated to the environment"). We assume water is allocated through water permits. Although economic water policies allow higher global economic returns from water use, it requests higher wastewater treatment investments. On the other hand, environmental water policies increase water dilution capacity and reduce both water abstraction and wastewater discharges, which contribute to protect water streams.

Water policies are subject to change. Given water plans are usually revised every 5 years, the dynamic programming model uses this interval as stage duration to allocate water (distributing water permits) along the planning horizon. However, from one stage to another the water policies driving the level of preference between environmental and economic water uses can be changed, allowing us to investigate how a given time trajectory of water policies fare along the time horizon. This is a relevant aspect given there may be different trajectories leading from current preferences in the watershed (current water policy) to a future where there is a given target on water and environmental quality (future water policy). The questions are:

how to get there? Should preference in allocating water to the environment be changed abruptly or more gradually? should it be conducted early in the planning horizon or delayed?

The third definition is the ***allocation strategy***. The allocation strategy defines how water is allocated among economic water users to maximize the economic benefit minus wastewater treatment costs. It should be noted that while both definitions of water policy and allocation strategy involve water allocation, we prefer to separate economic benefits from environmental ones and draw a Pareto frontier to allow users to pick the desirable water policy, which will set the main principle for water allocation (i.e. higher preference to maintain the environment or to economic returns). Having this policy as boundary condition, our approach defines the allocation strategy, which will determine the "best" plan of action distributing the water permits across the watershed, considering: (a) existing water quality targets in different river reaches, (b) changing water quality along multiple river reaches according to local flow and river conditions and (c) wastewater treatment costs. Given water left in the river to fulfill environmental water demands has several intangible benefits, some of which are either difficult/contentious to value economically (although the growing economic techniques to value intangible benefits) or for which there is no data, such separation avoids controversy. At this point, the users can readily identify the tradeoffs resulting from alternative water policies in the Pareto front.

2.2.2 Model configuration

In order to configure the VISTA, the following input elements are defined: (1) planning horizon; (2) total allocable flow along the planning horizon; (3) water policy trajectory; (4) water quality targets; (5) users that take part in the allocation process; and (6) development policy.

These elements are organized in three modeling sub-routines: (I) Temporal allocation, (II) Water policy and user sector allocation and (III) User spatial allocation. The sub-routines interact by using GAMS and MATLAB software. Figure 1 and 2 demonstrate the modelling sequence and main data.

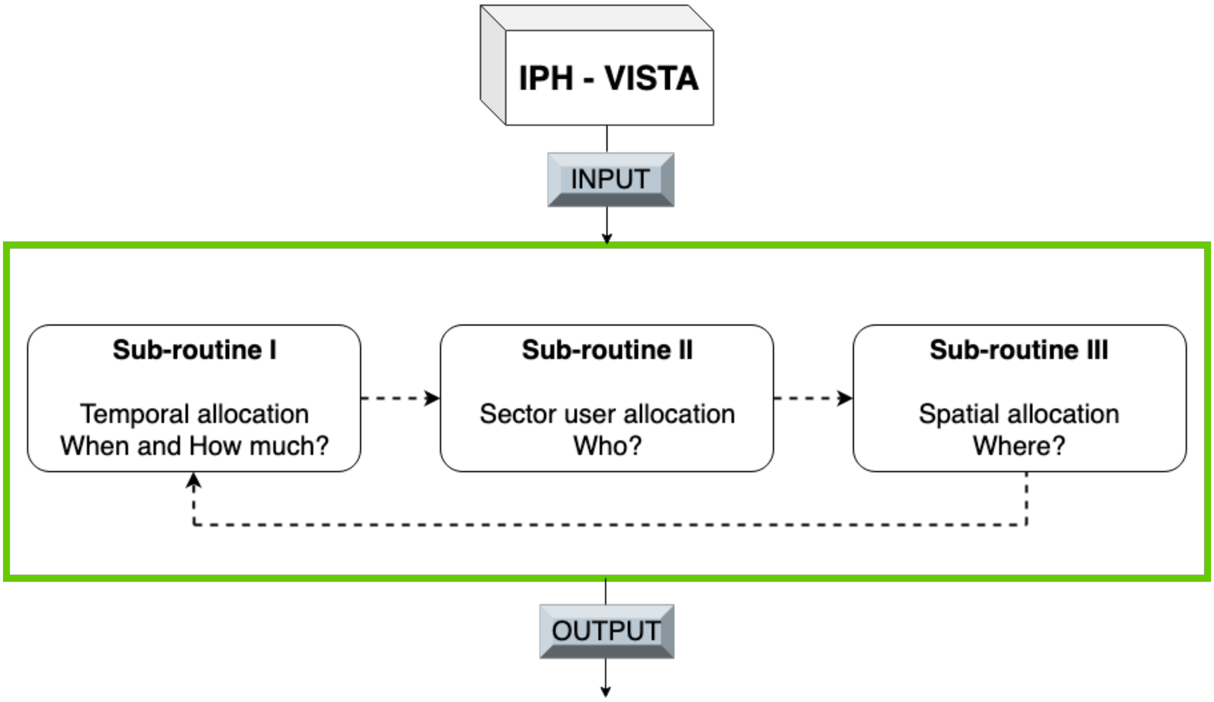

Figure 1: Modelling sequence.

The total allocable flow is the river flow with a given return period (flow duration value). Instead of using a time series or an implicit stochastic approach, we resorted to a single flow duration value directly associated with the probability of exceedance, which clearly indicates the risk assumed by the water planner (e.g. Q_{90}, $Q_{7,10}$ etc). Also, the water agencies in Brazil, where the model is applied, issue water permits having a given river flow duration value as a reference, thus making the approach more readily applicable to the water management instrument used (water permits). Finally, while we have adopted a single flow duration value throughout the planning horizon for simplicity, the approach allows for it to change from one stage to another, thus enabling incorporation of potential climate change impacts in local hydrology. The latter, however, is out of the scope of this paper.

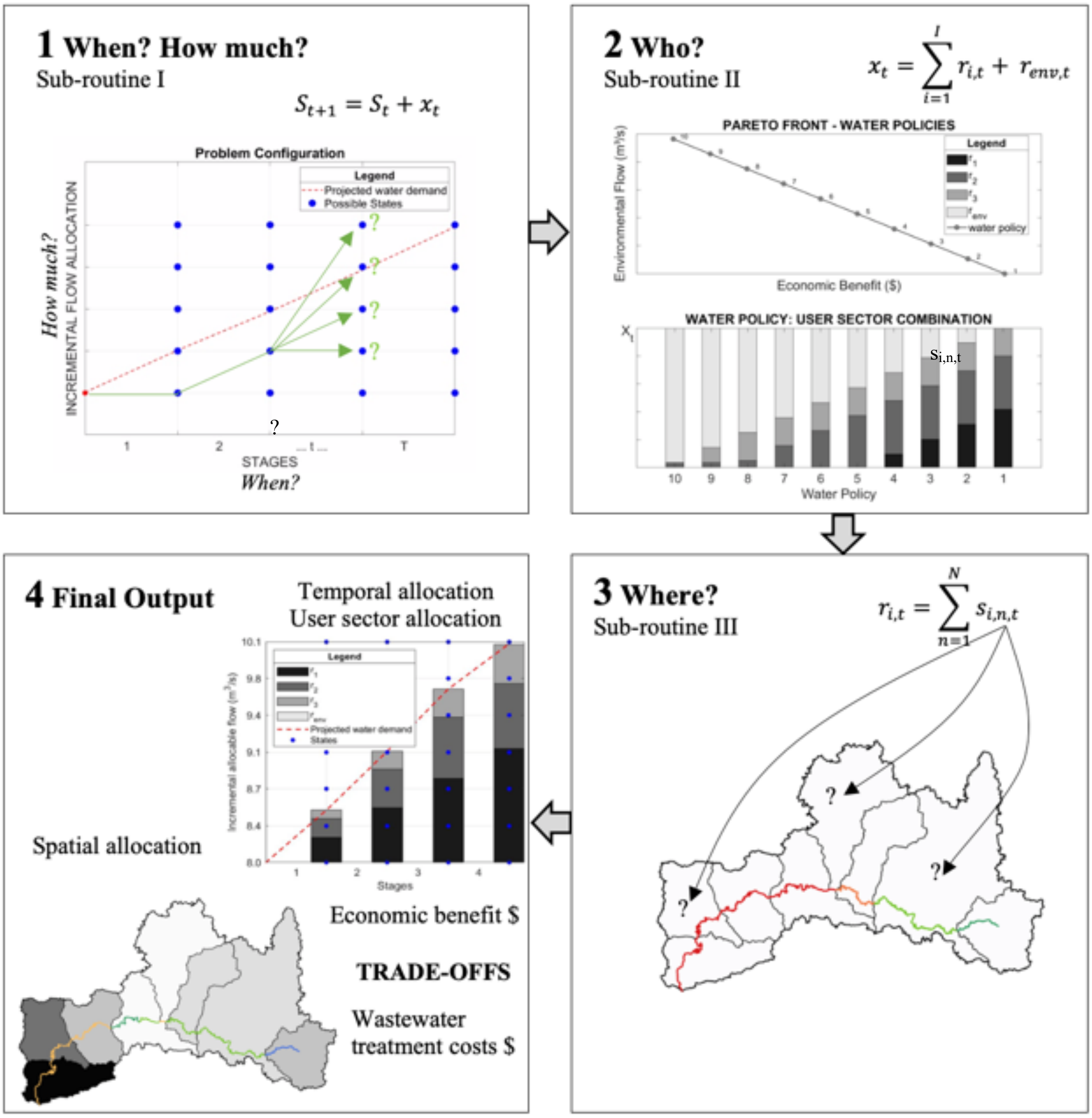

Figure 2: Sub-routines and final output data.

Modeling sub-routine I: Temporal allocation (when and how much?)

The first problem modeled is the allocation of water permits in time, having the allocable flow as upper bound. The total planning period T is discretized in stages t and the allocable flow S_{max} is discretized in states S_t in order to break the main problem into sub-problems, which are solved by a discrete dynamic programming algorithm to find the optimal allocation trajectory, having economic benefit maximization as objective function. The decision variables are the total amount of water that will be allocated in each stage x_t through water permits.

For each stage t it is possible to increase the incremental flow allocation (water permits concessions) in x_t increments, varying from zero to the maximum allocable flow S_{max} (Figure 3). The state variables S_t represent the total allocated flow at the beginning of Stage t, which increased by the incremental flow allocation x_t, result in the allocated flow at the end of Stage t, S_{t+1} (1).

$$S_{t+1} = S_t + x_t \qquad \forall\, t \tag{1}$$

$$S_{t+1} \leq S_{max} \qquad \forall\, t \tag{2}$$

$$0 \leq x_t \leq S_{max} - S_t \qquad \forall\, t \tag{3}$$

The VISTA represents water demands with separate water economic value functions and user's water activity level growth functions. Water activity level refer to acreages planted, industrial output and total population, which changes in time according to development policy assumed. The water economic value functions represent the economic demand of each user sector (e.g. industrial, agricultural and urban).

The upward sloping line in Figure 1 represents the total system projected water demand, which is the sum of the product of each sector activity level at each stage by a reference water use rate (flow units/hectare and industrial output and per capita consumption) (17). The reference water use rate is how much water users would consume at the urban water tariff values (for urban users) and at the point where marginal water costs equal marginal water benefits (for industrial and irrigated agriculture users). For the industrial sector, as the output widely varies according to the facility type, applying the water activity level growth as percentage value directly to the current industrial water demand in the watershed leaves the water use rate as an implicit.

The model cannot allocate more water than the total system projected water demand in a given stage, but it can allocate less. In the latter case, the economic benefit to the user is smaller and it is calculated individually to each user based on its economic water demand. Hence, while the *projected water demand* estimates the maximum amount of water the users would be willing to withdraw (and its change in the planning horizon), the *economic water demand* describes the variation of the water value to the user, for different quantities used.

Hence, the decisions over time of how much additional water will be allocated depends both on the user's projected water demands and on the economic value of the water, which vary through time. Each stage t has a correspondent projected water demand D_{pr_t}.

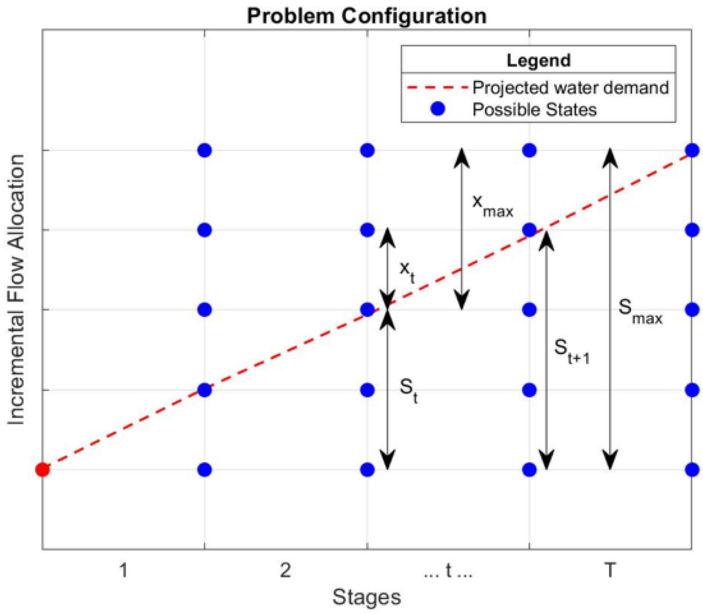

Figure 3: Graphic representation of the problem configuration.

The economic benefits from a given incremental flow allocation x_t, along with the corresponding wastewater treatment cost, are calculated through the next sub-routines II and III resulting in a final economic benefit $C_t(S_t,x_t)$ which is relayed back to the Dynamic Programming routine. Hence, for each discrete incremental flow allocation x_t and each stage t, modeling sub-routines II and III are executed.

The Discrete Dynamic Programming algorithm uses a forward moving procedure (4).

$$F_t(S_{t+1}) = maximize[(C_t(S_{t+1} - x_t, x_t) + (F_{t-1}(S_{t+1} - x_t)] \quad \forall\, t \tag{4}$$

Where:

$C_t(S_t, x_t)$ is the economic benefit gained by the incremental flow allocation x_t departing from S_t

$F_t(S_{t+1})$ is the economic benefit accumulated at each stage t and state S_t

Modeling sub-routine II: Water policy and user sector allocation (who?)

The second problem modeled is the distribution of the incremental flow allocation x_t (total water permits) through the user sectors in the system, discretized by the decision variable r. The index i represents the economic user sector (e.g. urban, industrial, agricultural; $i = 1, 2, ..., I$), and the index env represents the portion of the incremental flow allocated to the environment (5).

$$x_t = \sum_{i=1}^{I} r_{i,t} + r_{env,t} \qquad \forall\, i, t \tag{5}$$

There are several possible combinations of r_i that result in the same increment x_t. In order to solve the problem (who will receive water and how much), a multi-objective linear programming (MOLP) is run for each flow increment decision x_t, considering two objectives: (OF_1) maximize benefits of economic water use (6) and (OF_2) increase the flow allocated to the environment (7). The MOLP is solved through the augmented ε-constraint approach, an improved variation of the ε-constraint (MAVROTAS, 2009).

$$OF_1\text{: maximize } [\sum_{i=1}^{I} B_{(r_{i,t})} \quad \forall\, t] \tag{6}$$

$$OF_2\text{: maximize } [r_{env,t} \quad \forall\, t] \tag{7}$$

Where $B_{(ri)}$ is the economic benefit function of the respective user. We have adopted concave economic benefit functions, which were represented as stepwise linear functions in the MOLP algorithm (8 to 14).

$$B_{(ri)} = \sum_{j=1}^{J} slope_{i,j} . rr_{i,j} \qquad \forall\, i, j \tag{8}$$

$$h_{\max i} = \frac{-b}{2a} \qquad \forall\, i \tag{9}$$

$$delta_i = \frac{h_{\max i}}{J} \qquad \forall\, i \tag{10}$$

$$rr_{i,j} \leq delta_i \qquad \forall\, i, j \tag{11}$$

$$r_i = \sum_{j=1}^{J} rr_{i,j} \qquad \forall\, i, j \tag{12}$$

$$b_{i,j} = a.(j.delta_i)^2 + b.(j.delta_i) + c \qquad \forall\, i, j \tag{13}$$

$$slope_{i,j} = \frac{b_{i,j} - b_{i,j-1}}{delta_i} \qquad \forall\, i, j \tag{14}$$

For a given user i, water is allocated through the different j segments producing a benefit (8) which is the product of the amount allocated to the segment ($rr_{i,j}$) by the segment slope ($slope_{i,j}$). As each segment has a finite size, $rr_{i,j}$ is limited by (11), which calculates the size of each segment based on the maximum demand for the user $h_{\max i}$ and the number of segments J. The slope of each segment is calculated through (13) and (14), which approximates the whole function with a 2nd order polynomial. For this, the position of the vertices of each segment ($b_{i,j}$ and $b_{i,j-1}$) are calculated based on the parameters of the polynomial (a, b, c) and the size of the segment, followed by the calculation of the slope of the segment ($slope_{i,j}$). Finally, the summation of the water allocated to all segments must be equal to the total allocated to user i, r_i (12).

The model constrains the maximum allocation allowed for each economic user sector i to its projected water demand $d_{pr_{i,t}}$ (15). As urban water demand combines potable and non-potable uses, a lower bound constraint is also set for the urban sector in order to ensure the minimum coverage of potable demand (16). The sum of all user's sector projected water demands results in the total system projected water demand (17). The flow allocated to the environment r_{env} can receive any value throughout all grid flow increments possibilities.

$$r_{i,t} \leq d_{pr_{i,t}} \qquad \forall\, i, t \tag{15}$$

$$r_{i,t} \geq d_{min_{i,t}} \qquad \forall\, i = urban, t \tag{16}$$

$$D_{pr_t} = \sum_{i=1}^{I} d_{pr_{i,t}} \qquad \forall\, t \tag{17}$$

Where:

D_{pr_t} is the total system projected water demand at each stage t

$d_{pr_{i,t}}$ is the projected water demand for each user i at each stage t

$d_{min_{i,t}}$ is the lower bound on urban demand for i = *urban* use at each stage t

As a result, the MOLP algorithm produces a discrete pareto front with m points, labeled from 1 to 10, each one associated with a flow allocated to the environment (m^3/s) and an economic benefit ($) (Figure 4). The points represent different water policies options: closer to 1, further to the right in Figure 2, means more water available to be allocated to economic uses (i.e. water policy with stronger preference for economic monetary returns of direct water use in agriculture, industry and urban). Closer to 10, further to the left in Figure 4, means less available water to be allocated to economic uses and more left in the river to fulfill environmental water demands and protection (i.e. water policy with stronger preference for environmental quality).

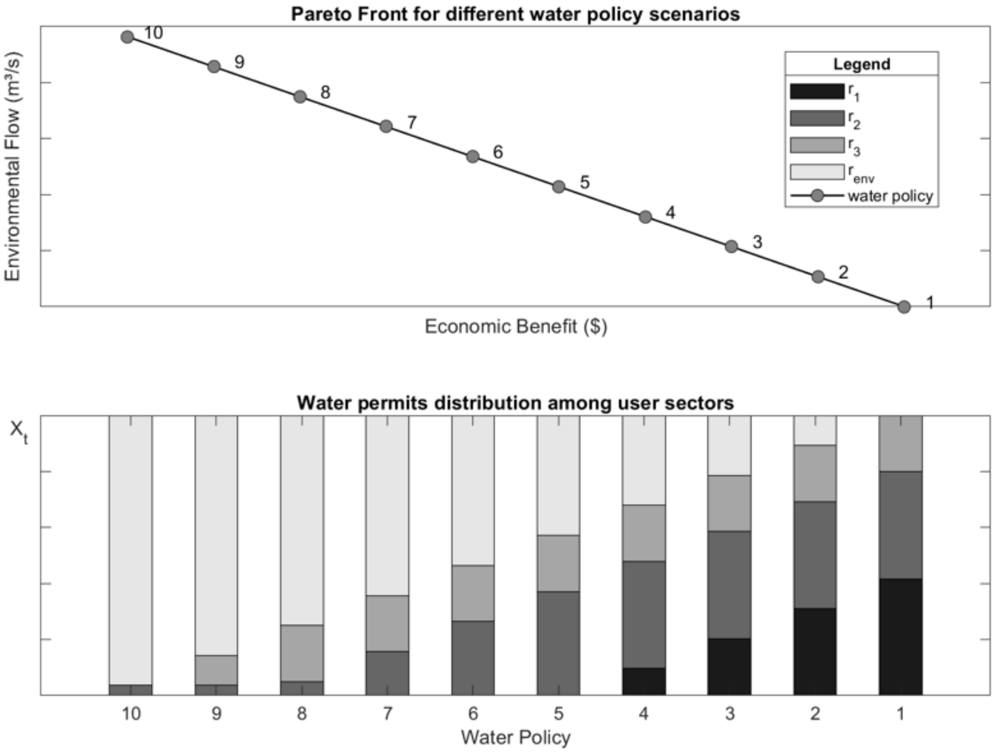

Figure 4: Representation of the Pareto front and user's combination produced by MOLP.

Given this is a multi-objective problem, a sequence of points defining a trajectory of water policies through the stages must be selected beforehand, so it can guide the dynamic programming model, allowing different water policies trajectories to be tested and evaluated. While the sheer number of combinations of different trajectories may indicate this could be an overly laborious approach, the main objective here is not to optimize the water policy itself. Rather, we want to propose a method to integrate an existing proposed water policy with other water management instruments (i.e. water permits and water quality standards) to allow users perceive the associated trade-offs and negotiate adjustments to the policy at hand. One example is to define the existing policy as close as possible to the current water allocation in the watershed, define a final "target" water policy at the end of the planning horizon and then test a few policy trajectories along the planning horizon to reach that target.

Modeling sub-routine III: Spatial water allocation strategy (Where?)

After identifying the incremental flow allocation to users *i* from the pareto frontier produced in the modeling sub-routine II, VISTA answers the question of where the water permits should be issued (and thus water allocated) across the watershed. This defines the third problem: the distribution of the incremental flow allocation to different users r_i, spatially discretized by the decision variable s_i across the watershed (18).

$$r_{i,t} = \sum_{n=1}^{N} s_{i,n,t} \qquad \forall\, i,t \qquad (18)$$

At this point, local water conditions become relevant, as water withdrawals and wastewater discharge resulting from flow allocation will affect river quality. Furthermore, different river reaches may be subject to different water quality standards (or water quality targets defined in the water resources plans).

Water quality standards are defined according to the designated uses of the water bodies (i.e. public drinking water supply, recreation, protection, among others) and water quality criteria to support and maintain the designated uses (i.e. maximum level for the concentration of quality parameters such as nitrogen, phosphorus, BOD, thermotolerant coliforms, among others). In order to establish priority constituents in reducing pollution level and protect the designated use in a specific water body in the watershed, water quality targets are set as a regulatory instrument.

Usually, a water quality simulation analysis is performed in order to check the requirements to achieve or maintain the criteria of the proposed designated use and set the water quality targets. However, this analysis is commonly based on a given reference flow, not considering the consequences of long-term future water allocations. Therefore, the definition of a water quality target (which is a water management instrument) lack a connection with the dynamic water permitting process that allocates water in the system (which is another water management instrument).

Thus, it is necessary not only to track the responses of a given incremental flow allocation solution in the water quality of a given river reach, but also to integrate into this solution existing water quality targets. The spatial allocation sub-routine fulfills this role, using an optimization/simulation water quality algorithm based on a non-linear programming approach. It searches for least cost spatial water allocation within the watershed through calculation of the effects of withdrawals and discharges on the river quality, along with wastewater treatment costs.

Finally, in watersheds with critical water pollution problems, long-term intermediate quality standards are usually defined as it allows to plan intermediary actions and provide incremental investments before achieving final targets. Hence, intermediate water quality targets can be defined for each stage along the planning horizon in the model.

For water quality simulation, the spatial allocation sub-routine incorporates contaminant transport and depuration equations, assuming some simplifications, such as: steady-state, exclusion of evaporation mechanism, constant temperature, and exclusion of the diffusion/aspersion term of the transport equation. Another simplification to the spatial allocation sub-routine is the consideration of the impacts of all withdrawals and discharges in the main river channel only (tributaries are not modeled). This limitation does not offer a significant impact to the results as the most relevant water users are located along the main river.

The main river is thus divided into reaches or control volumes n, sharing similar hydro and geology features, and with specific flow availability and quality constraints (Figure 5). User's withdrawals and discharges are represented as lateral contributions in the transport equation, and all loads W located within each control volume are summed and characterized as a single punctual discharge at the end of each volume control. The resulting mass balance for a control volume n is represented by equation (19).

$$Q_n . C_n = Q_{n-1} . C_{n-1} + W_{in,n} - W_{out,n} - k_n . C_n . V_n \tag{19}$$

Where:

Q_n is the volumetric flow at the end of reach n [L^3 T^{-1}]

C_n is the parameter concentration at the end of reach n [M L^{-3}]

W_{in} is the inflow mass loading at reach n [M T^{-1}]

W_{out} is the outflow mass loading at reach n [M T^{-1}]

k_n is the parameter decay rate at reach n [T^{-1}]

V_n is the control volume n [L^{-3}]

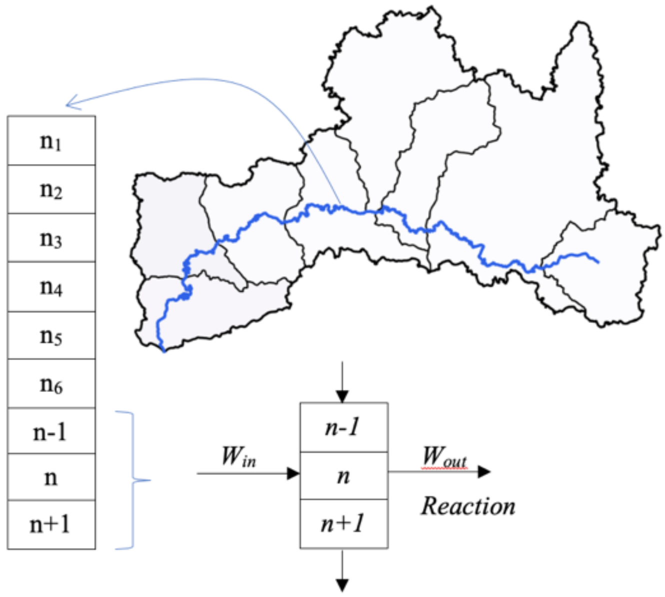

Figure 5: Division of the main river into control volumes *n* and mass balance representation for a control volume *n*.

In order to test the suitability of the proposed water quality simulation model, it was applied in the Sinos River, main river of the study watershed, in the reaches which water quality, withdrawals and wastewater discharge monitoring data is publicly available (108 km length from the total 198 km - see chapter 4 for further details). Water quality was simulated thought incorporation of withdrawals and sewage discharges from current urban users of the watershed. The water quality parameters evaluated were Biochemical Oxygen Demand (BOD) and Thermotolerant Coliforms, since they are representative of domestic sewage pollution. The results were compared to the ones obtained by modeling the same river reaches and conditions using the hydrodynamic modeling software HEC-RAS 5.05 (USACE, 2016), previously calibrated with available topobatimetry data, satellite images and water quality data from Sinos River monitoring network.

Figure 6 shows the results comparing the simplified model with the same scenario modeled by HEC-RAS. As expected, the simplified model is not able to represent all water quality variations along the river, since it calculates the concentration just at the end of each volume control and the river is discretized in just a few reaches. Even though, the simplified model was able to follow the general concentration behavior.

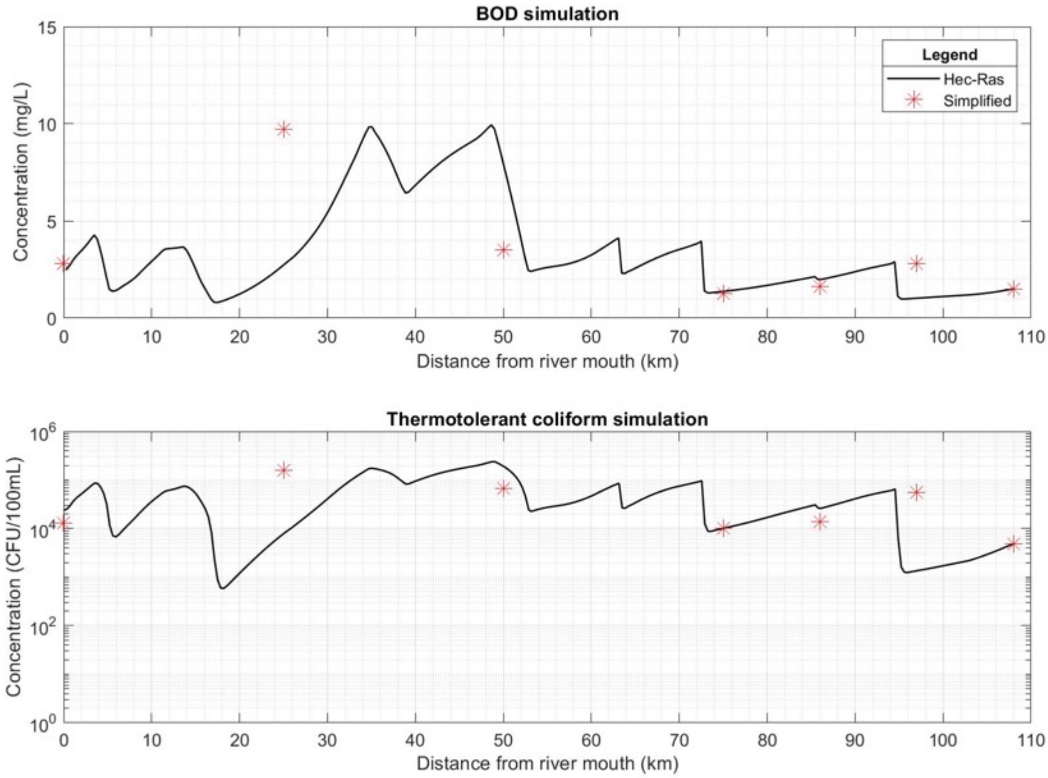

Figure 6: Water quality simulation results for BOD and thermotolerant coliforms.

To model the inclusion of new water users, or the increase in existing uses in the system (i.e. issuing new water permits in a given river reach) the mass transport equation is integrated to a non-linear optimization model. Thus, the water withdrawal at stage *t* from user *i* located at reach *n* (represented as the spatial water allocation $s_{i,n,t}$) is set as a decision variable in the mass transport equation. Hence, the return flow from this user $ret_{i,n,t}$ represents the aggregation of non-point source pollution in the drainage area (when considering agricultural areas) and point source wastewater discharges (when considering industrial and urban users, as example).

Representing the total load to be removed in order to meet the water quality targets at each stage *t* and reach *n*, a wastewater treatment flow $Q_{ww_{n,t}}$ is set as an additional decision variable in the model. The product of the wastewater treatment flow $Q_{ww_{n,t}}$ by a concentration removal efficiency of the parameter *P* modeled results in the total load removed from the system through wastewater treatment techniques (24). The wastewater treatment method reflects on the removal efficiency of parameter *P* (i.e. primary, secondary, tertiary or advanced treatment).

While non-point source pollution generated by agriculture can be partially removed through soil management practices (e.g. erosion control), and the costs of those practices could be associated to a removal efficiency and added to the model, we currently lack reliable data to include it in the model, thus it is limited to point source pollution removal through wastewater treatment.

To find the wastewater treatment cost resulting from the modeling load removal, the wastewater flow $Q_{ww_{n,t}}$ is then related to a cost value function $Cost(Q_{ww_{n,t}})$. As objective function (OF3), the model seeks to minimize the total cost *TCost* (21).

The modeling sub-routine III is thus defined by equations (20) through (31).

Objective Function:

$$\text{OF}_3\text{: minimize } [TCost] \tag{20}$$

Subject to

$$TCost = \sum_{n=1}^{N} Cost(Q_{ww_{n,t}}) \qquad \forall\, n, t \tag{21}$$

Where:

$Cost(Q_{ww_{n,t}})$ is the cost resulting from wastewater treatment at reach *n* and at stage *t* [$]

$TCost$ is the total cost resulting from wastewater treatment in the watershed [$]

Mass transport equations:

$$Q_{n,t}.C_{n,t} = Q_{n-1,t}.C_{n-1,t}.F + W_{n,t} + W_{natural_n} \qquad \forall\, n, t \tag{22}$$

$$W_{natural_n} = (Q_{p_n} - Q_{p_{n-1}}).C_{natural_n} \qquad \forall\, n \tag{23}$$

$$W_{ad_{n,t}} = \sum_{i=1}^{I} ret_{i,n,t}\,.\,s_{i,n,t}.D_i.F - \sum_{i=1}^{I} s_{i,n,t}\,.\,C_{n-1,t}.F - Q_{ww_{n,t}}.(D_{NT} - D_T).F \qquad \forall\, i, n, t \tag{24}$$

$$W_{n,t} = W_{n,t-1} + W_{ad_{n,t}} \qquad \forall\, n, t \tag{25}$$

$$F = e^{-k.t_{travel_n}.LP} \qquad \forall\, n \tag{26}$$

Where:

Index *n* represents the river reaches division (n = 1, 2, …, n, N)

$Q_{n,t}$ is the resultant flow at reach *n* and at stage *t* [M^3T^{-1}]

$C_{n,t}$ is the water quality parameter concentration at reach *n* and at stage *t* [$M\ L^{-3}$]

$W_{n,t}$ is the total load of parameter *P* resulting from user's discharges at reach *n* and at stage *t* [$M\ T^{-1}$]

$W_{natural_n}$ is the natural river load of parameter *P* at reach *n* [M T^{-1}]
$W_{ad_{n,t}}$ is the additional load of parameter *P* resulting from new user's discharges at reach *n* and at stage *t* [M T^{-1}]
Q_{p_n} is the flow with *p* probability of exceedance of reach *n* [M^3T^{-1}]
$C_{natural_n}$ is the natural river concentration of a parameter *P* at reach *n* [M L^{-3}]
$ret_{i,n,t}$ is the fraction of the flow withdrawal from user *i* located at reach *n*, that returns during stage *t*
$s_{i,n,t}$ is the flow withdrawal from user *i* located at reach *n* and at stage *t* [M^3T^{-1}]
D_i is the concentration of parameter *P* resulting from the discharge of user *i* [M L^{-3}]
D_{NT} is the concentration of parameter *P* resulting from urban user discharge when not treated [M L^{-3}]
D_T is the concentration of parameter *P* resulting from urban user discharge after wastewater treatment [M L^{-3}]
F is the depuration factor
k is the parameter decay rate [T^{-1}]
t_{travel_n} is the travel time of reach *n* [T]
LP is the decimal fraction position of the discharge within the reach *n* length*
** We assumed that all user's discharges are located 0.2 from the end of each reach n (equation* 24*). However, to calculate the depuration resulting from the mass transport from reach (n-1) to reach (n) (equation* 22*), the LP equals 1.*

Other constraints:

The spatial allocation is constrained at each reach *n* by: (a) an upper bound on the water permit allocation for each user sector (27); (b) water quality requirements, here named as water quality targets (28); (c) water mass balance at the reach (29 and 30); and (d) user's water quality requirements (31).

$$r_{i,n,t} \leq 0.5 . r_{i,t} \qquad \forall\, i,t \qquad (27)$$

$$C_{n,t} \leq \; C_{target_{n,t}} \qquad \forall\, n,t \qquad (28)$$

$$Q_{residual_{n,t}} = \; Q_{residual_{n,t-1}} - \sum_{i=1}^{I} r_{i,n,t} \qquad \forall\, i,n,t \qquad (29)$$

$$Q_{residual_{n,t}} \geq 0 \qquad \forall\, n,t \qquad (30)$$

$$r_{i,n,t} = 0 \;\because\; C_{n,t} \;\geq\; C_{requirement_i} \qquad \forall\, i,n,t \qquad (31)$$

Where:

$C_{target_{n,t}}$ is the target concentration of the water quality parameter P [M L^{-3}]

$Q_{residual_{n,t}}$ is the residual flow available to be allocated [$M^{-3}T$]

$C_{requirement_i}$ is the user water quality concentration requirement for water quality parameter P [M L^{-3}]

The main purpose of the spatial location module is to find which part of the river supports withdrawals and discharges with the lowest cost. It doesn't mean that it is not possible to allocate in other parts of the river, but, if done so, it will have higher cost and other water quality implications.

Thus, while sub-routine II (water policy and user sector allocation) searches for the non-dominated water allocation policies among different user sectors considering maximization of economic benefits and maximization of water allocation to the environment, sub-routine III (spatial water allocation) searches for the water allocation throughout the watershed that minimizes the treatment costs considering local river flow and quality conditions in different river reaches.

The sub-routine III uses the solution from sub-routine II and refines it spatially. A current limitation is the lack of feedback from sub-routine III to II, so that by maximizing benefit and minimizing cost separately does not warranty we will produce an overall optimal solution.

However, this limitation is attenuated by four aspects. First, the wastewater treatment cost in sub-routine III is only considered for urban use. For Industrial water users, we assumed the water quality regulations are followed to meet water quality thresholds for wastewater discharges, and the related cost is built into the industrial net benefit function of water use in sub-routine II. Second, in the real-world problem, at least in the Brazilian context, the decision to allocate water to urban users does not follow a cost-benefit rationale: cities will be warrantied additional water supplies according to their growth pattern, even under higher costs. In this context, an optimal model solution indicating that it could be economically better to allocate water to another city elsewhere because of higher benefits or lower treatment costs would be of little use. Third, given sub-routine III has the local water quality target as constraint, it imposes the necessary treatment level and costs, which can be passed on to users in their water bill, signaling local scarcity. Finally, all costs and benefits from sub-routines II and III are relayed back to the dynamic programming sub-routine (there is a feedback loop in this part) which adjusts the total allocation x_t if necessary. Despite such aspects, the inclusion of a feedback loop between sub-routines II and III should be subject of future model improvements.

2.3 Application

To test the model to integrate the water management instruments of water permitting and water quality targets along with a prescribed water policy and a broader development policy, an area of study was selected. This selection was based on the following criteria: (a) data availability, (b) relevance of water quality issues and (c) existence of competing water demands by different sectors for water permits.

2.3.1 Study area

The study area is the Sinos River basin, located in the northeast region of state of Rio Grande do Sul, Southern Brazil (Figure 7). The watershed total area is 3,696 km^2, which includes, totally or partially, the territory of 32 municipalities with an estimated population of 1,350,000. Although occupying just 1.3% of the total state territory, the Sinos River Basin plays a major role in the economy and development of the region, being responsible for generating approximately 21% of the State Gross Domestic Product (GDP) (Sinos, 2018). The main water demand activities are associated to urban supply (35%), industrial (11%), and rice irrigation practices (53%) (Profill, 2013). The strong and fast urban and industrial development during the last decades was not followed by necessary investment in pollution control, resulting in critical pollution issues, and positioning the Sinos River as the fourth most polluted of Brazil (IBGE, 2010).

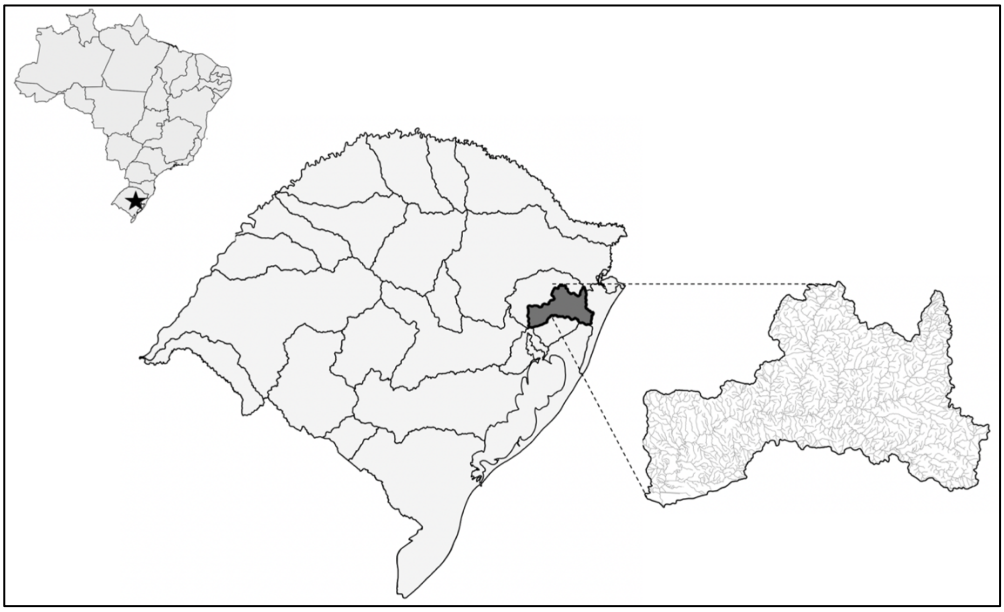

Figure 7: Sinos River Basin.

2.3.2 Input data and modeling assumptions

Several assumptions are necessary to avoid unnecessary modeling complexity and to overcome limitations on data availability. While such assumptions impose limitations to the accuracy of the results presented here, we highlight the main purpose of this work is to demonstrate the modeling concept adopted to integrate the different water management instruments.

Planning Horizon

The planning horizon adopted in this study considers a long-term period of twenty years discretized in four stages of five years each (2014-2034). This period is the same adopted by the water resource plan of the Sinos River Basin to propose programs and actions, which provides the same base to compare and discuss different alternatives (Profill, 2014a).

Total Allocable Flow

The study adopted an allocable flow following the current water permitting criteria, which allows up to 70% of Q_{90} (14.04 m^3/s) to be allocated for consumptive users (Profill, 2014a). While VISTA could use any value as maximum allocable flow, adopting the current criteria as boundary condition provides results that are closer to the present conditions, and thus easier to communicate and discuss with water users and managers. It should also allow a smoother transition to different water allocation solutions in the watershed. This study just considers allocation and use of superficial water, since it corresponds for approximately 97% of total water supplied to demands in the watershed (Sinos, 2010).

Water Policy Trajectories

The water policies vary from strictly economic (1) to strictly environmental (10). Given the large number of combinations and possible water possible trajectories, we have selected a group that represents possible major trends trajectories: from constant water policy trajectories along the planning horizon (i.e. 1-1-1-1 and 9-9-9-9) to trajectories shifting from economic to more environmental policies (i.e. 1-1-1-9; 1-1-9-9; 1-3-6-9; and 1-9-9-9). The main objective here is not to optimize the water policy itself. Rather, we want to propose a method to integrate an existing proposed water policy with other water management instruments.

Water Quality classification and water quality target standards

Water bodies are classified according to a Brazilian federal law, based on the expected uses of their water (Brasil, 2005). There are separate classes and each class has its own upper bounds for concentration of a list of different constituents. The first class is termed “special” and it has the strictest limits, meaning that while the quality is the highest among the other classes, the land use in the watershed draining to this water body is also the most limited. This class is followed by other four, ranging from 1 to 4 (the higher the number, the higher the concentration levels allowed). Class 4 has the highest limits for concentration of constituents and a water body in this class has very few uses aside from navigation. There is no class above 4, so that this class presents a wide range of concentrations from near class 4 limit to high level values. By law, urban water supply systems can only withdraw from water bodies below class 4. One of the activities during the preparation of a water resources plan in a given watershed is to determine the current water quality class for its water bodies (classification procedure) and the future expectations of the water users towards this water quality (target water quality setting procedure). Both the classification procedure and the target water quality setting procedure may result in different classes depending on the river reach or tributary. For example, a river crossing a densely occupied metropolitan area may be currently adhering to class 4 limits (and classified as class 4), but the population may want to see it improved in the future, and hence define as a target to reach class 3 or 2, within a given planning horizon (intermediate targets can also be set). In this example however, it might be unreasonable to set a future target class to 1 or "special" given it would require a severe change in the pollution control and licensing, and most likely it would never happen. In any case, there is always a trade-off when a higher water quality class is set for the future during the classification procedure, and knowing such tradeoff is a key element in the planning process, given the classification procedure involves participation and negotiation between users in the watershed committee. Finally, there can be no retreat to a lower water quality class. Once a given target is approved by the watershed committee, future revisions on the water resources plans can set targets that are either equal or of higher quality.

The classification procedure used by the Water Resources Plan of Sinos River Basin to classify the main water bodies in the watershed took into account five water quality parameters: Biochemical oxygen demand (BOD), dissolved oxygen (DO), thermotolerant coliforms, phosphorous, and nitrogen. The main course of the Sinos River is current classified as class 4 in its major length mainly due to phosphorous and thermotolerant coliforms concentrations exceeding class 4 thresholds.

Untreated domestic sewage discharges are a main pollution concern in the Sinos River Basin, since only 4.5% of its total population has sewage collection and treatment (Concremat, 2014). Based on this specificity, the water resource plan chose BOD, DO and thermotolerant coliforms as the parameters to set water quality targets, mainly to represent domestic pollution and also for requiring wastewater treatment solutions that are less expensive than if it were focusing on parameters removal requiring advanced and more costly treatment levels (i.e. phosphorous). Thus, this study adopted thermotolerant coliforms as the parameter to be used in the water quality module (sub-routine III), as its concentration in the study watershed is more critical than BOD.

We adopted a removal efficiency compatible with secondary treatment level (for biodegradable organic matter removal) followed by a disinfection process (for pathogens removal), which is the most applied wastewater treatment method by utility companies in Brazil. For thermotolerant coliforms, we assumed a removal efficiency of 99%, which considering an untreated concentration of 1.10^7 FCU/100mL, results in a post treatment concentration of 1.10^5 FCU/100mL (values attending references ranges as Metcalf and Eddy (2003). The main river was divided in 9 reaches, according to hydro and geology features of the study watershed (see chapter 4 for further details).

Table 1 presents the main variables used by VISTA for water quality simulation and optimization (sub-routine III), detailing both the current classification (column 2) and the future target class (column 3).

Table 1: variables used by VISTA in the water quality simulation and optimization module (sub-routine III).

Main river division	River reach	Water quality classification[a]	Water quality targets[a]	Reach length delta (km)	Travel time[b] (t_{travel}) (d)	Flow availability (Q_{90}) (m^3/s)
upper	9	2	1	198 - 168	0.77	1.80
	8	2	2	168 - 138	1.25	7.76
	7	4	2	138- 108	0.75	8.50
medium	6	4	3	108 - 97	0.30	12.82
	5	4	3	97 - 86	0.44	13.81
	4	4	3	86 - 75	0.33	15.02
lower	3	4	3	75 - 50	1.25	16.86
	2	4	3	50 - 25	1.85	18.48
	1	4	3	25 - 00	5.36	20.05

[a]Thermotolerant Coliforms thresholds for each water class (CFU/100mL):
class 1 ≤ 200; class 2 ≤1000; class 3 ≤ 4000; class 4 > 4000 (Brasil, 2005)
[b]For reaches 1 to 6 (lower and medium division) t_{travel} was obtained thought HEC-RAS modeling (where topobatimetry data was available) and for reaches 7 to 9 (upper division), t_{travel} was obtained applying manning equation using, length and elevation variables from SRTM digital elevation data (see chapter 4 for further details).

Water User Sectors

The user's sector definition took into consideration the main water uses of the watershed, which are: irrigated agriculture r_1, urban r_2, and industrial r_3 as economic users. The environmental use is represented by r_{env}.

Development Policy

Development policies affect the region economically, which is likely to affect the activity level growth rate adopted to calculate the projected water demand used by the model. The analysis of a development policy and its relationship with the activity level growth is out of the scope of this study, hence we adopted growth rates based on the past watershed development scenario (as described in the projected water demand section). However, the model and its application can take advantage of other studies exploring scenarios of activity level growth resulting from different development policies, incorporating this information into the integrated analysis of water management instruments proposed here.

User's projected water demand and economic demand functions

The water demanded by the economic users is a combination of their activity level (number of people, area planted, industrial output) and their unit water use rate (consumption per capita, per hectare and per output). The activity level is assumed a given growth trend (resulting from a development policy), and it is based on recent sectoral data. The unit water use rate is also based on reference published values.

The water allocation among economic water users (sub-routine II) is driven by economic value functions. The values were corrected at each stage by an interest rate of 9% per year. The economic benefits are calculated as the area beneath the users' marginal net benefits curves (for irrigated agriculture and industrial) and willingness-to-pay curves (for urban use). We assume these curves represent the users' economic water demands, so that the calculated areas reflect their consumer surplus. There are two major assumptions to treat how users' economic water demands change in time, described as follows.

Urban water demands and use (r_2)

In this paper, urban use of water refers to household (residential or domestic) as the most significant use, plus public and commercial uses. For the urban sector, as the sector grows, new users enter the system willing to pay as much as the existing users, given public water supply is a natural monopoly and users have no other significant alternative. Here, the payment refers to the water tariffs charged by the public utilities. Hence, as the sector grows, there is a change in the economic water demand (the whole curve shifts to the right – Figure 8).

Between 1991 and 2010, the state of Rio Grande do Sul population increased 17%, according to population census data IBGE (2010b). In the same period, Sinos River Basin's population increased 29%, a higher proportion compared to the State population growth, mainly provided by industrial expansion and urbanization. IBGE projections point out a population growth of 5.9% for the planning horizon (2014-2034) in the state of Rio Grande do Sul until it starts its declination by the year 2035. Considering the particular economic and urban characteristics of the watershed, where its economy is strongly driven by industrial activities, we adopted the geometric rate method based on the watershed population to estimate the population growth. The results indicate a population growth of 22% by the year 2034, with an annual average growth of 1.1%. The value is higher than the State projection but can be justified by the following industrial expansion in the area and possible migration from other state regions.

The urban projected water demand along the planning horizon was created multiplying the projected population growth (water activity level) by the reference water use rate of 226 L/day/capita (current average per capita rate presented in the Water Resource Plan diagnosis) (Table 2). The non-potable fraction of the total urban water demand is variable according to regional and local specificities. Lopes; Fernandes; Dornelles (2017) present values raging from 44% to 72% for non-potable use in Brazil, assuming 50% in their study. We adopted a value of 80% as the minimum coverage (lower bound at equation 16) for urban users. Such conservative estimate prevents reallocating too much water away from urban demands, which would be unfeasible in practice. The remaining 20% is actually within the loss rate in the supply network, which means it is a water volume that could be reallocated without affecting the population, Thus, for each stage t, the water distribution is constrained by a minimum guarantee (32).

$$D_{min_{i,t}} \leq 0.8\, D_{pr_{i,t}} \quad \forall\, i = urban, t \tag{32}$$

The willingness-to-pay for water is assumed here to represent the urban sector economic water demand. The point expansion method was used to estimate the willingness-to-pay for water, inferring an empirical demand function from an observed price-quantity point and an assumed price elasticity (Griffin, 2006). We adopted the retail company basic residential rate of 5.21 R\$/m^3 as observed price-quantity point (Corsan, 2018), a correspondent demand of 3.57 m^3/s (current total urban water demand in the watershed (Profill, 2013)), and a price elasticity value of -0.4 (Magalhães et al., 2016). The marginal benefit function was obtained applying the variable elasticity form described in (Griffin, 2006).

In Brazil, water losses reach a percentage of 38%, being divided between physical losses (leakages and losses during withdrawal, treatment, reservation, distribution, etc), and nonphysical losses (pilferage, meter error, lack of measure) (Trata Brasil, 2018). Although nonphysical losses don’t generate revenue to the retail company, users still take advantage from water use benefits. For this reason, we assumed that just physical losses reduce the total economic benefit and must be taken apart from the economic benefit estimation. Considering a percentage of 18% as physical losses, the corrected flow used in the point expansion method that generates economic benefit is 2.93 m^3/s.

The value refers to the willingness-to-pay not only for water itself but also for the services involved (i.e. capturing, transporting, treating, and storing), also called as at-site value (YOUNG AND LOOMIS, 2014). We assumed that urban water is priced to fully recover supply costs so that the average utility revenue can be subtracted from the total willingness-to-pay. The result is the net consumer surplus imputable to raw water or at-source urban water value, and it allows to properly compare urban demand with demand for instream uses or for raw water withdrawn (as in irrigated agriculture or industrial uses) (YOUNG AND LOOMIS, 2014).

Finally, population growth induces natural shifts to the economic water demand curve, as new consumers enter in the system willing to pay the same amount (Griffin, 2006). For this reason, each stage of the model was associated to a different economic water demand curve, by shifting the function from each stage successively outward to follow the new population growth (Figure 8). According to the results, marginal values ranged from 0 R$/m³ to 2.81 R$/m³.

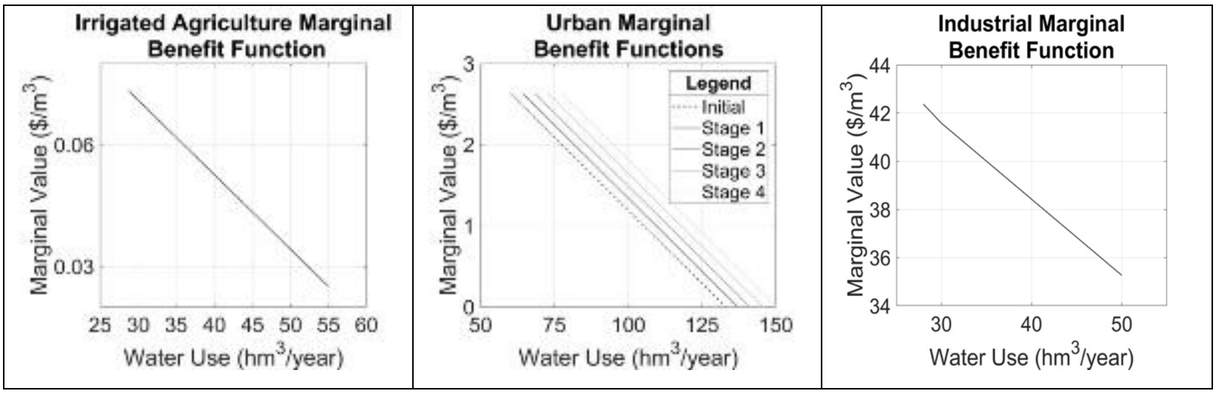

Figure 8: marginal benefit functions per stage for each use sector.

The integration of the economic water demand curves resulted in the benefit value functions for urban water use (at-source) (33 to 37).

$$B(r_{2,t}) = -1.41.10^{-7}.(r_{2,t})^2 + 13.03.(r_{2,t}) \qquad \forall\, t = 0 \tag{33}$$

$$B(r_{2,t}) = -1.41.10^{-7}.(r_{2,t})^2 + 13.97.(r_{2,t}) \qquad \forall\, t = 1 \tag{34}$$

$$B(r_{2,t}) = -1.41.10^{-7}.(r_{2,t})^2 + 14.55.(r_{2,t}) \qquad \forall\, t = 2 \tag{35}$$

$$B(r_{2,t}) = -1.41.10^{-7}.(r_{2,t})^2 + 15.17.(r_{2,t}) \qquad \forall\, t = 3 \tag{36}$$

$$B(r_{2,t}) = -1.41.10^{-7}.(r_{2,t})^2 + 15.84.(r_{2,t}) \qquad \forall\, t = 4 \tag{37}$$

Irrigated agriculture water demands and use (r_1)

For the irrigated agriculture sector, as it grows, new users join the system taking up properties and production conditions that are increasingly less favorable compared to previous users (e.g. farther from the river, less favorable topography and soil conditions. According to IRGA (2017), rice production in the RS state can yield the same output (aprox. 13,000 kg/ha) with different amounts of water applied, depending on the soil conditions (the most favorable soil conditions can result in the same yield with almost half the water) with indicates that diminishing marginal returns is a reasonable assumption as the best suitable areas are occupied first. Thus, we assume new users will face decreasing willingness-to-pay for water, along with marginal benefits. Hence, the marginal net benefit curve is assumed here to represent the agricultural sector economic water demand. As the sector grows, there is a variation in the quantity demanded for water (we move along the economic water demand curve towards the right).

We adopted the historical data of rice crop area from the Rice Institute of Rio Grande do Sul State (IRGA, 2017) in order to estimate the potential rice irrigation area growth in the watershed. The State rice planted area (ha) data was analyzed from 1990 to 2017 and extrapolated for the future planning horizon. The projection was then analyzed regarding the physical conditions of the watershed to sustain such growth according to the potential for expansion of irrigated areas (2030) reported by the Federal Irrigation Atlas (ANA, 2017). The results show a potential growth for the watershed of 25% by the year 2034, with an annual average growth of 1.25%.

The projected water demand for the irrigated agriculture sector along the planning horizon was created multiplying the given crop area expansion estimation (water activity level growth), by the reference water use rate of 8,500 m^3/ha/cropping season, as presented in the Water Resource Plan diagnosis. Final results are presented in (Table 2).

The irrigated agriculture marginal net benefit function used by VISTA was derived from a previous study involving the Santa Maria River Basin, which is located in the center region of the State and it is characterized for rice and soybean cultivation (Mattiuzi, 2018).

In Mattiuzi (2018) the author applied an agricultural production model, adapted from the Statewide Agricultural Production (SWAP) model (Howitt et al., 2012) with the objective to create marginal net benefit curves for different regions inside the watershed (Santa Maria River Basin). The marginal net benefit curves are the result of a combination of several factors, such as water demand by culture, yield, selling price, costs of land, water and labor. In order to generate them, the availability of water on SWAP was varied from 99% to 50% in successive model runs, and after each run the Lagrange multiplier from the water availability constraint was recorded. The Lagrange multiplier indicates the water shadow value. The resulting functions present linear form with lambda values ranging from 0.02 R$/m³ to 0.31 R$/m³. At 99% availability of water, when the users have access to almost all water they need, lambda values ranged from 0.02 R$/m³ to 0.10 R$/m³.

Through the marginal net benefit functions, we calculated the correspondent elasticity value (ε) for each water availability constraint along each curve in Mattiuzi (2018) (38). The elasticity variations along the curves in Mattiuzi (2018) present similar numeric values regardless the marginal net benefit curve, so those were used to compose a new single marginal benefit function specific for Sinos River Basin case, where the projected irrigated agriculture water demand at the final stage (t=4) was considered as maximum water demand for irrigated agriculture purposes in the Sinos River Basin (point where the marginal net benefit is close to zero). In the new marginal benefit function the economic water values range from 0.08 to 0.28 R$/m3, which are close to those obtained for the Santa Maria River Basin. The domain of this function goes from 50% to 99% of the maximum demand, which is the range we assume farmers can adjust production and technology to cope with more, or less, water availability. If farmers have less than 50% of their maximum demand they might do something else entirely (e.g. sell the property) and the proposed function would no longer represent their marginal water value.

$$\varepsilon = \frac{dQ/Q}{dP/P} \tag{38}$$

Where: Q refers to quantity and P refers to price.

The irrigated agriculture marginal net benefit function used by VISTA has a linear form, and the area beneath it results in the total net benefit function in the quadratic form (39).

$$B(r_{1,t}) = -3.49.10^{-9}.(r_{1,t})^2 + 0.479.(r_{1,t}) \tag{39}$$

Industrial water demands and use (r_3)

Like the irrigated agriculture sector, we assumed that new industrial users or expansion of the current ones join the system taking up conditions less favorable compared to previous users, as the best suitable areas for industrial installation are occupied first. Thus, we assume diminishing marginal return to water, and new users will face decreasing willingness-to-pay for water, along with marginal benefits. Hence, as the sector grows, there is a variation in the quantity demanded for water (we move along the economic water demand curve towards the right).

Industrial activities often face significant changes through time, mainly due to market oscillations and technologies progress, which makes it difficult to project long-term activity growth. In addition, the industrial water use depends heavily on the facility type and process. The water demand in the watershed due to industrial activities is mainly composed by chemical (refinery) and metallurgical facilities, followed by food and beverages (Sinos, 2010).

The average annual growth rate of industrial productivity (monetary growth per unit of time) by facility type in the period of 1996-2007 in the State of Rio Grande do Sul (FEE, 2010) indicates that facility types with positive growth during this period (i.e. computer components manufacturing, textile, furniture, vehicles, among others) the average annual growth observed was 5%. However, some facility types had negative growth (i.e. refinery and chemical industries, rubber, etc), which decreased the global average annual growth to 1%. In order to estimate the industrial productivity in the future planning horizon, we considered an intermediary annual growth rate of 2.4%, which leads to a potential industrial expansion in the study watershed of 48% by the year 2034.

Finally, the projected water demand along the planning horizon for the industrial sector was created by applying the percentage estimate of the industrial productivity growth (water activity level growth) to the current industrial water demand in the watershed (Table 2), which leaves the water use rate as an implicit value.

Industrial economic values for water use are likely to present a wide range of variability, as it depends on the particular type and design of the industrial facility (Lund & Redd, 1995). Due to its heterogeneity, and the lack of Brazilian studies assessing the economic value of water for industrial use, we estimated the industrial marginal net benefit function using the residual method described by (Young & Loomis, 2014). Hence, the marginal net benefit curve is assumed here to represent the industrial sector economic water demand. The method was, at first, applied to a particular type of industrial facility, and then extrapolated to the watershed data. The facility chosen for this study is a metallurgical type, since it makes part of the watershed industries mix and it was the only type in the watershed for which more detailed data was available.

For the particular facility case, the water value was isolated through the exhaustion theorem, which imposes that the amount of inputs of the company weighted by their value marginal products VMPs sums to the total value products TVP (Young & Loomis, 2014). Then, the total water amount consumed by the company (water availability) was restricted in small variations from 99% to 50% (for the same reasons as explained for the irrigated agriculture sector). The outcomes of each calculation of the company outputs and revenue were analyzed and recorded in order to find the consequent shadow value associated. According to the results, the marginal values for industrial water use varied from 128 to 161 R$/m^3. These marginal values are higher than the ones found for urban and irrigated agriculture uses, which is in line with what has been reported in the literature. CALVIN model (Jenkins, 2000) used marginal benefit curves with values ranging from 108 to 136 R$/m^3 obtained from penalty functions for the industrial sector.

Given the economic values for the water are provided by a single type of industry, and a particularly valuable one, its extrapolation to the whole watershed could lead to an overestimation of the water value to the industry sector and affect the water allocation solutions. To mitigate this issue and also to perform a sensitivity analysis, the water values were reduced to 25% of the total marginal value found, which is in the same order of magnitude of the contribution of this type of industry in the watershed GDP.

The marginal values when related to their correspondent water availability generated a marginal net benefit function in the linear form. Hence, the area beneath the curve resulted in the total net benefit function (40) for the integral values and (41) for the reduced values (sensitivity analysis).

$$B(r_{3,t}) = -6.15.10^{-7}.(r_{3,t})^2 + 195.(r_{3,t}) \tag{40}$$

$$B(r_{3,t}) = -1.23.10^{-7}.(r_{3,t})^2 + 39.(r_{3,t}) \tag{41}$$

Table 2: Projected water demand values.

Projected water demand (m^3/s)				
Stage	**Irrigated agriculture** r_1	**Urban** r_2	**Industrial** r_3	**Total**
Initial	5.3	3.6	1.2	10.1
1	5.6	3.8	1.3	10.8
2	6.0	4.0	1.4	11.4
3	6.3	4.2	1.6	12.0
4	6.7	4.4	1.7	12.7
Total growth	25%	22%	48%	26%

Wastewater Treatment cost function

The economic benefit function for urban water does not incorporate the costs related to the necessity of urban wastewater discharges treatment. This cost is handled separately in the modeling sub-routine III (Spatial water allocation strategy) with the objective of discounting the treatment costs from the benefit of urban water use. We adopted the basic residential rate of the retail company to collect and treat municipal wastewater (6.24 R$/m³) to compose the wastewater treatment cost function (Corsan, 2018) (equation 42).

The tariff value includes operational costs and long-term investment recoveries, which is compatible with the long-term planning horizon used in the modeling scenarios. Given most part of the retailer's companies adopts wastewater treatment plants with secondary treatment level, the wastewater treatment cost is also compatible with the removal efficiency adopted in the water quality modeling (sub-routine III).

$$Cost(Q_{WW_{n,t}}) = 6.24.(Q_{ww_{\mathrm{n,t}}}) \qquad \forall\, n, t \tag{42}$$

2.3.3 Modeling scenarios

The modeling scenarios include two time periods with a planning horizon of 20 years each: Past (1994-2014) and Future (2014-2034). Each modeling scenario must have a water policy for each stage t (composing a trajectory of water policies along the planning horizon) as well as a target for water quality, which imposes a constraint to the concentration of thermotolerant coliforms at each river reach n and each stage t. Table 3 summarizes the main variable data among the modeling scenarios.

We used the past scenario as a verification process in order to assess VISTA results consistency in regard with the past period observed water allocation trajectory (hindsight). This process was called model verification and it is bounded by a trajectory of water policies that has a strong preference for water allocation to economic uses, in detriment to environmental demands. This trajectory was chosen given it better reflects the past 20 years of occupation in the watershed, in which the water permits were issued to fulfill the demands (regardless of the economic benefits and broader environmental impacts), the cost of the urban wastewater infrastructure is not considered when deciding on water permits (in fact most cities discharge untreated sewage in the river without any penalty), and there were no water quality targets (the first water quality targets proposal was stablished in 2003, but it wasn't formalized and implemented).

Due to the lack of past data regarding water permits, we considered that the current observed water allocation attends its full water demands. Thus, to estimate the system water demand curve for the past scenario (red line in Figure 3), we adopted the current observed water allocation as the water demand at the final stage (t=4) and we estimated the water demand for the previous stages based on the observed water activity rates data, as described in section 2.3.2. This modeling scenario also had as targets for water quality the observed concentrations of thermotolerant coliforms at the final stage.

For the Future period, four main categories of scenarios were created, each one with a prescribed water policy trajectory and water quality targets:

Group (1): Scenario A follows a strictly economic water policy trajectory for water permits allocation, constrained to meet the water quality targets proposed by the Water Resources Plan;

Group (2): Scenarios B, C, D, and F follow diverse water policy trajectories changes along the planning horizon from more economic to more environmental choices, constrained to meet the same quality standards proposed by the Water Resources Plan;

Group (3): Scenario E follows an environmental water policy trajectory for water permits allocation, constrained to meet the water quality targets proposed by the Water Resources Plan, and

Group (4): Scenarios As and Es present a sensitivity analysis using a reduced industrial economic benefit function for a strictly economic water policy trajectory (Scenario As) and a strictly environmental water policy trajectory (Scenario Es).

Finally, an additional scenario where the model was forced to fulfill all demands regardless of economic benefits and costs is included (Nonintegrated scenario). This would be close to a fully nonintegrated application of the water management instruments (water permits, water quality targets and water policies defined in the water resources plan) simulating the current approach. The nonintegrated scenario has the same water quality targets as the remaining scenarios, but the allocation of water permits is done upon request and there is no water policy trajectory defined.

Table 3: Modeling scenarios.

Scenario	Description	Planning Period	Development Policy	Water Policy Trajectory[a]	Minimum Quality Standard Target[b]
Verification	Verification	Past	Observed water demand growth	1-1-1-1	Observed water quality data
Nonintegrated	Nonintegrated	Future	Projected water demand	none	Reach 1: c1-c2-c2 Reach 2: c3-c3-c3 Reach 3: c3-c3-c3
A	Strictly Economic	Future	Projected water demand	1-1-1-1	Reach 1: c1-c2-c2 Reach 2: c3-c3-c3 Reach 3: c3-c3-c3
B	Late Change	Future	Projected water demand	1-1-1-9	Reach 1: c1-c2-c2 Reach 2: c3-c3-c3 Reach 3: c3-c3-c3
C	Mid-Change	Future	Projected water demand	1-1-9-9	Reach 1: c1-c2-c2 Reach 2: c3-c3-c3 Reach 3: c3-c3-c3
D	Early Change	Future	Projected water demand	1-9-9-9	Reach 1: c1-c2-c2 Reach 2: c3-c3-c3 Reach 3: c3-c3-c3
E	Strictly Environmental	Future	Projected water demand	9-9-9-9	Reach 1: c1-c2-c2 Reach 2: c3-c3-c3 Reach 3: c3-c3-c3
F	Progressive Change	Future	Projected water demand	1-3-6-9	Reach 1: c1-c2-c2 Reach 2: c3-c3-c3 Reach 3: c3-c3-c3
As	Sensitivity Analysis	Future	Projected water demand	1-1-1-1	Reach 1: c1-c2-c2 Reach 2: c3-c3-c3 Reach 3: c3-c3-c3
Es	Sensitivity Analysis	Future	Projected water demand	9-9-9-9	Reach 1: c1-c2-c2 Reach 2: c3-c3-c3 Reach 3: c3-c3-c3

[a]1 is the highest preference to economic and 9 is the highest preference to environmental
[b] c1: class 1; c2: class 2; c3: class 3; c4: class 4 (CONAMA 357/2005)

2.4 Results

2.4.1 Model verification

Results obtained from the verification analysis show that, adopting an economic water policy trajectory (1-1-1-1) for the past scenario, the water allocation increments though time (stages) followed very close the past projected water demand curve and had a similar water distribution among users (Figure 9). The urban and industrial sectors had their full water demands covered, while the irrigated agriculture sector faced a reduction, resulting in a final 94% demand coverage. The lack of proper wastewater treatment investments and water quality targets in the past period allowed temporal and spatial water allocation by the VISTA mostly to fulfill demands, which explains the proximity between the modeling results and the observed data, as expected.

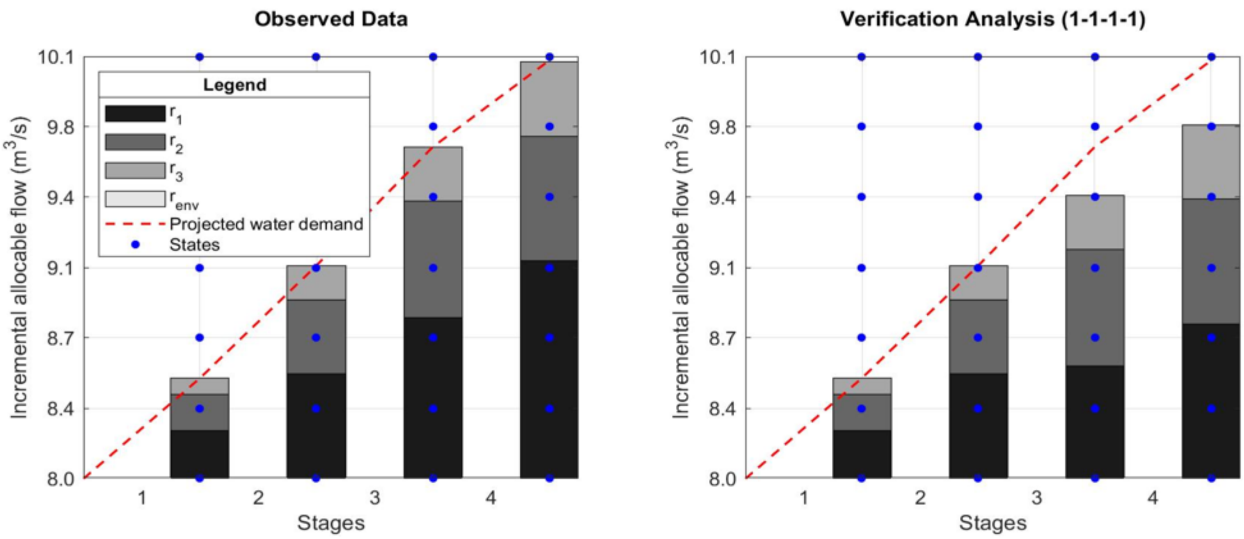

Figure 9: Verification modeling results (past scenario).

Although the absence of water quality targets, the past investments in wastewater treatment infrastructure (reaching only 4.5% of total population in the watershed) provided a small water quality improvement. To achieve this small improvement in the modeling verification scenario (constrained in the model by the observed water quality concentrations set as water quality targets), the model preferably reduced water demand coverage by 6% for the irrigated agriculture sector, rather than increasing wastewater treatment investments. Thus, the demand coverage reduction indicates that some level of water scarcity is economically optimal. In our work, this means that if water is delivered beyond this point, it will bring wastewater treatment costs higher than the economic benefits from using the water. Besides its intangible benefits, the flow allocated to the environment has clear economic advantages.

Regarding water quality results, measured by thermotolerant coliforms parameter, the model distributed the users along the river reaches resulting in the same current observed concentrations at the final stage. However, due to the optimization nature of the approach, which searches for the least cost spatial water allocation across the watershed, the modeling user's spatial distribution along the river reaches presented a small variation comparing to the observed data (Figure 10).

The spatial allocation variation led by lower costs optimization together with the water demand coverage reduction (for the irrigated agriculture sector) also resulted in a lower wastewater treatment flow necessary to meet the final thermotolerant coliforms concentration. The total wastewater treatment flow obtained (0.1 m^3/s) corresponds to 3.4% of the total urban water use at the final stage. The percentage obtained is close to the current 4.5% wastewater treatment coverage observed (0.13 m^3/s), which indicates the results are consistent with observations and that the model could be applied for future scenarios.

Finally, the verification process results suggest that the past blind decisions regarding water allocation and wastewater treatment investments weren't economically optimal. As the results indicate that some level of water scarcity is economically optimal, if more restricted water quality targets were imposed in the past period, different levels of water scarcity would be found.

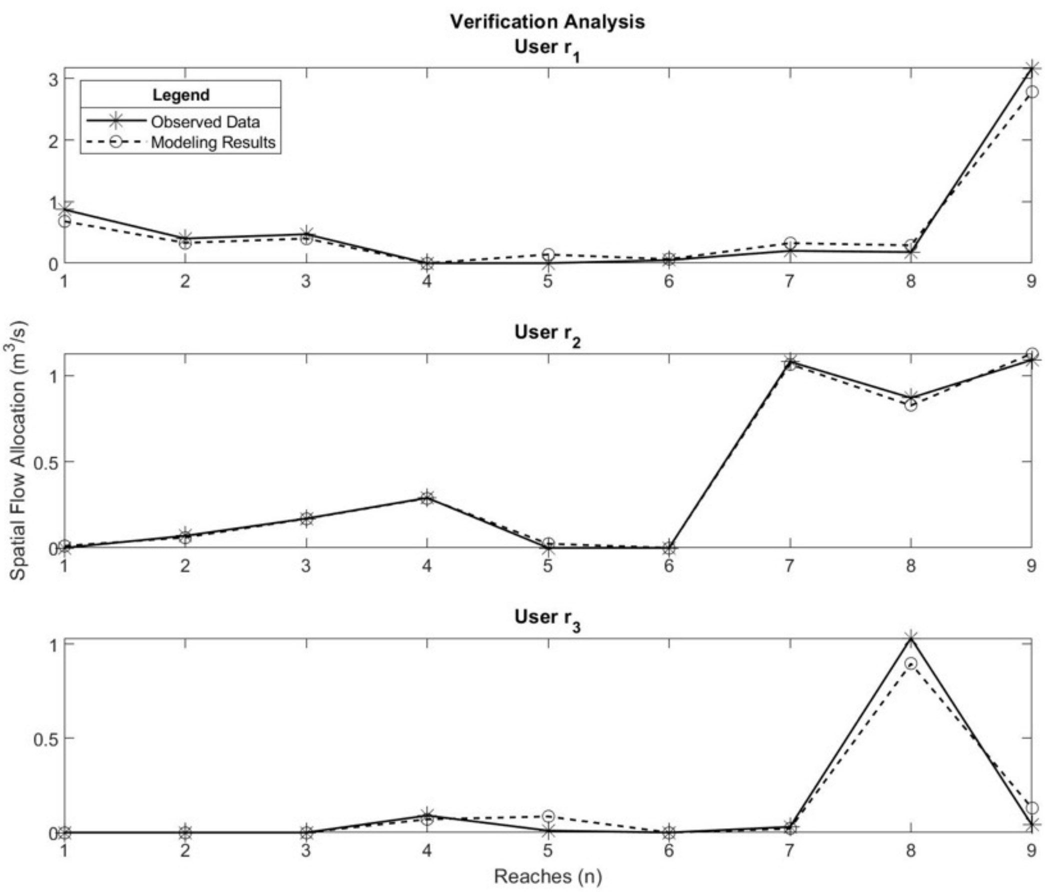

Figure 10: Accumulated user's spatial allocation at final stage (t = 4) for the model verification.

2.4.2 Future scenarios: temporal and economic results

Table 4 summarizes the main modeling results and Table 5 explores the economic trade-offs among the different modeling scenarios for the future planning horizon.

Table 4: Trade-off analysis among scenarios for future planning horizon.

Scenario	Total allocable flow to (r_{env})	Total allocable flow to (r_i)	Total wastewater treatment flow	Total wastewater treatment cost increment[a] (A)	Total economic benefit increment[a] (B)	Total resulting benefit (B-A)	Projected water demand coverage r_1	r_2	r_3
	m³/s	m³/s	m³/s	R$ million	R$ million	R$ million	%	%	%
Non integrated[b]	0	2.60	2.53	970	4,666	3,696	100	100	100
A (1-1-1-1)	0	1.40	2.52	959	4,664	3,705	82	100	100
B (1-1-1-9)	1.27	1.53	2.40	939	4,663	3,724	86	96	100
C (1-1-9-9)	2.5	0.99	2.30	909	4,649	3,740	81	92	100
D (1-9-9-9)	3.12	0.72	2.17	856	4,528	3,672	80	88	99
E (9-9-9-9)	3.45	0.40	2.01	759	3,969	3,210	80	82	94
F (1-3-6-9)	1.52	0.93	2.29	905	4,658	3,753	80	92	100
As (1-1-1-1)	0	1.40	2.52	959	1,596	637	82	100	100
Es (9-9-9-9)	3.45	0.40	2.01	759	1,390	631	80	82	94

[a]accumulated at final stage t = 4 (20° year)
[b]data simulating the total system projected water demand full covered.

Table 5: Economic trade-off analysis comparing to the nonintegrated scenario.

Scenarios[a]	Economic benefit change	Wastewater treatment cost reduction	Net economic gain	Water scarcity r1	r2	r3
	R$ million	R$ million	R$ million	%	%	%
A (1-1-1-1)	-3	10	8	18	0	0
B (1-1-1-9)	-3	31	27	14	4	0
C (1-1-9-9)	-17	60	43	19	8	0
D (1-9-9-9)	-138	113	-25	20	12	1
E (9-9-9-9)	-697	211	-486	20	18	6
F (1-3-6-9)	-9	65	57	20	8	0
As (1-1-1-1)	-3	10	8	18	0	0
Bs (9-9-9-9)	-212	211	-1	20	18	6

[a] Comparison between each integrated scenario and nonintegrated scenario.

Figure 12 demonstrates the incremental flow trajectory along the planning horizon for each modeling scenario together with the distribution of each increment among the user's sectors.

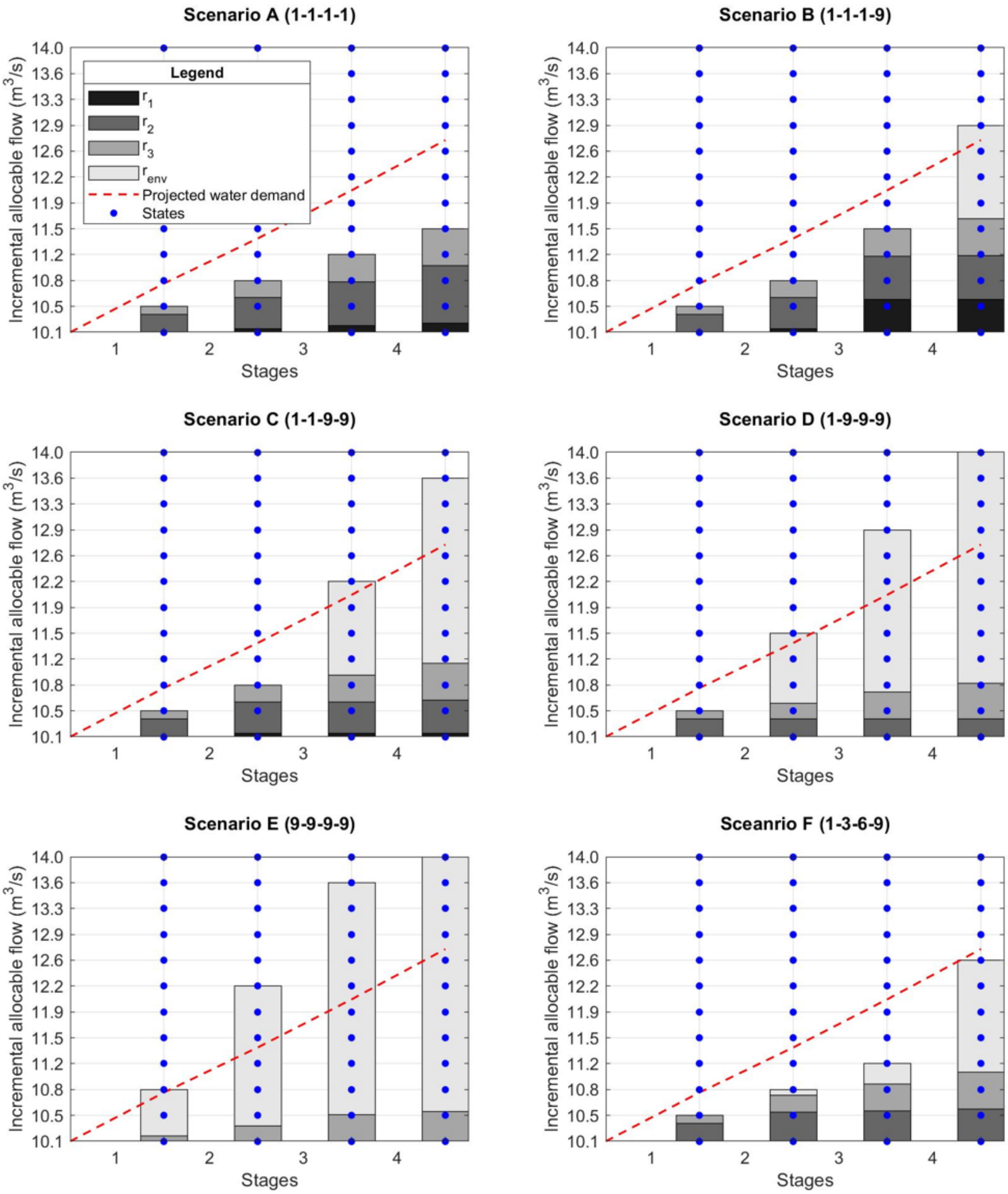

Figure 11: Temporal incremental flow allocation and user sector allocation.

Nonintegrated Scenario

Although having the highest economic benefit increment (R$ 4,666 million), the nonintegrated scenario also has the highest wastewater treatment cost (R$ 970 million), resulting in a final economic result of R$ 3,696 million (Table 4). The result is lower than scenarios A (1-1-1-1), B (1-1-1-9) and even C (1-1-9-9), which highlights the advantages of adopting an integrated approach to analyze water allocation strategies. Besides that, the lack of a water policy definition, as currently observed in nonintegrated scenarios, reduces the guarantee of attending environmental demands.

Strictly economic water policy trajectory - Scenario A

Scenario A results show that by adopting a strictly economic policy trajectory (1-1-1-1) for the future planning horizon, the water is preferably allocated below the projected water demand curve, which reinforces the previous analysis that some level of water scarcity is economically optimal.

Due to the past lack of investment in wastewater infrastructure and also the absence of some preference to environmental quality, great part of the wastewater treatment flow (load removal) necessary to meet water quality targets in the future horizon is used to reduce pollution to the target levels. The modeling results show a wastewater treatment flow raise by 95% from the initial stage to the last one (from 0.13 m^3/s to 2.52 m^3/s), which represents a total system load removal of 86%. The resulting removal is also compatible with the target population with sewage collection and treatment coverage (80%) set in the programs and actions proposals of the Water Resource Plan up to the final stage (Profill, 2014b).

Although the expressive necessity to increase the sewage collection and treatment infrastructure, the modeling results show the importance of interconnecting different instruments to support better decisions. Not attending full water demands in order to leave more water in the river to meet water quality targets together with proper wastewater treatment investments have better economic results than the economic benefits of attending full demand coverages. This highlights how the integration of the instruments is likely to improve economic efficiency. In the other hand, the nonintegrated application of the water management instruments, as it is currently done, can hide important outcomes.

The irrigated agriculture sector is the one whose water demand coverage is first reduced due to its lower resulting economic benefit, comparing to urban and industrial users. The demand coverage is reduced by 18% up to the last stage, totalizing 1.20 m^3/s as water scarcity. The total economic benefit loss over 20 years resulting from the irrigated agriculture water scarcity is 3 R$ million, and the money saved in wastewater treatment achieves 10 R$ million.

This result provides insight into water allocation opportunities. It is possible to mitigate the observed economic loss sustained by the irrigated agriculture sector through investments in efficient use of water, which could be made feasible through economic water management instruments (e.g. subsidies to users improve their technology and water charges). If the 1.20 m^3/s irrigated agriculture water scarcity are saved adopting strategies and technologies to improve the efficiency of water use (i.e. adopting new irrigation technologies, automating flow monitoring to avoid waste or leakages, among others), it would be possible to reach the same economic benefit level of a full demand coverage, but leaving more water in the river to fulfill environmental demands, dilution and protection, and also minimizing wastewater treatment investments in infrastructure (e.g. the irrigated agriculture sector achieves the same benefits, while using less water).

The water scarcity indicates the opportunity cost of the water and it is a reference for investments to improve the efficiency of water use in the watershed. As an example, the water permit form requests could include a percentage of investment in efficiency of water use in order to be approved. The flow coverage reduction in 18% (1.20 m^3/s scarcity) could serve as a reference efficiency target in the planning horizon, and the money saved in wastewater treatment (R$ 10 million) could dictate rates and taxes to encourage users to make proper investments.

Diverse water policy trajectories changes – Scenarios B, C, D, E

All Scenarios B, C, D, and E show some level of water scarcity in order to fulfill water policies with some level of preference towards environmental quality (less water is allocated to economic uses and more is left in the river to fulfill environmental water demands and protection).

Scenario B

The water policy trajectory (1-1-1-9) adopted in scenario B shows that, besides the irrigated agriculture sector (which receives a 14% reduction in future water permits), the urban sector also has its water demand coverage reduced by 4% (up to the final stage). These reductions are equivalent to 0.9 m^3/s and 0.19 m^3/s respectively. The users' sectors and the water demand coverage affected in this scenario is different from scenario A, mainly due to the abrupt water policy change from economic (1) to environmental (9) at the last stage.

Adopting a strong environmental water policy at the last stage causes the last incremental water allocation to be directed to the environment, leaving just a small percentage to be distributed among economic users (previous water permits issued in the past stages are maintained however). Hence, the incremental economic benefit at the last stage is decreased (in the case, only the industrial projected demand could be covered). To mitigate the economic loss resulting from this decision at the last stage (t=4) and increase the global economic benefit, more water permits are issued in the two last stages to fulfill the irrigated agriculture sector demand, besides full urban and industrial demands.

Although resulting in some level of water scarcity for both irrigated agriculture and urban users, scenario B results in a higher total economic return than the scenario where a strictly economic water policy trajectory is chosen (scenario A). The economic benefit loss of scenario B due to economic water scarcity is R$ 3 million but the wastewater treatment cost reduction is R$ 31 million, which results in a total net economic gain of R$ 27 million over 20 years (19 R$ million higher than scenario A) (Table 5).

Imposing a restriction on future increments in water permits to the urban sector may be, at first, controversial, and it will most likely meet criticism. However, this result indicates that providing urban water demands with unchallenged water supplies (as it is the case in Brazil) at the expense of other relevant economic demands and the environmental quality is not economically efficient. The results indicate that this could be compensated by reduction of physical losses and rational use. Physical losses are a critical issue in the Brazilian urban water supply infrastructure, and also in the study watershed. The Water Resource Plan includes in its action program (Profill, 2014a) a target reduction on the water supply distribution physical losses by 10% up to the last stage of the planning horizon modeled here, with an estimated investment of R$ 30 million. The 10% loss reduction would result in 0.35 m^3/s additional supply, which could meet either irrigated or urban demands and reduce economic losses. Taking the irrigated agricultural sector as example, if its demand coverage was increased by 0.35 m^3/s (from 5.76 m^3/s to 6.11 m^3/s) the economic benefit increment would be of R$ 7 million over 20 years. Considering the economic value of the water to the industrial sector, the same increase of 0.35 m^3/s represents an increment in the economic benefit of 100 million over 20 years. Although the water demand to the industrial sector is fully attended for scenario B, it would be a feasible result when adopting other development policies scenarios with higher industrial expansion objectives.

While not contributing to wastewater discharge pollution, hence not affecting the wastewater treatment cost, the losses influence the water supply costs. Lowering the marginal cost of each unit of delivered water causes the marginal cost curve to be shifted. Thus, urban users experience a net benefit gain in their consumer surplus.

On the other hand, rational use affects both water demand and wastewater discharge. For the urban sector, demand-reducing policies based on subsidies or investments in water use technology may induce shifts or rotations on the economic water demand curve, and not movements along the economic water demand curve (see GRIFFIN, 2006 for further details). Urban users still have access to the same water service but using less water they can save on their water bills. Hence, users achieve the same benefits, while using less water. To the system, the result is a gross gain resulting from the water supplied cost saved (the whole economic demand shifts inwards), combined with further savings in the wastewater treatment costs due to reduced discharges.

As example to accomplish this shift in demand, the regulations for urban expansion could require the urbanization projects to declare the efficient use of water. The water coverage reduction (0.19 m^3/s for the urban sector) could be used as a basis for the formulation of efficiency improvement targets (in the form of regulations) and the money saved in wastewater treatment investments (R$ 31 million) could provide assistance for encouraging investments (i.e. rebates, subsidies, among others).

Finally, the wastewater treatment investments saved due to water demand reduction could also be used as a subsidy to reduce wastewater treatment tariff for low income households, enlarging wastewater treatment access and coverage. The R$ 31 million saved in wastewater treatment up to the final stage represents a total reduction of 3.2% in wastewater treatment costs. Thus, this percentage reduction could be applied in the wastewater tariff. Again, the opportunity cost can be applied to justify investments in water efficiency and wastewater treatment leading to better environmental quality and protection, combined with higher economic benefits.

Scenario C

In scenario C (1-1-9-9), the water demand coverage among economic users is reduced even more, achieving a percentage coverage of 81% for irrigated agriculture sector and 92%

for urban sector (Table 4). The industrial sector still has its full demand covered. Although this reduction could lead to a misconception about severe economic losses, modeling results demonstrates that this level of water scarcity among economic users results the second-highest total benefit among all scenarios evaluated here (R$ 3,740 million over 20 years). This is an interesting result, considering that scenario C has a water policy trajectory that changes the preference towards the environment halfway through the planning horizon, and thus leaves out more water to the environmental demands if compared to 1-1-1-1 and 1-1-1-9.

The economic benefit loss of scenario C is R$ 17 million higher than a scenario attending full projected water demands but the wastewater treatment reduction achieves R$ 60 million, which results in a net economic gain of R$ 43 million (R$ 16 million higher than scenario B) (Table 5). These numbers allow good basis for setting long-term programs and actions in the watershed (as previously mentioned) encouraging rational use, environmental protection, and wastewater treatment actions. It also indicates that there is no economic compromise (trade-off) if the water policy is changed towards the environment up to 1-1-9-9 (albeit the distribution of the benefits and costs still needs to be addressed, which is not discussed here).

Scenarios D and E

Adopting water policies with even higher environmental preference (1-9-9-9 and 9-9-9-9) results in more water allocated to the environment, which further reduces water demand coverage. Differently from the previous scenarios, all water users' sectors are now affected, even the industrial one. Water demand coverages at final stage resulted in 80% (r_1), 88% (r_2), and 99% (r_3) for scenario D, and 80% (r_1), 82% (r_2), and 94% (r_3) for scenario E (Table 4). The total economic benefit achieves R$ 3,672 million for scenarios D and 3,210 R$ million for scenario E. The economic net benefit results in a loss of 25 R$ million for scenario D and 486 R$ million for scenario E (Table 5).

Although leaving more water in the river beyond 3.12 m^3/s affects economic benefit from water use and does not provide advantages in wastewater treatment cost reduction, it is important to highlight that it brings other benefits for the ecosystem (intangible benefits), which are out of the scope of this study but already highlighted in the literature. If those water policies should, or should not, be adopted remains a question for discussion in the watershed committee. The trade-offs calculated here should support those discussions by providing scientifically and technically sound information.

Scenario F

The scenario F (1-3-6-9) presents a smoother and progressive transition from a strong economic policy (1) at the first stage to a strong environmental one (9) at the last stage. This trajectory has explicit advantages comparing to the previous ones, among them: (a) it allows more time for users to adapt and incorporate water efficiency improvements facing future water permit restrictions, avoiding conflicts as the water coverages reduction are more gradual, and (b) the progressive change also provides better water demand coverage with higher net benefit results comparing abruptly changes. This scenario also resulted in the highest total economic benefit (R$ 3,753 million) indicating the best balance between water scarcity to users and wastewater treatment costs. The net economic gain achieves 57 R$ million, which is 14 R$ million higher than the scenario C (1-1-9-9) (Table 5). As seen in figure 9, this scenario suggests that water permits would not be issued up to the projected water demand (dotted line), instead, less water is allocated, which boosts the river assimilation capacity and reduces the wasterwater treatment costs (while still meeting the water quality targets prescribed).

Sensitivity analysis

The modeling scenarios performed in the sensitivity analysis (As and Es), which uses industrial benefit function whose original marginal water values were reduced to 25%, resulted in smaller economic benefit increments comparing to their base scenarios (A and E) (Figure 12). Table 4 shows a reduction from 4,664 R$ million to 1,596 R$ million between scenarios A and As and a reduction from 3,969 R$ million to 1,390 R$ million between scenario E and Es.

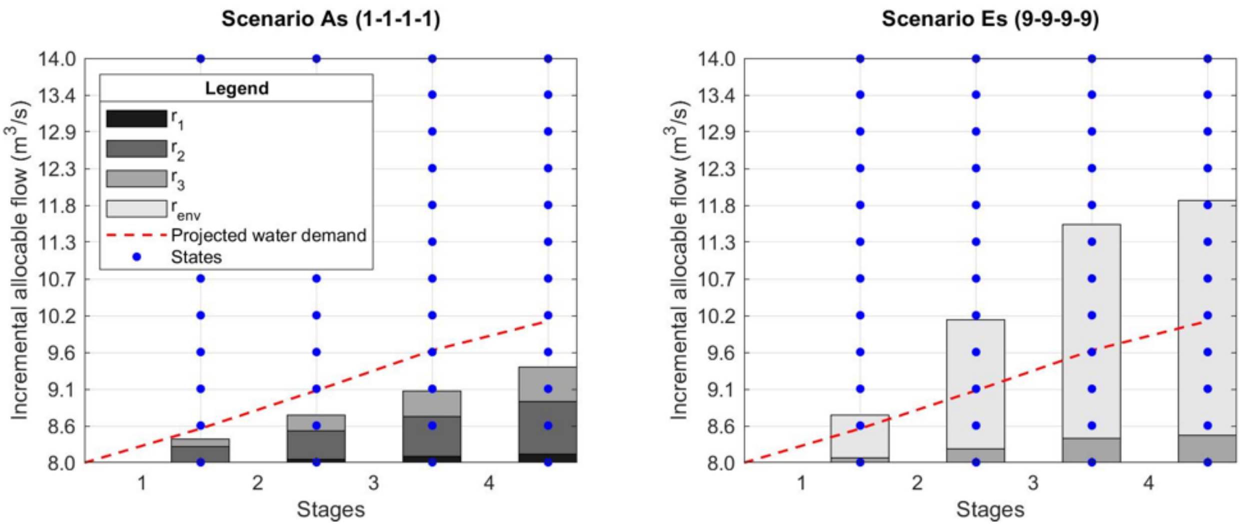

Figure 12: Scenarios: temporal incremental flow allocation and user' sector allocation.

The new industrial benefit function reduced the gap between wastewater treatment costs and economic benefit from use of water. The resulting economic benefit is R$ 637 for scenario As, and R$ 631 million for Scenario Es. Scenario As indicates that wastewater treatment costs are high enough to avoid water allocation to economic users, despite of having a lower economic benefit return comparing to scenario A.

In regard with water increments through time and distribution among users, the sensitivity analysis modeling results presented the same behavior comparing to their base scenarios (A and E). Water demand coverages at final stage remained in 78% (r_1), 100% (r_2), and 100% (r_3) for scenario As, and 75% (r_1), 78% (r_2), and 94% (r_3) for scenario Es.

2.5 Conclusion and future improvements

Reconciling economic development and environmental protection is still a challenge, but this study demonstrates that it is possible to achieve effective solutions and foster sustainable economic growth by integrating the water management instruments to the water allocation approach.

The integration of the water management instruments allows one to explore more efficient solutions to externalities that remain unaddressed in many water systems worldwide. Issuing water permits in other to fulfill economic demands, regardless of the evaluation of the economic benefits obtained by their use and wider environmental impacts, imposes several externalities. Discharges and withdrawals affect the assimilation capacity of the river and elevate the concentration level of pollutants, imposing externalities to downstream users. As an example, withdrawals from irrigated agriculture users can affect the assimilation capacity of the river, requiring downstream users to be more efficient in wastewater treatment in order to achieve the water quality targets. In the example of urban supplies in Brazil, water permits are issued to fulfill demands, which goes unchallenged at the expense of other relevant economic demands and the environment, regardless of how efficient the water is being used in the urban area and if the wastewater produced is being properly collected and treated.

While imposing a restriction on future increments in water permits may be controversial and it will most likely meet criticism, the modeling results show that fully meeting economic water demands is not an economically efficient solution. It brings wastewater treatment costs higher than the economic benefits from using the water and usually the downstream users pay for it. On the other hand, restricting water permits is likely to be more efficient economically, besides being an environmentally better option. As less water is abstracted from the river, more will be available to fulfill environmental demands and to assimilate other pollution loadings, also minimizing the necessary investment in wastewater treatment infrastructure. Thus, some level of water scarcity is economically optimal.

The water policy is a key element in this process, which allows to explore better solutions and reach these conclusions. In many studies, the environmental demand is represented as a physical constraint (e.g. as minimum flow) with little assessment of its socio-economic representation. In practice, there is no water policy definition, and hence little guarantee of environmental demand attendance, which leads to the traditional conflict between economic development and environmental quality. Bringing the water policy trajectory into the approach removes the limitation of the economic optimization not including other water values that are difficult or for which there is limited data to evaluate. By representing the society's preferences and priorities towards environmental quality, it enables negotiators to have a clear vision of environmental demands, and hence, its implications to economic users as previously mentioned. Thus, the water policy is a key element that, by being defined in the Water Resource Plans, it enables to set the directives to other instruments, such as temporal and spatial distribution of water permits, user sector, and water quality targets.

Finally, the proposed approach provides insight into water allocation opportunities. The economic loss caused by some level of water scarcity can be mitigated by adopting efficiency use of water actions (e.g. encouraging the adoption of water saving technologies, monitoring, reducing physical losses, among others), which could be made feasible through economic water management instruments (e.g. subsidies and water charges). In many situations such economic instruments, when applied, are based on poor knowledge of the real economic benefit they can provide, with no specific goal definition. Thus, the modeling results can provide a clear vision to help implementing such actions (target reduction, when and where it should be implemented, which user sector should be prioritized) and also provide insight of the amount to be invested or rebate in the form of subsidies.

Given the modeling limitations and insights, several improvements and recommendations are possible for continuing work:

a) Improving the marginal economic value function for the industrial sector, detailing and including more data from other facility types relevant to the watershed;
b) Incorporating potential climate change impacts in local hydrology into the approach, by varying the flow duration value throughout the planning horizon so that the model can explore an adaptation strategy for water allocation along the planning horizon;
c) Exploring different water quality targets and its implications;
d) Increasing the number of river reaches for better precision of the water quality results and incorporate other parameters in the water quality simulation equations (i.e. non-diffuse pollution);
f) Including decision variable on renewal of water permits for modeling saturated watersheds;
g) Define and include economic instruments into the approach;
h) Apply other indicators to measure indirect benefits, such as: employment and Gross Domestic Product (GDP).

2.6 Acknowledgments

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2.7 References

AHMADI, A. et al. Integrated Planning of Land Use and Water Allocation on a Watershed Scale Considering Social and Water Quality Issues**. Journal of Water Resources Planning and Management**, v. 138, n. 6, p. 671–681, 2012.

ANA. **Atlas irrigação: uso da água na agricultura irrigada**. 1. ed. Brasília: Agência Nacional de águas, 2017.

AZEVEDO, B. L. G. T. DE et al. Integration of water quantity and quality in strategic river basin planning. **Water Resources Planning and Management**, v. 126, n. April, p. 85–97, 2000.

BISWAS, A. K. Integrated water resources management: a reassessment. **Water International,** v. 29, n. 2, p. 248–256, 2004.

BRASIL. Resolução CONAMA 357. Ministério do Meio Ambiente. Brasília, DF, 2005.

BRASIL. Population census. Instituto Brasileiro de Geografia e Estatítica - IBGE, 2010.

CAI, X.; MCKINNEY, D. C.; LASDON, L. S. Integrated hydrologic-agronomic-economic model for river basin management. **Journal of Water Resources Planning and Management**, v. 129, n. 1, p. 4- 17 ST- Integrated hydrologic- agronomic- economi, 2003.

CONCREMAT. **Plano regional de saneamento básico: relatório do diagnóstico**. Porto Alegre, 2014.

CORSAN. **Estrutura tarifária sintética - consórcio Pró-Sinos**. Disponível em: <http://www.corsan.com.br/sistematarifario>. Acesso em: 11 dez. 2018.

DAVIDSEN, C. et al. Hydroeconomic optimization of reservoir management under downstream water quality constraints**. Journal of Hydrology**, v. 529, n. 1, p. 1679–1689, 2015.

FEE. **Três décadas de economia gaúcha: um panorama gráfico**. Porto Alegre: Fundação de Economia e Estatística, 2010.

GELT, J. Sharing Colorado River water: history, public policy and the Colorado River compact. **Water Resources Research Center**, v. 10, n. 1, 1997.

GEORGE, B. et al. An integrated hydro-economic modelling framework to evaluate water allocation strategies I: Model development**. Agricultural Water Management**, v. 98, n. 5, p. 733–746, 2011a.

GEORGE, B. et al. An integrated hydro-economic modelling framework to evaluate water allocation strategies II: Scenario assessment. **Agricultural Water Management**, v. 98, n. 5, p. 747–758, 2011b.

GHOSH, S. et al. Water allocation and management along the Santa Cruz border region. **Water Resources and Economics**, v. 19, n. September, p. 1–17, 2017.

GRAFTON, R. Q. et al. Optimal dynamic water allocation: Irrigation extractions and environmental tradeoffs in the Murray River, Australia. **Water Resources Research**, v. 47, n. 6, p. 1–13, 2011.

GRIFFIN, R. C. **Water resource economics: the analysis of scarcity, policies, and projects**. Massachusetts: The MIT Press, 2006.

HAROU, J. J. et al. Hydro-economic models: Concepts, design, applications, and future prospects. **Journal of Hydrology**, v. 375, n. 3–4, p. 627–643, 2009.

HEMMAT, A. et al. Dealing with Confict over Water Quality and Quantity Allocation? A Case Study. **Scientia Iranica**, v. 9, n. 1, p. 15–23, 2007.

HOWITT, R. E. et al. Calibrating disaggregate economic models of agricultural production and water management. **Environmental Modelling and Software**, v. 38, p. 244–258, 2012.

HU, Z. et al. A multi-objective optimization model with conditional value-at-risk constraints for water allocation equality. **Journal of Hydrology**, v. 542, p. 330–342, 2016a.

HU, Z. et al. Optimal allocation of regional water resources: From a perspective of equity-efficiency tradeoff. **Resources, Conservation and Recycling**, v. 109, p. 102–113, 2016b.

IBGE. **Indicadores de desenvolvimento sustentável: Brasil 2010**. Rio de Janeiro: 2010.

IRGA. **Área e produção de arroz no estado do Rio Grande do Sul**. Disponível em: <http://stirga2018-admin.hml.rs.gov.br/upload/arquivos/201805/22093809-producao-rs-serie-historica.pdf>. Acesso em: 11 dez. 2018.

JENKINS, M. W. **CALVIN Appendix B2: Urban value function documentation**, 2000.

KONDILI, E.; KALDELLIS, J. K.; PAPAPOSTOLOU, C. A novel systemic approach to water resources optimisation in areas with limited water resources. **Desalination**, v. 250, n. 1, p. 297–301, 2010.

LETCHER, R. A.; JAKEMAN, A. J.; CROKE, B. F. W. Model development for integrated assessment of water allocation options. **Water Resources Research**, v. 40, n. 5, p. 1–15, 2004.

LOPES, V. A. R.; FERNANDES, G.; DORNELLES, F. Performance of rainwater harvesting systems under scenarios of non-potable water demand and roof area typologies using a stochastic approach. **Journal of Cleaner Production**, v. 148, p. 304–313, 2017.

LUND, J. R.; CAI, X.; CHARACKLIS, G. W. Economic Engineering of Environmental and Water Resource Systems. **Journal of Water Resources Planning and Management**, v. 132, n. December, p. 399–402, 2006.

LUND, J. R.; REDD, R. U. Drought water rationing and transferable rations. **Journal of Water Resources Planning and Management**, v. 121, n. December, p. 429–437, 1995.

MAGALHÃES, A. S. et al. Quanto Vale a Água Que Usamos? Projeções Dos Impactos Econômicos De Restrições Ao Uso E Elevação De Preços Da Água Na Região Metropolitana De Belo Horizonte. **Revista de Economia** (Curitiba), v. 43, n. 2, p. 1–22, 2016.

MARTINSEN, G. et al. Joint optimization of water allocation and water quality management in Haihe River basin. **Science of the Total Environment**, v. 654, p. 72–84, 2019.

MATTIUZI, C. D. P. **Gestão integrada dos recursos hídricos: alocação otimizada com uso conjunto de água superficial e subterrânea para redução da escassez hídrica na bacia do Rio Santa Maria/RS**. Porto Alegre. Universidade Federal do Rio Grande do Sul (UFRGS), 2018.

MAVROTAS, G. Effective implementation of the e -constraint method in Multi-Objective Mathematical Programming problems. **Applied Mathematics and Computation**, v. 213, n. 2, p. 455–465, 2009.

METACALF AND EDDY, I. **Wastewater engineering: treatment and reuse**. Boston: McGraw-Hill, 2003.

MOLINOS-SENANTE, M. et al. A management and optimisation model for water supply planning in water deficit areas. **Journal of Hydrology**, v. 515, p. 139–146, 2014.

MOMBLANCH, A. et al. Using ecosystem services to represent the environment in hydro-economic models. **Journal of Hydrology**, v. 538, p. 293–303, 1 jul. 2016.

MORAES, M. et al. Modelo econômico-hidrológico integrado para gestão de bacias hidrográficas: abordagem piece-by-piece para incorporar a capacidade de autodepuração dos rios e limites de trofia dos reservatórios. **Revista Brasileira de Recursos Hídricos**, v. 13, n. 1, p. 43–55, 2008.

PROFILL. **Relatório final RT1 Fase inicial e Fase A**. Porto Alegre: 2013.

PROFILL. **Relatório técnico 2 - RT2 Fase B: complementação do enquadramento**. Porto Alegre: 2014a.

PROFILL**. Relatório Final RT3 Fase C: programa de ações.** Porto Alegre: 2014b.

PULIDO-VELAZQUEZ, M. et al. Hydro-economic river basin modelling : The application of a holistic surface – groundwater model to assess opportunity costs of water use in Spain. v. 6, 2008.

PULIDO-VELAZQUEZ, M.; ANDREU, J.; SAHUQUILLO, A. Economic Optimization of conjunctive use of surface water and groundwater at the Basin scale. **Journal of Water Resources Planning and Management**, v. 132, n. 6, p. 454–467, 2006.

ROOZBAHANI, R.; SCHREIDER, S.; ABBASI, B. Optimal water allocation through a multi-objective compromise between environmental, social, and economic preferences. **Environmental Modelling and Software**, v. 64, p. 18–30, 2015.

ROSEGRANT, M. W. et al. Integrated economic-hydrologic water modeling at the basin scale: The Maipo river basin. **Agricultural Economics**, v. 24, n. 1, p. 33–46, 2000.

SINOS. **Plano de Gerenciamento da bacia hidrográfica do Rio dos Sinos: Atividade 3.3 - síntese da situação atual dos recursos hídricos**. Porto Alegre, 2010.

SINOS. **Caracterização da bacia hidrográfica do Rio dos Sinos**. Disponível em: http://www.comitesinos.com.br/bacia-hidrografica-do-rio-dos-sinos. Acesso em: nov de 2018.

TRATA BRASIL, G. A. Perdas de água 2018 (SNIS 2016): desafios para disponibilidade hídrica e avanço da eficiência do saneamento básico. **Trata Brasil, Go associados**, v. 2018, n. Snis 2016, p. 68, 2018.

TU, M. Optimization of Water Distribution and Water Quality by Hybrid Genetic Algorithm. **Journal of Water Resources Planning and Management**, v. 131, n. 6, p. 431–440, 2006.

USACE, U. A. C. OF ENGINEERS. **HEC-RAS river analysis system User's Manual**. Davis, California: 2016.

WANG, X. et al. An eco-environmental water demand based model for optimising water resources using hybrid genetic simulated annealing algorithms. Part I. Model development. **Journal of Environmental Management**, v. 90, n. 8, p. 2628–2635, 2009.

WANG, Y.; YANG, J.; CHANG, J. Development of a coupled quantity-quality-environment water allocation model applying the optimization-simulation method. **Journal of Cleaner Production**, v. 213, p. 944–955, 2019.

WANG, Z. et al. Optimal Water Resources Allocation under the Constraint of Land Use in the Heihe River Basin of China. S**ustainability**, v. 7, n. 2, p. 1558–1575, 2015.

XU, J. et al. Intergenerational equity based optimal water allocation for sustainable development: A case study on the upper reaches of Minjiang River, China. **Journal of Hydrology**, v. 568, n. October 2018, p. 835–848, 2019.

YOUNG, R. A.; LOOMIS, J. B. **Determining the economic value of water: concepts and methods**. 2. ed. New York: RFF Press, 2014.

ZAMANI, H. et al. Strategic allocation of water conservation incentives to balance environmental flows and societal outcomes. **Ecological Engineering**, v. 127, n. October 2018, p. 160–169, 2019.

CAPÍTULO 3

Integrating water permits and quality targets to establish a long-term spatial water allocation strategy

3.1 Introduction

River water availability has long been used as the only constraint to issue water permits across the watershed. However, water use decisions have economic and water quality implications that are still little assessed in the present and long-term planning. The model VISTA through the *sub-routine III - User spatial allocation* uses water quality simulation and economic optimization integrated in the long-term planning (together with development policies), enabling to explore optimal spatial allocation of water permits at minimum wastewater treatment costs to accomplish river and user's water quality requirements. Thus, it provides a contribution to the field by filling the integration gap among water management instruments (water quality targets and water permits) and allowing to identify directives to water management instruments. When a new user requests a water permit, the river basin authority or department of water resources will check on the water policy directives to decide if that permit should be issued on that particular location, at that moment in time and in the required amount.

This article discusses the spatial water allocation resulting from the modeling *sub-routine III - User spatial allocation* of the VISTA model and its implications to the watershed water quality. Among the 4 main group of scenarios described in the *modeling scenarios* section of the first article *Reconciling water policies with broader economic development policies through integrated water management instruments*, we chose one scenario of each group to emphasize the discussion. Scenario A (1-1-1-1) representing strictly economic water policy trajectory, Scenario E (9-9-9-9) representing a strong environmental policy trajectory, and Scenario F (1-3-6-9) representing a gradual shift in the trajectory from economic to environmental water policies along the planning horizon. The scenarios representing the sensitivity analysis (As and Es) weren't analyzed since they resulted in the same spatial water allocation as their base scenarios (A and E).

The spatial results are shown in the form of maps organized by scenarios, water user's sector, and stages. The watershed is discretized into regions according to their contribution (withdrawals and discharges) to each reach *n* (Figure 12). Thus, each region represents the spatial allocation of water permits to users in that region. The resulting water quality from the spatial allocation of water permits is represented spatially by the variation of the concentration of a given constituent at each reach *n*. The constituent used in the analysis is thermotolerant coliforms, which has been verified as a key constituent in the case study watershed – Sinos River Basin).

3.2 Results

3.2.1 Spatial water allocation by user sector

Figure 13 represents the sum of all users' observed water demand at each watershed region (reference year 2014). The lower watershed (*n* = 1, 2, and 3) is the one with higher water demand due to its strong industrial and urbanization characteristics combined with irrigated rice production. The lower watershed represents 85% of the total urban water demand, 66% of the total water demand for irrigated rice production, and 92% for industrial purposes.

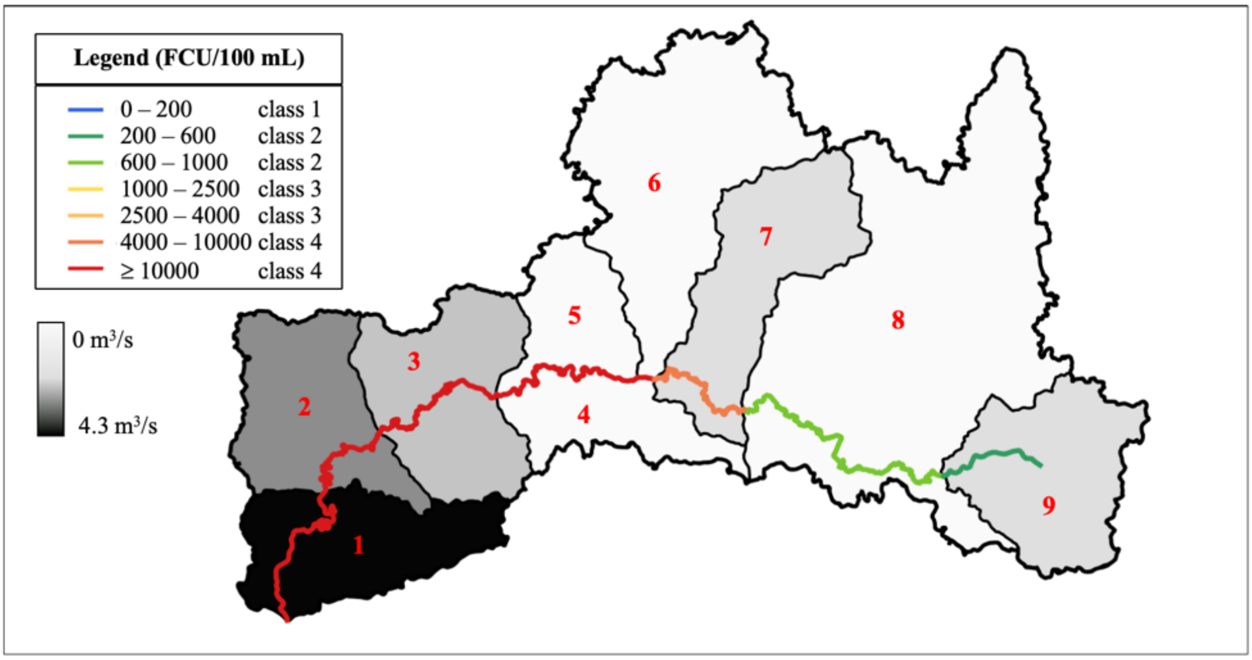

Figure 13: Representation of the observed water demand by watershed region and observed water quality concentration at the main river.

Figure 14 presents the optimal spatial allocation of water permits for each economic user sector *i* (irrigation, urban, and industrial) at each region. The darker the color, the more *new* water permits are issued (allocated) to the region. The maps are organized by scenario so that for each user sector *i,* is possible to analyze the spatial flow allocation increment *s* between the initial and the last stage. The intermediary stages are not shown since the smaller increments along the planning horizon would be of little use when comparing scenarios.

For the irrigated agriculture and industrial sector, the reaches 1 and 2 at the lower watershed are more favorable to the allocation of new water permits, despite of their higher pollution loading. The depuration capacity of these reaches together with less restricted water quality targets explains these results, as analyzed in the section 3.2.2. The industrial sector gets 100% of its projected water demand in scenarios A and F, which means new water permits are issued following the projected water demand growth. For scenario E however, some curtailment in the water permits takes place and demand coverage drops to 96% at the end of the planning horizon. Thus, the allocation of new water permits suffers little variation among scenarios. For the irrigated agriculture sector however, no new water permits are allocated along the planning horizon in Scenarios E and F. Only for scenario A there is the small increment in new water permits (0.12 m^3/s), which is barely perceptible comparing to the initial stage (observed demand).

For the watershed, these results mean future expansion of irrigated agriculture should be curbed (current water permits would be maintained, however) due to a combination of three drivers: (a) adoption of a water policy that places more priority in the environment, and consequently allocated less water to economic uses; (b) the water is reallocated to other economic demands (e.g. industry) for improved economic efficiency and (c) there are water quality targets to be met, which will be benefited if water withdrawals do not increase in the future.

For the urban sector, the allocation of new water permits is constrained by the user's water quality requirements. No new urban demand can withdraw water if the river has water quality class above 3. However, the current water permits for urban sector are maintained even if the reach is classified as class 4, since VISTA just considers new incremental flow allocation (although the current regulation imposes this restriction, withdraws under class 4 is a common reality in most part of the urban regions of Brazil). Another consideration here regards the water quality modeling limitation. The spatial allocation sub-routine considers the impacts of all withdrawals and discharges in the main river channel only (tributaries are not modeled). Thus, the water for urban use may be withdrawn from a tributary classified under a lower class, even if the main reach is classified as class 4. However, this limitation is attenuated by the fact that the main tributaries of the lower watershed are current also classified as class 4.

As the intermediary water quality targets of the upper watershed (reaches 7, 8, 9) meet urban requirements before the lower watershed (the lower watershed achieves class 3 just at the last stage), it is more favorable to receive new water permits to urban use, and thus some water permits are allocated to regions in the upper watershed.

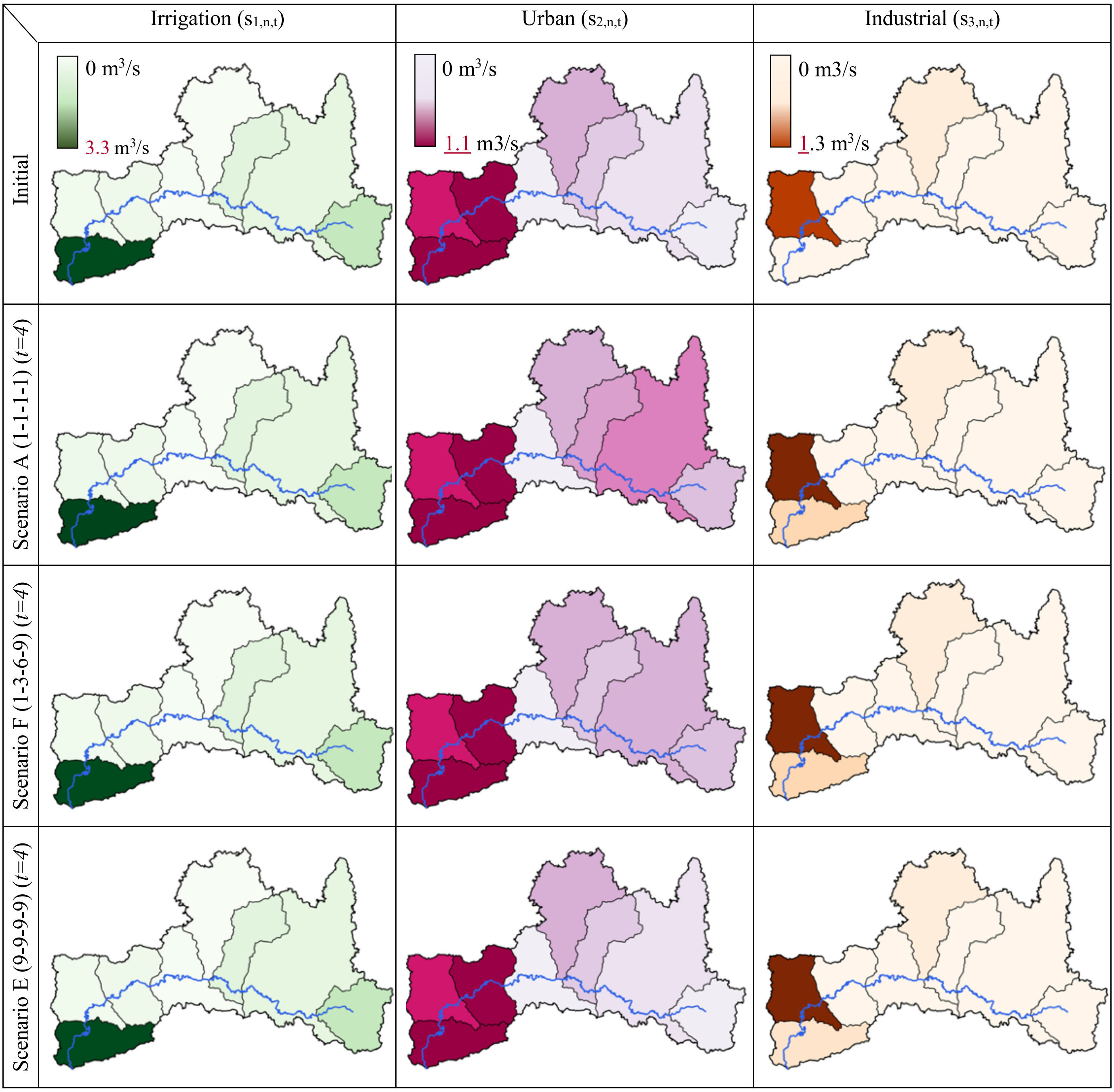

Figure 14: Spatial allocation of water permits.

3.2.2 Required load removal and water quality concentration evolution

Figure 15 shows the regions where higher investments in infrastructure must be done in other to achieve the prescribed intermediary and final water quality targets. The investments are for treating both the existing pollution and the pollution resulting from the allocation of new water permits) along the planning horizon (the darker the shaded area, the higher the wastewater flow treated). The maps are organized by scenario and stage. For each scenario, it is possible to visualize the wastewater treatment flow increments at each stage *t* together with the resulting water quality concentration at each reach *n*, depicted by the color of the river line.

The concentrations above water quality targets at all reaches *n*, with exception of reach 8, indicate that the load emissions are higher than the river depuration capacity at minimum flows (Q_{90}). Under such conditions, there is no water available to dilute additional loadings. Hence, in order to accommodate new users and meet the prescribed water quality targets, it is necessary to reduce the existing pollution levels in almost all reaches. An exception occurs at reaches 4 and 5 (at the middle watershed). These regions present smaller population, industrial and irrigation water uses, so that the concentration in the corresponding river reaches is mostly affected by users upstream (in both quantity and quality).

For all scenarios, the lower watershed (reaches 1, 2 and 3) is the one that requires the highest wastewater treatment investments, mainly due to the concentration of urban demands combined with very limited sewage collection and treatment infrastructure. Despite the concern with high pollution, these reaches are still more favorable to the allocation of new water permits, as previously observed in Figure 14. The backwater effect at the river mouth (Jacuí delta) reduces the hydraulic speed and contributes to increase the auto depuration capacity of the reaches 1 and 2.

Scenario A (1-1-1-1), which provides water permits for the most part of the economic projected water demand (82% for irrigation, and 100% for urban and industrial sectors), also has the highest investment in wastewater treatment compared to the other scenarios, achieving 2,52 m^3/s at a total cost increment of R$ 959 million (over the 20-year planning horizon). At the lower watershed (reaches 1, 2 and 3), the load removal follows the allocation of new water permits to industrial and irrigated agriculture sectors, combined with incremental wastewater improvements to treat the existing pollution and meet the intermediate water quality targets. No allocation of new water permits occurs at the middle watershed (reaches 4, 5, 6) so that this region only depends on the upstream reaches resulting quality. For the upper watershed, reaches 8 and 9 achieve the final water targets at the first stage but they need incremental wastewater treatment along the planning horizon due to the allocation of new water permits to urban users.

For Scenario F (1-3-6-9), which reflects a gradual shift to water policies with increasing preference for environmental quality, the allocation of new water permits is reduced, falling behind the projected water demand by a larger margin if compared to scenario A (irrigated agriculture has 20% of its demand unmet and the urban sector has 8% of its demand unmet). However, scenario F also demands smaller investment in wastewater treatment compared to scenario A. This can be verified in the upper watershed, in which scenario F achieves the same water concentration as scenario A, but with less investment in wastewater treatment due to its smaller water abstraction and wastewater discharges by users. Although new water permits are reduced along the planning horizon, under scenario F the lower watershed needs similar wastewater treatment investments as scenario A in order to remove the existing pollution.

For Scenario E (9-9-9-9), most of the investment in wastewater treatment must be done to remove the existing pollution. As the allocation of new water permits are severely reduced in order to follow a more environmental water policy trajectory (9-9-9-9), the river depuration capacity increases, along with less wastewater discharges to be treated. The water allocated to the environment achieves 2.01 m^3/s, reducing the total cost of new wastewater infrastructure from R$ 959 million in Scenario A to R$ 759 million in scenario E.

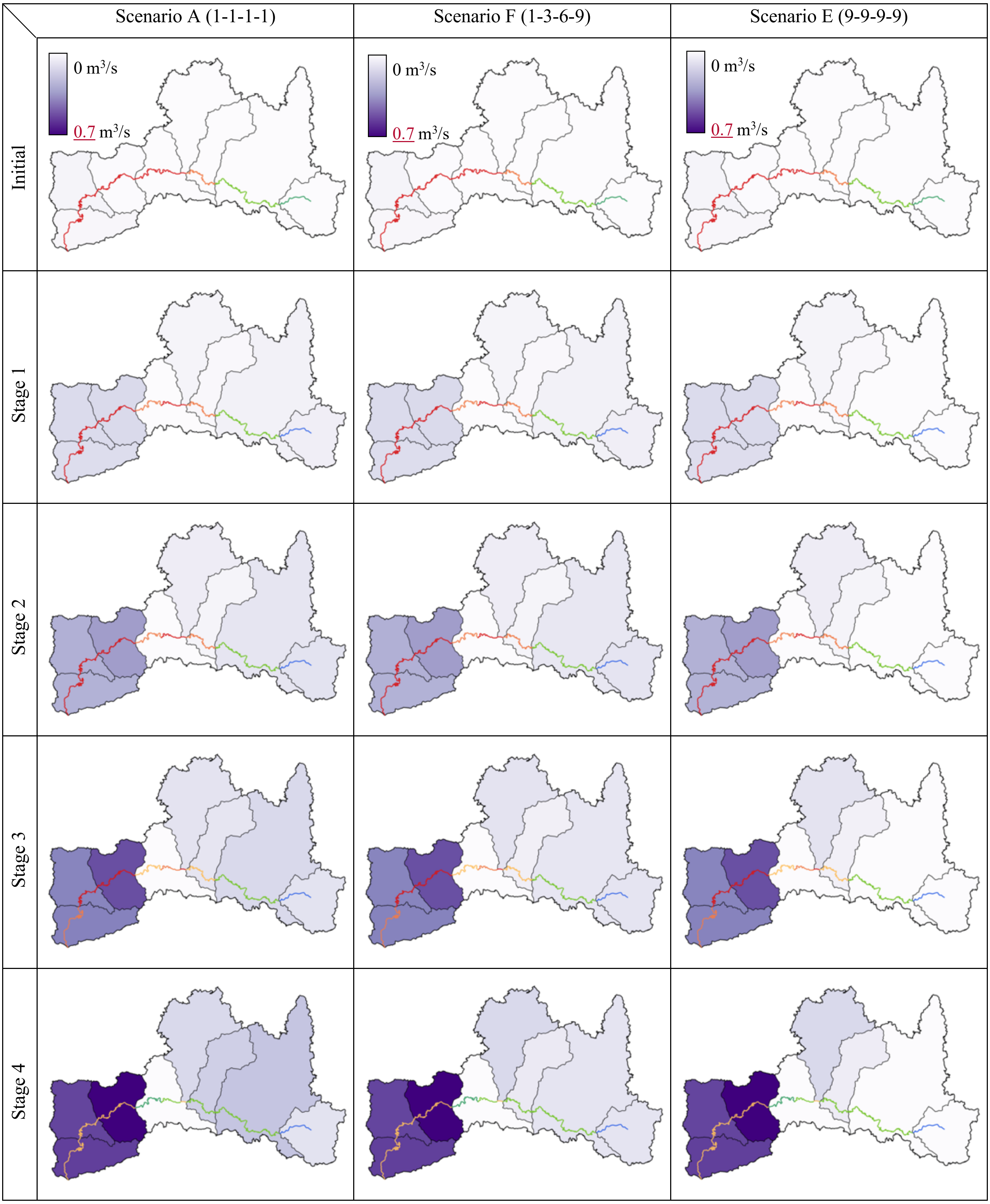

Figure 15: Load removal required and water quality concentration improvements.

3.2.3 Modeling concentration and water quality targets

Figure 16 shows the evolution of the concentration from the initial to the final stage comparing the modeling scenarios result and the water quality target prescribed by the River Sinos Water Resources Plan (Profill, 2014b). The reduction in the allocation of new water permits (which reduces demand coverage) not only meets the water quality targets but it achieves even better results. For reaches 4 and 5 (middle watershed), due to improved investment in wastewater treatment in the upper reaches, the water depuration capacity allowed it to achieve better water quality concentration than the water quality targets.

This indicates that the water quality targets proposed by the Water Resources Plan could be economically more efficient in reaches 4 and 5 (middle watershed) if they were stricter (class 2, as per the results, instead of class 3). While this apparently goes against conventional wisdom (after all, a less strict water quality target means more pollution accepted in the river and thus less treatment and costs) when analyzed within an integrated perspective with the other water management instruments (water permits in this case) the conclusion is different.

If reaches 4 and 5 arrive at class 3 (which is the proposed water quality target) it would be due to higher treated wastewater flow, which would be a consequence of an increased allocation of new water permits. However, increasing water allocation to economic users in reaches 4 and 5 would also bring water pollution (here measured as thermotolerant coliforms) to a level (and quantity) where the wastewater treatment needed to mitigate it and then achieve the water quality target is higher than the economic benefit brought by the water use, even at a slightly less strict water quality target (class 3).

In this case, it might be economically more efficient to either curb the demand (which is proposed here with a reduction in new water permits, which would result in reaches 4 and 5 arriving at class 2) or to look for wastewater treatment technologies that are less costly (and at least as efficient). By reducing the allocation of new water permits in reaches 4 and 5, the flow of wastewater produced is reduced, requiring less investment in treatment infrastructure. According to the model results, the savings in the infrastructure to properly treat and discharge the wastewater is higher than the economic benefit to the users who generated the wastewater. The negligence of such aspect is commonly found in Brazil, when the cost of eliminating the externalities (river pollution) is not considered when a development policy is laid out and water permits are issued to accommodate it at any cost. The result is the high pollution loading in the rivers.

These results show that the lack of properly assessing economic implications (both in terms of the economic value of the water and the costs to supply it and deal with the externalities of using it) and the lack of integration in the water management instruments (e.g. water quality targets and water permits) can result in decisions that are most costly. Also, adopting class 2 water quality target at reaches 4 and 5 would also contribute to the preservation of its ecosystem, which value has not been taken into account here. Finally, less or more restrictive water quality targets in a given region brings economic, social and environmental consequences. Thus, the modeling results provides a good perception of how far the proposed water targets are from an economically more efficient situation.

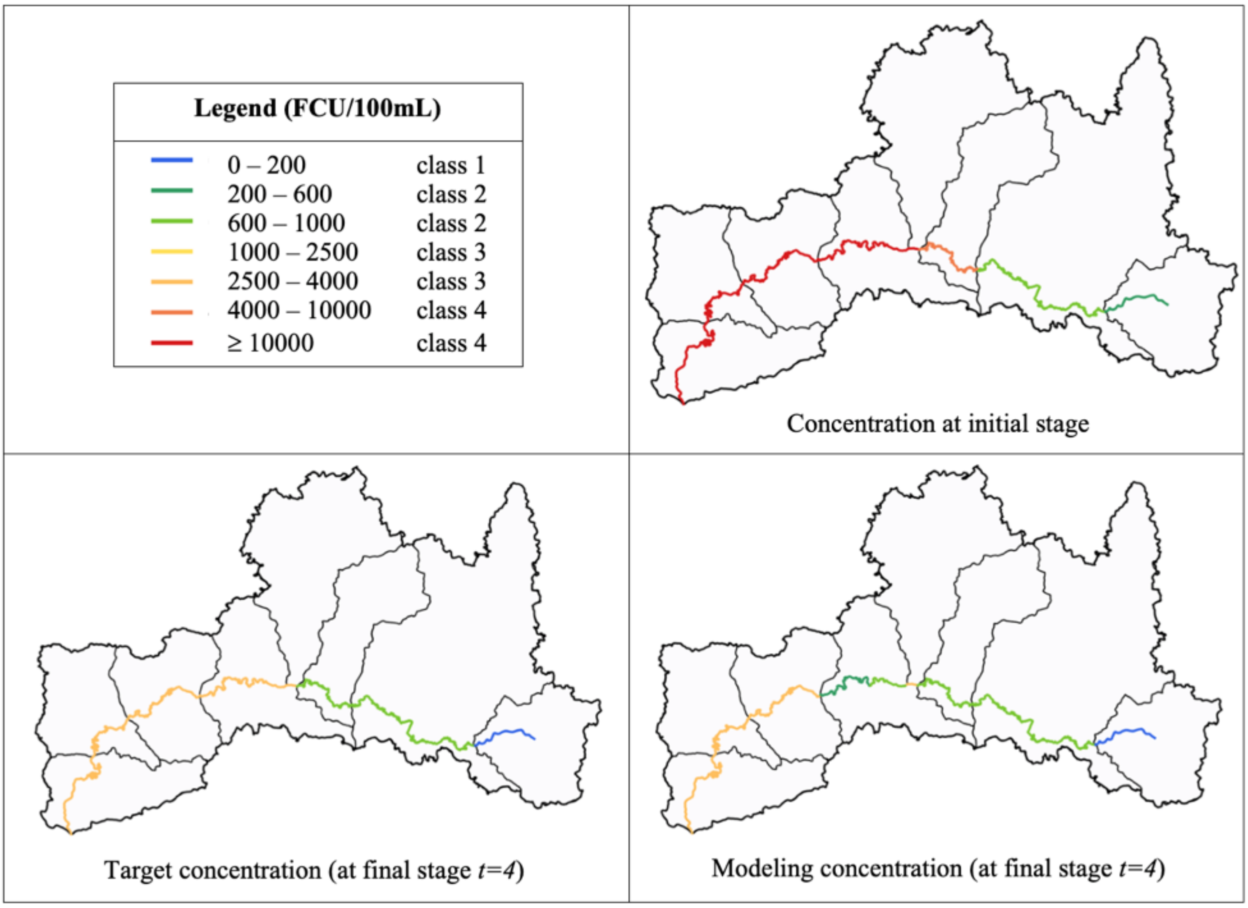

Figure 16: Water quality concentration at final stages.

3.3 Conclusion

The modeling results regarding optimal spatial water allocation allowed the identification of priorities for water pollution control investments, preservation, and economic development across the watershed. Allocating new water permits at some regions causes higher impacts on the water quality, requiring higher wastewater treatment investments in order to achieve the water quality targets or even to introduce more restrictive effluent emission thresholds.

Although having little influence in effluent emissions for the parameter analyzed in this study, allocating additional water permits to irrigated agriculture decreases the river dilution capacity. The modeling results showed that the upper and middle watershed are less favorable to irrigated agriculture, since it would result in higher concentration of pollution, requiring higher wastewater treatment investments to achieve the water quality targets. Thus, these regions are good examples where efficient use of water should be prioritized, so that fewer water permits would be necessary.

Scenario A resulted in irrigated agriculture allocated 18% less water than the total projected demand (representing 1.20 m^3/s). In order to mitigate the economic loss resulting from this scarcity, as proposed in article 1 (capítulo 2), strategies and technologies to improve the efficiency of water use should be adopted. The upper and middle division are good examples where efficient use of water should be prioritized. To allow irrigation expansion at these regions, water efficient use requirements should be imposed so that the curtailment on the new water permits would be mitigated by the water saved through application of such requirements.

Similarly, the installation of new industries in some regions should require more restrictive effluent discharges thresholds in order to mitigate the resulting elevation of the pollution and reducing of the dilution capacity. More restrictive water charges for water withdrawal or effluent emission thresholds could also be imposed at these regions in order to induce rational use and improve the efficiency of the water use and load removal efficiency.

The integration between different water management instruments, like the determination of water quality targets and its reflection on the issuing of water permits, is still little assessed. However, this study showed that such integration can bring in valuable insight to help finding long term water allocation strategies that are less costly to users and environmentally better, contributing to more effective water resources management.

3.4 Reference

PROFILL. **Relatório técnico 2 - RT2 Fase B: complementação do enquadramento**. Porto Alegre: 2014.

CAPÍTULO 4

Avaliação de um modelo simplificado de simulação da qualidade da água para o Rio dos Sinos

4.1 Introdução

A gestão de recursos hídricos é caracterizada pela natureza multidisciplinar, o que requer o emprego de diferentes (e avanço de novos) métodos para integrar aspectos técnicos, econômicos, ambientais, sociais e legais em uma abordagem coerente. Modelos hidrológicos geralmente utilizam técnicas de simulação, enquanto que modelos econômicos usam técnicas de otimização. As diferentes técnicas dificultam a incorporação das ferramentas necessárias para análise em uma única plataforma de avaliação. Desse forma, para atingir um nível de integração que permita um fluxo interativo e efetivo de informações entre os mesmos, diversas barreiras precisam ser superadas (Mckinney et al., 1999).

A avaliação de alocação de água em uma bacia hidrográfica, por exemplo, além dos aspectos econômicos relacionados pelo seu uso ou escassez, pode estar condicionada a restrições ambientais (disponibilidade hídrica e/ou de qualidade), o que faz necessária a avaliação conjunta de modelos econômicos e modelos hidrológicos ou de qualidade da água para simular os possíveis impactos decorrentes no processo de alocação.

Dentro da abordagem da modelagem hidroeconômica, as plataformas de simulação da qualidade de água mais sofisticadas (como a exemplo dos modelos QUAL-2E, HEC-RAS, entre outros) são comumente empregados com o objetivo de avaliar os impactos de alocações de água posteriormente ao processo de otimização. É o caso de Azevedo et al., (2000) que aplicaram o modelo QUAL-2E-UNCA para simular a qualidade da água de um modelo de otimização de alocação de água. Contudo, softwares de simulação como os apresentados muitas vezes apresentam dificuldade de serem empregados em análises integradas. A restrição quanto à livre comunicação e fluxo das variáveis de entrada e saída entre as diferentes plataformas empregadas para a avaliação ou mesmo pela necessidade dos processos de simulação e otimização se darem de forma simultânea podem ser citadas como exemplos de limitações. Dessa forma, a adoção de modelos de qualidade simplificados passa a ser uma alternativa para contornar essas barreiras.

Em Cai; Mckinney; Lasdon (2003) um modelo hidrológico-agronômico-econômico integrado é desenvolvido, o qual, além de alocar água entre os usuários de maneira a maximizar o benefício econômico, inclui simulação da qualidade da água através do constituinte salinidade. O mesmo é modelado na rede da bacia hidrográfica, por meio de balanço de massa, bem como nas zonas de raízes dos cultivos, resultando em penalidades de acordo com a concentração de salinidade efluente. Moraes et al. (2008) desenvolveram um modelo econômico hidrológico integrado a partir da fundamentação teórica do modelo proposto por

(Cai; Mckinney; Lasdon, 2003), de maneira a identificar a alocação ótima de água entre os diversos usuários da bacia hidrográfica do Rio Pirapama no estado de Pernambuco. Por apresentar problemas de poluição em diversos trechos relacionados à fertirrigação das áreas de cana, o modelo também incorpora a avaliação de qualidade da água, através dos parâmetros Demanda Bioquímica de Oxigênio (DBO) e Oxigênio Dissolvido (OD), por meio do emprego de equações de depuração Streeter-Phelps. Dessa forma, o modelo não só determina a alocação ótima de água por usuário, mas também o volume do efluente a ser introduzido como fertirrigação nas áreas plantadas de cana de acordo com requerimentos de qualidade.

Bandeira (2010) desenvolveu um modelo matemático de otimização de alocação de água entre as atividades de mineração de carvão e de cultivo de arroz irrigado na Bacia do Rio Sangão no estado de Santa Catarina sob consideração da disponibilidade hídrica para irrigação e despejo de efluentes. O modelo busca maximizar a renda líquida gerada, devendo os parâmetros de qualidade da água serem respeitados. Amorin Filho (2013) desenvolveu um modelo hidroeconômico para a bacia do rio Tapacura em Pernambuco, o qual é capaz de determinar a alocação econômica ótima de água para seus usuários existentes, respeitando restrições de quantidade e qualidade da água. Os resultados demonstraram que restrições de quantidade não comprometeram o atendimento das demandas requeridas, entretanto, ao ser executado o modelo com restrições de qualidade, houve uma redução nos benefícios dos usuários atuais, uma vez que alguns usos devem ser restringidos para que o rio possa se recuperar através do processo de autodepuração.

Molinos-Senante et al. (2014) propuseram um modelo de otimização para alocação de água que determina o volume de água a ser fornecido para cada usuário, com base em maximização do benefício, e identifica de que fonte o fornecimento deve ser realizado com base em requerimentos de qualidade. Davidsen et al., (2015) propuseram um modelo de otimização que permite comparar os impactos econômicos do cumprimento de vários graus de qualidade de água, considerando descarga de efluentes e tratamento de água no problema de alocação de água através da equação de Streeter-Phelps para calcular as concentrações de oxigênio dissolvido no rio.

Dentro deste contexto, este trabalho tem por objetivo avaliar o emprego de um modelo simplificado para simulação da qualidade da água na Bacia do Rio dos Sinos localizada no estado do Rio Grande do Sul, Brasil, possibilitando o seu emprego como parte integrante de modelos hidroeconômicos e no planejamento e gestão de recursos hídricos. A fim de testar a adequabilidade do modelo proposto, foram realizadas comparações com resultados obtidos para o mesmo cenário pelo programa computacional de modelagem hidrodinâmica HEC-RAS 5.05

previamente calibrado a partir de dados de topobatimetria, imagens de satélite e qualidade da água da rede de monitoramento do Rio dos Sinos. O programa também foi utilizado com o objetivo de avaliar as características hidráulicas do escoamento, tais como, velocidade e tempo de percurso, possibilitando a incorporação destes parâmetros nas equações do modelo simplificado.

4.2 Área de Estudo

A Bacia Hidrográfica do Rio dos Sinos localiza-se na porção leste do Estado do Rio Grande do Sul (Figura 17). O Rio dos Sinos deságua no Delta do Rio Jacuí, onde também afluem os Rios Caí e Gravataí. Área da Bacia é de 3.696 km^2, o que corresponde aproximadamente a 4,4% da área da Região Hidrográfica do Guaíba e a 1,3% da área do Estado do Rio Grande do Sul. 32 municípios estão localizados total ou parcialmente na bacia, abrigando uma população estimada em 1.350.000 habitantes. As principais demandas de água na bacia são para abastecimento urbano (35%), irrigação de arroz (53%) e usos industriais (11%) (Profill, 2014b).

De uma maneira geral, a bacia do Sinos pode ser dividida em três grandes compartimentos (Alto, Médio e Baixo) em que se destacam condições relativamente homogêneas de relevo e uso do solo. O Alto Sinos, em que são notadas as maiores altitudes, é delimitado desde as nascentes, a montante da sede urbana de Caraá, até o rio da Ilha. O Médio Sinos é formado essencialmente pelo segmento correspondente à Bacia do Rio Paranhana e contribuintes menores nas margens esquerda e direita. E por último, o compartimento do Baixo Sinos, que pode ser lançado a partir da região de Sapiranga e Campo Bom até foz, é onde estão localizadas as sedes urbanas das maiores cidades da Bacia (Novo Hamburgo, São Leopoldo, Esteio, Sapucaia do Sul e Canoas). A divisão hidrográfica em alto, médio e baixo sinos também é considerada pelo Plano de Bacia como base para a discretização em unidades de estudo (sub-bacias) e realização dos cálculos de disponibilidade hídrica e demandas.

O forte e rápido desenvolvimento urbano e industrial ocorrido nas últimas décadas, não acompanhado por investimentos compatíveis de controle de poluição, resultaram em problemas críticos de poluição. O Rio dos Sinos é posicionado como o quarto mais poluído do Brasil (IBGE, 2010), sendo que apenas 4.5% população urbana total da bacia conta com sistema de coleta e tratamento de esgotos (Concremat, 2014).

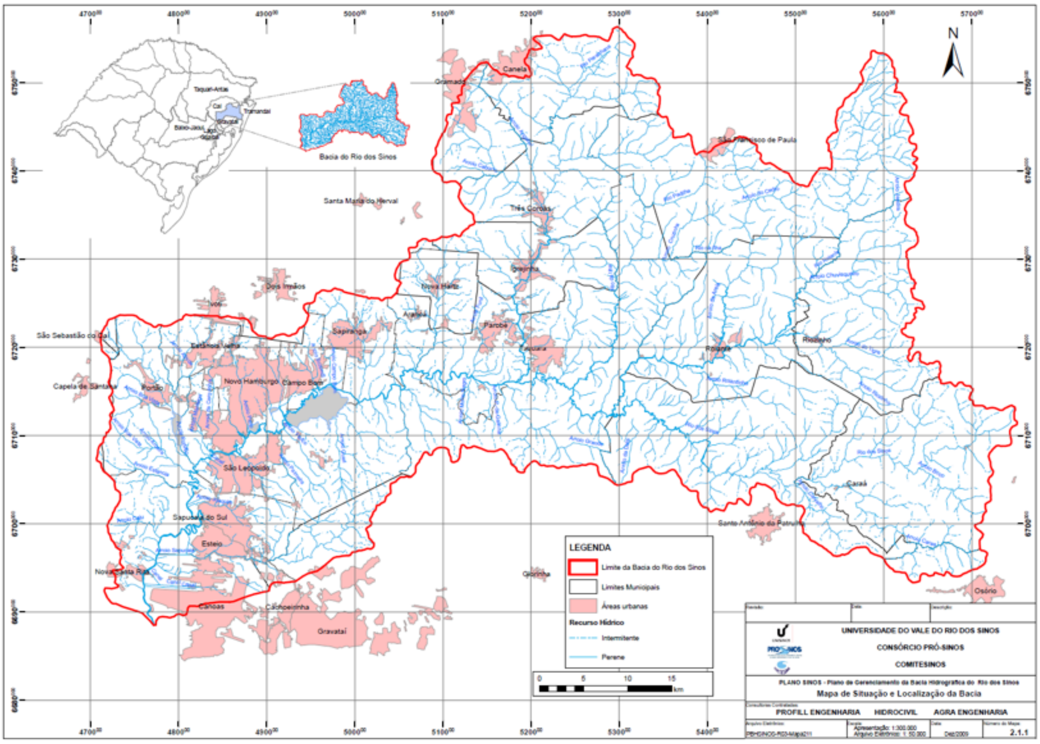

Figura 17: Situação e delimitação da Bacia hidrográfica do Rio dos Sinos.

(Fonte: PROFILL (2013)

4.3 Fundamentação teórica do modelo de simulação da qualidade

Segundo a Lei da Conservação, uma propriedade conservativa (energia, massa e momento) não pode ser criada ou destruída, apenas transferida ou transformada (Martin & McCutcheon, 1999). Modelos de qualidade da água são baseados neste princípio, podendo este ser expresso em termos quantitativos como uma equação de balanço de massa para um volume de controle finito englobando todas as transferências de matérias através das fronteiras do sistema e todas as transformações ocorridas dentro do sistema (43) (Chapra, 2008).

$$Acumulado = transporte \pm cargas\ externas \pm reações \tag{43}$$

A advecção e a difusão são os mecanismos responsáveis pelo transporte de massa. A advecção é resultante do fluxo unidirecional bem definido (movimento do fluido), enquanto que a difusão se refere ao movimento da massa devido ao movimento aleatório das moléculas pela existência de uma diferença de concentração (Martin & McCutcheon, 1999). Considerando um fluxo unidimensional em x, ao incorporar esses mecanismos na equação da conservação de massa, a mesma resulta na equação (44).

$$\frac{\partial C}{\partial t} = -u.\frac{\partial C}{\partial x} + D.\frac{\partial^2 C}{\partial x^2} \pm \: cargas\: externas \: \pm \: reações \qquad (44)$$

Onde: C = concentração da substância (M L^{-3}); t = tempo (T), u = velocidade (L^2 T^{-1}); D = coeficiente de difusão (L^2 T^{-1}).

Ao englobar as cargas externas que entram e saem do sistema, representadas por tributários e fluxos laterais, tais como o lançamento de efluentes e/ou captações de água, a equação de conservação de massa pode ser escrita conforme (45).

$$\frac{\partial C}{\partial t} = \frac{W_{in}}{V} - \frac{W_{out}}{V} - u.\frac{\partial C}{\partial x} + D.\frac{\partial^2 C}{\partial x^2} \pm \: reações \qquad (45)$$

Onde: W_{in} = carga da substância que entra no sistema (M T^{-1}); W_{out} = carga da substância que sai do sistema (M T^{-1}).

Por fim, uma substância ainda pode sofrer reações químicas e bioquímicas transformando-se em outros componentes, bem como pode sofrer mecanismos físicos de sedimentação e volatilização. As cinéticas das reações não conservativas podem ser expressas quantitativamente pela lei de ação das massas, a qual coloca que a taxa de variação da concentração é proporcional à concentração dos reagentes (Chapra, 2008). Considerando apenas um reagente, a equação pode ser representada conforme (46).

$$\frac{dC}{dt} = -k.C^n \qquad (46)$$

Onde: n = ordem; k = coeficiente de decaimento (T^{-1}).

Reações de ordem n = 0, 1 e 2 são as mais comumente empregadas em águas naturais, sendo os parâmetros de qualidade, Demanda Bioquímica de Oxigênio (DBO) e patógenos (coliformes termotolerantes ou fecais), por exemplo, representados por reações de primeira ordem. Resolvendo (4) analiticamente para n = 1, é possível obter a concentração no final do tempo t simulado (47).

$$C = Co.e^{-k.t} \qquad (47)$$

Onde: Co = concentração inicial da substância em t = 0 (M L^{-3}).

O coeficiente de decaimento da DBO (k_{DBO}) é composto por duas parcelas, uma referente à taxa de decomposição (k_d) e outra à taxa de sedimentação (k_s). O coeficiente de decaimento para patógenos (k_{col}) é, por sua vez, composto por três parcelas, mortalidade, perda devido radiação solar e perda por sedimentação (Chapra, 2008). De acordo com Chin (2012) os valores de k_d podem variar de 0,05 (rio não poluído) a 0,7 (efluentes não tratados), enquanto que para k_{col} a variação situa-se na faixa de 0,8 a 5,5 (Thomann & Mueller, 1987).

4.4 Descrição do modelo simplificado proposto

O modelo de simulação da qualidade da água proposto apresenta algumas simplificações, a fim de facilitar o seu emprego em rotinas de programação e consequente uso em modelos hidroeconômicos, entres outras ferramentas de gestão de recursos hídricos. A primeira delas é a consideração de que o mecanismo de transporte por difusão/dispersão tem uma representatividade muito inferior à advecção, sendo, portanto, excluído do equacionamento. A segunda consideração é em referência à avaliação do escoamento apenas em regime permanente (estacionário). Ou seja, é considerado que a massa se mantém constante ao longo do tempo (equilíbrio dinâmico), não havendo acúmulo no sistema. Por último, não são avaliadas alterações de temperatura nas reações cinéticas, mantendo a mesma como um parâmetro constante no modelo, bem como não são consideradas perdas por evaporação. Com base nas simplificações adotadas, a equação resultante do modelo pode ser expressa por (48).

$$0 = W_{in} - W_{out} - u.V.\frac{\partial C}{\partial x} - k.C.V \tag{48}$$

Ao dividir o corpo hídrico em segmentos ou volumes de controle *n* (Figura 18), o equacionamento resulta em (49).

$$Q_{n,n+1}.C_n = Q_{n-1,n}.C_{n-1} + W_{in} - W_{out} - k_n.C_n.V_n \tag{49}$$

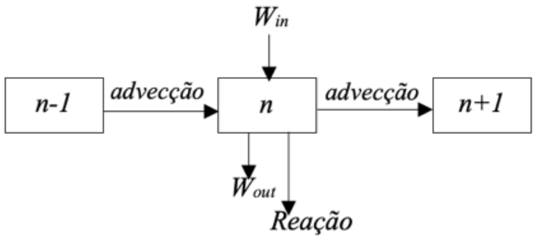

Figura 18: Balanço de massa simplificado para um volume de controle.
(adaptado de Chapra (2008))

As cargas externas *W* são representadas pelo produto das vazões resultantes de captações e lançamentos dos diferentes usuários com a sua concentração associada. As captações são discretizados pela variável *s*, enquanto que os lançamentos são resultantes de uma fração de retorno *ret*. O índice *i* caracteriza os diferentes usuários, como exemplo, urbano, irrigante e industrial. Todas as cargas situadas dentro de cada trecho *n* são somadas de forma a caracterizar um lançamento único pontual ao final do mesmo. Para representar o processo de autodepuração do rio, a concentração devido às cargas lançadas é corrigida pela equação (47).

Dessa forma, o transporte de massa é representado pelas equações (50) a (53).

$$Q_n . C_n = Q_{n-1} . C_{n-1} . F + W_n + W_{natural_n} \qquad \forall\, n \qquad (50)$$

$$W_{natural_n} = (Q_{p_n} - Q_{p_{n-1}}) . C_{natural_n} \qquad \forall\, n \qquad (51)$$

$$W_n = \sum_{i=1}^{I} ret_{i,n} . s_{i,n} . D_i . F - \sum_{i=1}^{I} s_{i,n} . C_{n-1} . F \qquad \forall\, i,n \qquad (52)$$

$$F = e^{-k . t_{travel_n} . LP} \qquad \forall\, n \qquad (53)$$

Onde:

Índice *n* representa a divisão do rio em trechos (*n = 1, 2, …, n, N*)

Q_n é a vazão resultante no final de cada trecho *n* [M^3T^{-1}]

C_n é a concentração da substância no final de cada trecho *n* [$M\,L^{-3}$]

W_n é a carga total da substância resultante de captações e lançamento no trecho *n* [MT^{-1}]

$W_{natural_n}$ é a carga natural (de base) da substância no trecho *n* [$M\,T^{-1}$]

Q_{p_n} é a vazão com probabilidade de excedência *P* no final do trecho *n* [M^3T^{-1}]

$C_{natural_n}$ é a concentração natural (de base) da substância no trecho *n* [$M\,L^{-3}$]

$ret_{i,n}$ é a fração decimal de retorno (lançamento) do usuário *i* no trecho *n*

$s_{i,n}$ é a vazão captada pelo usuário *i* no trecho *n* [M^3T^{-1}]

D_i é a concentração da substância no lançamento pelo usuário *i* [$M\,L^{-3}$]

F é o fator de depuração

LP é a fração decimal que representa a posição do lançamento dentro do trecho *n*

Como exemplo, assumiu-se que os lançamentos são realizados a 0,2 decimais do comprimento total do trecho (LP = 0.2). Contudo, para calcular a depuração da carga resultante do transporte entre os trechos *n-1* e *n*, o LP deve ser igual a 1.

4.5 Hec-Ras 5.05

O Hec-Ras é um programa computacional gratuito que permite simular escoamentos em rios e canais em regime permanente e não permanente, bem como realizar estudos de análise de transporte de sedimentos e qualidade da água. Entre os constituintes de qualidade da água passíveis de simulação nesta versão estão: algas, oxigênio dissolvido, demanda carbonácea de oxigênio dissolvido, ortofosfato dissolvido, fósforo orgânico dissolvido, nitrogênio amoniacal dissolvido, nitrito dissolvido, nitrato dissolvido e nitrogênio orgânico dissolvido. O programa ainda permite a modelagem de constituintes arbitrários através da inserção de parâmetros relacionados às taxas cinéticas de decaimento ou crescimento (USACE, 2016).

4.6 Metodologia

4.6.1 Cenário avaliado

O cenário atual da qualidade da água do Rio dos Sinos em períodos críticos de baixa vazão foi simulado em ambos modelos (HEC-RAS e Simplificado) através da incorporação dos lançamentos de esgotos sanitários dos municípios integrantes da bacia e vazões mínimas de permanência Q_{90}. Os constituintes de qualidade da água avaliados foram DBO e Coliformes termotolerantes por serem representativos quanto à poluição doméstica. Não foram avaliados lançamentos industriais ou outros de ordem difusa.

4.6.2 Dados de entrada

Os trechos do Rio dos Sinos utilizados nas simulações englobam a divisão médio e baixo sinos, com uma extensão total de 108 km. A escolha deu-se pela disponibilidade de dados topobatimétricos nos mesmos (levantamento realizado como parte integrante dos estudos do

Plano de Bacia do Rio do Sinos), bem como por ser mais representativos quanto a concentrações populacionais (maiores captações e lançamentos de efluentes).

A qualidade da água do Rio dos Sinos foi avaliada com base nos dados da rede de monitoramento gerenciada pela Fundação Estadual de Proteção Ambiental do Rio Grande do Sul (FEPAM) disponibilizados pela própria entidade. O Rio dos Sinos conta com 12 pontos de monitoramento, sendo que deste total 9 estão distribuídos ao longo da calha na extensão de 8 e 96 km da foz. O período de monitoramento ocorre desde 1990, porém trabalhou-se apenas com as médias dos dados de 2005 a 2015, por ser um período mais recente, com boa representatividade e que não apresenta grandes desvios populacionais que poderiam ocasionar variações acentuadas de carga orgânica na bacia.

De forma a ser simulada condição mais restritiva de qualidade da água (período de estiagem), as vazões adotadas na simulação foram as mínimas de permanência Q_{90} de acordo com estudo apresentado no diagnóstico do Plano de Bacia. O estudo foi baseado na série de vazões diárias observadas para o período de 1965 a 2007 da estação Fluviométrica de Campo Bom e transferidas espacialmente através da metodologia de vazão específica. Como condição de jusante ocasionada pela oscilação de nível no Delta do Jacuí e Lago Guaíba considerou-se o nível que poderia apresentar maior condição de restrição, nível baixo, H = -0,2 m (Profill, 2014b). A tabela 6 apresenta os dados de entrada empregados.

Tabela 6: Dados de entrada para o modelo simplificado.

Divisão Principal	Trechos	Comprimento dos trechos (km)	Vazão disponível (Q_{90}) (m^3/s)	Tempo de percurso t_{travel} (d)
Médio	6	108 - 97	12,82	0.30
	5	97 - 86	13,81	0.44
	4	86 - 75	15,02	0.33
Baixo	3	75 - 50	16,86	1.25
	2	50 - 25	18,48	1.85
	1	25 - 00	20,05	5.36

4.6.3 Modelagem da qualidade da água

Por meio de imagem orbital LANDSAT 8 o perfil do rio dos sinos foi traçado no programa HES-RAS, sendo, em seguida, inseridas as 23 seções transversais resultantes do levantamento topobatimétrico ao longo da calha menor do rio. As seções foram ainda interpoladas a uma distância de 50 m, a fim de melhor cobrir as variações do terreno. Os dados de vazão mínima (Q_{90}) foram distribuídos ao longo de sua extensão de acordo com a localização do exutório de cada sub-bacia correspondente. A simulação hidráulica possibilitou encontrar os valores de velocidade em cada seção interpolada e assim obter o tempo de percurso entre as mesmas. Este tempo foi então posteriormente utilizado no modelo simplificado na equação de decaimento (53).

Para a simulação da qualidade da água decorrente das cargas domésticas de esgoto sanitário, os lançamentos foram calculados de acordo com a população residente na bacia (Plano de Bacia) e distribuídos ao longo de cada trecho *n*, conforme posicionamento das manchas urbanas. Os municípios de Parobé, Araricá, Sapiranga e Campo Bom foram representados de forma pontual, enquanto que os demais (Novo Hamburgo, Estância Velha, São Leopoldo, Portão, Sapucaia do Sul, Esteio e Canoas) foram representados por lançamentos distribuídos. Foram considerados valores de geração de esgoto per capita de 160 L/hab.dia, concentração de DBO de 312 mg/L e Coliformes termotolerantes 1.10^7 UFC/100 mL (Von Sperling, 2007).

Como concentração de base da bacia dos parâmetros de qualidade avaliados (condição natural do rio sem lançamento de cargas), foram utilizadas as concentrações médias da estação de monitoramento SI096 da Fepam, a qual localiza-se próxima à cabeceira do comprimento simulado.

Realizada a calibração do modelo de qualidade no programa HEC-RAS, foram utilizados os valores de tempo de percurso de cada trecho *n* e os parâmetros cinéticos (k_{DQO} e k_{col}) resultantes para simular o mesmo cenário pelo modelo simplificado elaborado em MATLAB.

O Rio do Sinos foi dividido em um total de 9 trechos com diferentes comprimentos. Primeiramente dividiu-se o rio em 3 trechos principais, alto, médio e baixo sinos, conforme descrito no Plano de bacia. Posteriormente cada trecho foi ainda subdividido em outros trechos de igual comprimento totalizando 9 trechos. Destes, apenas os 6 primeiros foram utilizados na modelagem (correspondentes ao baixo e médio sinos). Os volumes de controle (divisão de trechos *n*) adotados no modelo simplificado são maiores que os empregados por um modelo computacional, porém destaca-se que, uma vez validado para a divisão proposta, a discretização em mais trechos pode ser facilmente incorporada no modelo simplificado.

As cargas situadas dentro de cada trecho *n* foram somadas de forma a caracterizar um lançamento único e pontual. Seguindo a mesma lógica, as distribuições das vazões foram simplificadas de maneira a atender aos tributários como um único fluxo lateral ao final do trecho *n*.

Os resultados obtidos foram comparados, aplicando o coeficiente de determinação r^2 (54), com o objetivo de testar a adequabilidade da simplificação para o caso proposto (Waseem et al., 2017). Por fim, também foram utilizadas as métricas *Nash-Sutcliffe Efficiency* (NSE) e *relative root mean square error* (RRMSE) (Krause et al., 2005), a fim de verificar variações entre métricas (tabela 7).

$$r^2 = \left(\frac{\sum_{n=1}^{N}(O_n - \bar{O})(P_n - \bar{P})}{\sqrt{\sum_{n=1}^{N}(O_n - \bar{O})^2}\sqrt{\sum_{n=1}^{N}(P_n - \bar{P})^2}} \right)^2 \tag{54}$$

Onde: O = Concentração observada; P = Concentração prevista; $\bar{O} =$ média das concentrações observadas; $\bar{P} =$ média das concentrações previstas.

A fim de avaliar o peso que a parcela de transporte difusivo exerce na equação do transporte de massa, uma vez que a mesma foi desconsiderada no modelo simplificado, foram realizadas simulações com ambos modelos (HEC-RAS e Simplificado) inferindo que os constituintes se comportam como substâncias conservativas (não sofrem depuração). Ou seja, em ambos os modelos, o coeficiente k de decaimento foi igualado a zero. A modelagem da qualidade utilizou vazões mínimas Q_{90}, as quais, por consequência, resultam em velocidades baixas e proporcionam melhor base para análise das influências da difusão no transporte de massa.

4.7 Resultados

4.7.1 Calibração do modelo em HEC-RAS

A Figura 19 apresenta os resultados do ajuste do modelo hidrodinâmico em HEC-RAS para o parâmetro DBO e Coliformes termotolerantes. O modelo conseguiu reproduzir satisfatoriamente os principais comportamentos das concentrações observadas nos pontos de monitoramento, sendo os maiores incrementos observados entre os quilômetros 30 e 50 do rio, devido aos lançamentos das cidades de Novo Hamburgo, Estância Velha, São Leopoldo e Portão. No trecho inferior (30 km até a foz), mesmo havendo contribuições significativas de cargas domésticas das cidades de Sapucaia do Sul, Esteio e Canoas, observou-se um decaimento acentuado das concentrações. Devido ao remanso do Delta do Jacuí, há uma redução da velocidade e consequente aumento do tempo de percurso o que pode influenciar a ocorrência deste decaimento. Comportamento semelhante também foi obtido na simulação da qualidade da água realizada no Plano de Bacia (Profill, 2014b).

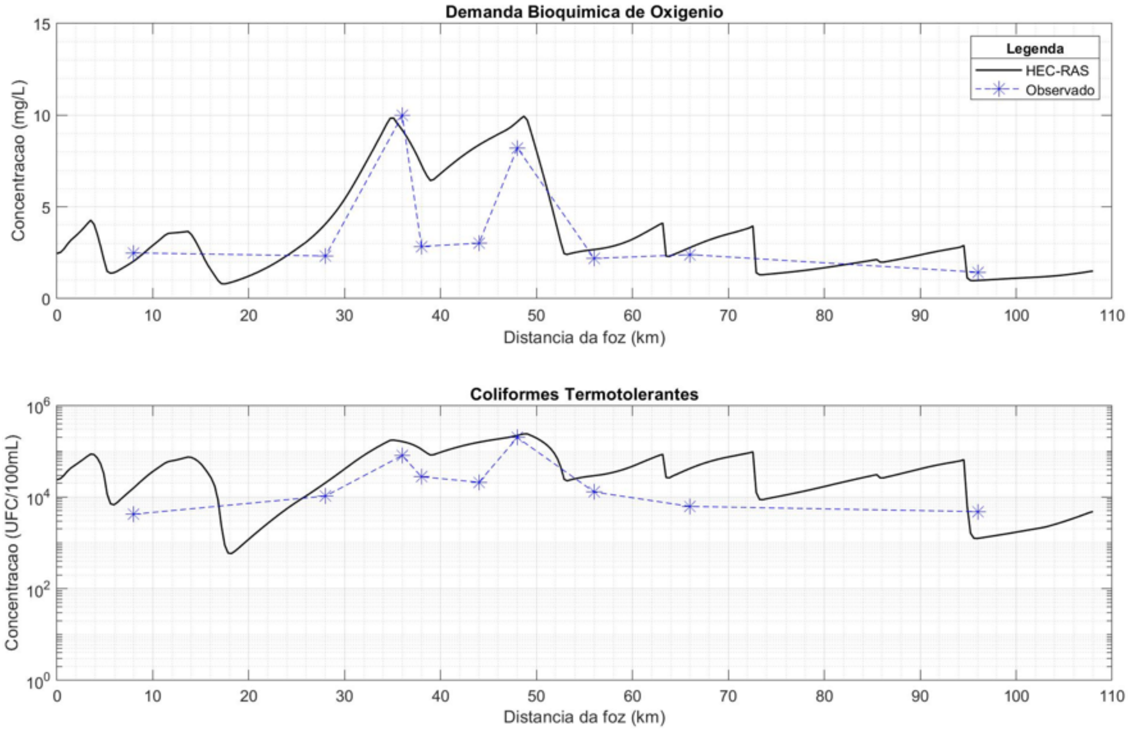

Figura 19: Resultado do ajuste do modelo simulado em HEC-RAS com as médias das respectivas concentrações da rede de monitoramento.

O valor do coeficiente de decaimento (k_{DBO}) para DBO obtido na calibração foi de 1,2 d^{-1}, sendo 0,7 d^{-1} referente à taxa de decomposição (k_d) e 0,5 d^{-1} referente ao termo de remoção por sedimentação (k_s). Ambos se apresentam dentro da faixa de valores citados pela literatura. O coeficiente de decaimento para patógenos (k_{col}) obtido pela calibração foi de 2 d^{-1}, também em acordo com a faixa de variações apresentada pela literatura.

O coeficiente de correlação r^2 para o parâmetro DBO resultou no valor de 0,61, enquanto que para Coliformes Termotolerantes o valor foi de 0,65. Devido ao uso de vazões mínimas Q_{90} na modelagem, os valores das concentrações resultantes podem ser maiores que às médias observadas utilizadas na comparação. Dessa forma, é proposta como análise futura adotar concentrações observadas que sejam mais próximas às condições de vazões mínimas Q_{90} utilizadas na modelagem. Para isso, faz-se necessário a avaliação de dados fluviométricos em conjunto com os dados de qualidade, o que está fora de escopo deste estudo.

4.7.2 Avaliação da parcela difusiva do transporte de massa

Através da figura 19 é possível verificar que a parcela difusiva exerce baixa influência para o caso estudado, sendo que o modelo simplificado consegue acompanhar a tendência de aumento na concentração dos parâmetros avaliados, DBO e coliformes termotolerantes, ao longo de sua extensão da nascente à foz (aqui avaliados de forma conservativa). As principais variações observadas se dão devido ao somatório das cargas e lançamentos ocorrem um único ponto dentro de cada trecho.

O valor do coeficiente r^2 resultante da comparação entre as variáveis resultantes do modelo simplificado e HEC-RAS é de 0.93 para ambos parâmetros de qualidade avaliados. O valor indica uma boa correlação, conforme também visualizado na figura 20.

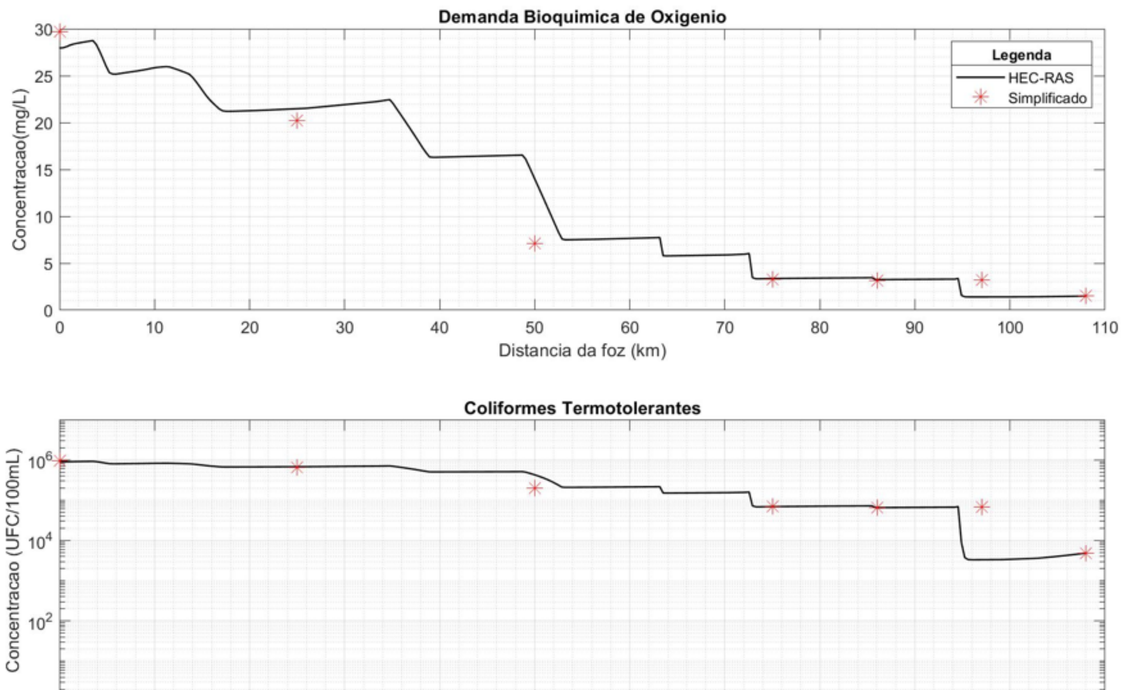

Figura 20: Avaliação do comportamento considerando a inexistência da parcela referente ao transporte difusivo de massa no modelo simplificado.

4.7.3 Comparação das simulações de qualidade da água pelo modelo simplificado e HEC-RAS

A Figura 21 apresenta a comparação entre o modelo calibrado em HEC-RAS e o resultado da simulação simplificada realizada em MATLAB utilizando os valores de k e tempo de percurso do modelo calibrado em HEC-RAS. Como esperado, é observado um leve deslocamento dos picos entre os modelos, principalmente pelo motivo do modelo simplificado considerar o somatório das cargas de cada trecho lançadas em um único ponto.

O deslocamento entre os picos, principalmente o situado a 25 km da foz, resultou numa baixa correlação para ambos parâmetros (r^2 igual a 0,05). Apesar desta variação, é importante destacar que o modelo simplificado consegue acompanhar o aumento e variações de concentrações ao longo da calha de maneira bastante condizente, o que demonstra a coerência das simplificações utilizadas no transporte de massa.

Ao se comparar a correlação das distâncias em que as mesmas concentrações ocorrem entre modelos (como exemplo, a concentração de 9,7 mg/L de DBO ocorre no km 25 para o modelo simplificado e no km 35 para o modelo HEC-RAS), o coeficiente r^2 resulta em 0.93. A diferença de 10 km é inferior ao comprimento total do trecho igual a 25 km adotado no modelo simplificado. Isso demonstra que, apesar do deslocamento no espaço, o resultando se encontra dentro do comprimento total do trecho avaliado. Caso o resultado da concentração seja mais importante que o posicionamento da mesma ao longo do rio, a divisão em 6 trechos proposta é o suficiente. Todavia, se for desejado aumentar o ajuste entre a concentração e o seu posicionamento no espaço, sugere-se aumentar a divisão do rio em mais trechos, a fim de que as cargas dos lançamentos sejam melhor posicionadas. Este último está fora do escopo deste estudo, mas é sugerido para futuros trabalhos.

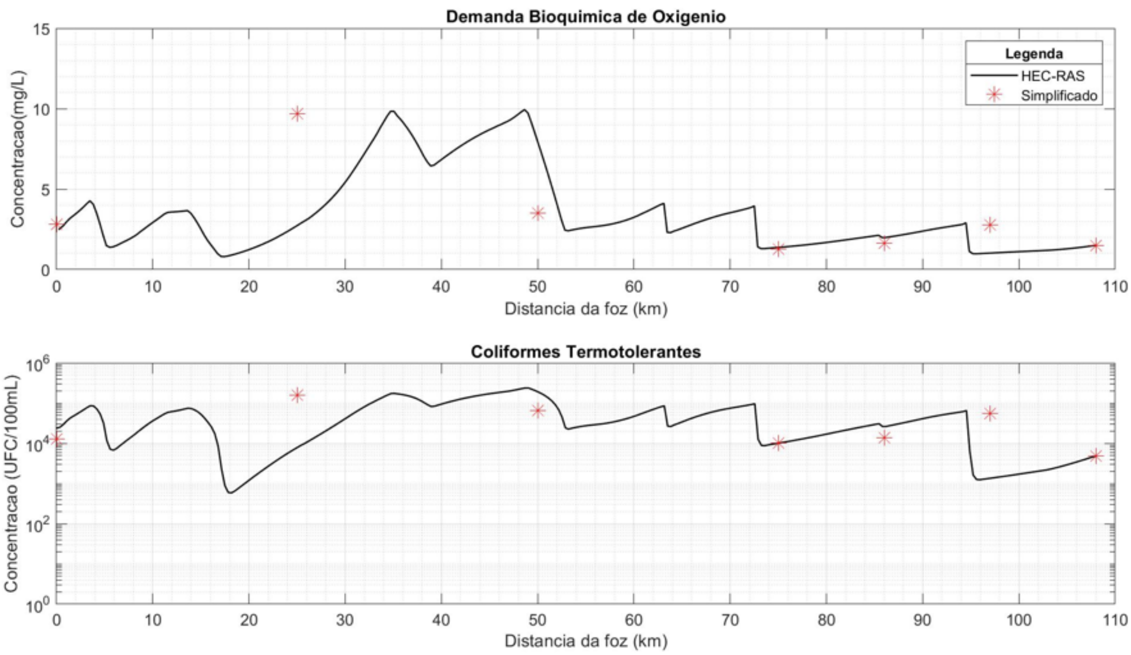

Figura 21: Comparação entre os resultados do parâmetro DBO e coliformes termotolerantes obtidos pelo modelo Hec-Ras e o modelo simplificado.

Tabela 7: Comparação dos resultados entre diferentes métricas.

Comparação	r^2	NSE	RRSME
DBO - calibração	0,61	0,34	0,60
CF - calibração	0,65	-0,15	1,59
DBO – sem reação	0,93	0,92	0,26
CF – sem reação	0,93	0,93	0,29
DBO – entre modelos (concentração)	0,05	-0,26	1,15
CF – entre modelos (concentração)	0,05	0-0,99	1,25
DBO – entre modelos (distância)	0,93	0,91	0,15
DBO – entre modelos (distância)	0,98	0,98	0,07

4.8 Conclusão e recomendações

O presente artigo apresentou a base para formulação e simplificação de um modelo de qualidade da água para o Rio dos Sinos. A adequabilidade do modelo simplificado foi testada através da avaliação do mesmo cenário pelo programa computacional de modelagem hidrodinâmica HEC-RAS 5.05, previamente calibrado a partir de dados de topobatimetria, imagens de satélite e qualidade da água da rede de monitoramento do Rio dos Sinos. Os resultados mostrarem-se satisfatórios quanto às simplificações consideradas. A parcela responsável pelo transporte difusivo de massa apresentou baixa influencia no resultado final, mesmo sendo modelado para vazões mínimas, como é o caso da Q_{90} adotada neste estudo.

O modelo simplificado conseguiu acompanhar as variações de concentração, porém com leve deslocamento da sua posição no espaço (comprimento do rio). A correlação entre os resultados apontou que a divisão em 6 trechos adotada no modelo simplificado não foi suficiente para acompanhar o mesmo comportamento das cargas observados no espaço pelo modelo HEC-RAS.

Desta forma, recomenda-se para futuros trabalhos: (a) discretizar o rio em mais trechos para que os lançamentos possam ser melhor posicionados no espaço; (b) avaliar o processo de calibração utilizando vazões observadas coerentes com vazões mínimas, através da compilação de dados de estações fluviométricas e da rede de monitoramento da qualidade da água; (c) comparar os resultados utilizando as classes da CONAMA 357 (2005) – tabela de contingência.

Por fim, destaca-se que o modelo, dentro de suas limitações, é de fácil aplicação em rotinas de programação diversas, o que facilita o seu emprego em modelos integrados de planejamento e gestão de recursos hídricos, como o apresentado no artigo do capítulo 2, *Reconciling water policies with broader economic development policies through integrated water management instruments.*

4.9 Referências

AMORIM FILHO, C. A. G. DE. **Alocação Econômica Ótima na Bacia do Rio Tapacurá-Pernambuco sob Diferentes Cenários Climáticos**. Tese de doutorado – Centro de Ciências Aplicadas. Universidade Federal de Pernambuco, 2013.

AZEVEDO, B. L. G. T. DE et al. Integration of water quantity and quality in strategic river basin planning. **Water Resources**, v. 126, n. April, p. 85–97, 2000.

BANDEIRA, A. DE P. F. **Alocação Econômica de Recursos Naturais no Contexto de Bacia Hidrográfica: Aplicação na Bacia do Rio Sangão em Santa Catarina.** Porto Alegre: Tese de doutorado. Universidade Federal do Rio Grande do Sul, 2010.

CAI, X.; MCKINNEY, D. C.; LASDON, L. S. Integrated hydrologic-agronomic-economic model for river basin management. **Journal of Water Resources Planning and Management**, v. 129, n. 1, p. 4- 17 ST- Integrated hydrologic- agronomic- economi, 2003.

CHAPRA, S. C. **Surface water-quality modeli**ng. 2. ed. Illinois: Waveland Press, 2008.

CHIN, D. A. **Water-quality engineering in natural systems: fate and transport processes in the water environment**. 2. ed. New Jersy: John Wiley and Sons, 2013.

CONCREMAT. **Plano regional de saneamento básico: relatório do diagnóstico**. Porto Alegre, 2014.

DAVIDSEN, C. et al. Hydroeconomic optimization of reservoir management under downstream water quality constraints. **Journal of Hydrology**, v. 529, n. 1, p. 1679–1689, 2015.

IBGE. **Indicadores de desenvolvimento sustentável: Brasil 2010**. Rio de Janeiro: 2010.

MARTIN, J. L.; MCCUTCHEON, S. C. **Hydrodynamics and transport for water quality modeling**. CRC Press, 1999.

MCKINNEY, D. C. et al. **Modeling Water Resources Management at the Basin Level: Review and Future Directions**. SWIM Paper ed. Sri Lanka: International Water Management Institute (IWMI), 1999.

MOLINOS-SENANTE, M. et al. A management and optimisation model for water supply planning in water deficit areas. **Journal of Hydrology**, v. 515, p. 139–146, 2014.

MORAES, M. et al. Modelo econômico-hidrológico integrado para gestão de bacias hidrográficas: abordagem piece-by-piece para incorporar a capacidade de autodepuração dos rios e limites de trofia dos reservatórios. **Revista Brasileira de Recursos Hídricos**, v. 13, n. 1, p. 43–55, 2008.

PROFILL. **Plano de Bacia do Rio dos Sinos: Relatório final RT1 Fase inicial e Fase A**. Porto Alegre, 2013.

PROFILL. **Plano de Bacia do Rio dos Sinos: Relatório técnico 2 - RT2 Fase B - complementação do enquadramento**. Porto Alegre, 2014.

THOMANN, R. V.; MUELLER, J. A. **Principles of surface water quality modeling and control**. New York: Harper an Row Publishers, 1987.

USACE, U. A. C. OF ENGINEERS. **HEC-RAS river analysis system User's Manual**. Davis, California: 2016.

VON SPERLING, M. **Estudos e modelagem da qualidade da água de rios**. 1. ed. Belo Horizonte: UFMG, 2007.

WASEEM, M. et al. A review of criteria of fit for hydrologycal models. **International research journal of engineering and technology**, v. 4, n. 11, p. 1765–1772, 2017.

KRAUSE, P.; BOYLE, D. P.; BÄSE, F. Comparison of different efficiency criteria for hydrological model assessment. **Advances in Geosciences**, v. 5, p. 89–97, 2005.

CAPÍTULO 5

Conclusões gerais

5.1 Conclusões gerais

Esta dissertação de mestrado foi desenvolvida a partir da hipótese de que *integrar os instrumentos de gestão sob uma visão de planejamento de longo prazo aumenta a efetividade na gestão dos recursos hídricos, auxiliando também a reconciliar o desenvolvimento econômico com a qualidade ambiental*. Os principais aprendizados obtidos com o desenvolvimento deste trabalho são abaixo discutidos.

A avaliação integrada dos instrumentos de gestão dos recursos hídricos permite explorar soluções mais eficientes para as externalidades que permanecem sem resposta em muitos sistemas de água. Emitir outorgas de maneira a satisfazer plenamente demandas econômicas, independentemente da avaliação dos benefícios econômicos auferidos pelo seu uso e impactos ambientais mais amplos (combate à poluição), não se mostrou uma solução economicamente eficiente. É o caso, por exemplo, do usuário urbano, o qual dificilmente tem seu total atendimento contestado, mesmo em detrimento de outras demandas relevantes (tanto ambientais como econômicas), independentemente de quão eficiente a água está sendo usada, bem como se os efluentes produzidos estão sendo coletados e tratados adequadamente.

Algumas regiões da bacia podem ser mais favoráveis que outras na distribuição de outorgas devido aos diferentes impactos na qualidade da água e condições do rio. Dessa forma, além de possibilitar a formulação de diretrizes quanto à distribuição de outorgas (*quanto*, *quando*, *quem recebe* e *onde outorgar*), as reduções no uso de água apontadas pelo modelo como necessárias para se obter um desempenho economicamente mais eficiente, tanto temporalmente como espacialmente, também permitem definir diretrizes relacionadas a prioridades no combate à poluição, uso eficiente da água, e emprego de instrumentos econômicos explorando o custo de oportunidade da água.

A abordagem integrada também permite verificar o distanciamento das metas de enquadramento de soluções ambientalmente e economicamente mais eficientes. Metas de enquadramento menos restritivas não necessariamente produzem maiores benefícios econômicos. Quando as externalidades são consideradas na avaliação, restringir outorgas (conservação da água) em determinadas regiões pode ser uma solução economicamente mais eficiente do que tratar a poluição gerada (internalização dos custos).

A política hídrica entra como elemento chave estratégico neste processo. Deixar mais água no rio de maneira a atender a demandas ambientais e garantir uma maior proteção ambiental apresenta diversos benefícios que vão além da representação como restrição física. Assim, a definição de uma política hídrica traz uma visão clara dos *trade-offs*, tanto ambientais como econômicos, associados a diferentes níveis de preferências e prioridades em relação à qualidade ambiental e ao desenvolvimento econômico.

Por fim, a integração entre diferentes instrumentos de gestão da água, como a determinação de uma política hídrica associada a políticas de desenvolvimento econômico pelos Planos de Bacia, a determinação de metas de qualidade da água e sua reflexão sobre a emissão de outorgas, ainda é pouco avaliada. No entanto, este estudo mostrou que esta integração pode trazer uma visão valiosa para ajudar a encontrar estratégias de alocação de água a longo prazo que sejam menos onerosas para os usuários e ambientalmente melhores, contribuindo para uma gestão mais eficaz dos recursos hídricos.

CAPÍTULO 6

Referências

6.1 Referências mencionadas na introdução

ALMEIDA, C. C. DE. Outorga dos direitos de uso de recursos hídricos. **Revista Jus Navigandi**, v. 8, n. 1, p. 61, 2003.

ANA. **Implementação do enquadramento em bacias hidrográficas no Brasil / Sistema nacional de informações sobre recursos hídricos – Snirh no Brasil: arquitetura computacional e sistêmica**. Brasília, DF: Agência Nacional de águas, 2009.

ANA. **Planos de recursos hídricos e enquadramento dos corpos de água.** Brasília, DF, 2013.

DAVIDSEN, C. et al. Hydroeconomic optimization of reservoir management under downstream water quality constraints**. Journal of Hydrology**, v. 529, n. 1, p. 1679–1689, 2015.

GELT, J. Sharing Colorado River water: history, public policy and the Colorado River compact. **Water Resources Research Center**, v. 10, n. 1, 1997.

GEORGE, B. et al. An integrated hydro-economic modelling framework to evaluate water allocation strategies I: Model development. **Agricultural Water Management**, v. 98, n. 5, p. 733–746, 2011a.

GEORGE, B. et al. An integrated hydro-economic modelling framework to evaluate water allocation strategies II: Scenario assessment. **Agricultural Water Management**, v. 98, n. 5, p. 747–758, 2011b.

HAROU, J. J. et al. Hydro-economic models: Concepts, design, applications, and future prospects**. Journal of Hydrology**, v. 375, n. 3–4, p. 627–643, 2009.

LETCHER, R. A.; JAKEMAN, A. J.; CROKE, B. F. W. Model development for integrated assessment of water allocation options. **Water Resources Research**, v. 40, n. 5, p. 1–15, 2004.

MOMBLANCH, A. et al. Using ecosystem services to represent the environment in hydro-economic models. **Journal of Hydrology**, v. 538, p. 293–303, 1 jul. 2016.

MORAES, M. et al. Modelo econômico-hidrológico integrado para gestão de bacias hidrográficas: abordagem piece-by-piece para incorporar a capacidade de autodepuração dos rios e limites de trofia dos reservatórios. **Revista Brasileira de Recursos Hídricos**, v. 13, n. 1, p. 43–55, 2008.

OECD. Governança dos Recursos Hídricos no Brasil. Paris: OECD Publishing, 2015.

PENGELLY, C. et al. **Managing Water as a constraint to development with decision-support tools that promote integrated planning: The case of the Berg Water management Area**. South Africa: 2017.

PULIDO-VELAZQUEZ, M. et al. Hydro-economic river basin modelling : The application of a holistic surface – groundwater model to assess opportunity costs of water use in Spain. v. 6, 2008.

TILMANT, A. et al. Economic valuation of benefits and costs associated with the coordinated development and management of the Zambezi river basin. **Water Policy**, v. 14, n. 3, p. 490–508, 2012.